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Regional patterns and controls of ecosystem salinization with grassland afforestation along a rainfall gradient

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[1] Vegetation change affects water fluxes and influences the direction and intensity of salt exchange between ecosystems and groundwater. In some conditions it can also lead to an intense accumulation of salts in soils and aquifers, as has been documented for the conversion of native grassland to tree plantations in the plains of Argentina, Hungary and Russia. In this paper we present a hierarchical framework to predict salt accumulation following vegetation change that is based on climatic, hydrogeological and biological factors. We evaluated this spatially explicit framework in temperate South America using a network of 32 pairs of adjacent plantation and grassland stands studied with detailed field measurements and remotely sensed imagery from MODIS. Our sites cover a broad precipitation gradient (770 to 1500 mm a^{-1}) and are underlain by shallow water tables (<2.5 m of depth). At the regional scale, geoelectric surveying revealed that the salinization of plantation soils depended strongly on climate, occurring only where the annual water balance (mean precipitation-Penman-Monteith potential evapotranspiration) was <100 mm a^{-1} ($p < 0.0001$, $n = 24$). At the local scale, we observed that groundwater salinities observed under ~50-year old plantations of different species were associated with their tolerance to salinity ($p < 0.001$, $n = 10$). Salinization occurred rapidly where rainfall was insufficient to meet the water requirements of tree plantations and where groundwater use compensated for this deficit, driving salt accumulating in the ecosystem. A general understanding of the vegetation-groundwater relationship will help predict and manage the negative and positive consequences of groundwater use from stand to regional levels of analysis.

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1. Introduction

1.1. Problem and Background

[2] Terrestrial vegetation, which strongly influences the global hydrological cycle, has the capacity to modify water resources, ecosystem productivity, and vadose zone salinity from stand to regional scales. Transpiration, the principal component of evapotranspiration over most land surfaces, is strictly linked with carbon assimilation [*Monteith*, 1988]. Vegetation effects on water fluxes, and accompanying solutes, across the continuum of ecosystem-vadose zone-aquifer, also affect the direction and magnitude of salt exchange between ecosystems and groundwater [*Schofield et al.*, 2001]. As a result of this influence on water fluxes, vegetation changes that alter evapotranspiration strongly, such as shifts between herbaceous and woody vegetation [*Zhang et al.*, 2001] can leave an important imprint on salt distribution and accumulation patterns with potential feedbacks on ecosystem functioning. In this paper we characterize the regional patterns of soil salinization that accompany the establishment of tree plantations on native grasslands and their link with groundwater consumption along a broad climatic gradient in temperate South America.

[3] Grassland afforestation, mainly with fast growing species such as eucalypts and pines, has expanded rapidly in the last decades in South America, motivated in part by broad governmental incentives [*Wright et al.*, 2000]. Numerous estimates of primary productivity in the region suggest higher values in the plantations compared to the native grasslands they replace [*Deregibus et al.*, 1987; *Frangi et al.*, 2000; *Jobbágy et al.*, 2006; *Piñeiro et al.*, 2006]. However, accompanying this higher carbon gain in the plantations is also a higher evaporative water loss, ultimately determined by the structure of trees and forests.

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Under wet conditions, the high aerodynamic conductance of trees lets them exchange water vapor with the atmosphere at rates up to 10 times higher than possible for shorter-statured vegetation [*Calder*, 1998] allowing a more exhaustive use of rainfall inputs. Under dry conditions, the deeper root distribution that trees often possess [*Canadell et al.*, 1996] provides them with access to water sources (e.g., vadose zone moisture, groundwater) that are often inaccessible to herbaceous plants [*Calder*, 1998]. Evidence for these increased evaporative water losses that follow afforestation includes drier soils and vadose zones [*Sapanov*, 2000], reduced water yield [*Jackson et al.*, 2005], and the elimination of groundwater recharge and the onset of discharge under afforested plots within herbaceous landscapes [*Heuperman*, 1999; *Jobbágy and Jackson*, 2004].

[4] Based on the higher water use of trees compared to herbaceous vegetation, afforestation has been proposed as a feasible management option for lowering the water table in regions with shallow groundwater [*Schofield*, 1992]. While tree establishment can provide increased productivity and beneficial ‘‘bio-drainage’’ where shallow saline water tables constrain crop production, continuous groundwater use by trees may also introduce a long-term, negative feedback on forest growth associated with salt accumulation, arising from salt exclusion during water uptake by roots [*Heuperman*, 1999; *Jobbágy and Jackson*, 2004, 2007]. Groundwater and soil salinization after afforestation has been reported in several regions around the world, for different types of trees (deciduous and evergreen, conifers and broadleaf) and in a broad range of climates [*George et al.*, 1999; *Heuperman*, 1999; *Jobbágy and Jackson*, 2004; *Nosetto et al.*, 2007; *Sapanov*, 2000; *Vertessy et al.*, 2000]. However, the controls on this salinization and the context in which it is likely to occur remain poorly understood.

[5] Australia presents an interesting example of the opposite shift in vegetation, where the replacement of vast areas of woodlands by herbaceous species led to an increase in hydrological recharge [*George et al.*, 1997]. This process raised the water table and moved deeply stored salts to the surface [*Pierce et al.*, 1993]. Current estimates suggest that the reversion of this process would only by achieved by reforesting 70–80% of the watershed [*George et al.*, 1999].

[6] In South America the opposite process (i.e., grassland afforestation) is taking place at high rates [*Geary*, 2001]. A better understanding is urgently needed for afforestation planning, especially because grassland afforestation is likely to grow across South America and other regions, partially motivated by the role of forests as carbon sinks [*Wright et al.*, 2000].

1.2. The Context of Salinization: Mechanisms and Predictions

[7] We propose a hierarchy of climatic, hydrogeological, and biological factors that help predict the onset and rates of salt accumulation in afforested grasslands (Figure 1). On the basis of this hierarchy we identify mechanisms of soil and water salinization and predict patterns at the regional, landscape and stand levels that can be evaluated in the field. This theoretical framework can then be applied to other vegetation shifts that alter rates of evapotranspiration.

[8] At the regional scale, the annual climatic water balance (precipitation - potential evapotranspiration) strongly influences salinization (Figure 1). If this balance is positive, occasional contributions of groundwater to evapotranspiration (discharge) will be offset by precipitation inputs to groundwater (recharge). In this case a net downward water flux means that salinization will be unlikely, although temporary groundwater use and salt accumulation could occur during dry periods. On the other hand, if the water balance is negative and groundwater has the potential to offset this deficit, its contributions may exceed recharge, causing a net upward flux of water and accompanying solutes. Salinization will proceed faster where water balances are more negative and groundwater is saltier [*Schofield et al.*, 2001].

[9] Hydrogeological factors also affect salinization from landscape to regional scales (Figure 1). Climate interacts with lithology and geomorphology to determine the presence and depth of phreatic groundwater and accompanying salts [*Domenico and Schwartz*, 1990]. Hilly landscapes with massive bedrock are less likely to provide widespread access to groundwater than flat sedimentary plains. Through its influence on the hydraulic conductivity of sediments, lithology also dictates the maximum rates at which groundwater flow to plants can be sustained. While the hydraulic resistance of clay sediments yields groundwater at rates that are likely to be orders of magnitude lower than vegetation demand, even with a water table gradient of several meters, sandy sediments can match this demand with only slight water table depressions [*Jobbágy and Jackson*, 2004; *Sapanov*, 2000]. Thus lithology and geomorphology act as filters on climate, restricting the extent of salinization to areas where groundwater can be accessed and used at significant rates by plants.

[10] Biological factors dictate the intensity of salinization and its location across the landscape, as well, by influencing maximum evapotranspiration and salinity tolerance, interacting with the large-scale factors mentioned above (Figure 1). Afforestation can shift the actual water balance from positive to negative, resulting in net groundwater discharge and salt accumulation [*Jobbágy and Jackson*, 2004]. Vegetation also dictates maximum salinity values in places where net groundwater discharge occurs. The combination of groundwater absorption and salt exclusion by roots eventually raises groundwater salinity to a concentration that hinders further groundwater uptake [*Morris and Collopy*, 1999]. Different salinity tolerances for tree species may determine how long a species can continue to use groundwater as salinity increases. Once the salinity tolerance is reached, further groundwater uptake is minimal for stands of that species.

[11] In this paper we characterize the process of salinization that accompanies grassland afforestation in the Río de la Plata Grasslands of South America and use that setting to evaluate the theoretical framework proposed above (Figure 1). In a previous study conducted in the region, *Jobbágy and Jackson* [2003] evaluated the role of hydrogeological factors on salinization, showing that afforestation

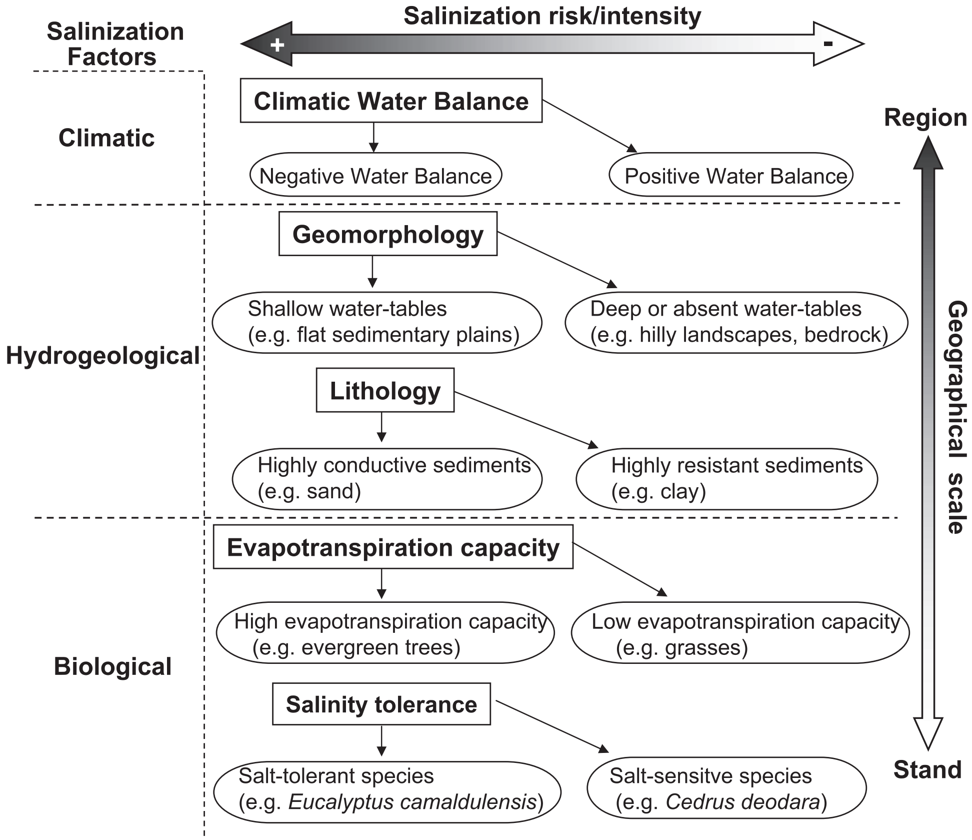

Figure 1. Hierarchical framework for predicting salinization following vegetation changes. Climatic water balance defines the possibility of salinization at the regional scale. Where rainfall does not meet vegetation water needs and groundwater has the potential to offset this deficit, salinization can proceed. Hydrogeological factors (geomorphology and lithology) affect salinization from landscape to regional scales, restricting salinization to areas where groundwater can be accessed and used at significant rates by plants. Biological factors dictate the intensity of salinization across the landscape through the regulation of evapotranspiration rates and salinity tolerance thresholds.

led to a stronger groundwater salinization on intermediate texture soils (silty) than in clayey and poorly conductive soils or in sand and highly permeable soils. In this work we explore the salinization process across a climatic gradient, integrating previous work on salinization controls into a more regional framework that accounts for climatic and biological influences.

[12] The Río de la Plata Grasslands offer vast areas with shallow water tables and highly conductive sediments [*Soriano et al.*, 1991; *Tricart*, 1973], which provide groundwater access to vegetation. On the basis of our theoretical framework outlined above, we made two predictions: (1) As the water balance becomes more negative, soil salinity will show increasingly higher concentrations under plantations compared to adjacent grasslands. In humid areas salinization will be prevented by a positive water balance that cannot be reversed by trees; below a certain water balance threshold, however, plantations will be able to trigger this switch from recharge to discharge and salinization will proceed. (2) Tree species strongly influence soil and groundwater salinity driven by their respective salinity tolerances. Under homogeneous conditions, species with higher salinity tolerances will lead to stronger salinization than less salt-tolerant species.

[13] We tested both predictions across 32 field sites spanning a ~700 mm a^{-1} range of annual water balance. The regional influence of climate on salinization was explored in pairs of adjacent grassland-eucalyptus stands, and the effects of tree species was evaluated in ten contiguous experimental plots at one of our study sites. Our measurements were complemented by an integrative estimate of water and carbon fluxes in grasslands and eucalyptus plantations based on a remotely derived vegetation index.

2. Materials and Methods

2.1. The Region

[14] The Rio de la Plata Grasslands of Argentina and Uruguay extend between latitude 28°18′S and 39°05′S and longitude 56°02′W and 64°48′W (Figure 2), with an elevation of less than 300 m above sea level in most of the area [*Soriano et al.*, 1991]. Climate is temperate, with

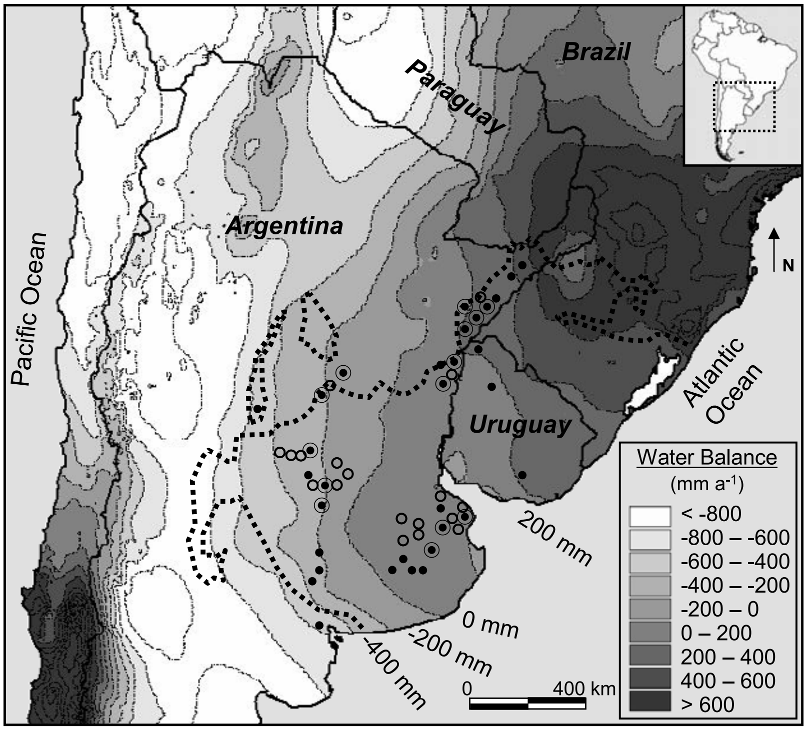

Figure 2. Study region showing the sites where salinity (circles) and satellite (dots) analyses were performed. The climatic water balance (mm a^{-1}), calculated as mean annual precipitation minus Penman-Monteith potential evapotranspiration, is also displayed. Dotted lines surround the original grassland area of the Río de la Plata Grasslands.

mean annual temperatures varying between 15°C and 21°C. Rainfall ranges from ~600 mm a^{-1} in the SW to 1600 mm a^{-1} in the NE part of the region, and Penman-Monteith potential evapotranspiration ranges from ~800 mm a^{-1} to ~1200 mm a^{-1} (period 1961–1990, CRU database) [*New et al.*, 2002], defining dry subhumid and humid climatic types [*Burgos and Vidal*, 1951]. Much of the region is a vast, flat plain, although isolated rock outcrops and hills are occasionally found [*Soriano et al.*, 1991]. Soils, developed from loess-derived sediments, are mainly Mollisols, with Alfisols and Vertisols found toward the east part of the region and with Entisols toward the west [*Soriano et al.*, 1991]. Currently, most of this originally treeless region is dominated by a combination of native grasslands, pastures and annual crops. However, tree plantations of different exotic species (eucalypts, pines, and poplars, among others) that were originally established for shade, windbreak and landscaping are present across the region. Today, commercial afforestation is increasing rapidly and is expected to continue its expansion in the coming decades [*Wright et al.*, 2000].

2.2. Study Sites and Field Work

[15] To evaluate the effect of climate on salinization, we assessed soil salinity at 32 pairs of adjacent *Eucalyptus camaldulensis* plantations and grassland stands across ranges of ~720 mm a^{-1} (770 to 1490 mm a^{-1}) of annual rainfall and ~700 mm a^{-1} (−350 to 350 mm a^{-1}) of annual climatic water balance (Figure 2). Plantations were 0.5–70 ha in size and were never fertilized nor irrigated. Each pair of stands was located at the same topographic position and shared the same soil type, as confirmed by direct observations in the field. Topographic micro-depressions were avoided in the selection of the study sites. Grassland stands were neither fertilized nor irrigated but were typically grazed. In order to remove possible age effects [*Nosetto et al.*, 2005], we selected plantations >15 years of age. All sites had shallow water tables (<2.5 m of depth) and medium-textured sediments, the most common situation in the region.

[16] We assessed soil salinity through electromagnetic induction, which allows the measurement of apparent soil electrical conductivity (ECa) and is a direct measure of soil salinity. In this electromagnetic technique, a time-varying magnetic field is created in the soil from a transmitter coil located at or above the ground surface. This magnetic field induces small currents in the soil, which generate a secondary magnetic field that is registered by a receiver coil. The ratio between the primary and secondary field is a linear function of soil conductivity [*McNeill*, 1992]. The instrument we used (EM38, Geonics Ltd.) has an intercoil spacing of 1 m and operates at a frequency of 14.6 kHz. We used it in the vertical dipole mode, which provides an effective measurement depth and width of ~1.5 m when the instrument is placed on the ground, with the highest sensitivity at ~0.4 m away from the instrument [*Nosetto*

Table 1. List of Tree Species Included in our Analysis and the Maximum Salinity Values That They Tolerate as Derived From our Literature Analysis[a]

Tree Species	Maximum Salinity Reported, dS m^{-1}	Reference
Eucalyptus camaldulensis	35	*House et al.* [1998]
Eucalyptus melliodora	33.6	*House et al.* [1998]
Eucalyptus cinerea	15	*Taws* [2003]
Pinus radiata	18.9	*Stirzaker et al.* [2002]
Cedrus deodara	5.8	*Wu et al.* [2001]
Cupressus sempervirens	17.7	*Benito et al.* [2005]
Casuarina cunninghamiana	25.7	*House et al.* [1998]
Quercus robur	18.9	*Spennemann and Marcar* [1999]
Populus deltoides	10	*Stirzaker* [2002]
Maclura pomifera	10	*Swift* [1997]

[a]Values obtained from soil extracts were transformed to equivalent groundwater salinity according to *Tanji and Kielen* [2003].

et al., 2007; *Sudduth et al.*, 2001]. In April 2006–May 2006, 10 to 20 measurements were taken at each stand along a 20–40 m long transect. Transects were located ~40 m away from the plantation-grassland edge.

[17] Apparent soil electrical conductivity may also be influenced by soil texture and moisture content. Our sites minimize textural effects, being located on loessic sediments dominated by the silt fraction in most cases. Six sites were sampled to explore soil texture homogeneity between paired stands and we found that sand, silt and clay contents (hydrometer method, *Bouyoucos* [1962]) did not differ significantly between paired plantation and grassland stands at any given site ($p > 0.10$, $n = 3$, depth = 0–100 cm). Moisture effects on our apparent conductivity measurements were likely minimized by the fact that sites were sampled during a wet period (April–May) in which soil moisture contents showed little differences between grassland and plantation stands [*Nosetto*, 2007]. At the same sites in which soil texture was measured, soil gravimetric water content showed no significant differences between grassland and tree plantation sites ($p > 0.10$, $n = 3$, depth = 0–100 cm). Finally, we tested the accuracy of the electromagnetic measurements in twelve calibration plots, where mineral soil was sampled above the water table at depth intervals of 0–10, 10–20, 20–30, 30–50, 50–100, 100–150 cm for electrical conductivity analysis in the laboratory. A strong correlation was obtained between electromagnetic measurements and soil electrical conductivity measured in 1:2 soil-water extract ($r^2 = 0.91$, $p < 0.01$).

[18] To isolate the influence of tree species on groundwater salinization, we also compared groundwater chemistry beneath tree plantations of ten different species at a single site (Estación Forestal y Vivero Francisco P. Moreno, América, Buenos Aires, Argentina). Plots 0.5–4 ha in size were located <1000 m from each other in a homogenous landscape with medium textured Mollisol soils, a ~2.5 m-deep water table, and a subhumid climate (780 mm a^{-1} of precipitation and −277 mm a^{-1} of water balance). The species selected for this study spanned a broad range of salinity tolerance (5 to 35 dS m^{-1}; see Table 1) and included broadleaf evergreens (*Eucalyptus camaldulensis, E. cinerea* and *E. melliodora*), broadleaf deciduous trees (*Quercus robur*, *Maclura pomifera* and *Populus deltoides*), conifers (*Pinus radiata*, *Cedrus deodara and Cupressus sempervirens*), and a non-coniferous species with needle-type leaves (*Casuarina cunninghamiana*). These plantations were 40–54 years old and had basal areas of 40–66 m^2 ha^{-1} at the time of our measurements. All plantations had similar management regime; no fertilization or irrigation had been applied to the plantations, which also were never thinned or pruned. Understory plants were absent under broadleaf evergreen and coniferous species, but broadleaf deciduous species hosted a sparse understory dominated by *Ligustrum lucidum*. Herbaceous species in the understory were generally absent. We evaluated salinization by sampling groundwater beneath each plantation for electrical conductivity, chloride, and sodium analyses. Boreholes located ~40 m inside the edge of each stand were made with a hand auger (10–cm outside diameter). Electrical conductivity was measured with a conductivity meter automatically corrected for temperature (Orion Model 115, Orion Research, Inc., MA, USA). Sodium and chloride concentrations were determined with ion-selective electrodes [*Frankenberger et al.*, 1996; *Talibudeen*, 1991].

2.3. Satellite Analysis

[19] We also evaluated the influence of vegetation type (tree plantation versus grassland) and groundwater access using remotely sensed data. We compared the Enhanced Vegetation Index (EVI) from MODIS imagery, an index of primary productivity [*Xiao et al.*, 2005], and consequently of water use [*Monteith*, 1988], across 32 pairs of adjacent eucalyptus plantations and native grasslands. These plantation and grassland pairs covered a gradient of ~1000 mm a^{-1} of annual rainfall (580 to 1600 mm a^{-1}) and ~1050 mm a^{-1} of climatic water balance (−550 to 500 mm a^{-1}) (Figure 2). Fourteen of these pairs coincided with those used for the salinization analysis. Six additional sites were located in areas with bedrock at ~2–3 m below the surface, preventing groundwater access, and twelve were located in flat areas with shallow water tables, as confirmed by topographic maps and field observations. In two cases suitable adjacent grasslands were absent and cultivated pastures were used for comparison.

[20] Compared to the Normalized Difference Vegetation Index (NDVI), the most commonly used vegetation index, EVI has higher sensitivity in high biomass situations [*Huete et al.*, 2002]. Its improved vegetation monitoring arises through a decoupling of the canopy background signal and a reduction of atmospheric influences [*Huete et al.*, 2002]. In forests with high biomass, NDVI tends to saturate asymptotically while EVI remains sensitive to canopy variations [*Huete et al.*, 2002]. EVI combines data for the near infrared, red and blue bands of the electromagnetic spectrum [*Huete et al.*, 1994, 2002].

[21] For the period 2000 to 2005, we used the MOD13Q1 product which derives from the daily surface reflectance product (MOD09 series), corrected for molecular scattering, ozone absorption, and aerosols [*Vermote et al.*, 2002]. It represents a 16-day composite with a spatial resolution of 250 m. The MODIS algorithm applies a filter to the

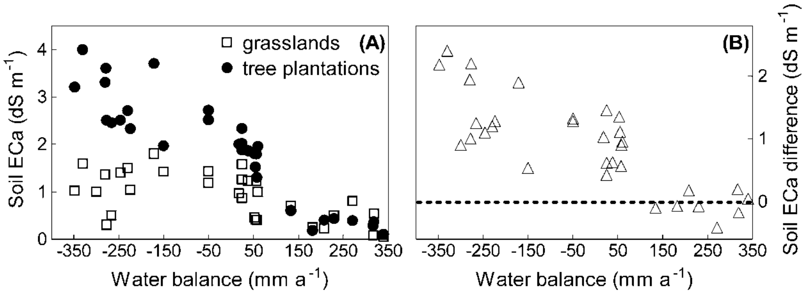

Figure 3. Soil salinity patterns in grasslands and plantations across a regional climatic gradient. The association between apparent soil electrical conductivity in grasslands and plantations (a) as well as the difference between both vegetation types (b) with the climatic water balance is shown. Apparent soil electrical conductivity (ECa) was measured using an electromagnetic induction instrument with an integration depth of ~1.5 m (n = 10–20 per stand). The climatic water balance was calculated as the difference between mean annual precipitation and Penman-Monteith potential evapotranspiration.

databased on quality, cloud cover, and viewing geometry, and only high quality, cloud-free, filtered data are retained for compositing [*Huete et al.*, 2002].

2.4. Data Analysis

[22] To evaluate the climatic control on salinization at the regional scale, we explored the relationship between apparent soil electrical conductivity and the annual climatic water balance (WB), calculated as the difference between mean annual precipitation and Penman-Monteith modified by FAO potential evapotranspiration [*Allen et al.*, 1998]. Climatic data were obtained from the CRU CL 2.0 database (1961–1990) elaborated by the UK Climate Research Unit with a spatial resolution of 10 min [*New et al.*, 2002]. Because most plantations were between 20 and 50 years of age at the time of measurement (April 2006–May 2006) and the CRU database comprises the 1961–1990 period, there is a large overlap between plantation lifetime and water balance data.

[23] We related apparent soil electrical conductivity with the WB through linear regression models and compared them, using the Akaike Information Criteria [*Akaike*, 1974], with the following threshold model:

$$y = (\mathrm{a} + \mathrm{b}^* x)^*(x \le \mathrm{c}) + (\mathrm{a} + \mathrm{b}^*\mathrm{c})^*(x > \mathrm{c})$$

where y and x represent apparent soil electrical conductivity and WB, respectively, and a, b and c describe the intercept, the slope and the inflection point of the curve, respectively. The terms $x \le$ c and $x >$ c define conditions in which the term equals to 1 when true, and 0 when false.

[24] The influence of tree species on salinization was assessed exploring the association between observed groundwater electrical conductivity and maximum values tolerated by the species. We expected that higher salinity tolerance would lead to higher groundwater electrical conductivity observed in the field. We obtained the maximum salinity values tolerated by each species after surveying the literature (Table 1). When these values were obtained from soil extracts they were transformed to equivalent groundwater electrical conductivity according to *Tanji and Kielen* [2003]. For some analyses we also grouped tree species into categories of eucalyptus, conifer, and deciduous and compared their associated groundwater electrical conductivity values using one-way ANOVA, followed by Tukey's test.

[25] We explored groundwater access and vegetation effects on EVI values pairing sites without groundwater access to nearby sites with access to groundwater (confirmed by field observations) and performed a two-way ANOVA followed by Tukey's test. We analyzed the spatial relationships between EVI annual averages for the period 2000–2005 using the annual climatic water balance. Although the CRU climatic database and the EVI values correspond to different time periods (1961–1990 versus 2000–2005), mean annual precipitation values from the CRU database agreed with those obtained at eight meteorological stations across the precipitation gradient during 2000–2005 (r^2 = 0.98, n = 8, slope and y-intercept not significantly different from 1 and 0, respectively, $p > 0.10$). At one site (Tandil) we computed the EVI monthly average, and we evaluated its temporal association with monthly precipitation (2000–2005), considering different integration periods (1 to 12 months of integration).

3. Results

3.1. Salinization Patterns and Controls

[26] As predicted, soils were increasingly saltier in tree plantations compared with native grasslands and were strongly influenced by the annual climatic water balance. The mean apparent electrical conductivity of soils (0–150 cm of depth) in plantations was twice as high as in grasslands (1.84 versus 0.96 dS m^{-1}, $p < 0.0001$, n = 32, Figure 3a). Soil apparent electrical conductivity decreased linearly with the climatic water balance in both types of vegetation (y = 1.82–0.005 * WB, r^2 = 0.83, $p < 0.0001$; and y = 0.88 – 0.0014 * WB, r^2 = 0.33, $p < 0.001$, for plantations and grasslands, respectively, n = 32). The slope of the relationship, however, was significantly higher in tree plantations ($p < 0.0001$). The same analyses using precipitation as an explanatory variable showed similar relation-

Table 2. Groundwater Chemistry Under Different Trees Species and a Grassland Stand in America (35.5°S, 62.9°W, Argentina)[a]

Tree Species	EC, dS m^{-1}	Chloride, ppm	Sodium, ppm
Eucalyptus camaldulensis	27.1	11,425	5710
Eucalyptus melliodora	25.6	9566	4098
Eucalyptus cinerea	15	5632	3225
Pinus radiata	6.2	2585	354
Cedrus deodara	5.4	2060	467
Cupressus sempervirens	6.4	2175	521
Casuarina cunninghamiana	14.9	5460	2219
Quercus robur	17.5	7299	850
Populus deltoides	4.2	1362	408
Maclura pomifera	2.15	312	299
Grassland	1.5	3.8	244

[a] Samples were taken at the water table from boreholes located ~40 m inside the edge of each stand. Electrical conductivity was measured with a conductivity meter automatically corrected for temperature and sodium and chloride concentration were determined with ion-selective electrodes.

ships but lower goodness of fit (r^2 = 0.71 and 0.38 for plantations and grasslands, respectively).

[27] We observed different rates of salinization by plantations across the climatic water balance gradient. Tree establishment triggered substantial soil salinization where the annual climatic water balance was less than ~100 mm a^{-1}, with increasing salinization intensity (difference in apparent soil electrical conductivity between plantation and neighboring grassland) toward the arid end of the climatic gradient (Figure 3b). Below this climatic water balance threshold, plantation soils doubled the mean apparent electrical conductivity of grasslands soils (2.34 versus 1.14 dS m^{-1}, $p < 0.0001$, n = 24), whereas no difference was observed above this threshold (0.34 versus 0.39 dS m^{-1}, $p > 0.10$, n = 8). Using a statistical threshold model, the climatic water balance explained 67% of salinization variance across the entire climatic gradient, though the goodness of fit was similar for a linear model, which explained 66% of the variability (ratio of probabilities between linear and threshold model = 3.1, Akaike Information Criteria).

[28] Different tree species strongly affected groundwater salinities, supporting our second prediction (Table 2). Although these forest stands had similar age, basal area, and management regime, electrical conductivity of the groundwater varied by more than an order of magnitude among the different species (from 2.15 to 27.1 dS m^{-1}, for *M. pomifera* and *E. camaldulensis*, respectively) and tracked their levels of salinity tolerance (Figure 4a). Eucalypts triggered a stronger salinization than conifers or deciduous trees ($p < 0.05$, Figure 4b). Chloride concentrations in the groundwater showed even larger differences among the different species and ranged from 312 ppm for *M. pomifera* to 11,425 ppm for *E. camaldulensis* (Table 2). Similarly, sodium concentrations ranged between 299 ppm (*M. pomifera*) to 5710 ppm (*E. camaldulensis*). Even the species showing the weakest salinization (*M. pomifera*) increased groundwater chloride concentration by ~100x compared to a neighboring grassland (312 versus 3.8 ppm, Table 2).

3.2. Patterns of Primary Productivity

[29] Tree plantations displayed higher primary productivity than grasslands, as suggested by their consistently higher EVI values (Figure 5a). At sites with access to groundwater, mean annual EVI values in plantations were 29% higher than in neighboring grasslands (0.45 versus 0.35; $p < 0.0001$, n = 26), with differences amplified toward the extremes of the climatic gradient (Figure 5a). When groundwater was inaccessible the differences declined, with tree plantations increasing EVI by 13% (0.43 versus 0.38; $p < 0.05$, n = 6). Although the climatic water balance explained a large proportion of EVI variability, the best explanatory models differed between both types of vegetation. While in plantations annual EVI increased linearly with the water

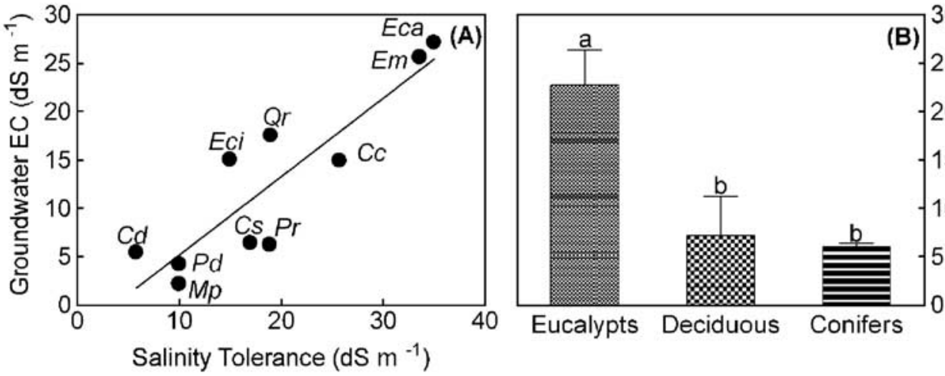

Figure 4. Groundwater salinity patterns under stands of different tree species. The relationship between salinity tolerance and measured groundwater electrical conductivity of the tree species is shown (panel a; $r^2 = 0.79$, $p < 0.001$, n = 10). Values were grouped as Eucalyptus (n = 3), Deciduous (n = 3) and Conifers (n = 3) and compared using one-way ANOVA followed by Tukey's test. (Letters show significant differences at $P < 0.05$) (b). Species analyzed were: *Eucalyptus camaldulensis (Eca)*, *E. melliodora (Em)*, *E. cinerea (Eci)*, *Quercus robur (Qr)*, *Maclura pomifera (Mp)*, *Populus deltoides (Pd)*, *Pinus radiata (Pr)*, *Cedrus deodara (Cd)*, *Cupressus sempervirens (Cs)* and *Casuarina cunninghamiana (Cc)*.

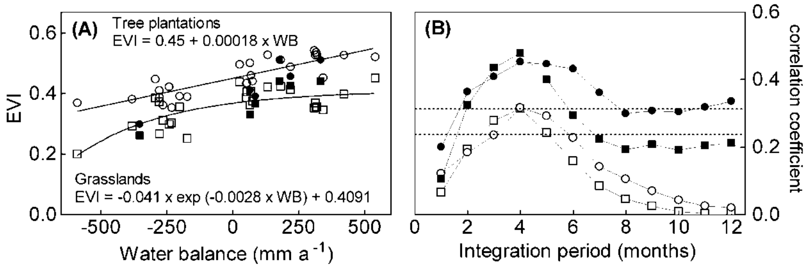

Figure 5. Spatial relationships between water balance and enhanced vegetation index (EVI) (a), and temporal association between precipitation and EVI values of tree plantations and grasslands for varying time integration periods at Tandil (b). EVI data were extracted and averaged from MODIS satellite images for years 2000–2005. Linear and curvilinear models were adjusted in plantations (r^2 = 0.73, n = 26) and grasslands (r^2 = 0.55, n = 26) with groundwater access. For panel B, Spearman correlation coefficients of a linear model relating monthly EVI (n = 70) and the number of months of cumulative precipitation data prior to image acquisition are shown. Correlation levels corresponding to significances of P = 0.05 and P = 0.01 are indicated with horizontal lines.

balance, in grasslands, EVI tended to reach a plateau, and the use of a saturation response model resulted in a better fit (ratio of probabilities between saturation and linear model = 3.3, Akaike Information Criteria).

[30] Although annual averages of EVI did not differ between sites with and without access to groundwater for any vegetation type ($p > 0.05$, n = 6), strong differences emerged when we separated the data for relatively dry (WB < 100 mm a^{-1}) and humid (WB > 100 mm a^{-1}) sites. Plantations had significant higher EVI values with groundwater access in the dry sites ($p < 0.05$, n = 3) but not in the humid ones ($p > 0.10$, n = 3). Temporal analysis at Tandil showed that changes in monthly EVI values of tree plantations and grasslands were best explained by the rainfall registered in the previous 3–6 months; with stronger associations at the site without access to groundwater (Figure 5b).

4. Discussion

4.1. The Context of Salinization in the Río de la Plata Grasslands

[31] Based on the results of our regional analysis, we have shown that climate dictates the onset of salinization in afforested grasslands at the regional scale. Solutes accumulate when rainfall is insufficient to meet vegetation water requirements and groundwater makes up the deficit, acting as a net source of water and salts to the ecosystem. The pattern of salinization that we observed in afforested grasslands showed no salinization above an annual climatic water balance of 100 mm a^{-1}, but below this threshold, salinity increased linearly (Figure 3). This pattern suggests that around this climatic boundary trees are actually achieving a "zero" balance in which they switch the hydrological regime from a net recharge to a net discharge situation, triggering salinization. On the basis of this threshold relationship, rainfall would match the evaporative demand of the plantations of the region when it approaches ~1100 mm a^{-1}, in agreement with independent evapotranspiration estimates of ~1200 mm a^{-1} at eucalypt plantations with shallow water tables within the region [*Jobbágy and Jackson*, 2004; *Nosetto et al.*, 2005]. These observations point to a water limitation below 100 mm a^{-1} of annual water balance that is ameliorated when groundwater is available (Figure 5a), but coupled tightly to salt accumulation.

[32] Groundwater salinity patterns under mature stands of different tree species showed that tolerance to salinity is a key biological driver of salinization at the stand scale, influencing salt accumulation levels in the groundwater-vadose zone system (Figure 4). Through groundwater use and salt exclusion by roots, trees accumulate salts in the vadose zone and groundwater [*Jobbágy and Jackson*, 2007; *Morris and Collopy*, 1999]; if no mechanism helps to remove those salts, their concentrations can increase to a level that constrains further groundwater uptake by the trees [*Thorburn et al.*, 1995]. The relationship that we observed, with the groundwater under more salt-resistant species having higher salinity values than under less salt-resistant species (Figure 4a), suggests that the salinity tolerance of the species defines the maximum salinity level reached in groundwater and soils. The higher tolerance of *Eucalyptus*, *Quercus* and *Casuarina* species could be due to a combination of effective salt exclusion mechanism by roots and better osmotic adjustment through the synthesis and accumulation of osmolytes [*Adams et al.*, 2005; *Niknam and McComb*, 2000]. Other mechanisms, such as salt uptake and excretion through foliar glands, become important in *Tamarix sp.* [*Thompson et al.*, 1969].

[33] Sporadic salt leaching together with lateral groundwater flow can help remove accumulated salts from some ecosystems, allowing sustained groundwater consumption by vegetation [*Morris and Collopy*, 1999; *Thorburn et al.*, 1995]. Although some flushing may exist in the Río de la Plata Grasslands associated with periods of high rainfall, lateral exports occur slowly because of the low topographic and piezometric gradients in the region [*Soriano et al.*, 1991]. Moreover, these subtle regional gradients are overwhelmed by the water table depressions generated by plantation discharge, driving lateral movements toward the

interior of the afforested stands and preventing salt removal [*Jobbágy and Jackson*, 2004, 2007]. In addition to Río de la Plata Grasslands, other grassland regions with shallow groundwater, particularly those in flat sedimentary landscapes like the steppes of western Siberia [*Bazilevich*, 1965], the Great Plains of western Canada [*FAO*, 1991] or the Carpathian basin (Hungary, Romania, Serbia, and Slovakia) [*Toth and Rajkai*, 1994], could also be vulnerable to tree-induced salinization. It is interesting to note, however, that in subhumid to semiarid regions where a very shallow water table promotes direct soil water evaporation and surface salt accumulation, as it is common in deforested areas of Australia [*Barret-Lennard*, 2002], afforestation could be beneficial. In these cases, tree establishment will likely lead to deeper water tables and salt accumulation zones [*Nosetto et al.*, 2007; *Schofield*, 1992].

[34] Solute diffusion, a mechanism that is often ignored in many analyses (e.g., *Thorburn et al.* [1995]), may represent an alternative pathway of salt removal [*Morris*, 1999]. Since the rate at which this process takes place is dictated by salt concentration gradients (Fick's law, *Bresler et al.* [1982]), salt-tolerant species that can cope with higher concentrations would lead to steeper salt gradients and, consequently, higher diffusion rates. For instance, if the salt concentration equilibrates with background salinity one meter below the water table [*Jobbágy and Jackson*, 2004; *Nosetto et al.*, 2007], *E. camaldulensis*, the most salt resistant species evaluated in this study, would develop a diffusive gradient of 16.3 $\mathrm{Kg\ m^{-3}\ m^{-1}}$. Considering a diffusion coefficient of $2 \times 10^{-5}\ \mathrm{cm^2\ s^{-1}}$ and a volumetric water content of 50% [*Morris*, 1999], this gradient promotes a downward diffusion flux of solutes of 0.98 $\mathrm{g\ m^{-2}\ d^{-1}}$, which would be balanced by an upward convective flux (i.e., groundwater uptake) of $\sim$1 $\mathrm{mm\ d^{-1}}$. With the same reasoning, *M. pomifera* would generate a diffusive gradient of 0.4 $\mathrm{Kg\ m^{-3}\ m^{-1}}$ and an upward groundwater flux of only $<$0.1 $\mathrm{mm\ d^{-1}}$. On the basis of this analysis, more salt-resistant species would lead to higher salt accumulation and soil degradation, but would be able to maintain relatively high rates of groundwater use and primary production.

[35] In agreement with previous studies [*Engel et al.*, 2005; *Jobbágy and Jackson*, 2004], our observations point to significant groundwater use by plantations of the region, but tree productivity nevertheless showed a strong response to rainfall inputs, suggesting some water limitations to growth (Figure 5). One possible cause of this discrepancy may be the high salt concentration of groundwater (e.g., $\mathrm{EC_{gw}} = 27.1\ \mathrm{dS\ m^{-1}}$ for the *E. camaldulensis* plantation, Table 2). Because of increasingly negative osmotic water potential, groundwater use by trees may be unable to fully compensate for rainfall deficits, constraining groundwater use as salts build-up. Declining groundwater consumption by vegetation as salinity increases [*Thorburn et al.*, 1995], as well as important responses of transpiration rates to precipitation inputs observed in forests with shallow water tables [*Cramer et al.*, 1999; *Engel et al.*, 2005], supports this observation and suggests that even with access to groundwater, vegetation may experience periodic water deficits. Additionally, the beneficial effect of rainfall on groundwater-fed plantations may be associated with soil nutrient availability, since higher rainfall and soil-water content could enhance rates of soil organic matter decomposition and mineralization [*Kladivko and Keeney*, 1987].

4.2. Management Implications and Climate Interactions

[36] Our study suggests that groundwater consumption poses both opportunities and risks for forestry and land management. On the one hand, productivity is enhanced with groundwater consumption; on the other hand, salt build-up can threaten long-term productivity as well as the integrity of soil and water resources. A combination of management and planning strategies from stand to regional levels are available in order to take advantage of the benefits of groundwater consumption while minimizing the risk of salinization. At the regional level, the climatic water balance is a crucial variable defining expected production rates as well as the risk of salinization and could be incorporated into afforestation zoning. Plantations located toward arid regions and with saltier groundwater will likely intensify salinization, shortening the time in which the species salinity threshold is achieved and leading to more rapid declines in growth rates [*Schofield et al.*, 2001; *Thorburn et al.* 1995, Figure 3]. Although salt removal will be unlikely if the negative water balance persists in plantations, removal may be promoted by rotating plantations with crops or pastures [*Stirzaker et al.*, 2002], which would allow flushing of accumulated salts after tree harvesting and yield a less negative water balance at the landscape scale. At the stand scale, selecting tress species with lower evaporative capacity, for instance deciduous or coniferous trees [*Farley et al.*, 2005; *Myers et al.*, 1996], may also decrease the risk of salinization. In contrast, management techniques that increase production rates (e.g., fertilization, high tree density, *Hubbard et al.* [2004]) will increase water use and the risk of salinization.

[37] Our findings stem from observations made in isolated plantations ($<$1 $\mathrm{km^2}$) established in a matrix of herbaceous ecosystems (native grasslands, pastures, annual crops). Tree plantations covering larger areas ($>$10 $\mathrm{km^2}$) may display different groundwater consumption and salinization patterns if feedbacks with hydrology and climate develop. For instance, extensive grassland afforestation may significantly reduce groundwater replenishment at the landscape scale, leading to widespread water table depressions that constrain groundwater consumption by the plantations. This possibility is suggested by observations in reforested regions of SW Australia, where widespread water table depression is achieved when $>$70–80% of the watershed is reforested [*George et al.*, 1999]. Large-scale afforestation could also influence the regional climate, as suggested by observations and modeling experiments of the opposite land use change (i.e., replacement of forest by pastures and crops) in Amazonian and Cerrado ecosystems [*Durieux et al.*, 2003; *Hoffmann and Jackson*, 2000; *Shukla et al.*, 1990]. Because forests usually show higher evapotranspiration rates than herbaceous vegetation [*Nosetto et al.*, 2005; *Zhang et al.*, 2001], a more humid atmosphere would be a likely outcome accompanying this land use shift. This forcing could decrease the risk of salinization if it results in higher rainfall

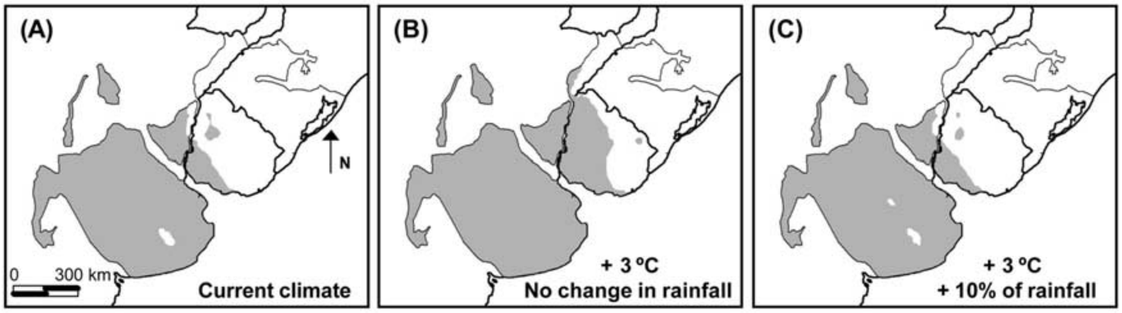

Figure 6. Grassland areas that are potentially prone to salinization following afforestation, based on current (a) and plausible climatic scenarios (B and C) for the 2080 decade [*Barros et al.*, 2006]. An annual water balance of 100 mm a^{-1} was considered as the threshold below which grassland afforestation triggers salinization (see Figure 3). Climatic scenarios included (b): 3°C increase in annual temperature with no annual rainfall change; (c): 3°C increase in annual temperature and 10% increase in annual rainfall. Hydrogeological factors constraining groundwater use were not considered for the areas definition.

and/or lower vapor pressure deficit, leading to a more positive water balance.

[38] Larger scale agents of climate change such as warming by elevated concentrations of greenhouse gases may modify the range of grasslands areas that are prone to salinization. According to the climatic water balance threshold that we defined in our study, a 3°C temperature raise would expand areas with potential salinization risk by 15% (Figure 6), engulfing zones where the fastest rates of afforestation are currently taking place (e.g., Corrientes province (Argentina) and Uruguay). However, if this temperature increase is accompanied by a 10% increase in annual rainfall, the area at risk of salinization would decrease by ~1% (Figure 6). Modeling experiments that explicitly consider cross-scale interactions and feedbacks mechanisms will help to uncover the reciprocal relationships between climate, hydrology and vegetation. A continuous update of regional climate will help reassess areas at risk.

[39] We have shown how vegetation interacting with climate triggers salinization from stand to regional scales by altering the direction and intensity of water fluxes across the ecosystem-groundwater continuum. Groundwater consumption, rarely studied in ecological systems, emerges as a key process explaining salt accumulation and shifts in productivity after tree establishment in native grasslands of the Río de la Plata. A general understanding of the relationship between vegetation and groundwater use will help land managers and policy makers maximize stand productivity while managing the negative consequences of groundwater consumption at scales ranging from individual stands to entire regions.

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