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RESEARCH ARTICLE

Ecohydrological transformation in the Dry Chaco and the risk of dryland salinity: Following Australia's footsteps?

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Abstract

During the last century, the massive conversion of Australian dry forests to crops and pastures triggered the massive soil and groundwater degradation process known as dryland salinity. Currently, South American Chaco's dry forests are undergoing a similar transformation, leading global deforestation rates. The goal of this study was to review existing ecohydrological information about natural and cultivated systems in the Chaco to assess the dryland salinity risks. We review deep soil water, salt stocks, and groundwater recharge from agriculture or native dry forests stands located in a precipitation range of 450–1100 mm. We complement this with water table level records and geoelectric profiles together with personal observations. We use data from 15 Australian studies for comparison. Strong salt leaching, especially after 20 years of forest clearance, indicates the onset of deep drainage following forest conversion to agriculture in the Dry Chaco. Water stocks were more than double in the cleared stands compared to their dry forest pairs, and recharge rates were up to two order magnitude higher in agricultural areas. Although lower atmospheric salt deposition, younger sediments, and relatively high water-consuming agricultural systems in the Dry Chaco attenuate salinization risks compared to Australia, the very flat topography and related shallow water table levels of the South American region could make groundwater recharge and salt mobilization processes more widespread and difficult to manage. The lack of awareness among the general public, farmers, and decision makers about this issue amplifies the problem, making land management plans for the Argentine dry forest territories essential.

KEYWORDS

Australia, deep drainage, deforestation, dry forests, dryland salinity, South American Chaco

1 | INTRODUCTION

More than half of the dry forests of the world are located in South and Central America and despite representing a large percentage of their forested territory, they have received much less attention than their humid counterpart (Sánchez-Azofeifa et al., 2005). Climate change, fires, and cultivation are the main threats for this biome where only a small fraction is subject to protection (Miles et al., 2006).

Adaptations to drought such as the presence of a deep root system, strong osmotic adjustment, and the coexistence of deciduous or perennial sclerophyllous leaf habits among native species result in an exhaustive use of rainfall by vegetation with almost zero deep drainage and groundwater recharge in most dry forests (Jackson et al., 1996; Marchesini, Fernández, & Jobbágy, 2013; Wilcox, Breshears, & Allen, 2003). The partition between evapotranspiration and deep drainage is not only regulated by vegetation characteristics but also by soil texture and rainfall seasonality (Jobbágy, Nosetto, Contreras, Jackson, & Calderon, 2009) although in most dry forests occupying flat sedimentary landscapes, deep drainage remains low or absent even under highly seasonal rainfalls and for a broad range of soil textures.

In natural circumstances in these systems, salts, including those deposited by rain and dust and those derived from in situ mineral weathering, remain in the soil profile because the exhaustive use of rainfall by plants minimises leaching. As a result, a characteristic salt concentration peak around the bottom of the active root zone usually develops (Phillips, 1994; Scanlon, 1991). The elimination of trees and shrubs modifies water fluxes, reducing the evapotranspirative losses and favouring deep liquid losses through deep drainage (Scanlon, 2005). As a result, salts become leached away from the unsaturated zone towards the water tables as groundwater recharge starts. In the long term and when forest clearing occurs at large scale, water table levels raise, mobilizing salts to the ground and causing what is broadly known as secondary salinization or "dryland salinity" (in opposition to salinity brought by irrigation), an irreversible process that triggers root osmotic stress, ion-toxicity and anoxia, often translating into vegetation die-off (Turnbull et al., 2012; Walker, Gilfedder, & Williams, 1999).

Climate and hydrogeology can also play an important role controlling the dryland salinity process. Intermediate aridity conditions seem to facilitate the process given that towards humid climatic conditions (precipitation > potential evapotranspiration) salt accumulation would not take place and under very dry conditions (precipitation < <potential evapotranspiration) water excesses would be rarely developed (Nosetto et al., 2013).

The lithology and geomorphology also affects salinization risks through their effects on the depth of phreatic groundwater and accompanying salts and on the hydraulic conductivity of sediments. Flat sedimentary landscapes are more likely to promote shallow water tables, and thus salinization risks, than hilly landscapes with massive bedrock. Highly conductive sediments can supply groundwater for evapotranspiration process at high velocity, increasing salinization rates (Jobbágy & Jackson, 2004). Faults also have considerable impact on groundwater movement, if these structures are occupied by low permeability materials, they act as barriers to water flow, which contribute to increase shallow water table levels (Please, Watson, & Evans, 2002).

One of the most well-known and documented cases of dryland salinity has been taking place in Australian native dry forests and scrublands during the last century (George, Nulsen, Ferdowsian, & Raper, 1999; Turnbull et al., 2012). By the end of 1980, almost 40% of Australian dry forests were converted to crops and pastures (Bradshaw, 2012). In the southwest part of the country, for example, this land use change caused the salinization of more than 2 million hectares of productive lands (Lambers, 2003). The Australian dryland salinity is a worldwide known example of the strong influence of vegetation in regulating water dynamics and how land cover changes induced by humans can drastically alter it. Water tables rises as consequence of perennial vegetation clearing have also been reported for other areas of the world such us South-western Niger (Favreau et al., 2009) and the North American plains (Scanlon, 2005). A comprehensive review of the increments in groundwater recharge and its causes at global scale has been reported by Scanlon (2006).

1.1 | Land use changes in the dry forests of South America

The American Chaco is one of the last large-remnant dry forests of the world and the second forest mass of South America after Amazonia. Historically, land uses in the Chaco were limited to extensive livestock production and semi-industrial or manual logging for charcoal. However, in a similar way to Australian dry forests, but much more recently, the Chaco has experienced a fast and large scale transformation driven by the establishment of pastures and summer crop monocultures (Houspanossian, Giménez, Baldi, & Nosetto, 2016). Currently, deforestation rates in the Chaco lead South American rankings and are among the highest of the world (Baldi, Guerschman, & Paruelo, 2006; Boletta, Ravelo, Planchuelo, & Grilli, 2006; Viglizzo et al., 2010). Growing demand for grain from external markets as well as the implementation of new agricultural technologies are some of the factors that explain the fast expansion of agriculture in this area (Jobbágy, 2010). Houspanossian et al. (2016) estimated that in 1 decade, the cleared area for pastures and agriculture of the Chaco doubled from 6.9 to 12.5 million hectares. Gasparri and Grau (2009) also estimated that in the last decade, 1.4 million hectares were cleared for agriculture in the dry portion of the Chaco, one of the most active deforestation focus of the region.

A few studies have evaluated the ecohydrological impact of dry forest clearance and cultivation in South America (Amdan, Aragón, Jobbágy, Volante, & Paruelo, 2013; Jayawickreme, Santoni, Kim, Jobbágy, & Jackson, 2011; Santoni, Jobbágy, & Contreras, 2010), being confined to specific study areas. For example, a study in dry forests of central Argentina indicated that the elimination of perennial woody vegetation with deep root system and longer active phenology shortened the growing season by 3 months and reduced evapotranspiration by 30% (Marchesini, Fernández, Reynolds, Sobrino, & Di Bella, 2015). Given the current conditions and expected transformation of Chaco forests and considering the similarities with the Australian dry forests transformation, it is critical to assess the risks of dryland salinization of the Chaco. With this goal, we review information about key ecohydrological variables (e.g., recharge rates, changes in soil water, and salt stocks) from agriculture or native Chaco dry forests stands located in a precipitation range of 450–1,100 mm $year^{-1}$. Data from water table monitoring and geoelectric profiling studies together with other field observations by researchers and farmers and reports were used as supplementary evidence. The hypothesis that guided our study was that Chaco dry forests make a complete use of soil water allowing the accumulation of salts in the vadose zone and disturbances such as forest clearing and cultivation prevent the exhaustive use of water, increasing deep drainage and salt leaching. We expect higher water content and deep drainage rates and reduced salt pools in disturbed areas. The magnitude and rates of changes in vadose salt and water stocks would provide an initial assessment of the risks of dryland salinity in these territories.

2 | DESCRIPTION OF THE CHACO AND METHODOLOGY

With an extension of more than 1 million km^2, this sedimentary plain located between the Argentinean Pampas and the Amazonian lowlands, covers parts of Argentina, Paraguay, and Bolivia. Despite its large precipitation range, vegetation shows similar physiognomy dominated by xerophytic trees and shrubs adapted to drought and high temperatures. The region is characterized by the presence of alluvial fans from five rivers, which have transformed the landscape during the late quaternary (Sayago, 1995). Soils are the result of aeolian and alluvial episodes. Loess is predominant in the region although paleochannels with coarse sediments and fields dunes are common in the West Chaco.

The main groundwater recharge sources in the Chaco plains are allochtonous rainfalls, originating in the Andean piedmont and intermountain valleys. Some local recharge has been identified in sandy areas occupied by paleochannels (Iriondo, 1993). The ultimate discharge areas are on the east part of Chaco, on the Paraguay and the Parana River (Pasig, 2005).

Three main environmental attributes shape the ecohydrology of the Chaco region: first, the exceptionally flat topography with slopes <0.1% that limit runoff and horizontal groundwater fluxes (Jobbágy et al., 2008). Second, the negative climatic water balance as consequence of high temperatures and limited rainfall inputs. Third, despite the flat topography, there is a regional transport of water and sediments from the eastern Andean slopes towards the Paraguay and Parana rivers subject to frequent channel shifts that has created a dense network of paleochannels. These attributes favour the development of shallow water table levels if local recharge is favoured as suggested by global groundwater depth models (Fan, Li, & Miguez-Macho, 2013).

We compiled and analysed existing studies on deforestation and deep soil moisture or salt loads and drainage estimates in the Chaco region performed over the last 10 years. These studies encompassed six Argentinean provinces, between 34°W 64°S and 22°W 63°S (Figure 1). Rains in the region occur mostly between October and March, and mean annual temperatures vary from 24 to 30 °C. High thermal amplitudes are observed in the west while in the east, more humid conditions define a milder climate (Iriondo, 1993). The whole area is characterized by species such as Quebracho colorado (*Schinopsis lorentzzi*), Quebracho blanco (*Aspidosperma quebracho-blanco*), and *Prosopis* species (*Prosopis caldenia*, *P. flexuosa*, and *P. alba*) with a diverse shrub and small-tree stratum that includes *Larrea divaricatta*, *Lycium chilense*, *Condalia microphylla*, *Capparis atamisquea*, *Ziziphus mistol*, and *Geoffroea decorticans*, among others. *Sporobolus*, *Aristida*, *Stipa*, *Poa*, and *Pappophorum* are the main grass genera that dominate the understory. Irrigation is not common in the area. Some of the remnant dry forests are grazed by domestic animals with a rather low density.

We collect information from paired stands with contiguous dry forest and agricultural plots where recharge rates were estimated on the basis of chloride mass balances, either by the residual moisture flux (CMB, Phillips, 1994) and/or by the tracer front displacement method (CFD, Walker, Jolly, & Cook, 1991). For most of the stands, recharge rates were obtained from studies already published (Amdan et al., 2013; Gimenez, Mercau, Houspanossian, & Jobbágy, 2014; Jayawickreme et al., 2011; Marchesini et al., 2013; Santoni et al., 2010), but some recharge rates were also calculated from raw data of soil water and chloride content. Deposition rates for the area were obtained following Santoni et al. (2010) and Amdan et al. (2013). In all cases, soil water content and chloride storage were obtained from manual boreholes done down to 6–10 m of depth, in deforested and contiguous remnant dry forests.

Given that agriculture stands presented different age since they were deforested, we classified as "<20" those new stands that have been cleared and cultivated for 20 years or less and ">20" those old stands cleared 21 years ago or more, including situations subject to 90 years of cultivation. None of the studied stands was ever irrigated. Most of the measurements were performed between 2009 and 2015. Apart from soil cores, we also used information from national land use reports, geoelectric surveys, ground water monitoring, and local farmer's reports. Detail information for each site is showed in supplementary information (Table S1).

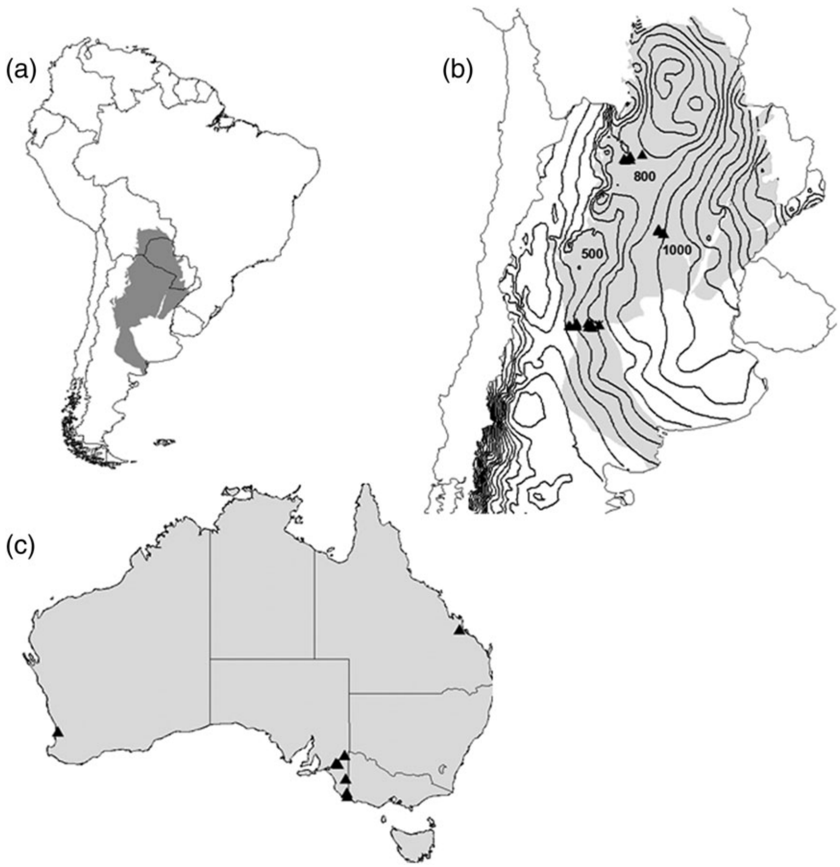

FIGURE 1 (a) South American map showing dry forests in grey. (b) Detail of Chaco and Espinal dry forests including the sampling sites and 100-mm isohyets for the whole region. (c) Australian map showing the sampling sites from bibliography used in this study

2.1 | Differences and similarities between Chaco and Australian dry forests

In order to compare our results with the Australian dryland salinity case, we compiled 15 original studies focused on deep drainage or recharge and salt mobilization following forest clearance in Australia, mainly in the southern part of the country (Table S2). Recharge rates, water table variation, time elapsed since clearing, and original concentration of chlorides in the soil profile, among other variables, were contrasted between these two areas with the purpose of finding patterns and similarities that let us envisage if the hydrological changes observed in Chaco might follow the same patterns of dryland salinity observed in Australia.

3 | RESULTS AND DISCUSSION

3.1 | Changes in chloride stocks in Chaco dry forests after clearance

Significantly higher chloride stocks were found under Chaco's forests soils when compared with their contiguous cultivated stands. Forest stands located in new agriculture areas (<20 years) showed the highest chloride content in their profiles with an average of 41.5 kg m^2 (Figure 2). On the other hand, old agriculture stands (>20) showed the lower chloride stock, 10 times less than their dry forest pairs. Interestingly, dry forest stands located next to old agricultural sites also showed less chloride when compared to the new agriculture stands (Figure 2, Table 1). This pattern could be the consequence of two effects: on one hand and since agriculture in Argentina has advanced from central-east to northwest regions (Viglizzo et al., 2010), new agriculture areas are located in soils where chloride stocks are naturally higher but on the other hand, it is probably that those remnant forest stands that subsisted next to old agricultural areas have been exposed to the effects of fragmentation for longer time, showing more signs of degradation due to the marginal influence of their deforested stands. A similar pattern has been observed in remnant or protected Australian eucalyptus forests fragmented long time ago, which show signs of degradation by manual logging and urban development such as roads and trails (Bradshaw, 2012).

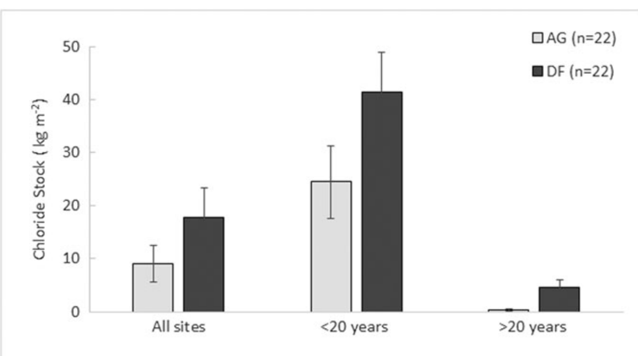

FIGURE 2 Chloride stocks for dry forests and agriculture stands for all sites (n = 22) and separated by years since clearing: less than 20 years (<20, n = 8) more than 20 years (>20, n = 10) and their contiguous dry forest pairs. Chloride's sample depth for all stands up to 6 m. Some stands were not included due to missing data

Soil chloride stocks in agriculture stands decrease sharply with years elapsed since clearing. An "inflexion point" around 20 years is evidenced (Figure 3), where chlorides have been almost completely leached down from the profile. For example, cultivated areas cleared 90 years ago showed 0.36 kg m^{-2} of soil chloride compared to 23 kg m^{-2} stored in those with only 6 years of cultivation. These results agree with data from geoelectrical transects and soil cores obtained in agriculture areas of Central Argentina where high deep drainage leached 95% of the salts in the profile. These transects also showed high electrical conductivity (indicating a large presence of salts) in dry forests and younger agriculture sites (<6 years) but the absence of this signal in agriculture areas cleared more than 20 years ago (Jayawickreme et al., 2011).

3.2 | Water stocks and recharge rates

All cleared stands, independently of their age, showed higher soil water stocks than their dry forest counterparts (Figures 4 and 5, $p < .05$). Agricultural stands showed on average 37% higher soil water content than dry forests (Figure 4). However, this difference between dry forests and cleared stands was lower in new cleared areas (+20%) compared to old cleared areas (+50%; $p < .05$). Although with large differences between methods, drainage estimations by both methods were at least two order of magnitude higher in cleared stands compared to remnant dry forests (Figure 5). Recharge rates with the CMB method approached 11 mm $year^{-1}$ and 0.07 mm $year^{-1}$ for cleared and dry forests stands, respectively (Table 1). For the CFD, recharge rates in deforested plots were even higher than those estimated by CMB method, with new cleared areas duplicating the recharge rates of old deforested ones (Figure 5 and Table 1). The differences between estimations can be explained by the assumptions and ways in which both methods calculate recharge. The CMB method assumes a steady-state flux (Scanlon, 1991) which a priori we know it cannot be validated for the new agricultural stands. Additionally, the CFD method considers the displacement of the chloride peak (compared to the natural-pristine situation) and divides it by the years elapsed since clearing, resulting in a larger number for the new agriculture (<20 years) than for the old cleared stands (>20 years). In other words, recharge estimations could be assumed valid for the comparison between cleared versus dry forests stands, but they should be taken with cautions if the comparison is done for those stands recently cleared.

3.3 | The Australian sites

Most of the Australian sites reviewed in this study were located on the southern and western part of the country (Figure 1), where eucalyptus "Mallee" forests were replaced by pastures and crops more than 60 years ago (Williams, Walker, & Hatton, 2002). The soils in this region varied from limestone-karstic to sand dunes, and most of the sites presented high chloride deposition, averaging 7.8 g m^{-2} $year^{-1}$, and large but variable chloride stocks (up to 25,000 mg L^{-1}). Recharge rates in the native vegetation sites that we reviewed were very low

TABLE 1 Chloride content, water stock, recharge rates, and rain ranges for all site conditions

Condition	Chloride stock (kg m^{-2})	Water stock (mm)	Recharge CMB (mm year^{-1})	Recharge CFD (mm year^{-1})	Rain range (mm year^{-1})
AG-all sites	9.1	1,245.9	11.0	28.2	502–1,100
Dry forest-all sites	17.8	774.8	0.07		502–1,100
AG <20 years	24.5	1,436.6	1.8	43.8	525–980
Dry forest (by AG < 20 years)	41.5	1,171.1	0.02		525–980
AG > 20 years	0.4	1,079.0	18.0	21.4	502–1,100
Dry forest (by AG > 20 years)	4.7	573.4	1.05		502–1,100

Note. AG = agricultural sites (n = 22); CMB = residual moisture balance; CFD = tracer front displacement; DF = Dry forest (n = 22).

Note. Chloride's sample depth for all stands ranged from 6 to 10 m. There were no bias between stand condition and location and the depth of the sample. Stand details are given in supplementary information (Table S1). AG < 20 (n = 8): sites cleared less than 20 years ago, AG > 20 (n = 10): sites cleared 21 years ago or more.

and averaged 1 mm year^{-1}, and they increased up to 90 mm year^{-1} after clearance. Except for one site in the north-east of the country, for these sites, rains occur in winter, which may increase the chances of deep drainage due to low evapotranspiration rates during this period.

3.4 | Chaco versus Australia

Important differences are observed when key ecohydrological variables are compared between Australian (South and Western regions) and Argentina Chaco forests. For example, chloride deposition and deep drainage rates are higher in Australia than in Argentina. Chloride deposition in Australia exceeds by one order of magnitude the deposition in Chaco (Table 2), which is explained by the proximity to the ocean of most cultivated lands in Australia. Yet, chloride stocks are not that lower in Argentina, with modal values (20,000 mg L^{-1}) halving the Australian ones and maximum values surpassing them. Another difference between both regions is the time elapsed since clearing, which is almost half for Chaco than for Australian forests (30 versus 60 years, Tables 1, S1, and S2). Land use after deforestation is also different between regions, while in Australia, dry forests and bushlands were converted to pastures and winter crops, in Argentina the Chaco forests turned to pastures and summer crops. Rainfall ranges are similar between areas, although Australian sites present a lower rainfall limit.

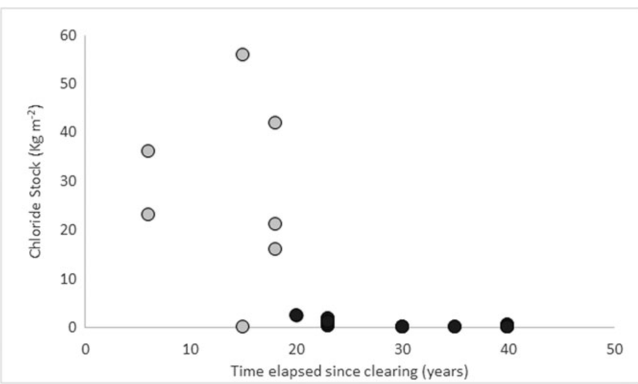

FIGURE 3 Chloride stock (kg m^{-2}) versus time elapsed since clearing for all agricultural stands (n = 18). Some stands were not included due to missing data. Dark circles represent stands cleared more than 20 years ago

3.5 | Are these results enough evidence to predict that Chaco dry forest will face a dryland salinity process in the near future?

In terms of geology and contrary to what is observed in Australian lands, the presence of younger sediments in Chaco would determine smaller salt stocks. Table 2 indicates that Chaco forests show similar chloride stocks in their profiles than the Australian ones. At this point, it is important to note that other ions apart from chloride could also be playing an important role in determining the total salt stock, especially in Chaco soils where the ocean source has less influence. Jayawickreme et al. (2011) found that in Chaco soils, sulfate is one of the most abundant anions, and although it is spatially more variable, its concentration can be up to 8 times higher than chloride. Thus, the amount of sulfate (not considered in this review) in Chaco soils could be an additional source that contributes to increase the risk of dryland salinity in Chaco.

On the other hand, Chaco soils are more fertile than Australian ones characterized by low pH, very low phosphorous, and metal toxicity (Lambers, Brundrett, Raven, & Hopper, 2010). This important aspect of soil fertility may determine higher primary productivity and sustain more intensive cropping systems, which may translate into higher water use by crops and less residual water in the soil in Chaco. Remote sensing assessments of the productivity of croplands under similar climates in Australia and Argentina support this (Baldi &

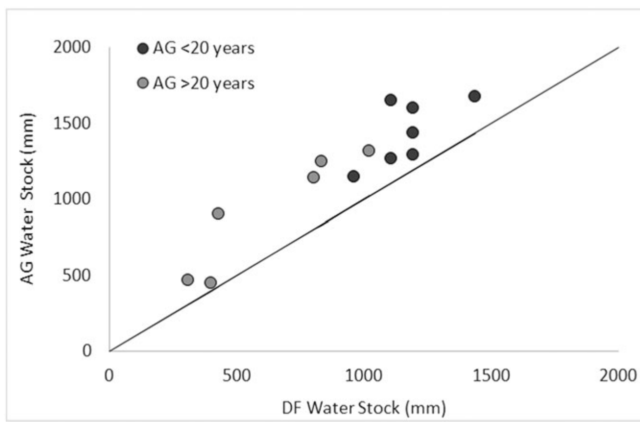

FIGURE 4 Water stocks for each dry forest (DF) or cleared pair (AG; n = 14). Some stands were not included due to missing data. Sample depth for soil water was 6 m for all stands. The black line represents the 1:1 situation. AG, agricultural sites; DF, Dry forest

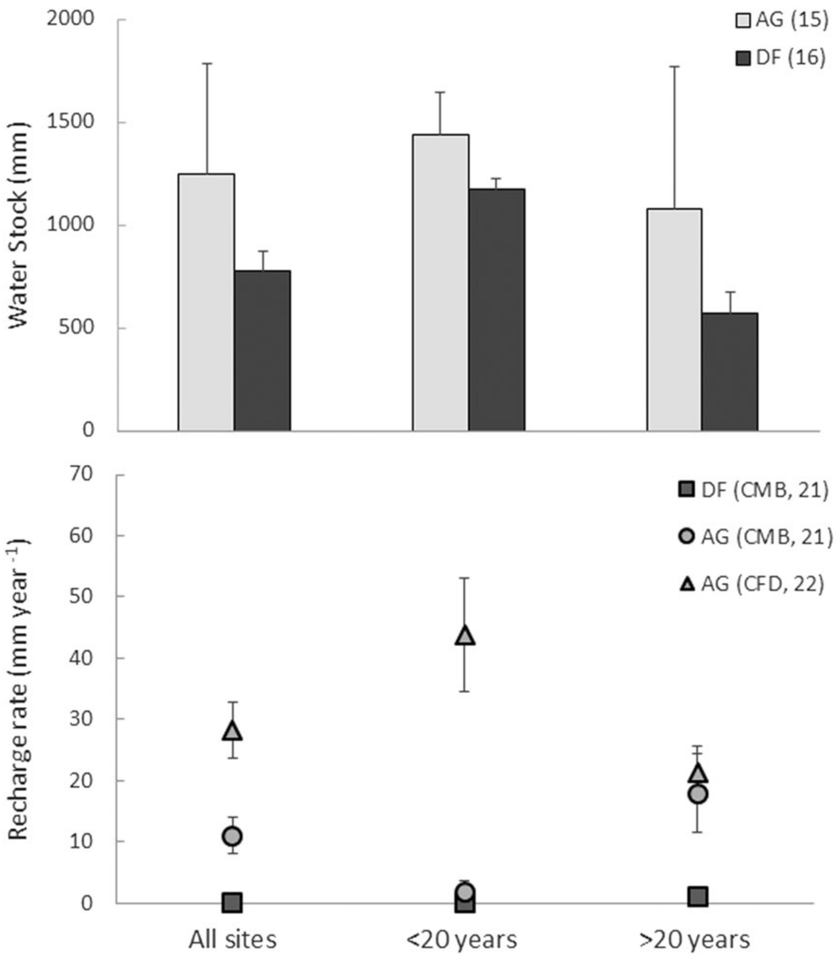

FIGURE 5 Water stock (above) and recharge rates (below) for agriculture (AG) and dry forests (DF) stands. Recharges rate were estimated by the CFD (triangles) and CMB (squares [AG] and circles [DF]) methods. AG, agricultural sites; CMB, residual moisture balance; CFD, tracer front displacement; DF, Dry forest

TABLE 2 Comparison of ecohydrological variables between South and Western Australia (n = 15) and Northwest and Central Argentina regions (n = 22)

	South and Western Australia (15)	Northwest and Central Argentina (22)
Chloride in native forest soils (mg L^{-1}, 0–12 m depth)	1,000–25,000 mg L^{-1} 10–50 kg m^2	1,080 to 19,400 0.15 to 59 kg m^2
Recharge in native forests (mm year^{-1})	0.04–12	0.01–5.2
Recharge in cleared stands (mm year^{-1})	3–250	0.01–34
Water table depth (m)	2–70	4–40
Chloride deposition (g m^{-2} year^{-1})	0.8–31.9	0.22–0.46
Time elapsed since clearing (years)	30–100	6–90
Rainfall range (mm)	250–885	447–1,100
Native vegetation	Mallee (*Eucaliptus* spp.) and native Brigalow (*Acacia* spp.)	Chaco dry forest (*Prosopis* spp.)
Current vegetation/land use	Pastures and winter crops	Pastures and summer crops

Note. Number of sites per group are given in brackets. References and details for each stand are given in supplementary information (Tables S1 and S2).

Jobbágy, 2012). Another important aspect mentioned before is the rain seasonality. Winter crops and pastures are more widespread in Australia than in dry Chaco where summer and oil or industrial crops are dominant (Baldi, Verón, & Jobbágy, 2013). Thus, in the Australian areas analysed in this study (South-western and South-eastern), precipitation occurs mainly in winter where low evapotranspiration contributes to increase drainage occurrence. All the aspects mentioned above would play an important role decreasing the risk, or slowing the process, of dryland salinity in the Chaco.

Agricultural practices in the Chaco, however, show important features that potentially could facilitate and even accelerate the process of dryland salinity with regard to Australia. No-till agriculture is widespread in Argentina and reduces water losses by evaporation also, the increasing trend of delaying crop sowing dates to store more water in the soil during fallow and to reduce crop water needs to minimize the risk of crop loss, suppose a higher risk of water underuse particularly in high-rainfall years (Gimenez et al., 2014). Additionally, the prevalence in the territory of a single summer crop (mainly soybean) in detriment of pastures or double cropping systems would also favour water recharge when water is underused even when evapotranspiration is high (Nosetto et al., 2012). The water table depth is also a key variable determining salinization risks; a global modelling analysis (Fan et al., 2013) suggests that water table tends to be naturally shallower in Chaco than in Australia, showing potentially higher chances of reaching the surface in less time. Finally, at regional scale, low regional hydraulic gradients, low saturated conductivity of the sediments, and the poorly developed river incisions could play an important role preventing the liquid evacuation of raising the groundwater system, favouring its evaporative loss at the surface and the associated salinization problem.

3.6 | Possible early signs of dryland salinity in Argentina

Although the problem of dryland salinity is barely known among farmers in the Chaco plains and only few of them envisage it as a threat, evidence of water table raise and salinization has been recorded by different actors across the region. The Argentinean salinity network has recently emphasized the necessity of studies that explore the magnitude and the extension of this phenomenon in Argentina. A shorter history of intensive agriculture in this region, less than 30 years compared to more than 60 years in Australia (Table 2), also defines a smaller time frame for detecting changes in water table levels, still the conversion operates at a much faster pace in the Chaco (Baldi & Jobbágy, 2012). Given the time lag existing between recharge increase and the first evident symptoms of salinization, it becomes difficult to perceive the problem of dryland salinity early enough to revert it. Before salt reaches the surface, and unless water tables are being monitored, farmers and landowners cannot detect the process occurring below the ground, what makes the problem more difficult to revert (Beresford, Phillips, & Bekle, 2001).

In north Argentina, the early signs of this problem are emerging, in the area of Pozo Betbeder, Santiago del Estero where researchers recorded seven times higher groundwater electric conductivity in areas deforested 25 years ago compared to nearby dry forest areas (Marasas & Sogo, 2011). In this province, the existence of large plains areas plus the massive dry forest clearing have contribute to increase the risk of large-scale soil salinization. Data from the Tucuman province, in the Northwest of the country, also indicated that one third of its territory has been exposed to some process of land degradation such as soil salinity and flooding (Puchulu, 2008). The geological, topographical but mainly climatic conditions in this area have contributed to the natural salinization; however, this phenomenon has exponentially increased due to deforestation and irrigation in areas with shallow water tables.

The results presented in this study highlight the importance of monitoring hydrological processes after land clearing, especially in areas prone to rapid salinization as those where water table are already close to the surface. At the same time, some similarities with the Australian salinity case alert about the chances to suffer this process in the near future since the replacement of dry forests by agriculture keeps going. In this context, it is essential the application of effective plans of land management in Argentinean dry forest territories in order to avoid further hydrological problems.

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SUPPORTING INFORMATION

Additional Supporting Information may be found online in the supporting information tab for this article.

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