

Groundwater use and salinization with grassland afforestation

ESTEBAN G. JOBBÁGY*† and ROBERT B. JACKSON†‡

*CONICET, INTA San Luis, Ruta 7 y Ruta 8, Villa Mercedes, 5730, San Luis, Argentina, †Department of Biology, Duke University, Durham, NC 27708 USA, ‡Nicholas School of the Environment and Earth Sciences, Duke University, Durham, NC 27708 USA

Abstract

Vegetation changes, particularly transitions between tree- and grass-dominated states, can alter ecosystem water balances and soluble salt fluxes. Here we outline a general predictive framework for understanding salinization of afforested grasslands based on biophysical, hydrologic, and edaphic factors. We tested this framework in 20 paired grassland and adjacent afforested plots across ten sites in the Argentine Pampas. Rapid salinization of groundwater and soils in afforested plots was associated with increased evapotranspiration and groundwater consumption by trees, with maximum salinization occurring on intermediately textured soils. Afforested plots (10–100 ha in size) showed 4–19-fold increases in groundwater salinity on silty upland soils but < twofold increases on clay loess soils and sand dunes. Two years of salinity and groundwater measurements at a 40 ha *Eucalyptus camaldulensis* plantation revealed that the plantation reduced groundwater recharge, underwent groundwater discharge on > 50% of the days, and depressed the water table 38 cm on average compared to the adjacent grassland. Soil cores and vertical electrical soundings indicated that $\approx 6 \text{ kg m}^{-2}$ of salts accumulated close to the water table and suggested that salinization resulted from the exclusion of fresh groundwater solutes by tree roots. Groundwater use with afforestation in the Pampas and in other regions around the world can enhance primary production and provide a tool for flood control. However, our framework and experimental data also suggest that afforestation can compromise the quality of soils and water resources in predictable ways based on water use, climate, and soil texture.

Keywords: ecohydrology, evapotranspiration, forest hydrology, pampas, phreatophytes, soil salinity, water table dynamics, woody encroachment

Received 19 December 2003; revised version received and accepted 24 February 2004

Introduction

Changes in vegetation, particularly transitions between grasslands and forests, alter ecosystem water balances and soluble salt fluxes. Compared with grasslands, forests often have greater leaf area and canopy roughness as well as root systems that access deeper water sources (Kelliher *et al.*, 1993; Canadell *et al.*, 1996; Calder, 1998; Jackson, 1999; Schenk & Jackson, 2002). These changes usually result in higher evaporative water losses (Kelliher *et al.*, 1993; Zhang *et al.*, 2001), lower moisture content in the soil and vadose zone (Calder *et al.*, 1993, 1997; Chen *et al.*, 2003), and reduced

groundwater recharge (Le Maitre *et al.*, 1999). At watershed scales, these ecological differences can also affect water yields from tree- and grass-dominated catchments. Paired watershed experiments typically show increased streamflow when forested watersheds are replaced by pastures and grassland (Sahin & Hall, 1996) and decreased streamflow when grassland watersheds are afforested (Scott & Lesch, 1997).

By affecting the amount of water that moves through the vadose zone (i.e. groundwater recharge), grassland–forest transitions can also alter patterns of salt accumulation and transport from stands to regional scales. The most well-known example is the salinization that accompanied the large-scale conversion of woodlands and forests in Australia. In vast areas the replacement of native Australian trees by herbaceous

Correspondence: Esteban G. Jobbágy, tel./fax +54 2652 490269, e-mail: egj@duke.edu

species raised the water table and mobilized deeply stored salts to the topsoil (Schofield, 1992; Pierce *et al.*, 1993). Current estimates suggest that the reversion of this process would only be achieved by reforesting 70–80% of the watershed (George *et al.*, 1999). A similar large-scale salinization mechanism has been proposed for deforested areas of the Sahel (Culf *et al.*, 1993).

The opposite transition, a vegetation shift from grassland to forest, also has the potential to alter groundwater and salt dynamics (Jackson *et al.*, 2000). Afforestation and tree invasion or encroachment affect many grasslands globally (Jackson *et al.*, 2002), especially in the Southern Hemisphere (Rudel & Ropel, 1996; Richardson, 1998; Geary, 2001), but the effects on groundwater and salt fluxes remain poorly understood. Soil and groundwater salinization have been linked to tree establishment on grass-steppes in Russia and pastures in Australia (Heuperman, 1999; Vertessy *et al.*, 2000; Sapanov, 2000), with recharge interruption and groundwater use by trees proposed as potential causes (Heuperman, 1999; Sapanov, 2000). In this paper, we report rapid soil and groundwater salinization with the afforestation of humid grasslands in the Pampas of Argentina. We use this system to evaluate a general ecohydrological framework that predicts the conditions and mechanisms under which tree establishment in grasslands leads to salinization. This general predictive framework should also help researchers predict the

conditions under which both grassland afforestation and deforestation can lead to salinization.

A general framework for predicting salinization

We propose that the salinization of afforested grasslands results from shifts in the water balance and groundwater flow patterns (Fig. 1). In humid grasslands like those in the Pampas, evapotranspiration (ET) < precipitation (P) and hydrological recharge takes place through deep drainage (Fig. 1; surface run-on or runoff are negligible in the Pampas, but our framework can easily be adapted to include these terms). After trees establish and ET increases, hydrological recharge is substantially reduced. Where groundwater is available, trees reverse the relationship between ET and P, initiating a net discharge regime (ET > P) in which groundwater use exceeds deep drainage. The discharge regime is sustained by groundwater recharge and lateral transport from the surrounding grassland. Groundwater use through transpiration leaves dissolved salts behind, either through root exclusion in the soil or in the trees after absorption. If this net discharge regime is sustained, salts accumulate in the system, including the plants, soil, vadose zone, or groundwater. We propose that this process will occur only if three conditions are met: (a) grasslands allow a moderate net hydrological recharge that can be re-

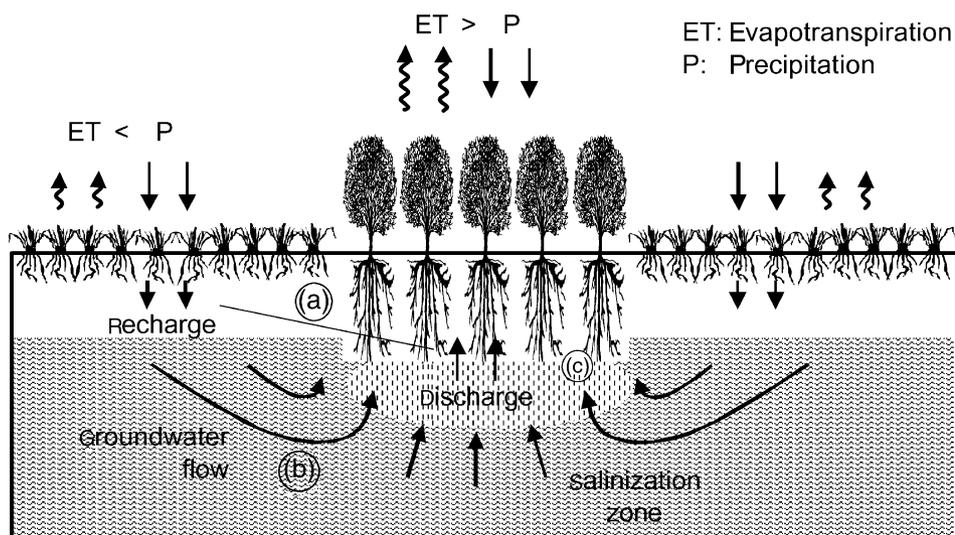


Fig. 1 Hypothetical mechanism of soil and groundwater salinization following the afforestation of grasslands. Tree establishment reverses the net vertical flux of groundwater when groundwater absorption exceeds recharge. Groundwater exits the system through transpiration leaving its salt load in the afforested stand. This process is predicted to occur if the following ecohydrological conditions are met: (a) grasses allow a moderate net hydrological recharge that is fully interrupted by trees, (b) the terrain allows a sustained horizontal flow of groundwater and dissolved salts towards the tree stand, (c) trees access groundwater. The framework described here does not include surface run-on or runoff because these terms are negligible at our sites in the Pampas but can easily be adapted to include them.

versed after tree establishment, (b) the terrain and soil conductivity allow a sustained horizontal flow of groundwater and dissolved salts towards the tree stands, and (c) trees access groundwater (Fig. 1).

Using this framework we test predictions locally and regionally in afforested grasslands of the Pampas. At the regional scale, we predict that salinization will be greatest in intermediately textured soils which are most likely to fulfill conditions (a) and (b) simultaneously. Under the humid climate of the Pampas (800–1000 mm yr⁻¹ mean annual precipitation), condition (a) is unlikely to be met in coarse-textured soils where groundwater recharge and salt-leaching rates can be high. Condition (b) is unlikely on fine-textured soils where poor saturated hydraulic conductivity constrains the lateral transfer of groundwater and accompanying salts from grasslands to plantations. Groundwater access, condition (c), is common in the Pampas with water tables typically <6 m below the surface. Species effects may also be important, particularly those species able to take up water directly from saturated soils (e.g. *Eucalyptus camaldulensis* and *Populus deltoides*). However, salinization may be more general if capillary transport moves sufficient groundwater to the unsaturated soil above the water table.

To evaluate these predictions we assess salinization across 10 afforested sites (20 grassland-plantation pairs) with different soil/sediment types and tree species. At three of these sites representing a sediment texture gradient, we explore aspects of groundwater transport and use by plants. At the local scale we then evaluate the mechanisms of salinization and predict that tree stands will have drier soil, reduced water table recharge after rain, and deeper water table levels than adjacent grasslands. We expect maximum salt accumulation in the zone of groundwater absorption by trees at depths close to the water table.

Methods

The study region

The Pampas occupy a vast sedimentary loess plain in South America (Zárate, 2003). The native vegetation of this temperate humid region is dominated by grasses and was originally treeless (Soriano, 1991; Prieto, 1996). The extremely flat topography and the imprint of wind erosion during the Quaternary created a poor drainage network which, combined with the current humid climate (800–1000 mm mean annual precipitation), results in shallow groundwater across most of the region (Tricart, 1973). Saline/sodic soils are frequent in low areas of groundwater discharge but are absent in uplands (Tricart, 1973; INTA, 1989; Soriano, 1991). Most

of the region is still dominated by a combination of native grasslands, pastures, and annual crops. Tree-dominated stands, however, are present as a result of European settlement. Starting in the 1800s, plantations ranging from 0.1 to 100 ha were established across the region primarily for shade, windbreak, and aesthetics. Such plantations were rarely fertilized or irrigated. Today, commercial afforestation with pines and eucalyptus is increasing rapidly, with counties in Argentina and Uruguay near the Uruguay River having as much as 30% of their area covered by trees (MAGP, 1998; SAGPyA, 2000). Afforestation rates are expected to increase dramatically in the coming decades (Wright *et al.*, 2000).

Regional study

To evaluate the ecohydrological conditions favoring salinization after tree establishment, we measured groundwater salinity in 20 adjacent plantations and grassland pairs at 10 sites with different soil/sediment types including pure loess on plains, loess/granite sequences on hills, and quartz-calcareous sand dunes on the coast, as well as multiple tree species (Table 1). Our sites covered a broad range of soil textures, including loess with silty-clay to clay texture in the Rolling Pampas (Chascomús and La Plata), loess with silty to sandy loam texture in the Flooding and Inner Pampas (Castelli, San Narciso, Guerrero, Dolores, and América), and Atlantic coastal sand dunes (Pinamar and Cariló). (INTA, 1989; Zárate 2003). Sites were ≈2 to ≈550 km away from the Atlantic Ocean. We included one site on a hilltop at Tandil where a thin layer (<0.5 m) of silty loess overlays Precambrian granite, preventing access of roots to groundwater. We sampled plantations of *Eucalyptus camaldulensis* and *Populus deltoides* with the capacity to take up water from saturated soil (Tewari, 1993; Bell & Williams, 1997). We also included pine species such as *Pinus halepensis*, *P. taeda*, and *P. pinaster* which are intolerant to anoxic soil environments and unable to use water from saturated soil (Richardson, 1998). All sites were located in uplands and served as hydrologic recharge zones before afforestation, except for two *Populus deltoides* plantations in lowlands at San Narciso (Table 1).

Plantations were >10 ha in size and were never fertilized or irrigated, as confirmed with land managers and consistent with regional practices. Biomass was not harvested from the stands except at Guerrero and Pinamar, where 20–50% of the Eucalypts were cut previously. Grassland stands were neither fertilized nor irrigated but were typically grazed (see Jobbágy & Jackson, 2003 for further details). We sampled groundwater in grassland/forest pairs 50–100 m on each side

Table 1 Groundwater salinity under tree plantation and control stands across a range of geological settings in the Pampas

Sediment type	Site	Position	Soil type/water depth	Date	Tree species	Age (years)	EC (dS m ⁻¹)	Control stand	EC (dS m ⁻¹)	EC ratio
Silty clays	Chascomús	Upland	Argiaquol 2–6 m	14/11/01	<i>E. camaldulensis</i>	46	1.63	OP	1.39	1.2
				14/11/01	<i>Populus deltoides</i>	46	1.71	id	1.39	1.2
<i>Buenos Aires Formation</i>	La Plata	Upland	Vertic Argiudol 1.5–4 m	14/11/01	<i>Pinus taeda</i>	46	1.51	id	1.39	1.1
				18/7/03	<i>E. camaldulensis</i>	12	1.19	NG	0.98	1.2
Silty and sandy loams	Castelli	Upland	Thapto Hapludol 1–3 m	21/9/01	<i>E. camaldulensis</i>	50	14.51	OP	0.72	20.2
				21/9/01	<i>E. globulus</i>	50	4.55	id	0.72	6.3
<i>Junin Formation</i>	San Narciso	Lowland	Natracuol 0.2–1.5 m	18/7/03	<i>E. globulus</i> - *	dead	0.99	id	0.56	1.8
				21/9/01	<i>Pinus halepensis</i>	50	4.42	NG	0.72	6.2
				18/7/03	<i>Celtis tala</i> [†]	>50	4.91	NG	0.56	8.8
				8/3/03	<i>P. deltoides</i> #1 S [‡]	23	4.52	NG	0.83	5.5
				18/7/03	W		4.44	NG	0.81	5.5
				8/3/03	<i>P. deltoides</i> #3 S	23	4.05	NG	2.10	1.9
Medium sands	Guerrero	Upland	thapto Hapludol 1–3 m	18/7/03	W		3.64	NG	1.75	2.1
				8/3/03	<i>P. deltoides</i> #4 S	23	2.18	NG	1.79	1.2
				18/7/03	W		1.00	NG	1.51	0.7
				2/2/02	<i>E. camaldulensis</i>	96	14.29	NG	1.01	14.1
				8/6/03	<i>E. camaldulensis</i>	42	18.80	OP	0.63	29.8
				8/6/03	<i>Quercus robur</i>	42	10.80	id	0.63	17.1
Atlantic Dune fields	Cariló	Dune slope	Quartzipsamment 2–5 m	3/3/02	<i>E. camaldulensis</i>	30	1.03	NG	0.54	1.9
				15/7/01	<i>Pinus pinaster</i>	30	0.99	NG	0.56	1.8
Rocky subsoil Hills	P Medanos	Dune slope	Quartzipsamment 3–7 m	23/12/01	<i>E. globulus</i>	55	0.65	BD	0.36	1.8
				23/12/01	<i>Pinus pinaster</i>	55	0.58	BD	0.36	1.6
				23/12/01	<i>Pinus pinaster</i>	16	0.62	BD	0.43	1.4
Rocky subsoil Hills	Tandil	Hilltop	Lithic Hapludol Not accessible	8/9/01	<i>E. camaldulensis</i> [§]	50	0.08	NG	0.06	1.4

Sites are grouped according to sediment texture from fine to coarse. Tandil has massive rock <0.5 below the surface. In some sites more than one tree plantation was sampled. Electric conductivity values (EC) of water measured in the field indicate salinity. The electrical conductivity ratio (EC ratio) is the EC of the tree plantation divided by the EC of the grassland control. In all cases plantation and control stands were <200 m away. The textural class of sediments, topographic position, and soil type (USDA 1998) are indicated for each site. Control stands are old pastures (OP), native grasslands (NG), and bare dunes (BD). Some control stands are shared by two or more plantations (id).

**E. globulus* and control stand sampled for second time 2 years after trees were windblown.

[†]Spontaneous grove of small tree *Celtis tala*.

[‡]*Populus deltoides* stands sampled at the end of summer (S) and in mid winter (W).

[§]No access to groundwater. EC values correspond to ephemeral streams.

of the boundary, making boreholes with a hand auger (10 cm outside diameter) that were 50–100 cm deeper than the water table. Salinity was measured with an electric conductivity meter automatically corrected for temperature. To evaluate salinization at Tandil where groundwater was inaccessible, we sampled soils for salinity down to bedrock and also water in two ephemeral streams that originated on the plantation and in a nearby grassland 48 h after rainfall.

We evaluated the effect of afforestation on groundwater fluxes and its interaction with soil texture in *E. camaldulensis* and grassland stands at Chascomús (silty-clay loess), Castelli (silty loam loess), and Pinamar (sand dunes). There, we measured groundwater levels during late spring and summer of 2001 in paired boreholes located in the tree and grassland stands 100 m from the plantation boundary. Surface elevation measured using a water level was used to calculate the absolute gradient of groundwater depth between grassland and plantation boreholes. We examined diurnal fluctuations in groundwater levels as an indicator of groundwater absorption (White, 1932; Freeze & Cherry, 1979). For this purpose we monitored water levels with high temporal (30 min) and altitudinal (<3 mm) resolution during 2–4-week periods without rainfall using pressure transducers (model PS9800, Instrumentation Northwest, Kirkland, WA, USA) connected to permanent dataloggers (CR10, Campbell Scientific Instruments, Logan, UT, USA). At all sites we measured saturated hydraulic conductivity of the sediments *in situ* using well tests (auger-hole method, Amoozegar & Warrick, 1986). We bracketed the potential rates that

capillary rise could achieve under tree plantations at each site with a simple simulation model (*Upflow*, Raes & Deproost, 2003). The inputs for this model were vegetation type, root distribution, soil textural class, saturated hydraulic conductivity, atmospheric water demand, water table depth, and topsoil moisture content. To examine scenarios of capillary rise, we selected forest vegetation and limited rooting depth to 1 m, assuming no direct access of tree roots to groundwater. We ran parallel simulations using two different root distributions, a depth-decreasing function reported by Jackson *et al.* (1996) and a homogeneous distribution. Soil texture was obtained from soil surveys and saturated hydraulic conductivity from our field measurements. We assumed a high water demand (5 mm day^{-1}) in order to find potential rates of capillary transport with soil water at -1.5 MPa . Capillary rise was simulated for a hypothetical water table located 1 or 3 m below tree roots.

Mechanistic local studies

To explore the mechanism of salinization after tree establishment (Fig. 1), we studied changes in soil moisture, groundwater levels, and salt accumulation for 2 years in a 40 ha plantation and surrounding grasslands at Castelli, Buenos Aires province ($-36^{\circ}02.0'$, $-57^{\circ}50.3'$). The site has a mean annual temperature of 15.3°C and mean annual precipitation of 980 mm. This upland, loamy site was planted with *E. camaldulensis* in 1951 at $3 \times 2 \text{ m}$ spacing (Table 1, Fig. 2), with trees neither thinned nor managed after planting.

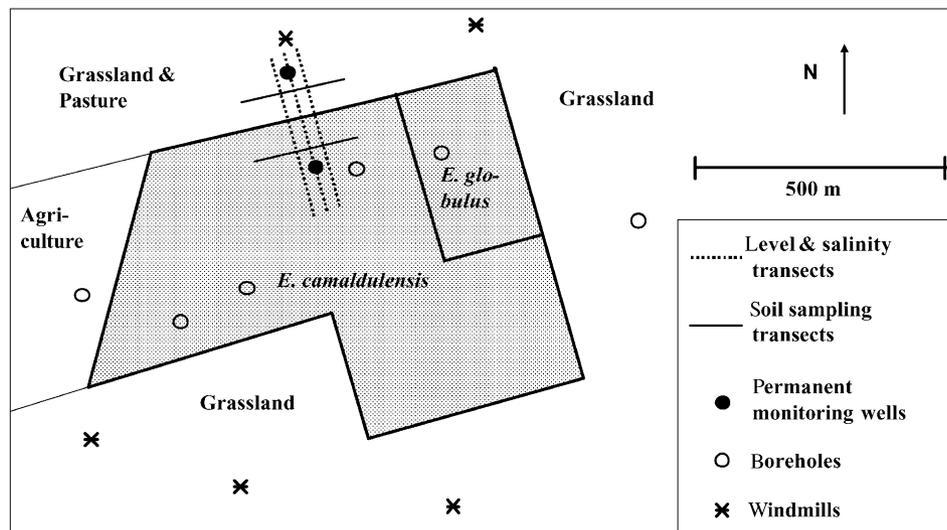


Fig. 2 Plantation and grassland groundwater sampling points at Castelli. Groundwater level and salinity were recorded in 45 boreholes along transects (dotted lines). Two permanent sampling wells were monitored during a 2-year period. Other sampling points for water salinity involved additional boreholes and existing windmills pumping water from 6 to 15 m depth.

Current density is 783 stems ha⁻¹ with a mean tree height \approx 45 m. There is a very sparse understory of the small tree *Celtis tala*. The surrounding area is covered by native grasslands and improved pastures with the typical composition and structure of upland communities in the Pampas, a mixture of C3 and C4 grasses and no bare soil (Perelman *et al.*, 2001). We recorded rainfall throughout the study period with a tipping bucket pluviometer connected to a datalogger in the native grassland. Longer term precipitation records beginning in 1950 were obtained from the town of Castelli, 7 km from the study site.

We compared soil moisture dynamics in the grassland and Eucalypt plantation using time domain reflectometry (TDR). We randomly located three soil pits (1 m wide \times 1 m deep) 50 m from the grassland-forest edge in each stand and measured soil volumetric water content at 5, 15, 30, 50, and 100 cm depth using a TDR sensor with four 6-cm-long probes (ML2x Theta-Probe, Delta-T Devices, Burwell, UK). We took five measurements horizontally at each depth along the wall of the soil pit. One time measurements were repeated in six opportunities (June 2001 – January 2002).

Changes in groundwater depth between the grassland and plantation were characterized using a network of 45 boreholes located in three parallel 350 m transects in September 2002 (winter) and along the central transect in January 2003 (summer) (Fig. 2). These measurements were complemented with a 2-year-long record of groundwater levels in boreholes in the grassland and plantation along the central transect 100 m from the boundary (Fig. 2). At several intervals during this 418-day period we monitored water levels with high temporal (30 min) and altitudinal (<3 mm) resolution using pressure transducers calibrated every two months.

Salt accumulation in groundwater, soil, and vadose zone was evaluated in three ways. Groundwater salinity was measured just below the water table in the network of 45 boreholes described above and was further quantified using 11 deeper boreholes and windmills within the plantation and adjacent grassland (Fig. 2). Groundwater salinity at greater depth along grassland-plantation transects was characterized using vertical electrical soundings (VES). In VES, a current is introduced into the ground using electrodes and the corresponding electric field is measured. Resistance values are obtained from Ohm's law, $R = V/I$, where V is the measured Potential difference and I is the introduced current. These soundings, which characterize the resistance of the terrain to the flow of electrical current, have a variable depth of integration that is controlled by shifting the horizontal spacing of two emitting electrodes. In addition to substrate type, the resistivity of the terrain is negatively associated with

groundwater salinity. We used a Schlumberger arrangement of electrodes with 180 m of maximum aperture (\approx 90 m of penetration). We performed 26 VES along three grassland-forest transects in January 2003 with 50-m VES spacing in distal zones of the transects and 10 m spacing close to the plantation edge (Fig. 2), using an inverse 1-D modeling routine to estimate the thickness and resistivity of the terrain assuming a three-layer structure (Schlumberger software, MicroFEM Development and Support, Amsterdam, The Netherlands).

At a more detailed scale, we quantified the amount and location of salts in the soil/vadose zone of adjacent grassland and plantation stands. Mineral soil was sampled between January and March 2003 in 0.5 m intervals to 6 m depth with a 10 cm diameter auger at four positions in each stand. Sampling points were randomly located along transects parallel to the plantation boundary, 50 m towards the grassland and forest interior (Fig. 2). Bulk density, estimated every 50 cm, ranged from 1.31–1.45 Mg m⁻³ and did not differ significantly between stands. We obtained clean samples of 6-m deep groundwater after casing and sealing the soil sampling holes with PVC pipe and bentonite plugs. Approximately 30 L of water were pumped before each water collection. Root presence/absence was recorded using soil cores and during borehole drilling. Soil samples were analyzed for electrical conductivity and Cl concentration using a 1:2 soil/water ratio shaken for 5 min and equilibrated 24 h. Chloride was measured in the laboratory using an ion-selective electrode (Frankenberger *et al.*, 1996).

Results

Regional study

As predicted by the ecohydrological framework in Fig. 1, groundwater salinization in afforested stands was greater in intermediately textured sediments and lower in coarse and fine soils (Table 1). All upland sites had fresh groundwater (<1.4 dS m⁻¹) under grasslands, confirming the non-saline nature of the original environment. These sites experienced different degrees of salinization according to their soil texture (Table 1). Along a textural gradient, the ratio of groundwater salinity of *E. camaldulensis* plantations compared to adjacent grasslands increased from 1.2 in clay soils (Chascomús and La Plata) to 14–30 in sites with silty to sandy loam sediment (Castelli, Guerrero, América), and back to only 1.9 in the sand dunes (Pinamar) (Table 1). At the hilly site (Tandil), where massive bedrock and topography impeded groundwater use, *E. camaldulensis* showed no salinization of stream water (Table 1) or soils (data not shown – see Jobbágy, 2002).

More detailed observations of groundwater dynamics helped to reveal how sediment texture interacted with salinization. Although tree plantations affected groundwater water levels at all sites, significant groundwater use occurred only in the intermediately and coarse-textured sites (Table 2). Saturated hydraulic conductivity was lowest at the fine-textured site (Chascomús), increased ≈ 25 -fold in the intermediately textured site (Castelli) and increased ≈ 500 -fold in the coarse-textured site at Pinamar (Table 2). According to Darcy's law, groundwater flow increases linearly with the saturated hydraulic conductivity of the phreatic aquifer and the hydraulic gradient or water table depression developed by the trees. Therefore, the larger water table depression observed in the fine-textured site was not sufficient to compensate for its low saturated hydraulic conductivity. Only with a water table depression of ≈ 15 m would the tree plantation at this site have matched the conditions for lateral groundwater and salt transport (and for groundwater use and salt accumulation) observed in the intermediately textured site (condition (b), Fig. 1).

Diurnal fluctuations of groundwater levels were evident only under plantations in the intermediately and coarse-textured sites (Table 2). No fluctuations were observed under grasslands at any site. The consistent pattern of nighttime rise and daytime decline

in groundwater depth under *E. camaldulensis* plantations was sustained during 2-week summer periods without rain at Castelli and Pinamar and averaged 62 and 15 mm daily fluctuations at each site, respectively (Table 2; actual water fluctuations are lower, as adjusted for the specific yield of the soil). Simulations of capillary flow suggested that intermediate- and coarse-textured sediments were able to convey water to tree roots in unsaturated soil 1 m above the water table at a speed that matches typical transpiration rates in forests (Table 2). Mean annual increments of wood biomass increased from $8.1 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ at the fine-textured to 12.6 and $11.4 \text{ Mg Ha}^{-1} \text{ yr}^{-1}$ or $\approx 50\%$ at the intermediately and coarse-textured sites, respectively, also suggesting higher water consumption at these sites (Table 2).

The comparison of plantations with different tree species indicated that salinization can occur under tree species of contrasting tolerance to anoxic soil conditions. In all upland sites with intermediate sediment texture (Castelli, Guerrero, America), salinization was detected under every tree species sampled, including *Pinus halepensis* and other species that are unable to take up water from saturated soil (Table 1). Repeated groundwater measurements of tree plantations in Castelli provided information about the reversibility of the salinization process. We found no changes in groundwater salinity between active and dormant

Table 2 Observed and expected effects of tree plantations on groundwater dynamics along a sediment texture gradient

Variable	Units	Site		
		Chascomús	Castelli	Pinamar
Sediment texture		Silty clay	Silty loam	Sand
Salinization of groundwater *	EC ratio	Low (1.2)	High (20)	Low (2)
Saturated hydraulic conductivity	m day^{-1}	0.04	1	20
Groundwater depression [†]	m	1.72	0.6	0.1
Groundwater fluctuation [‡]				
Tree plantation	mm	no	80	15
Grassland	mm	no	no	no
Capillary rise (simulated)				
Water table 1 m below roots	mm day^{-1}	0.7	4.1	>5
Water table 3 m below roots	mm day^{-1}	0.1	0.4	0.5
Plantation age	Years	47	50	41
Timber biomass [§]	Mg ha^{-1}	383	632	467

Eucalyptus camaldulensis plantations at three sites illustrate the gradient (see Table 1 for plantation details). Saturated hydraulic conductivity (K), and depression, diurnal fluctuations, and salinization of groundwater under tree plantations are shown. Capillary rise for each sediment type was calculated assuming two depths between groundwater and roots using the Upflow model (see Methods and Raes & Deproost, 2003).

*Salinization characterized as plantation:grassland groundwater electrical conductivity ratio.

[†]Absolute grassland-forest level difference in boreholes located 100 m away from the plantation edge and 200 m from each other. Values recorded in 1/12/2002, 1/8/2003, and 11/7/2001 at Chascomús, Castelli, and Pinamar, respectively.

[‡]Mean level differences between night-time maximum and day-time minimum in plantation boreholes.

[§]Obtained from tree diameter, height, and wood density measurements at each site (see Jobbágy & Jackson, 2003).

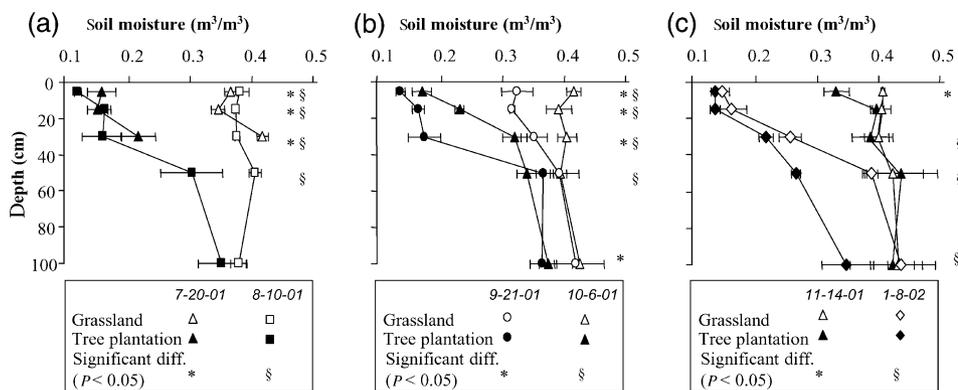


Fig. 3 Soil moisture in the grassland and plantation stands at Castellí. Volumetric water content values were measured at six dates and five soil depths using time domain reflectometry (see Methods). Bars correspond to standard deviation ($n = 3$ in all cases).

periods in two plantations of the deciduous *Populus deltoides* (Table 1), suggesting that the interruption of groundwater use during the winter was not sufficient to leach or dilute salts. In contrast, a decline of groundwater salinity was noticed 2 years after a *E. globulus* stand was felled by wind, indicating a partial reversion of salinization in longer time frames (Table 1).

Mechanistic local studies

We evaluated the mechanisms of salinization in a *E. camaldulensis* plantation and its surrounding grassland at Castellí (Table 1). In agreement with the hypothesis that tree plantations increase evapotranspiration, we found significantly drier soils in the plantation compared to the grassland during most of the sampling time (Fig. 3). Volumetric moisture contents converged for the grassland and the plantation only in November 2001, when soils of both stands approached field capacity at the end of spring rains, and in January 2002, when the topsoil dried out in both stands (Fig. 3c). Soil at 1 m depth remained close to field capacity ($\approx 0.38 \text{ m}^{-3} \text{ m}^{-3}$) most of the time for both stands.

Despite the slightly higher elevation of the plantation (10 cm on average), absolute groundwater levels were 30–60 cm lower than in the grassland (Fig. 4a). The water level gradient was greatest at the plantation edge, where it achieved 1 cm m^{-1} slope during summer (Fig. 4a). During the 2-year measurement period groundwater depth was lower by an average of 38 cm under the tree plantation (Fig. 5). This period was much wetter than normal, with rainfall of 1502 and 1243 mm yr^{-1} in years one and two, compared with a mean annual precipitation for 1950–2002 of 987 mm yr^{-1} (Fig. 5a). Groundwater levels attained maximum height and minimum differences between grassland and plantation after periods of heavy rainfall in the spring of 2001

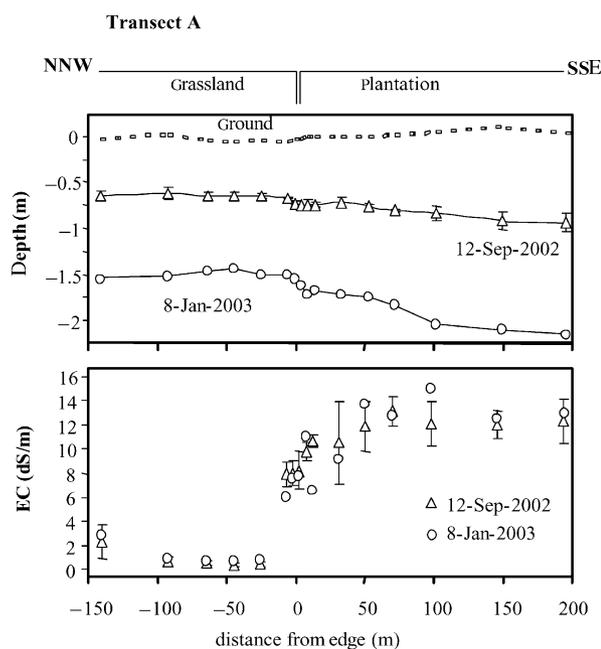


Fig. 4 Groundwater levels and salinity along the grassland-forest transect at Castellí. In the winter (September 2002), sampling values from three parallel transects 50 m apart were averaged and standard deviations are indicated. In the second date only the central transect was measured. Ground levels correspond to the central transect.

and fall of 2002 (Fig. 5b). However, even in these very wet years, water levels dropped during summer, when maximum differences between grassland and plantation were observed.

We hypothesized that salinization occurred because trees initiate a net discharge regime in which groundwater use exceeded recharge. Detailed time courses of groundwater levels measured between August 2001

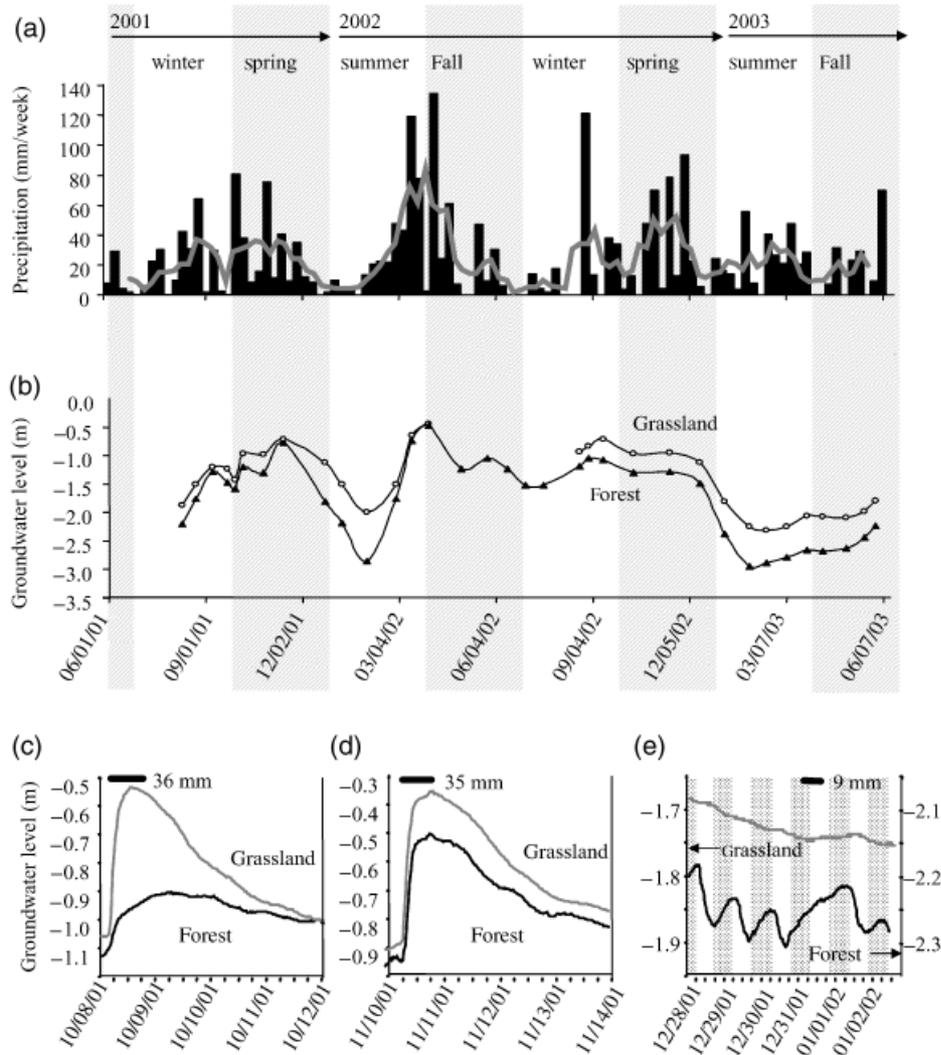


Fig. 5 Precipitation and water table dynamics at Castelli. (a) Weekly precipitation (bars) and 4-week moving average (grey line). (b) Water table levels were measured at permanent monitoring wells (see Fig. 1). Levels correspond to the absolute elevation of the water table with respect to the ground level at the plantation edge. (c–e) Water table levels from permanent monitoring wells measured with high temporal resolution (30 min). (c) Depicts the typical responses after large rain events. (d) Shows the only event after which both grassland and plantation wells showed immediate rise of the groundwater. E. Shows the diurnal fluctuation pattern observed in the plantation during $\approx 50\%$ of the recording period.

and April 2003 showed that trees reduced groundwater recharge and initiated a sustained discharge regime that was absent under grasses (Fig. 5c–e). Recharge interruption was shown by the patterns of groundwater rise following large rainfall events ($>20 \text{ mm day}^{-1}$). After these events, groundwater depth increased immediately in the grassland but not in the forest, as shown for October 2001 (Fig. 5c). During 306 days of paired high-frequency measurements of water table depth, 16 recharge events were detected in the grassland but only one in the tree plantation. In this recharge event, groundwater rose simultaneously in both stands

after the rain of November 10, 2001, at the end of the wettest spring on record (Fig. 5d).

Discharge of groundwater by trees was shown by sustained diurnal fluctuations of the water table only under the plantation (Fig. 3e). Consistent patterns of nighttime increase and daytime decrease of at least 10 mm were observed on 65% of the 418 days of measurements. These fluctuations averaged 49 mm over the whole period (33 and 62 mm during winter and summer days, respectively). Small rain events, as on December 31, 2001, interrupted fluctuations and caused daytime groundwater levels to rise in the tree

plantation without recharging the grassland (Fig. 3e), most likely due to a cessation of groundwater use by trees.

Groundwater was 10–20 times saltier beneath the plantation than in surrounding grasslands ($P < 0.00001$; paired t -test). Electrical conductivity measured at the water table averaged only $\sim 0.7 \text{ dS m}^{-1}$ in the grassland but was 6–15 dS m^{-1} under the plantation (Fig. 4b). Additional samples from two boreholes and five windmills in the surrounding grassland ranged from 0.6–0.8 and 1.1–1.2 dS m^{-1} , respectively. In contrast, samples from four boreholes inside the plantation were 7–19 dS m^{-1} . VES revealed that the terrain under the grassland had uniformly high resistivity and fresh groundwater down to 20–35 m depth (Fig. 6). In contrast, the plantation had a $\sim 4 \text{ m}$ thick layer of low-resistivity saline groundwater under it (Fig. 6). Below 20–35 m and down to the maximum sounding depth of $\approx 90 \text{ m}$, resistivity was uniformly low, indicating deep brackish water under both vegetation types (Fig. 6). The salty layer found at the surface of the aquifer in the plantation overlaid a freshwater layer and was not connected with deep salty water. It is important to note, however, that VES obtained at the core of the plantation on transects A and C indicated higher salinity in the intermediate layer and a shallower upper boundary for the salty aquifer in the innermost sampling point.

In agreement with VES, soil cores showed large salt accumulation in the rooting zone of *E. camaldulensis* (Fig. 7). The deepest tree roots in the plantation were found at 5–6 m depth, coincident with the upper boundary of a fully reduced (gley) sediment layer of blue-greenish color present in all grassland and forest cores, suggesting the long-term maximum watertable depth at the site (Fig. 7b). Cl concentrations were low ($< 15 \text{ mg kg}^{-1}$) throughout the whole profile in the grassland, but were an order of magnitude higher between 2.5 and 3 m depth under the plantation (1300–2200 mg kg^{-1}). The plantation stored $3.8 \pm 0.8 \text{ kg m}^{-2}$ Cl in the soil column down to 6 m of depth ($\approx 6 \text{ kg m}^{-2}$ of salts), 85% of which was between 0.5 and 3 m (Fig. 7a). Groundwater Cl concentrations had inverse vertical gradients in the grassland and the plantation, as expected from recharge and discharge regimes, respectively. In the grassland, groundwater had concentrations of $6.5 \pm 1.3 \text{ g m}^{-3}$ Cl at the water table and $42 \pm 27 \text{ g m}^{-3}$ at 6 m depth. In the plantation, chloride concentrations in groundwater were $2472 \pm 880 \text{ g m}^{-3}$ at the water table and $264 \pm 50 \text{ g m}^{-3}$ at 6 m depth.

Discussion

The conversion of grasslands to plantations in the Pampas triggered intense soil and groundwater salinization in areas with intermediate texture sediments, the

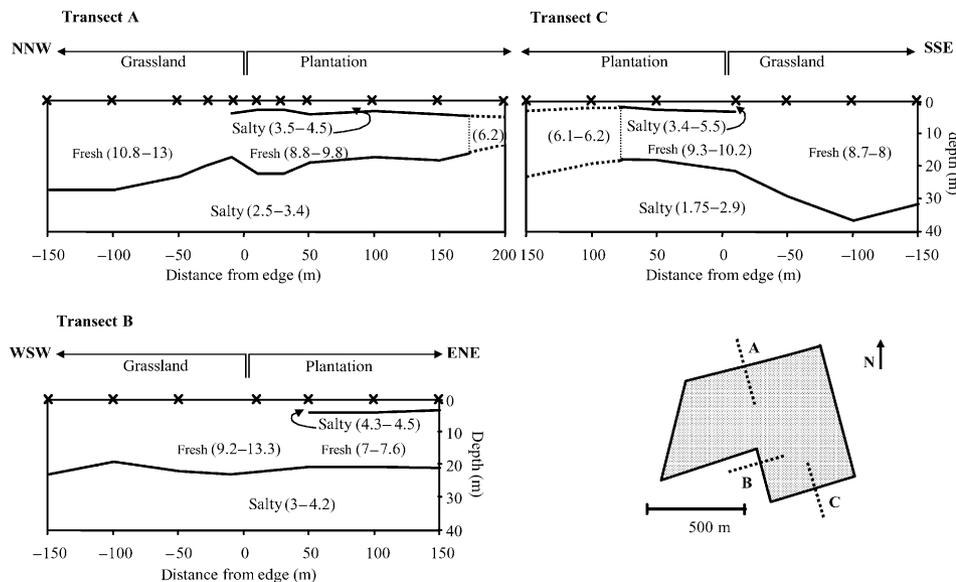


Fig. 6 Resistance to the flow of electrical current in the ground (resistivity) along three grassland–forest transects at Castelli. The location of 26 vertical electrical soundings (VES) performed in January 2003 is indicated with crosses. Each VES was used to infer a 1-D ground resistivity profiles using an inverse modeling technique. Lines represent the modeled boundaries of layers with contrasting resistivity and values show the range of modeled resistivities for each layer in Ohms per meter ($\Omega \text{ m}^{-1}$). Dotted lines depict three individual VES that were not properly explained by a three-layer model (error $> 10\%$) and had lower resistivity in the intermediate zone. Lower resistivity is associated with higher water salinity.

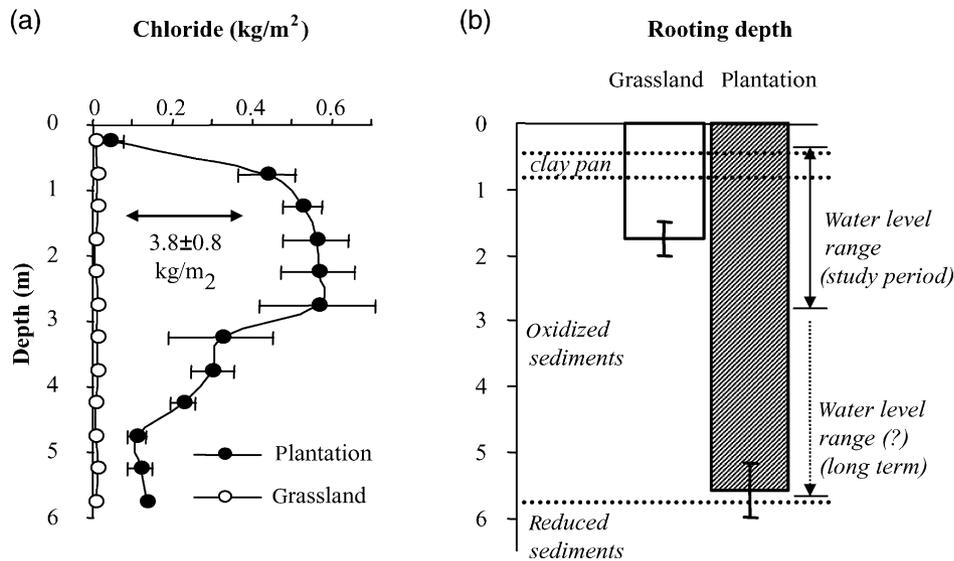


Fig. 7 Soil chloride storage and maximum rooting depth in grassland and forest stands at Castelli. (a) Mean total Cl storage per square meter in each 0.5 m depth interval is indicated, with bars representing standard deviation ($n = 4$). (b) Mean maximum rooting depth and standard deviation were obtained from presence/absence observations during soil sampling cores and borehole augering ($n = 6$). The position of the clay pan (b horizon) and the top of a reduced (gley) sediment layer is shown, as well as the range of water level fluctuations throughout the study period.

most common soil type in the region. We hypothesized that salinization occurred as a result of deep drainage reduction and groundwater absorption by trees, something that was expected to occur where (a) initial vegetation allowed a moderate net hydrological recharge that was fully interrupted by trees, (b) the terrain allowed a sustained horizontal flow of groundwater and dissolved salts towards tree stands, and (c) trees could access groundwater (Fig. 1). We found groundwater salinization in tree plantations where these conditions were met. All tree species triggered salinization on intermediately textured soils, including some that do not tolerate anoxic soil conditions (Table 1). Simulations suggested that capillarity could have conveyed sufficient groundwater (and salts) to roots in the unsaturated soil (Table 2). Salinization was smaller or absent where conditions (a), (b), or (c) failed, including in sand dunes (a failed), in fine loess where saturated water transport was probably limited by low conductivity (b failed), or in hilltop soils overlaying massive bedrock where groundwater was not accessible (c failed) (Tables 1 and 2).

Our mechanistic local study confirmed that tree plantations in the Pampas have a high evaporative potential realized through groundwater use. Despite the extremely rainy conditions that prevailed during the study (the wettest 2-year period in 50 years of records), the Castelli plantation was able to reverse vertical groundwater fluxes, maintain low levels of soil

moisture, and interrupt recharge almost completely, discharging groundwater during more than half of the study period (Fig. 5). The diurnal fluctuations of groundwater level presented here showed a strong correlation with independent estimates of groundwater discharge based on sapflow measurements on trees and both approaches suggested that during the study period the groundwater consumption in the Castelli plantation was 350 mm yr^{-1} (Engel *et al.*, 2004). A Cl balance at Castelli also suggested a similar groundwater use of 290 mm yr^{-1} by the plantation (Engel *et al.*, 2004). According to our capillary rise simulations and water table depth records, grass roots could have absorbed groundwater from the capillary fringe during most of the study period but did not. This suggests that water demand rather than access/supply determined higher evaporative water losses in tree plantations (Calder, 1998). The high coupling of tree canopies with the atmosphere could have interacted with advective energy imports from surrounding areas, increasing the evaporative potential of afforested stands in the Pampas; such a relationship has been suggested for forests compared to herbaceous canopies in Britain (Calder, 1998). Nonetheless, the deeper root systems of trees in the Pampas could enhance access to water during dry periods as it does in regions with lower or more seasonal precipitation inputs (Nepstad *et al.*, 1994; Calder *et al.*, 1997; Kleidon & Heimann, 1998).

Vertical patterns of salt accumulation at Castelli suggest that salinization was predominantly caused by the exclusion of fresh groundwater solutes by tree roots, whereas alternative mechanisms of salt transport, such as salt cycling by trees or salty groundwater intrusion from greater depth, played a lesser role. Salts at the Castelli plantation accumulated in the vicinity of the water table, where groundwater absorption and salt exclusion are expected to occur (Fig. 7). Salt cycling by trees would have developed a zone of high salinity in surface soil, which was not observed (see Jobbágy & Jackson, 2001). The detection of low Cl and Na concentrations in the biomass, throughfall, and litterfall of trees at this site in a previous study provides additional evidence against salt cycling (Jobbágy & Jackson, 2003). This mechanism, however, may be important where salt-concentrating species such as *Tamarix* spp. are established (Di Tomaso, 1998). Deep salty aquifers are common in sedimentary plains and may intrude overlaying fresh aquifers after afforestation (see Sapanov, 2000). Although vertical electrical soundings confirmed the presence of a salty aquifer at 25–30 m, they showed no connection between this water and the salty zone close to tree roots (Fig. 6). Direct measurements of groundwater at the base of the rooting zone in the plantation showed that it had ≈ 10 times less salt than at the water table (see Fig. 7 and results section). These observations suggest that the salinization triggered by trees was independent of salty groundwater intrusion from greater depths throughout most of the plantation. The core of the tree stand, however, apparently had experienced an intrusion of deep salty water, as suggested by the higher elevation of the upper boundary of the salty aquifer and the higher salinity of the intermediate zone.

How far can salinization proceed in the Pampas and other regions of the world? We suggest that the salinization process will ultimately face a biological limitation based on the tolerance of tree species to salinity. The combination of groundwater uptake, salt exclusion, and lack of leaching, would eventually raise groundwater salinity to a point beyond which no further uptake is possible. This limit will depend on the ability of individual plants to sustain water absorption as salinity increases (e.g. through osmotic adjustment) and may imply different maximum levels of groundwater salinity depending on the species involved. Remarkably, the highest groundwater salinity levels that we found were achieved by *E. camaldulensis* plantations, the most salt-tolerant tree species covered by our study. These values were slightly lower than 20 dS m^{-1} , the salinity tolerance threshold reported for the species (Morris *et al.*, 1998). This negative and self-limiting feedback of salt exclusion on water consump-

tion has been identified in mangroves, and the ability of such trees to absorb and excrete salts is an effective strategy to cope with the problem (Passioura *et al.*, 1992). A more drastic limit to salinization may be imposed by tree death, something that can occur when episodic stresses such as drought or waterlogging affect heavily salinized plantations. This may have occurred at the Castelli plantation during the spring of 2001, when *E. camaldulensis* trees died extensively within a 0.5 ha depression, a week after groundwater attained its highest level on records, reaching the surface of that small area of the plantation interior.

The results on groundwater use and salinization presented here are for 10–100 ha tree plantations embedded in a matrix of grassland vegetation that subsidizes lateral groundwater replenishment. What would the effects of more massive afforestation or tree encroachment in the Pampas be? Extensive tree establishment could compromise groundwater replenishment at the landscape scale, making its use transient and producing a widespread depression of groundwater. A landscape with deep and salty groundwater would be a likely outcome of a massive tree establishment in the Pampas and is what occurs naturally under forests with similar precipitation levels in flat landscapes of SW Australia (Schofield, 1992; Pierce *et al.*, 1993). In deforested areas of SW Australia, where groundwater has risen near the surface, widespread watertable depressions are achieved only after >70–80% of a watershed is reforested (George *et al.*, 1999); smaller scale reforestation can lead to localized groundwater use and salinization as we found in the Pampas (Vertessy *et al.*, 2000; see also Heuperman, 1999 for W Australia). Other grassland or cropland regions with shallow groundwater, particularly those in flat sedimentary landscapes like the eastern Chaco in South America (Sayago, 1995), the steppes of western Siberia in Russia (Bazilievich, 1970), or the Hortovágy grasslands in Hungary (Tóth & Rajkai, 1994), are likely to meet these conditions and could be vulnerable to tree-induced salinization.

Our study demonstrates that vegetation change can have a profound effect on groundwater hydrology and salinization. In the case of grassland to forest conversions in the Pampas, this ecohydrological interaction has both positive and negative aspects. On the one hand, groundwater use by plantations enhances primary production and could provide a tool for flood control; on the other, plantation establishment may compromise the fertility of soils, the sustainability of forestry and other land uses, and the quality of water resources. Identifying the mechanisms and environmental conditions that allow sustained groundwater use by trees is critical for achieving sustainable forestry

in grasslands and for predicting the consequences that this land use change will have on hydrology and climate.

Acknowledgements

We thank Ram Oren, Gaby Katul, José Paruelo, Zbigniew Kabala, Claudio Ghera, Roberto Fernandez, Will Pockman, and Vic Engel for the discussion of the data and ideas presented here and suggestions for improvement. Pedro Gundel, Carlos Villardi, Pablo Roset, Marcelo Noretto, Edgardo Meleguer, and Rubén Lewin provided valuable help in the field. This work was supported by grants from the InterAmerican Institute for Global Change (SGP 004 and CRN 012), the Fundación Antorchas, NSF, and the Mellon foundation. EGJ has a career fellowship from CONICET – Argentina.

References

- Amoozgar A, Warrick AW (1986) Hydraulic conductivity of saturated soils: field methods. In: *Methods of soil analysis – Part 1: Physical and Mineralogical Methods* (ed. Klute A), pp. 735–770. Soil Science Society of America, Madison.
- Bazilievich NI (1970) *The Geochemistry of Soda Soils*. Israel Program for Scientific Translation, Jerusalem.
- Bell DT, Williams JE (1997) Eucalyptus ecophysiology. In: *Eucalypt Ecology: Individuals to Ecosystems* (eds Williams JE, Woinarski JCZ), pp. 168–196. Cambridge University Press, Cambridge.
- Calder IA (1998) Water use by forests, limits and controls. *Tree Physiology*, **18**, 625–631.
- Calder IR, Hall RL, Prasanna KT (1993) Hydrological impact of *Eucalyptus* plantation in India. *Journal of Hydrology*, **150**, 635–648.
- Calder IR, Rosier PTW, Prasanna KT *et al.* (1997) *Eucalyptus* water use greater than rainfall input – possible explanation from southern India. *Hydrology and Earth System Sciences*, **1**, 249–256.
- Canadell J, Jackson RB, Ehleringer JR *et al.* (1996) Maximum rooting depth of vegetation types at the global scale. *Oecologia*, **108**, 583–595.
- Chen CR, Condon LM, Davis MR *et al.* (2003) Seasonal changes in soil phosphorus and associated microbial properties under adjacent grassland and forest in New Zealand. *Forest Ecology and Management*, **177**, 539–557.
- Culf AD, Allen SJ, Gash JHC *et al.* (1993) Energy and water budgets of an area of patterned woodland in the Sahel. *Agricultural and Forest Meteorology*, **66**, 65–80.
- Di Tomaso JM (1998) Impact, biology, and ecology of saltcedar (*Tamarix* spp.) in the southwestern United States. *Weed Technology*, **12**, 326–336.
- Engel V, Jobbágy EG, Jackson RB (2004) Water use characteristics of expanding *Eucalyptus* plantations in central Argentina (in preparation).
- Frankenberger WT, Tabaitabai MA, Adriano DC *et al.* (1996) Bromine, chlorine and fluorine. In: *Methods of Soil Analysis – Part 3 Chemical Methods* (eds Sparks SL, Page AL, Helmke PA, Loeppertke RH, Soltanpour PN, Tabaitabai MA, Johnson CT, Sumner ME), pp. 833–868. Soil Science Society of America, Madison.
- Freeze RA, Cherry JA (1979) *Groundwater*. Prentice-Hall, Englewood Cliffs, NJ.
- Geary TF (2001) Afforestation in Uruguay – study of a changing landscape. *Journal of Forestry*, **99**, 35–39.
- George RJ, Nulsen RA, Ferdowsian R *et al.* (1999) Interactions between trees and groundwaters in recharge and discharge areas – a survey of Western Australian sites. *Agricultural Water Management*, **39**, 91–113.
- Heuperman A (1999) Hydraulic gradient reversal by trees in shallow water table areas and repercussions for the sustainability of tree-growing systems. *Agricultural Water Management*, **39**, 153–167.
- INTA (1989) *Mapa de suelos de la provincia de Buenos Aires –Escala 1:500.000*. Instituto Nacional de Tecnología Agropecuaria, Buenos Aires.
- Jackson RB (1999) The importance of root distributions for hydrology, biogeochemistry, and ecosystem functioning. In: *Integrating Hydrology, Ecosystem Dynamics and Biogeochemistry in Complex Landscapes* (eds Tenhunen JD, Kabat P), pp. 219–240. John Wiley & Sons Ltd, Chichester.
- Jackson RB, Banner JL, Jobbágy EG *et al.* (2002) Ecosystem carbon loss with woody plant invasion of grasslands. *Nature*, **418**, 623–626.
- Jackson RB, Canadell J, Ehleringer JR *et al.* (1996) A global analysis of root distributions for terrestrial biomes. *Oecologia*, **108**, 389–411.
- Jackson RB, Schenk HJ, Jobbágy EG *et al.* (2000) Belowground consequences of vegetation change and their treatment in models. *Ecological Applications*, **10**, 470–483.
- Jobbágy EG, Jackson RB (2001) The distribution of soil nutrients with depth: Global patterns and the imprint of plants. *Biogeochemistry*, **53**, 51–77.
- Jobbágy EG (2002) *The imprint of plants on the vertical distribution of soil resources*. PhD dissertation, Duke University, Durham.
- Jobbágy EG, Jackson RB (2003) Patterns and mechanisms of soil acidification in the conversion of grasslands to forests. *Biogeochemistry*, **54**, 205–229.
- Kelliher FM, Leuning R, Schulze ED (1993) Evaporation and canopy characteristics of coniferous forests and grasslands. *Oecologia*, **95**, 153–163.
- Kleidon A, Heimann M (1998) A method of determining rooting depth from a terrestrial biosphere model and its impacts on the global water and carbon cycle. *Global Change Biology*, **4**, 275–286.
- Le Maitre DC, Scott DF, Colvin C (1999) A review of information on interactions between vegetation and groundwater. *Water SA*, **25**, 137–152.
- MAGP (1998) *Anuario estadístico agropecuario*. Ministerio de Agricultura y Producción, Montevideo.
- Morris J, Mann L, Callopy J (1998) Transpiration and canopy conductance in a eucalypt plantation using shallow saline groundwater. *Tree Physiology*, **18**, 547–555.
- Nepstad DC, De Carvalho CR, Davidson EA *et al.* (1994) The role of deep roots in the hydrological and carbon cycles of amazonian forests and pastures. *Nature*, **372**, 666–669.
- Passioura JB, Ball MC, Knight JH (1992) Mangroves may salinize the soil and in so doing limit their transpiration rate. *Functional Ecology*, **6**, 476–481.

- Perelman SB, Leon RJC, Oesterheld M (2001) Cross-scale vegetation patterns of Flooding Pampa grasslands. *Journal of Ecology*, **89**, 562–577.
- Pierce LL, Walker J, Dowling TI *et al.* (1993) Ecohydrological changes in the Murray-Darling basin. III. A simulation of regional hydrological changes. *Journal of Applied Ecology*, **30**, 283–294.
- Prieto AR (1996) Late Quaternary vegetational and climatic changes in the Pampa grassland of Argentina. *Quaternary Research*, **45**, 73–88.
- Raes D, Deproost P (2003) Model to assess water movement from a shallow water table to the root zone. *Agricultural Water Management*, **62**, 79–91.
- Richardson DM (1998) *Ecology and Biogeography of Pinus*. Cambridge University Press, New York.
- Rudel T, Ropel J (1996) Regional patterns and historical trends in tropical deforestation, (1976–1990): A qualitative comparative analysis. *Ambio*, **25**, 160–166.
- SAGPyA (2000) Primer inventario nacional de plantaciones forestales en macizo. Secretaría de Agricultura, Ganadería y Pesca Argentina. Buenos Aires – Argentina. *Revista Forestal*, **20**, 1–9.
- Sahin V, Hall J (1996) The effects of afforestation and deforestation on water yields. *Journal of Hydrology*, **178**, 293–309.
- Sapanov MK (2000) Water uptake by trees on different soils in the northern caspian region. *Eurasian Soil Science*, **33**, 1157–1165.
- Sayago JM (1995) The Argentinian neotropical loess: an overview. *Quaternary Science Reviews*, **14**, 755–766.
- Schenk HJ, Jackson RB (2002) The global biogeography of roots. *Ecological Monographs*, **72**, 311–328.
- Schofield NJ (1992) Tree-planting for dryland salinity control in Australia. *Agroforestry Systems*, **20**, 1–23.
- Scott DF, Lesch W (1997) Streamflow responses to afforestation with *Eucalyptus grandis* and *Pinus patula* and to felling in the Mokobulaan experimental catchments, South Africa. *Journal of Hydrology*, **199**, 360–377.
- Soriano A (1991) Río de la Plata grasslands. In: *Natural Grasslands: Introduction and western Hemisphere* (ed. Coupland RT), pp. 367–405. Elsevier, Amsterdam.
- Tewari DN (1993) *Poplar*. Surya Publications, New Delhi.
- Tóth TS, Rajkai K (1994) Soil and plant correlations in a solonchic grassland. *Soil Science*, **157**, 253–262.
- Tricart JL (1973) *Geomorfología de la Pampa Deprimida. Base para los estudios edafológicos y agronómicos*, Vol. XII. Colección Científica, INTA, Buenos Aires.
- Vertessy R, Connell L, Morris J *et al.* (2000) *Sustainable hardwood production in shallow watertable areas*. RIRDC publication 00-163, Rural Industries Research and Development Corporation, Barton.
- White WN (1932) *A method of estimating groundwater supplies based on discharge by plants and evaporation from soil*. USGS Water-Supply Paper 659-A. USGS, Washington.
- Wright JA, Di Nicola A, Gaitan E (2000) Latin American forest plantations – opportunities for carbon sequestration, economic development and financial returns. *Journal of Forestry*, **98**, 20–23.
- Zárate MA (2003) Loess of southern South America. *Quaternary Science Reviews*, **22**, 1987–2006.
- Zhang L., Dawes WR., Walker GR. (2001) Response of mean annual evapotranspiration to vegetation changes at catchment scale. *Water Resources Research*, **37**, 701–708.