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The ecohydrological imprint of deforestation in the semiarid Chaco: insights from the last forest remnants of a highly cultivated landscape

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Abstract:

The semiarid Chaco plains present one of the highest rates of forest clearing and agricultural expansion of the world. In other semiarid plains, such massive vegetation replacements initiated a groundwater recharge and salt mobilization process that, after decades, raised regional water tables and salts to the surface, degrading agricultural and natural ecosystems. Indirect evidence suggests that this process (known as dryland salinity) began in the Chaco plains. Multiple approaches (deep soil profiles, geoelectric surveys and monitoring of groundwater salinity, level and isotopic composition) were combined to assess the dryland salinity status in one of the oldest and most active agricultural hotspots of the region, where isolated forest remnants occupy an extremely flat cultivated matrix. Full vadose moisture and chloride profiles from paired agriculture-forest stands (17 profiles, six sites) revealed the following: a generalized onset of deep drainage with cultivation (32 to >87 mm year^{-1}), full leaching of native chloride pools (13.7 ± 2.5 kg m^{-2}) down to the water table after >40 years following clearing and differential groundwater table rises (0.7 to 2 m shallower water tables under agriculture than under neighbouring forests). Continuous level monitoring showed abrupt water table rises under annual crops (up to 2.6 m in 15 days) not seen under forests or pastures. Varying deep drainage rates and groundwater isotopic composition under agricultural plots suggest that these pulses are strongly modulated by crop choices and sequences. In contrast to other dryland salinity-affected areas of the world, forest remnants in the study area (10–20% of the area) are not only surviving the observed hydrological shifts but also sustaining active salty groundwater transpirative discharge, as evidenced by continuous water table records. The overall impact of these forest remnants on lowering neighbouring water tables would be limited by the low hydraulic conductivity of the sediments. As highly cultivated areas of the Chaco evolve to new hydrological conditions of shallower saline water tables, innovative crop rotations that minimize recharge, enhance transpirative discharge and tolerate salinity will be needed. Copyright © 2016 John Wiley & Sons, Ltd.

KEY WORDS agriculture expansion; dryland salinity; deep drainage; groundwater recharge; land clearing

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INTRODUCTION

The semiarid Chaco is one of the most active agricultural frontiers of the world (Grau *et al.*, 2008; Gasparri *et al.*, 2013; Vallejos *et al.*, 2015). Over the recent decades, this ecoregion, originally dominated by xerophytic forests and subordinate grasslands, has been intensively cleared to grow crops and pastures (Zak *et al.*, 2004; Clark *et al.*, 2010). The agriculture expansion started in the most humid edges of the region and is now rapidly moving towards drier and more fragile areas (Grau *et al.*, 2005; Gasparri and Grau, 2009). The production of crops in semiarid plains entails unique challenges given that even a small change in any of the ecohydrologic controls (climate, soil and vegetation) may alter significantly the water balance, with consequences difficult to revert. Around 15.8 Mha of Chaco dry forests (i.e. 21% of the region) have already been cleared with marked local differences in the time and severity of deforestation (Vallejos *et al.*, 2015). The rate and extent of these land-cover changes indicate a need to diagnose and anticipate the potential eco-hydrological consequences of agriculture expansion in the region.

In other semiarid sedimentary plains of the world, the massive replacement of dry forests by annual crops or pastures strongly disrupted the regional water balance and progressively led to almost irreversible salinization processes (Peck and Williamson, 1987; Allison *et al.*, 1990; George

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et al., 1997; Zhang *et al.*, 2001; Scanlon *et al.*, 2006; Jobbágy *et al.*, 2008). This phenomenon, globally known as dryland salinity (as it occurs without irrigation), has been particularly important in SW and SE Australia where the clearance of native eucalypt forests since the end of 19th century increased deep drainage rates, generating groundwater recharge and a general rise of water table levels (Peck and Williamson, 1987; Allison *et al.*, 1990). As a result, high amounts of salts naturally stored in the vadose zone of the soil for millennia were leached by drainage and gradually moved with the water table towards the soil surface where they concentrated by evaporation and root uptake, affecting not only crops and pastures but also the survival of native vegetation remnants (George *et al.*, 1997; Cartwright *et al.*, 2004). The area affected by dryland salinity in Australia has been estimated to be 5.7 Mha in 2001 and would reach 17 Mha by 2050 (NLWRA, 2001). Similar processes have been reported in a lesser extent in other semiarid plains of the world (Worcester *et al.*, 1975; Daniels, 1987; Scanlon *et al.*, 2007a), and early signs of dryland salinity development are becoming evident in different zones of the South-American Chaco (Nitsch, 1998; Glatzle *et al.*, 2001; Jobbágy *et al.*, 2008; Amdan *et al.*, 2013) and Espinal (Santoni *et al.*, 2010; Jayawickreme *et al.*, 2011; Contreras *et al.*, 2013; Marchesini *et al.*, 2013).

Understanding the hydrologic impacts of recent land-cover changes in the semiarid Chaco is crucial to diagnose the potential risk of dryland salinity and for the planning of timely adaptation and mitigation strategies. In semiarid areas, native vegetation normally evapotranspires most of the incoming rainfall, generating nil liquid losses as runoff or deep drainage (Scanlon *et al.*, 2005a, b). The perennial lifecycle, deep root systems and the ability to extract water at very low water potentials are adaptative traits that confer native species a very effective water-capture capacity that limits the duration and depth of episodic pulses of high soil moisture (Scanlon *et al.*, 2005a; Seyfried *et al.*, 2005). In the absence of deep drainage, the salts that have been deposited in the soil for millennia have accumulated in the vadose zone below the root zone. The replacement of native forests with annual crops or pastures alters many vegetation parameters necessary to maintain the no-drainage condition. As crops present shallower root systems and shorter and more seasonal lifecycles, the patterns of soil water storage and consumption are modified, and the frequency and magnitude of drainage events increase (Scanlon *et al.*, 2005b; Radford *et al.*, 2009). The occurrence of drainage, where it was negligible before, causes the leaching and mobilization of stored salts deeper in the vadose zone. Eventually, drainage pulses and dissolved salts reach the water table, and successive drainage pulses generate groundwater recharge and the rise of saline water tables.

Besides the high deforestation rates, the semiarid Chaco presents conditions that may increase the dryland salinity hazard, including an extremely flat topography, high stocks of salt in the soil profile and high variability in annual rainfall (Jobbágy *et al.*, 2008; Amdan *et al.*, 2013). Although land-use changes may have an immediate effect on drainage rates, their more severe hydrologic consequences could take a long time (decades or centuries) to become evident and harmful. Cumulative drainage pulses must wet the entire vadose zone to field capacity to cause recharge, and saline water tables must get close to the soil surface to affect the growth of crops and native species (Scanlon *et al.*, 2007b; Radford *et al.*, 2009). So, in this sense, it is possible that different stages of the dryland salinity process could be developing unnoticed in different parts of the Chaco region (Grundy *et al.*, 2007).

In this work, we assessed the dryland salinity risk from an ecohydrological perspective in one of the most active and intensively transformed agricultural hotspots of the semiarid Chaco (Bandera, Santiago del Estero province, Argentina), where the consequences of land-cover changes would presumably be in a more advanced stage. The main questions that guided our work were as follows: (1) To what extent have deforestation and cultivation (annual crops and pastures) triggered the onset of deep drainage? (2) Is groundwater recharge taking place? What is the magnitude and timing of drainage and recharge? (3) How do these hydrological shifts affect salt transport and damage to crops and forest remnants? and (4) To what extent are these effects modulated at the landscape level by the spatial and temporal interactions among different cover types? By answering these questions, we expect to understand the risks, rates and mechanisms of dryland salinity of our study region and the other multiple deforestation fronts of the Chaco that are following its fate.

MATERIALS AND METHODS

Study area

The study area covers approximately one million hectares in the agricultural zone of Bandera in the SE edge of the semiarid Chaco (General Taboada and Belgrano departments in Santiago del Estero province, Argentina). It hosts one of the main agricultural clusters of the region, which presents some of the highest deforestation rates of the last decades (Vallejos *et al.*, 2015). Around 70% of the area has been transformed for production purposes, with a mean clearing rate of 1.7% of the area per year since 1990 (Figure 1). Soybean is the dominant crop (60–70% of the annual crops area), followed by maize, sorghum, cotton, sunflower and wheat. *Panicum maximum* and *Cenchrus ciliaris* are the

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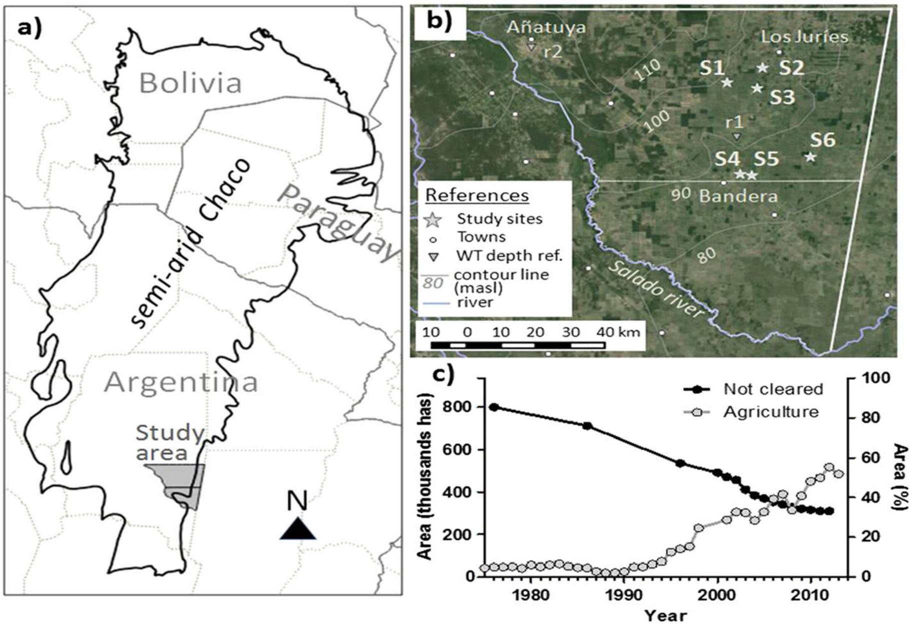

Figure 1. (a) Location of the study area in the semiarid Chaco. (b) Location of study sites. (c) Evolution of not cleared and annual crop areas in the study region (1975–2013). Not-cleared area was obtained from Vallejos *et al.* (2015), and agriculture area was calculated from SIIA data of individual summer crops records. In (b), the location of two railway stations with historical water table (WT) records is indicated: r1, Sanavirones st. (WT depth, −8.2 m; WT salinity, 12.3 g l^{-1}; Aug 1916); r2, Añatuya st. (WT depth, 10 m, 1937; Dr. Alfredo Martín, pers. comm)

main pastures sown for beef cattle production. Dry forest remnants represent <20% of the area, concentrated mostly in marginal zones for agriculture towards the west, and only distributed sparsely as tree corridors and isolated patches across the most intensively cropped zone.

Climatic conditions are highly variable with a mean annual rainfall of 975 mm year^{-1} (cv 30%) and a potential evapotranspiration (ET$_0$) of 1450 mm year^{-1} (c.v. 6%), according to the Priestley–Taylor method modified by Ritchie (1998). Rainfall concentrates in summer (~72% between November and March) and ET$_0$ exceeds rainfall year-round except in March and April. Mean annual temperature is 21 °C, with mean minimum of 7 °C and mean maximum of 33–34 °C in the coldest and warmest months, respectively. Precipitation in the study area increases from west (<800 mm year^{-1}) to east (>900 mm year^{-1}) (INTA, 1978).

The topographic gradient is very low (regional slope < 0.1%), without prominent forms or clear networks of surface water evacuation, except for the Salado River that flows NW–SE across the western edge of the area. Local micro-relief forms include palaeo riverbeds, interfluvial plains and hollows. Soils are predominantly mollisols and alfisols of silty loam to silty-clay loam texture (INTA, 1978).

Native vegetation varies with the micro-relief. Open forests of the extended plains have an overstorey predominantly of *Aspidosperma quebracho-blanco*, *Prosopis nigra*, *Cercidium australe* and *Jodina rhombifolia*; the understorey is dominated by *Acacia aromo*, *Larrea divaricata* and *Condalia microphylla*; and *Digitaria californica* dominates the herbaceous storey. In more low-lying areas (concave plains, which are also more susceptible to salinization), the forests are dominated by *Allenrolfea vaginata*, *Prosopis ruscifolia*, *P. nigra*, *A. quebracho-blanco*, *Cereus* sp. and *Opuntia quiscaloro*. Palaeo river beds are dominated by *Elionurus muticus* grasslands (INTA, 1978).

Sampling sites

In order to analyse differences in groundwater recharge associated with land-cover changes, in Oct 2012, we sampled five sites (S1 to S5) across the study area with paired plots that included a dry forest remnant (10–200 ha) and an adjacent crop plot (40–80 ha) cleared before 1972, as confirmed by Landsat images. As dry forest plots (DF) were meant to reflect the pristine condition of their respective agriculture (AG) counterparts, both sampled plots at each site share the same soil type and topographic position. Another site (S6) was sampled in May 2013 to deepen the analysis in the temporal and spatial dimensions. S6 included a transect with a forest patch surrounded by two agriculture plots that were deforested more recently (1995 and 1993, for the western AG$_W$ and eastern AG$_E$ plot, respectively). A *P. maximum* pasture (P) was added later to S6 (Aug 2014) to represent a more recent situation of land-cover change (cleared in 2008) of different land use.

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Vadose zone and groundwater observations

At each plot, soil samples were taken from the whole vadose zone (50-cm intervals) down to the water table. A 10-cm-diameter hand auger was used to perform unscreened boreholes in each sampling site. Soil samples were homogenized in the field and immediately sealed in polyethylene bags (double bag for each sample), identified and kept in cool and dark conditions until laboratory analyses (1–4 days later). The depth of water table from the ground surface was registered, and groundwater samples were taken (bailed from the bottom of the borehole after being purged twice and refilled with groundwater by lateral transport) and carefully sealed for isotopic and other laboratory analyses.

Soil laboratory analyses included gravimetric water content (WC_g), electric conductivity ($EC_{1:2}$) and chloride concentration of the soil solution (Cl_{St}). Gravimetric water content was measured by weighing soil subsamples (150–200 g) before and after oven drying at 105 °C to constant weight. Subsamples for $EC_{1:2}$ and Cl_{St} determinations were air dried and sieved (2-mm particles) to prepare 1:2; soil-deionized water extracts were shaken (12 h) and allowed to stabilize (12 h) to allow the solids to decant. Electrical conductivity of the extract was measured with a conductimeter that automatically performs temperature corrections (Orion 105A+, Orion Research, Inc., MA, USA). Chloride concentration of the extract ($Cl_{1:2}$) was measured with a solid state ion-selective electrode (Orion 94-17BN, Thermo Electron Corporation), calibrated every six samples using known chloride concentration solutions. Chloride concentration in the soil pore water (Cl_{St}) was calculated by correcting for the dilution factor. The amount of chloride stored at different depths of the vadose zone (Cl_{soil}) was obtained by the product of chloride concentration of the pore water by the weighted volumetric water content of the soil layer (WC_v), assuming a mean bulk density of 1.3 g cm^{-3}. Paired *t*-tests were employed to assess differences in soil moisture, vadose zone and groundwater salinity and chloride content, and water table depth, between AG–DF pairs.

Soil chloride profiles were employed to estimate deep drainage rates according to the chloride front displacement (CFD) method (Walker *et al.*, 1991), which is one the most successful methods to estimate drainage changes resulting from land cover variations in semiarid environments (Allison *et al.*, 1994). Chloride is a useful tracer of water movement because it is a highly mobile ion incorporated by atmospheric deposition, with negligible contribution from the soil parental material and minimal vegetation absorption. Natural ecosystems in semiarid regions generally present large chloride accumulations in the vadose zone as a result of long-term accumulation of atmospherically deposited chloride and nil deep drainage. The increased drainage that typically follows the conversion of natural to agricultural ecosystems usually results in the downward displacement of the chloride stocks. Drainage rate is calculated by multiplying the apparent velocity of the chloride front (vcf) by the average volumetric water content in the flushing zone (WCv) as represented in Equation 1, where the chloride front depth represents the depth at which chloride concentrations increase sharply from low (high drainage zone) to high concentration (little drainage zone) (Scanlon *et al.*, 2007b).

$$Dr = Vcf*\overline{WCv} = \frac{(z2 - z1)}{t2 - t1}*\overline{WCv} \qquad (1)$$

where $z1$ and $z2$ are the chloride fronts under native vegetation and under cultivation, respectively, and the time interval ($t2$–$t1$) is the time elapsed (years) since vegetation clearing to the moment of sampling.

Deep drainage refers to water fluxes below the root zone that have not yet reached the water table, whereas the term recharge is used once deep drainage has reached the water table (Scanlon *et al.*, 2007 a, b). So, in this sense, we recognized three hydrologic conditions for cleared plots: (i) recharge, where the chloride bulge is absent (full leaching of chloride) and the water table is higher than that of the respective forest plot; (ii) drainage, where the chloride front is below the root zone (assumed ~2 m for most crops; Dardanelli *et al.*, 1997) but did not reach the water table; and (iii) chloride displacement without drainage, where the chloride front displacement occurs within the root zone (<2 m depth). When recharge occurs, CFD provides only a lower bound on recharge rates given that the chloride front is absent and $z2$ takes the value of the original water table level (Scanlon *et al.*, 2007b).

Groundwater analyses included electric conductivity (EC) and stable isotopes composition. For conductivity measurements, groundwater samples were filtered (Whatman filter paper 0.4 μm) and measured with the same device employed in soil conductivity determinations. Groundwater stable isotopes composition [oxygen-18 (^{18}O) and deuterium (^{2}H)] was determined at the Grupo de Estudios Ambientales laboratory (San Luis, Argentina) with a L2130-i isotopic water analyser (Picarro Inc., California, USA; www.picarro.com). Isotopic data are reported in the standard delta notation in parts per thousand relative to the Vienna Standard Mean Ocean Water (Coplen, 1995), with a 1σ precision of <0.025‰ for $\delta^{18}O$ and 0.1‰ for $\delta^{2}H$. Positive values indicate the sample to be enriched in the heavy isotope species, and negative values indicate depletion in the heavy isotope species relative to the standard. As there are no local data on precipitation isotopic composition, data from the nearest International Atomic Energy Agency's Global Network of Isotopes in Precipitation (IAEA-GNIP)

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station were used to compute the local meteoric water line. Specifically, data of Miramar (<200 km to the south of the study area; 30.92S 62.97W) for the 1995–2002 period were obtained from the IAEA's Water Isotope System for Data Analysis, Visualization, and Electronic Retrieval interface (https://nucleus.iaea.org).

Geoelectric surveying

Electromagnetic induction (EMI) surveying was used as a reconnaissance tool to investigate the spatial variability of soil moisture/salinity across land covers and to validate the representativeness of soil coring samples (Scanlon *et al*., 1999; Gates *et al.*, 2011). EMI is a noninvasive technique that measures a depth-weighted average of electrical conductivity (i.e. apparent electrical conductivity, ECa), which varies primarily with clay content, water content and salinity (Scanlon *et al*., 1999). We performed geoelectric tomographies along crop–forest–crop transects (0.5 to 2 km long) in three of the study sites (S1, S5 and S6), with a portable electromagnetic conductivity meter (CMD-Explorer, GF_Instruments, Czech Republic) (Figure S1). In two of them (S1 and S5), we crossed the dry forest remnant from side to side, while in the other (S6), we merely surveyed its southern margin (1 km length, ≤1 m from the edge) because of difficulties of moving the EMI device through the thick vegetation. In S6, we also surveyed a pasture plot (80 ha) next to the transect.

We used the instrument in the vertical dipole mode to sense the upper ~6 m of the soil profile, taking measurements intervals of 3–4 m along the transects. In this mode, the instrument allows simultaneous measurements integrated at three depth ranges (0–2.2, 0–4.2 and 0–6.7 m). Also, we specifically measured ECa in most of the locations of soil collection (and others not included in this work) to calibrate electromagnetic measurements with field data. Correlation analyses ($n = 18$ boreholes; 10 AG, 7 DF and 1 P) were performed between the ECa measurements at the three depths, and WC_v, $EC_{1:2}$ and Cl_{soil} determined in laboratory and integrated at two depth intervals (0–2 and 0–5 m). ECa differences among covers in each site were assessed by analysis of variance.

Groundwater level monitoring

Submersible pressure transducers (HOBO water level logger; Onset Computer Corporation, Bourne, Massachusetts, USA) were installed in five monitoring wells (cased boreholes) to obtain continuous water table depth records from contrasting land covers, at high vertical (1.4 mm) and temporal (30-min intervals) resolutions. Two of them were installed in May 2013 in site 6, under the forest and western agriculture plots (S6_DF and S6_AG), while the others were added in Aug 2014, one in the pasture of site 6 (S6_P) and the others in the paired AG–DF stands of site 5 (S5_DF and S5_AG). An extra transducer was installed in site 6 to perform barometric corrections to water-level measurements.

The monitoring period included two full crop seasons for S6_DF and S6_AG (crop season: 1 Jun to 31 May of the following year), and only one in the rest of the plots. These seasons were relatively wet (914 and 1056 mm, for 2013/2014 and 2014/2015 crop seasons, respectively) when compared with those of the previous decade (mean of the 2003/2004–2012/2013 decade = 754 mm). Crop sequence in S6_AG included an 8-month fallow, followed by a maize crop (Jan 2013–Jun 2014) and then a 6-month fallow followed by a soybean crop (Dec 2014–May 2015). Measurements in S5_AG only included a soybean crop (Dec 2014–May 2015) and most of the previous 6-month fallow. The S6_P included a full growing season of a 6-year *P. maximum* pasture, which normally grows from September to July. Native vegetation grows actively from September/October to July/August.

Fluctuations in groundwater levels over time have been used to estimate recharge in unconfined aquifers (Healy and Cook, 2002). Assuming that rises in groundwater level are due to recharge water arriving at the water table, recharge can be estimated with the water-table fluctuation method (Equation 2):

$$R = S_y \frac{\Delta H}{\Delta t} \tag{2}$$

where S_y is the specific yield of the aquifer and ΔH is change in water-table height over the time interval Δt.

Equation 2 was applied for each water-level rise, where ΔH was set equal to the difference between the peak of the rise and the low point of the extrapolated antecedent recession curve at the time of the peak. This antecedent recession curve represents the trace that the well hydrograph would have followed in the absence of the precipitation event that triggered the water-table rise (Healy and Cook, 2002). An S_y value of 0.05 was determined in the laboratory as the mean difference in volumetric water content of soil samples between field capacity and saturation. This value is in agreement with the 0.06 value proposed by Johnson (1967) for silt loam soils and the 0.04 value computed by Loheide *et al.* (2005) for the readily available specific yield for a similar soil texture.

RESULTS

Salinity and water table changes associated to old clearings

The conversion of native dry forests to rain-fed agriculture increased deep drainage rates and triggered the recharge processes. While forest remnants displayed

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high salinity and chloride stocks through the vadose zone, evidencing the native zero-drainage condition, agriculture soil profiles presented less salty vadose zones and a partial to full depletion of the chloride stocks (Table I). In the root zone for most crop plots (0–2 m), salinity levels were reduced by ~80% ($EC_{1:2}$ 1.8 in DF *vs* 0.3 dS m^{-1} in AG) and chloride content by ~90% relative to soils under forest remnants, minimizing soil osmotic restrictions for crop growth. Differences were also evident in deeper layers of the vadose zone (2–5 m) where salinity levels were reduced on average by 50% and chloride content by ~75%. On average, agriculture presented 3% more volumetric water content in the vadose zone, but this difference was not statistically significant ($p > 0.05$).

The absence of a chloride front in the vadose zone of agriculture plots and the consistently higher groundwater levels compared with those of their respective forest are clear evidences of groundwater recharge in old clearings (Table I; Figure S2). The missing chloride front indicates that drainage flux and chloride leaching have reached the water table. Conservative estimates with the CFD method indicate that a mean drainage rate >32–50 mm $year^{-1}$ would have been needed to displace chloride fronts down to the water table. The water table depth in forest remnants ranged from 4.6 to 8 m, following the regional topography, and rose ~0.67 m on average following clearance (Figure 2a). Along with the water table rise, the recharge in agriculture plots reduced groundwater salinity (approximately –43%) and chloride content (approximately –67%) at the water-table level (Table I; Figure 2b).

Salinity and water table changes in recent clearings

Boreholes in plots that were deforested recently (site 6) displayed different stages in the development of the recharge process. The plot deforested earliest (the eastern plot in the transect, AG_E, cleared in 1993) presented clear signs of groundwater recharge including a shallower water table (+2.3 m), fresher groundwater (5.8 *vs* 32–35 dS m^{-1}) and a chloride-free vadose zone (Figures 3 and 4a). The western plot, AG_W, which was cleared 2 years later (1995), also had a shallower water table (+1.6 m), but a smaller difference in groundwater salinity (25 dS m^{-1} in AG_W *vs* 32 dS m^{-1} in DF; Figure 3) and a 2.5-m displacement of the chloride front within the vadose zone (~4.5 *vs* ~2 m; Figure 4b), which suggest an incipient recharge. Finally, the pasture plot cleared in 2008 presented only a slight water table rise (+0.5 m) and salinity decline (30.5 *vs* 32 dS m^{-1}) and a very short chloride front displacement (~1.5 *vs* ~1 m; Figure 4c). When the pasture was re-sampled the following year (2015), an additional chloride front displacement of 0.5 m was detected, indicating a downward water flux within the root zone that has not yet translated into deep drainage. No chloride displacement was detected for the forest in the same resampling (Figure 4c).

Estimates with the CFD method indicate recharge rates >87 mm $year^{-1}$ for AG_E, drainage rates of 40–48 mm $year^{-1}$ for AG_W and no drainage for the pasture. Although forest eastern and western edges samples presented no drainage, an ~80-cm displacement of the chloride front within the root zone would suggest a slight increase in downward water flux (around 5–7 mm $year^{-1}$) compared with that in the central sample (Figure 4a and

Table I. Mean volumetric water content (VWC), salinity ($EC_{1:2}$) and chloride stock (Cl) at different depth ranges of the vadose zone and water table depth and groundwater salinity and chloride content in paired forest–agriculture plots (sites S1 to S5).

	Dry forest	Agriculture	Mean difference	Significance
Soil				
$WCv_{0–2\,m}$ (%)	19.8 ± 2.0	23.0 ± 1.8	3.2	ns
$WCv_{2–5\,m}$ (%)	25.0 ± 3.6	28.2 ± 1.8	3.2	ns
$EC_{1:2\ 0–2\,m}$ (dS m^{-1})	1.8 ± 0.5	0.3 ± 0.2	−1.5	*
$EC_{1:2\ 2–5\,m}$ (dS m^{-1})	2.5 ± 0.5	1.3 ± 0.5	−1.3	*
$Cl_{0–2\,m}$ (kg m^{-2})	5.0 ± 1.6	0.4 ± 0.3	−4.7	*
$Cl_{0–5\,m}$ (kg m^{-2})	12.4 ± 3.3	2.0 ± 1.2	−10.4	**
$Cl_{0\,m\text{-}WT}$ (kg m^{-2})	15.9 ± 2.2	2.2 ± 1.2	−13.7	***
Groundwater				
Depth (cm)	630 ± 70	563 ± 71	−66.8	**
EC (dS m^{-1})	30.4 ± 1.8	17.2 ± 2.4	−13.2	**
Chloride (mg l^{-1})	12780 ± 730	4226 ± 1315	−8553	***

The last column presents the significance level of AG *versus* DF differences according to a paired *t*-test.
ns, not significant, $p > 0.05$.
*$p < 0.05$.
**$p < 0.01$.
***$p < 0.001$.

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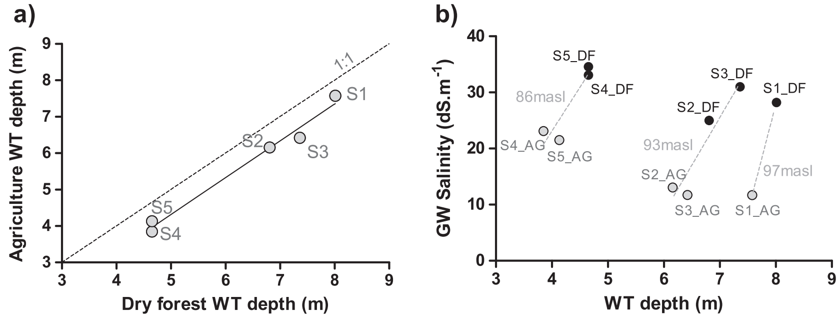

Figure 2. Changes in water table (WT) depth and groundwater (GW) salinity after >40 years of forest clearing in five study sites (S1 to S5). (a) Water table depth under agricultural plots in relation to its depth under the neighbouring forest plot. (b) Groundwater salinity in relation to water table depth for agricultural (grey markers) and dry forest (black markers) plots. Linear model in (a): AG WT depth = DF WT depth * 1.01–0.73; $r^2 = 0.98$; $p < 0.001$. Dashed lines in (b) show water table changes for sites grouped by their elevation altitude (metres above sea level)

b). Analysing spatially the transition between the western agriculture plot and the forest, abrupt changes in vadose zone and groundwater salt content were detected, but the change in groundwater level was more gradual (Figures 3 and 4b).

Geoelectric survey

Statistical differences in apparent electric conductivity (ECa) were detected among the land covers surveyed by EMI (Figure 5). ECa was mainly correlated to chloride content and, to a lesser extent, to $EC_{1:2}$ in the upper 2 m of the vadose zone (Table II). The highest contrast among land covers was found in site 5, where dry forest ECa values more than doubled those of the old-cleared agriculture plot (AG, cleared before 1972) and were also higher than those of a more recent clearing (AG*, cleared in 2007), which presented intermediate values. Transforming ECa measurements into chloride stocks by Equation 3, the forest presented $17.4 \pm 1.6\,\mathrm{kg\,m^{-2}}$ of chloride in the upper 5 m of the vadose zone, the old agriculture only $4.4 \pm 0.5\,\mathrm{kg\,m^{-2}}$ and the recently cleared $15 \pm 1.9\,\mathrm{kg\,m^{-2}}$, evidencing less chloride leaching by drainage. A similar but less contrasting pattern was found in site 1, where forest presented higher ECa than the surrounding agriculture plots (AG and AG*, cleared before 1972 and in 1978, respectively) but with smaller differences in chloride stocks (Cl_{soil} 0–5 m: 4.7 $\pm 0.9\,\mathrm{kg\,m^{-2}}$ in DF, $2.2 \pm 0.3\,\mathrm{kg\,m^{-2}}$ in AG and 2.8 $\pm 0.6\,\mathrm{kg\,m^{-2}}$ in AG*). Site 6 presented less contrasts among vegetation covers: The forest presented higher ECa values than the western agriculture plot (AG_W) but similar to those of the eastern one (AG_E) and lower than those of the southern pasture (P). As a result, small differences were estimated in the chloride stock below these covers (from $3.4 \pm 0.2\,\mathrm{kg\,m^{-2}}$ in AG_W to 6.1 $\pm 0.5\,\mathrm{kg\,m^{-2}}$ in P). A bias in Equation 3 would have avoided the expression of stronger contrasts among land covers in sites 1 and 6, where chloride stocks under forest tended to be underestimated.

The geoelectric survey supported the representativeness of the sampled boreholes. In general, most of the boreholes presented ECa values close to the median and between the 25th and 75th percentiles of their respective plots (Figure 5). The only exception was AG_E in site 6,

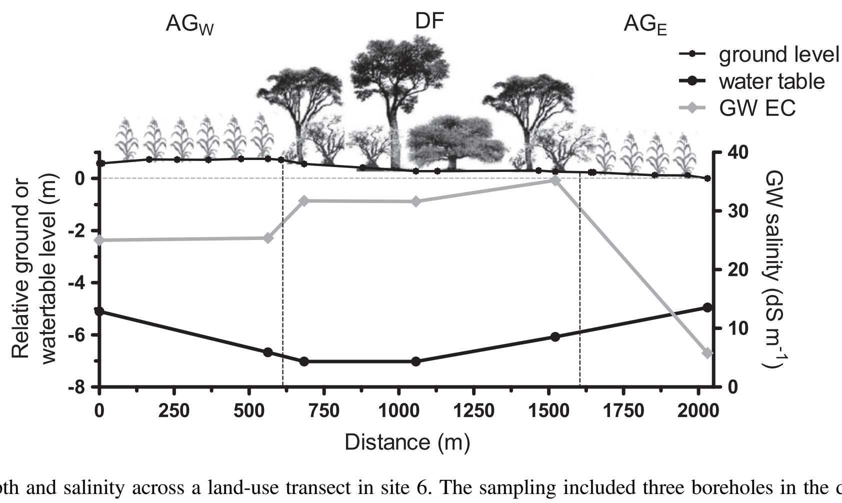

Figure 3. Water table depth and salinity across a land-use transect in site 6. The sampling included three boreholes in the dry forest patch, two in the western agricultural plot and one in the eastern one. Relative ground level measurements are also presented to discard topographic effects

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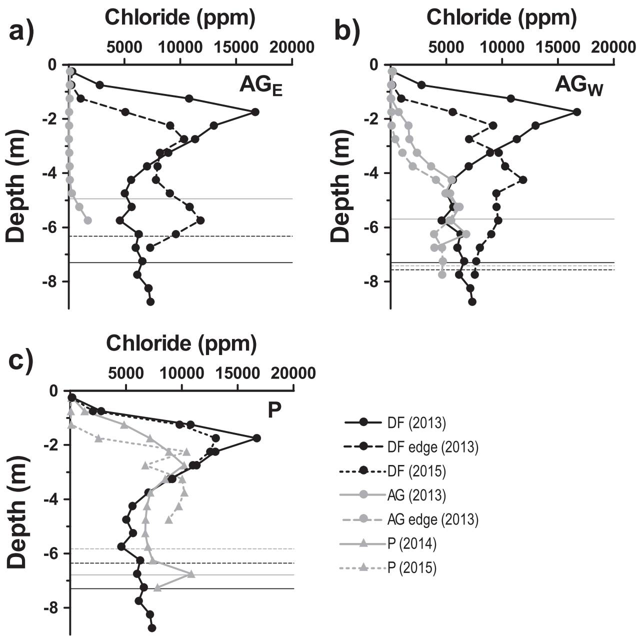

Figure 4. Soil-water chloride concentration profile of different across a land-use transect in site 6. (a) Eastern agricultural plot (AG_E) and dry forest E edge. (b) Western agricultural plot (AG_W) and dry forest W edge. (c) Pasture plot (P) south of the dry forest stand, sampled in two moments. In all plots, the central dry forest (DF) measurement was included as reference, and groundwater level measured in each borehole is presented as horizontal lines. Year of measurement is presented between brackets

which presented an ECa value much lower than the 25th percentile, indicating that the soil samples taken from this borehole did not adequately represent the general condition of the plot. The lower chloride content in this spot (1.6 kg m^{-2} *vs* a plot median of 5.1 kg m^{-2}) suggests that the borehole of AG_E was located in a micro-site of focused recharge, which could not be distinguished in the field at the moment of sampling, when a mature soybean canopy was present.

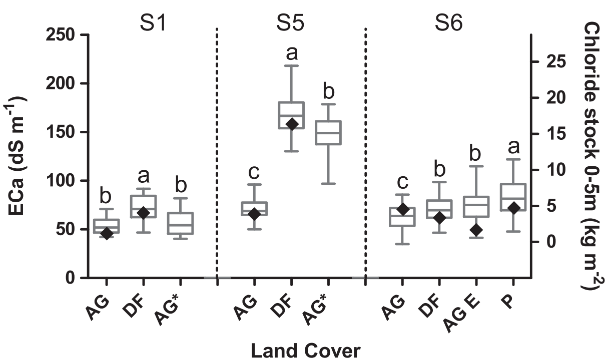

Figure 5. Apparent electrical conductivity (ECa) or chloride stock to a 5-m depth (estimated from ECa) under different land covers (AG, agriculture; DF, dry forest; P, pasture) in three study sites (S1, S5 and S6). Different letters (a, b and c) indicate significant differences among land covers ($p < 0.05$) for a one-way analysis of variance performed in each borehole site. Diamonds indicate the ECa value measured exactly in each sampling site. AG* are plots surveyed by EMI but not sampled for laboratory analyses

Isotopic analysis of groundwater

Differences in the isotopic composition of groundwater samples provide further evidences supporting the occurrence of episodic and asynchronous groundwater recharge in cleared plots and no recharge under forest remnants (Figure 6). Despite the high variability in rainfall isotopic composition ($-4.3 \pm 2.0‰$ $\delta^{18}O$ and $-22.9 \pm 15.6‰$ $\delta^{2}H$, mean and SD, respectively), groundwaters from different forest remnants were very homogeneous ($-5.0 \pm 0.2‰$ $\delta^{18}O$ and $-29.0 \pm 1.2‰$ $\delta^{2}H$). Comparatively, agriculture groundwaters displayed a more variable and lighter isotopic composition than those of forests (i.e. $-6.0 \pm 0.5‰$ $\delta^{18}O$ and $-33.40 \pm 3.0‰$ $\delta^{2}H$) with both situations showing little overlap. Differences in isotopic composition were also evident between groundwater samples of a same agriculture plot (S5_AG) taken at different moments. The pasture groundwater sample displayed intermediate isotopic values, within the isotopic ranges of forest groundwater ($-5.3‰$ $\delta^{18}O$ and $-30.3‰$ $\delta^{2}H$).

Water table fluctuations under different covers

Groundwater levels showed contrasting dynamics and response to rainfall events under different covers. Despite being deeper, water tables under forest displayed more frequent rises after rainfall events than their respective agriculture counterparts, which presented fewer and more discrete and abrupt rises (Figure 7). These contrasts were

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Table II. Pearson coefficient (r) for correlation analyses among electromagnetic measurements (ECa, three depth ranges) and laboratory determinations from field samples (WC_v, $EC_{1:2}$ and Cl_{soil}) integrated at two depth ranges of the vadose zone (0–2 and 0–5 m).

	WC_v (0–2 m)	WC_v (2–5 m)	$EC_{1:2}$ (0–2 m)	$EC_{1:2}$ (2–5 m)	Cl_{soil} (0–2 m)	Cl_{soil} (0–5 m)
$ECa_{0–2.2\ m}$	0.25 ns	0.48 ns	0.69**	0.38 ns	0.79***	0.83***
$ECa_{0–4.2\ m}$	0.25 ns	0.50*	0.67**	0.39 ns	0.77***	0.82***
$ECa_{0–6.7\ m}$	0.35 ns	0.56*	0.57*	0.35 ns	0.68**	0.74***

The linear model for the most correlated variables is presented in Equation 3.
ns, not significant.
*$p < 0.05$.
**$p < 0.01$.
***$p < 0.001$.

$$Cl_{soil}\ 0-5\ \text{m}\ (\text{kg m}^{-2}) = ECa_1\ (\text{dS m}^{-1}) * 0.135 - 5.01 \quad (r^2 = 0.69;\ n = 18;\ p < 0.001) \tag{3}$$

more evident along the more extended monitoring period of site 6. There, the forest (S6_DF) water table declined gradually, likely driven by transpirative discharge (i.e. groundwater consumption by plants) during the dry seasons (June to September) until the onset of the rainy season in October. Observations after the first rains suggest that water table under forest rises not as a result of direct recharge but responding to groundwater discharge interruptions and level equalization with the surrounding croplands, once dry forest species switched to the new fresh rainwater source available in the upper soil layers. During the wet season, the forest water table fluctuated alternating transient drops (discharge onset) and rises (discharge interruption) following all rainfall events. Conversely, the water table under agriculture (S6_AG) displayed slight or nil depth variations during the dry seasons, and part of the wet seasons, indicating the lack of discharge and recharge pulses. A slight but constant declining trend (0.5 mm day^{-1}) during the first crop season (2013/2014) showed no response to rain pulses, suggesting a slow subsurface discharge; yet, in March 2014, when the wetting front of successive rainfall events would have reached the water table, a gradual recharge process was initiated. In the following crop season (2014/2015), the combination of higher, more frequent and earlier rainfall inputs, together with lower atmospheric demand and a wetter antecedent soil moisture condition, favoured an abrupt recharge event that lifted the water

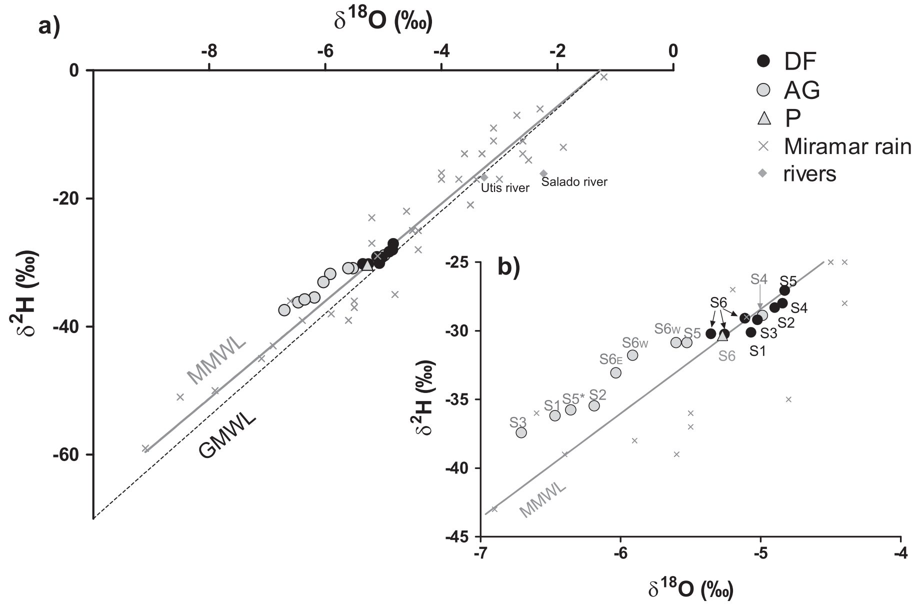

Figure 6. (a) δ^2H and δ^{18}O plot showing groundwater samples from different vegetation covers: dry forest (DF), agriculture (AG) and pasture (P). (b) Details of plot (a) where each sample is identified by its corresponding site (S1 to S6). The AG plot of site 5 was sampled in two moments (S5, 2012; S5*, 2014). Miramar meteoric water line (MMWL) was computed from isotopic composition of rainfall obtained from the IAEA GNIP database. GMWL, Global Meteoric Water Line

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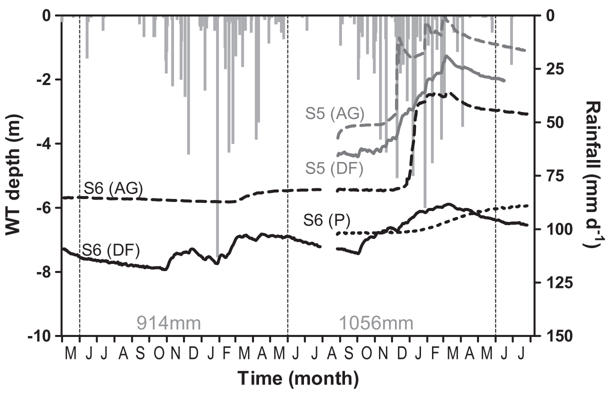

Figure 7. Water table dynamics under different land covers (DF, dry forest; AG, agriculture; P, pasture) in two study sites (S5 and S6). Observed period extended from May 2013 to July 2015 including two full cropping seasons (2013/2014 and 2014/2015)

table level 2.6 m up in only a month (from −5.4 m in 15 Dec 2014 to −2.8 m in 15 Jan 2015). Subsequent rains caused slighter water table rises (<0.2 m) in January–February that were followed by a gradual decline as rainfall frequency and amount declined. As a result, in 2 years, the water table under agriculture rose 2.7 m (−5.7 to −3.0 m) but only 1.1 m in the neighbour forest (from −7.5 to −6.4 m), despite the increase in the hydraulic gradient, which, on the other hand, seemed to have caused a gradual subsurface inflow below the pasture.

A similar dynamic took place in the agriculture plot of site 5 (S5_AG), where the shallower water table determined a more abrupt and anticipated recharge of 2.5 m in 2 weeks (from −3.2 to −0.7 m), which was followed by at least two more slighter recharge events during the wet season. Water table dynamics below the forest remnant in this site (S5_DF) seemed to be more coupled to that of agriculture, presenting a gradual but steady water table rise after each agriculture recharge event, to reach an equilibrium pressure head difference of ~1 m between covers. Groundwater under agriculture reached the soil surface once and remained within the direct evaporation discharge zone (0 to −2 m) during most of the wet season. According to the water table fluctuation method, cumulative recharge in S6_AG was 21 and 148 mm for the 2013/2014 and 2014/2015 crop seasons, respectively. The three recharge events in S5_AG accrued 213 mm for the 2014/2015 crop season, while the water table lift in S6_P corresponded to a total subsurface water inflow of 35 mm.

DISCUSSION

The complementary approaches applied in this study indicate that most of the processes that lead to secondary salinization of soils are taking place in the semiarid Chaco. The relatively old and widespread agricultural hotspot of Bandera seems to be transitioning from an original situation in which native vegetation evapotranspired most of the incoming rainfall, maintaining dry vadose zones and deep water tables (Figure 8a), into a new hydrological status in which most of the area generates not only deep drainage and local recharge but also evapotranspirative groundwater discharge and salt accumulation close to the soil surface (Figure 8c). So far, the water table has been deep enough not to affect crop growth and performance, and local farmers seem unaware of the salinity hazard (Figure 8b). However, during the last crop season (2014/2015), groundwater would have reached, perhaps for the first time, the critical depth to onset evapotranspirative discharge under shallow-rooted crops (~2 m; Nitsch, 1998; Nosetto *et al.*, 2009) in different spots of the study area. As long as the area subject to local recharge increases with deforestation and annual crop expansion over pastures (Houspanossian *et al.*, 2016), and water tables keep rising, reducing the thickness of the vadose buffer, the most visible symptom of dryland salinity, which involve waterlogging and surface salt precipitation, should become evident. To what extent current shallow and salty groundwater bodies are stimulating crop yields through water provision or limiting them through high salinity and waterlogging remains unclear but should be one of the first indicators to monitor if a fast response in the farming systems is sought.

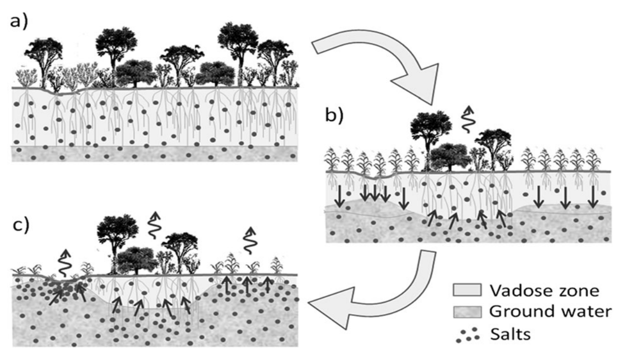

Figure 8. Conceptual scheme of the dryland salinity process in the Bandera area. (a) Pristine condition: dry forests consume exhaustively rainfall inputs, and deep water table is uncoupled from the atmosphere. (b) Current condition: salt leaching and groundwater recharge in cleared plots generate relatively fresh water lens above the naturally saline groundwater in recharge zones, while water table rising enhances evapotranspirative discharge and salt concentration beneath dry forest relicts. (c) Future condition: shallow water tables (<2 m) onset evapotranspirative discharge in cleared plots, which mobilize and concentrate soluble salts in the soil surface, particularly in microdepressions. As sensitive crops are affected, evaporative discharge becomes dominant. Forest relicts deal with shallower and saltier water tables possibly through the replacement of more sensitive species by more tolerant ones

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Onset of deep drainage

Addressing the first guiding question of this work (*To what extent deforestation and cultivation have triggered the onset of deep drainage?*), soil moisture and salinity profiles confirmed that the conversion of natural ecosystems to rain-fed agriculture/pastures in Bandera initiated deep drainage water fluxes that were nil before the vegetation change. High vadose salt and chloride loads and low water contents consistently found under the dry forest remnants indicate negligible drainage, as reported in other parts of the Chaco (Amdan *et al.*, 2013) and other semiarid plains of the world (Scanlon *et al.*, 2007a). Conversely, all the cleared plots presented full or partial chloride leaching evidencing downward water flux. Full flushing of chloride stocks took place after 40 years of cultivation in all sites, and earlier in low microsites that may have received focused recharge (as detected by geoelectric surveying). Salt leaching is still taking place where cultivation histories are shorter, particularly under pastures. Although the perennial pastures may slow down drainage and leaching fluxes, we observed a 0.5-m chloride front displacement under the pasture, which suggests that these systems may never achieve the no-drainage condition of forests.

Drainage rates of cleared plots, estimated with the chloride front displacement method, ranged from >32 to >87 mm year^{-1}, representing >3–9% of mean annual rainfall. These estimates were very similar to those computed by Amdan *et al.* (2013) in eastern Salta, an agriculture cluster in the opposite edge of the semiarid Chaco. Drainage differences among land covers were also supported by geoelectric surveying, which served to detect differences not only in the vadose zone chloride content of neighbouring parcels but also within the plots. The electromagnetic readings confirmed the representativity of point measurements in most of the boreholes, enhancing the reliability of soil-tracer drainage estimations, which are subject to spatial variability (Walker *et al.*, 2002). An interesting exception was the sample in S6_AG$_E$, which corresponded to a focused recharge spot within a plot that was still under the ongoing salt leaching stage (as shown by geoelectric profiling), highlighting the complementarity of these approaches under apparently homogeneous landscapes (Scanlon *et al.*, 1999; Dissataporn *et al.*, 2001).

Evidences of groundwater recharge

In response to the second guiding question (*Is groundwater recharge taking place? What is the magnitude and timing of drainage and recharge?*), we found evidences of ongoing recharge processes across the studied area. The most direct and conclusive evidences came from the abrupt water table rises registered under agriculture plots. Recharge according to the water table fluctuation method reached 150–200 mm year^{-1} in the last growing season, indicating much higher rates than those integrated along the whole cultivation history by the chloride front displacement method, which were in the order of 30–50 mm year^{-1}. This reflects the episodic nature of drainage/recharge fluxes in response not only to the variability in rainfall amount but also to its interaction with the crop sequence and the antecedent soil moisture condition (Tolmie *et al.*, 2011; Giménez *et al.*, 2015). Local groundwater recharge was not detected neither in the pasture nor in the forests, but levels under these covers rose in response to the generalized recharge in the surrounding matrix of annual crops.

Full leaching of chloride from the vadose zone and shallower water tables in comparison with forest remnants are indirect evidences of drainage fluxes recharging groundwater in old clearings (Cook *et al.*, 1989). Additional evidence came from groundwater chemistry and isotopic composition contrasts between cleared and non-cleared plots. While all forest remnants presented highly saline groundwater of relatively similar conductivity (EC: 31.3 ± 3.4 dS m^{-1}, mean and SD, respectively) and isotopic composition (−5.0 ± 0.2‰ δ^{18}O, −29.0 ± 1.2‰ δ^{2}H), agricultural groundwater presented lower and more variable salinity levels (17.6 ± 7.3 dS m^{-1}) and a more heterogeneous and isotopically depleted composition (−6.0 ± 0.5‰ δ^{18}O, −33.4 ± 3.0‰ δ^{2}H). The salinity and isotopic composition of the sampled pasture were very similar to those of the forest, confirming that the recently cleared pasture plot has not been subject to recharge yet. Two overlapping patterns emerge from the groundwater salinity observations. In the first one, across study sites, where water tables are located closer to the surface, groundwater is saltier; yet in the second one and opposing to this negative depth–salinity relationship, the onset of recharge under cultivation raises water tables but dilutes salts (Figure 2b). We find that these trends support the notion that under forests, there is no recharge and net discharge prevails, while under cultivation, recharge is the dominant flux. The transpirative discharge of forests is likely to be more intense where water tables are shallower (lower sites towards the south-east of the region), making groundwater under forest saltier (Nosetto *et al.*, 2013). The onset of cultivation, however, breaks this negative depth–salinity relationship by introducing a novel recharge flux, which is high enough in relation to the vadose salt pool that it supplies a solution of lower salinity than the phreatic aquifer (Cook *et al.*, 1989). Whether the current fresher aquifer found under croplands results from mixing and dilution of pristine groundwater or from the stratified settling of newly recharged water (Cartwright *et al.*, 2010) still needs to be resolved. The broad isotopic variability of agricultural groundwaters

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suggests a limited mixing, capable of preserving the signature of episodic and asynchronous precipitation-recharge events (Gonfiantini *et al.*, 1998; Yin *et al.*, 2011). On the contrary, the little variability of forest groundwater is typical of larger and deeper flow systems that are more resilient to temporal variations in recharge (Clark, 2015). Heavy rainfall events that tend to be more isotopically depleted seem to drive recharge under agriculture (Acheampong and Hess, 2000).

Implications on crops and dry forest remnants

Although water tables may be within the root zone of native vegetation, they are likely approaching shallower crop roots only very recently, which provides a preliminary and partial answer to the third guiding question of this work (*How do these hydrological shifts affect salt transport and damage to crops and forest remnants?*). Contrary to the Australian situation, where the rise of saline water tables has killed some of the native vegetation remnants (George *et al.*, 1997; Beresford *et al.*, 2004), forests remnants in the Chaco have been able to withstand high groundwater salinity levels (~30 dS m^{-1}) and even consume that source, as shown by our frequent water level observations and suggested by previous ecophysiological studies (Mitlöehner, 1998; Mitlöehner and Koepp, 2007). Pre-clearing records from an old railway station indicate that groundwater has always been relatively shallow and saline in the study area (water table depth, 8 m; groundwater salinity, 12.3 g l^{-1} at the Sanavirones station in 1906, Figure 1b), so native vegetation could have evolved under these conditions. On the contrary, the yield of the most common crops sown in the region are negatively affected by relatively low salinity levels (EC > 2–5 dS m^{-1}). It can be expected that the eventual rise of saline water tables would be detrimental for crop production (Maas, 1990), although so far, it may have benefited from the transient washing of salt stocks (Radford *et al.*, 2009). Moreover, contrary to what happened in Australia, where recharge augmented groundwater salinity, a progressive dilution of groundwater salinity seemed to occur in recharge zones. A gross extrapolation of the depth–salinity trend found under croplands suggests that in the highest sites, by the time water tables reach the crops root zone (~2 m), groundwater would be diluted enough to be harmless for crops osmotically, representing an extra source of fresh water. Nevertheless, under the semiarid conditions of the area, the presence of a shallow water table would surely lead to new processes of salt concentration by capillary transport and direct evaporation (Nosetto *et al.*, 2008), which could eventually limit crop and soil productivities (Cartwright *et al.*, 2004). A careful monitoring of water table levels and salt stocks and vertical distribution profiles is urgently needed throughout the region.

Interactions among land covers

This work not only contributes to our understanding of how vegetation changes alter vertical water transport along the soil–groundwater continuum but also provides information about surface and subsurface lateral flow, helping to address the last guiding question (*To what extent are these effects modulated at the landscape level by the spatial and temporal interactions among different cover types?*). Surface lateral flow and water redistribution within agricultural plots have been evidenced by our geophysical surveys and by the presence of patchy waterlogged areas observed in high-resolution images after heavy rainfall events (Figure S3). To what extent this lateral transport involves adjacent plots with different vegetation (i.e. surface water transfer from crops to forests, or between crops with different management/ phenology) is yet unclear. Subsurface lateral flow is evidenced by the gradual water table rise under forests and pasture plots that received no recharge but were subjected to increasing hydraulic gradients driven by agricultural recharge in the surroundings (Figure 7).

In the same way as distant agricultural recharge could raise water table levels under forests, forest transpirative discharge could contribute to lower water table levels in adjacent agriculture plots, particularly during dry periods. The magnitude of this influence depends on the hydraulic gradient generated between neighbouring forest and agriculture plots and their spatial configuration but is constrained by the low saturated hydraulic conductivity of the sediments in the study area (~0.16 m day^{-1}, according to a well test performed in the field, with the auger hole method; Van Beers, 1958). The current landscape configuration suggests that most of the area is prone to groundwater recharge, and little hydrologic control can be expected from the few and sparse forest remnants, as reported for reforestation initiatives in Australia, which required >70% of tree cover to be successful (Bari and Schofield, 1992).

Final comments

Evidences collected in this study indicate that dryland salinity is developing, until now unnoticed by farmers, in the area of Bandera. Although the most intense land-use changes took place just during the last 25 years, the naturally shallow (8–10 m) water table and the high proportion of the area sown with water-conservative crop schemes prone to drainage (Giménez *et al.*, 2014) may have led to a faster progress of the process compared with Australia. In fact, during the last crop season, groundwater would have reached, probably for the first time, a

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critical depth for evaporative discharge in many sites. If water tables remain shallow, the soil capacity to buffer further water excess would be diminished, and evaporation and capillary rise would lead to salt concentration in the soil surface. Although Bandera was purposely chosen to study the dryland salinity hazard in the Chaco, given its environmental and land-use history attributes, there are signs of dryland salinity in other spots of the region including the onset of deep drainage and salt leaching after deforestation in eastern Salta (Amdan *et al.*, 2013) or the rise of saline water tables after bush clearing in the Paraguayan Chaco (Nitsch, 1998; Glatzle *et al.*, 2001), which indicate that other agriculture fronts are going throw different chapters of the same story. Monitoring groundwater levels and salinity, salt distribution in soils and developing crop sequences that minimize recharge and have a chance to tolerate and use shallow salty water tables are urgent needs for farmers and policy makers in the broad and ever-changing plains of the semiarid Chaco.

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