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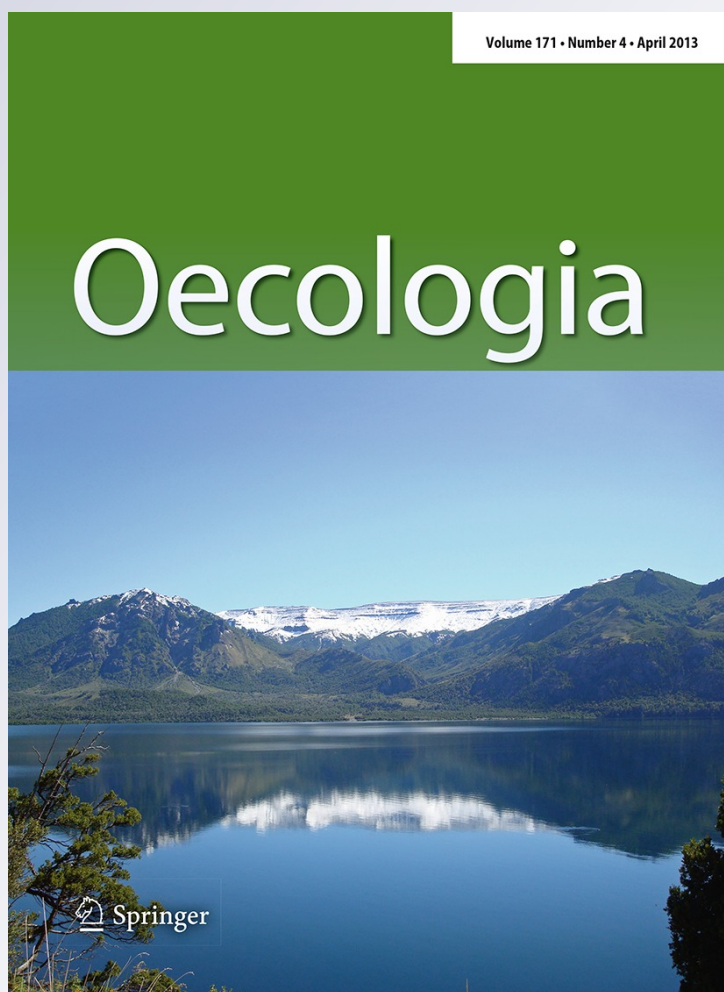
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Salt leaching leads to drier soils in disturbed semiarid woodlands of central Argentina

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Abstract Disturbances in semiarid environments have revealed a strong connection between water, salt and vegetation dynamics highlighting how the alteration of water fluxes can drive salt redistribution process and long-term environmental degradation. Here, we explore to what extent the reciprocal effect, that of salt redistribution on water fluxes, may play a role in dictating environmental changes following disturbance in dry woodlands. We assessed salt and water dynamics comparing soil-solution electrical conductivity, chloride concentration, soil water content (SWC) and soil matric and osmotic water potential (Ψ_m , Ψ_{os}) between disturbed and undisturbed areas. A large pool of salts and chlorides present in undisturbed areas was absent in disturbed plots, suggesting deep leaching. Unexpectedly, this was associated with slight but consistently lower SWC in disturbed versus undisturbed situations during two growing seasons. The apparent paradox of increased leaching but diminishing SWC after disturbance can be explained by the effect of native salt lowering Ψ_{os} enough to prevent full soil drying. Under disturbed conditions, the onset of deep drainage and salt

leaching would raise Ψ_{os} allowing a decline of Ψ_m and SWC. Soil water storage seems to be modulated by the presence (under natural conditions) and partial leaching (following selective shrub disturbance) of large salt pools. This counterintuitive effect of disturbances may be important in semiarid regions where deep soil salt accumulation is a common feature. Our results highlight the importance of water–salt–vegetation coupling for the understanding and management of these systems.

Keywords Chloride · Salt and water dynamics · Selective deforestation · Osmotic and matric potential · Woody vegetation

Introduction

In arid and semiarid regions, where plant cover is usually distributed in variable fractions of woody and grass life forms (Archer 1995; Aguiar and Sala 1999), the occurrence of disturbances that affect plant composition can produce changes in the hydrological and biogeochemical cycles (Chapin et al. 2000; Huxman et al. 2005; Jobbágy et al. 2012). Water is, by definition, limited in arid environments and plants tend to maximize its consumption (Eagleson 1982; Hatton et al. 1997; Contreras et al. 2011), matching evaporative losses (transpiration and direct soil evaporation) with rainfall inputs and making deep drainage virtually non-existent (Phillips 1994). Under these conditions, many natural dry areas have accumulated large pools of salts in deep soil/sediment layers resulting from centuries to millennia of exhaustive water uptake and salt exclusion by plant roots (Scanlon 1991; Walvoord et al. 2003; Santoni et al. 2010). The occurrence of unusually extreme rainfall events or long humid periods can cause increased

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deep drainage and partially leach these salts (Edmunds and Tyler 2002; Scanlon et al. 2006). Human disturbances, including interventions such as the removal of woody plants and the establishment of crops or pastures, can produce similar effects, when reductions in ecosystem productivity and water consumption lead to rising soil moisture levels and the onset of deep drainage and salt leaching (Walker et al. 1993; George et al. 1999).

These ecohydrological effects of vegetation and land-use management have been thoroughly investigated in certain regions of the world (Thorburn et al. 1991; Favreau et al. 2002; Scanlon et al. 2006). In Western and South-eastern Australia, for example, where large areas of dry forests were cleared for agricultural purposes after European settlement, tree clearing increased deep drainage and salt transport to the unsaturated zone. This process led to increased groundwater levels with the consequent salinization of productive areas (George et al. 1997; Eberbach 2003; Dorrough and Scroggie 2008). A similar pattern has been observed in Argentina after the conversion of native dry forests into croplands, which significantly increased soil water content (Santoni et al. 2010; Jayawickreme et al. 2011). These cases reveal an important coupling between water, salts and vegetation, in which initial perturbations affect water fluxes, which in turn influence salt dynamics and eventually alter vegetation, once groundwater and salts reach the surface. While this sequence of well-recognized effects acknowledges the influence of water dynamics on salt pools, the reciprocal effects, those of salt pools on water dynamics, have received less attention.

Many woodlands and dry forests of the world, mainly those located in flat sedimentary areas, host large pools of native salts in the first 10 m of their soil profiles (Scanlon 1991; Walvoord et al. 2003; Scanlon et al. 2006). Examples include woodlands in the sedimentary plains of Argentina (Jobbágy et al. 2008; Santoni et al. 2010), Southwestern US (Scanlon 2005), the Sahel (Favreau et al. 2002), the Murray River Basin in Australia (Thorburn et al. 1991; George et al. 1999), the Tengger desert in China (Wang et al. 2004), and several alluvial plains in India (Sukhija et al. 2003). Among other effects, soil salinity can affect water availability to plants through reduction of soil water potential (Ψ_s), the variable that best describes the direction of water movement along the plant–soil–atmosphere continuum (Lambers et al. 1998). Osmotic and matric potential (Ψ_{os} , Ψ_m) are two additive components of soil water potential: the matric component is related to the capillary forces in the soil matrix, while the osmotic component is related to the content of solutes in soil water. Under a lower osmotic potential, plant access to soil water is decreased due to a reduction in total soil water potential. On the other hand, as plant roots take up water from the soil, reducing its matric potential, they tend to increase the

concentration of any solutes that are not absorbed, simultaneously reducing the osmotic and the overall soil water potential. Individual plant responses to increases in soil salinity are well known (Zhu 2001; Barrett-Lennard 2003); the most common effects include reduced growth of young tissues and increased leaf senescence, leading to reductions of plant biomass and leaf area (Munns and Tester 2008). Although the effect of salinity on individual plants has been widely studied, the effects of soil salt reservoirs at the ecosystem level remain largely unknown.

In this study, we explore the role of deep soil salt pools shaping soil moisture dynamics in a semiarid woody ecosystem of west-central Argentina subject to large human disturbance. Our hypothesis was that in natural woodlands there is a complete use of soil water by vegetation (mainly via transpiration, with negligible deep drainage) allowing a sustained accumulation of salts in the soil profile; episodic disturbances such as the selective mechanical elimination of shrubs will prevent such integral use of water, increasing percolation and causing a partial leaching of salts from the soil profile (at least temporarily, until succession proceeds). Thus, reduced salt pools are expected in disturbed areas compared with those that maintain natural woodland (prediction 1). On the other hand, different disturbance effects on soil water storage can be anticipated depending on the relative importance of lower plant water consumption, by biomass reduction, versus the effect of the leaching of salts. In the short term, it is expected that lower plant water uptake could lead to higher soil water content in disturbed areas (prediction 2A), but in the longer term, recurrent salt leaching could alter the osmotic component of soil water potential enough to allow water uptake by plants to reduce average water content (prediction 2B). As far as salt stocks in the natural woodlands represent an important osmotic limitation on water consumption, mechanism 2B will become more important than 2A.

Taking advantage of a selective shrub removal operation that occurred during 2004–2006 in a large ranch in west-central Argentina, we studied soil salt and soil moisture dynamics observing three pairs of contiguous plots of undisturbed/natural and disturbed woodlands subject to roller-chopping clearing 1 year before our sampling. This setting was complemented with another group of areas cleared 2–3 years earlier.

Materials and methods

Study area and land-use changes

South America supports a high proportion of the dry forests and woodlands of the world, with the largest areas located in northern Brazil and north-central Argentina (Miles et al.

2006; Baldi and Jobbágy 2012). In the last 10 years, land-use change in these areas has been rapid, with more than 100,000 ha of natural woodlands cleared annually, mainly to grow soybean and other annual crops (Grau et al. 2005; Gasparri and Grau 2009), but also with the purpose of increasing forage and cattle production (Oesterheld 2005; Paruelo et al. 2005). In the driest part of this area, where agriculture is ecologically and economically less viable, a common practice is the mechanical removal of small-sized trees and shrubs using heavy rolls with blades (roller-chopping) with the purpose of promoting the establishment and growth of grasses (Aguilera et al. 2003). As a consequence of this disturbance, large areas of native forests are converted into savannas, i.e. become dominated by grasses and isolated large trees (Kunst et al. 2003).

The study site covered 20,000 ha of a ranch located in the Province of San Luis, Argentina (33.5°S, 66.5°W), in a transition zone between two adjacent vegetation types: the Monte creosotebush scrub to the west and the semiarid Chaco Dry Forest to the east. These areas included patches with different land uses, ranging from planted pastures (all woody vegetation cleared) to roller-chopped fields (only shrubs and small trees eliminated) within a matrix of undisturbed woodland. The tree canopy is dominated by two species, *Prosopis flexuosa* D.C. and *Aspidosperma quebracho-blanco* Schltdl. The understory consists of several shrub species as *Larrea divaricata* Cav., *Lycium chilense* Miers ex Bertero and *Condalia microphylla* Cav., and perennial grasses such as *Aristida mendocina* Phil. and *Trichloris crinita* (Lag.) Parodi. Plant growth occurs predominantly during the spring–summer season (November–March). Maximum and minimum green biomass stocks are found in early fall and late spring, respectively (Villagra et al. 2004). Average rainfall is 400 mm year⁻¹ (1989–2007; Salinas del Bebedero); 60 % of rainfall occurs in events exceeding 20 mm, and only 7 % in small events (<5 mm) (*P. Magliano*, personal communication). Soils are Regosols with a low content of organic matter (Peña Zubiate et al. 1998).

Experimental design

The study was based on the existence of areas that were subject to shrub removal (disturbed or “treatment” situation) and natural adjacent woodland plots (undisturbed or “control” situation). Field measurements were performed in three paddocks (>300 ha each) that had been subject to shrub removal in 2006 and their respective natural control plots situated <100 m from them. We also studied a temporal sequence of disturbed sites that had been cleared in 2004, 2005, and 2006, and their control plots (also <100 m from them). Some additional measurements, in particular those related to water dynamics, were also performed in

two pastures where all woody vegetation had been removed at least 10 years before; these were used as examples of areas with extreme disturbance and woody vegetation suppression. Distance between disturbed plots ranged from a few hundred meters, in the case of the temporal sequence, to 10 km in the case of plots disturbed in the same year.

The study area was covered by native forests until 2001, when rangeland interventions began; since then, all patches have a similar land use history with no intense logging activity or fires over the last decade. The slope in the area ranges between 1 and 2 %, and soil texture is sandy-loam with 50 % sand and 8 % clay. After roller-chopping disturbance, woody vegetation (trees and shrubs) cover was reduced from 77 to 22 % (Marchesini 2011). No soil compaction was detected after the disturbance: soil texture and bulk density were compared between treatment and control plots, and no differences were found ($n = 24$, $P = 0.94$).

Gravimetric and volumetric water content measurements

Three different approaches were used to compare soil water dynamics and soil water storage between disturbed and undisturbed plots: (1) gravimetric soil water content (GSWC) samples were taken in boreholes at regular depth intervals from 0.1 to 3 m; (2) volumetric soil water content (VSWC) was periodically recorded from 0.1 to 3 m using a TDR probe in fixed positions (access pipes); and (3) VSWC was also continuously recorded using TDR sensors from 0.2 to 1 m. While gravimetric measurements allowed us to directly monitor soil water dynamics at many sites, TDR probes provided more frequent, yet indirect, measurements at fixed positions, providing a better representation of temporal changes. Continuous records from TDR sensors offered a more detailed representation of soil water dynamics following individual precipitation events and their subsequent drying cycles. These three complementary methods provided more robust estimates of the effects of disturbance on soil water storage.

GSWC measurements were performed using a hand auger of 0.1 m internal diameter. Samples were collected at depth intervals of 0.5 m except for the first meter where depth ranges were: 0–0.1, 0.1–0.2, 0.2–0.5, 0.5–0.75, and 0.75–1 m. Samples were collected on ten occasions, from December 2006 to April 2010. Three boreholes per plot and nine samples per borehole were taken in all study plots, and 100-g subsamples were oven-dried for 48 h at 105 °C. Boreholes for GSWC measurements were located no more than 2 m from the access tubes for the TDR probe, thus allowing valid comparisons. GSWC data were multiplied by site-specific bulk density to convert them to VSWC and provide TDR calibration.

To estimate VSWC, we used a Trime-T3 TDR probe (IMKO Micromodultechnik). Three fixed tubes were installed in each disturbed/undisturbed plot ($n = 3$ per site type) and also on two pastures/natural woodland plots ($n = 3$ per site) 2 months before performing water measurements. The access tubes were buried and installed using a manual hand auger. VSWC measurements were performed systematically, once a month during the cold dry season (April–October) and every 15 days during the warm rainy season. The tubes were installed in August 2008 and soil water content was recorded until December 2009. Each access tube was located within each plot at equidistant position and covered with a small metal lid to prevent water inputs and other sources of contamination. Although soil salinity can interfere on TDR-based estimates of soil moisture (Muñoz-Carpena et al. 2005), a comparison with independent gravimetric measurements suggested that this interference was relatively low: linear regressions between volumetric SWC estimates resulting from both approaches across all plots and dates yielded a significant positive relationship ($P < 0.01$) with a slope that did not differ from 1.

VSWC measurements were also performed using TDR sensors (ECH₂O; Decagon) connected to dataloggers (21X; Campbell Scientific). These measurements were performed in three plots that were respectively; undisturbed, disturbed in 2004, and disturbed in 2006. Sensors were located at 0.2, 0.6; and 1 m depth in each plot and registered hourly values from March 2007 to January 2009.

Chloride budget and electrical conductivity measurements

Soil salt stocks were characterized through measurements of electrical conductivity (EC) and chloride concentration. Chloride in the soil profile represents a useful tracer of water movement. It is a highly mobile ion incorporated by atmospheric deposition, has a negligible contribution from the soil parent material, and its vegetation absorption is minimal in relation to the soil stock (Moore et al. 2010; Jayawickreme et al. 2011). Hence, chloride concentrations provide a valuable record of water depletion and the gradual decline of percolation rates with soil depth (Scanlon 1991; Scanlon et al. 2002). Soil samples measurements were carried out in three disturbed plots and in their adjacent undisturbed pair located 100 m from it, and in the sequence of disturbed sites during 2004–2006 (plots of 1, 2, and 3 years old) and in one undisturbed plot also located 100 m from the disturbed areas. Three boreholes per site were drilled using the same hand auger, the same depths (and at times the same samples) as for GSWC and electrical conductivity measurements. Chloride concentrations and EC were

measured in saturated extracts 2:1 water:soil in weight using an ORION 94-17 chloride ion selective electrode and an ORION 115 electrical conductivity meter (Thermo Electron). Chloride measurements were subsequently corrected to express them at the actual SWC of the original soil sample. The elapsed time between soil sample collection and the disturbance event was 1 year in the case of areas cleared in 2006, and 2 and 3 years in the case of those cleared in 2005 and 2004.

Water potential measurements

Effects of shrub removal on both water and salt dynamics were evaluated through their impact on water potential. We used standard equations requiring volumetric water content and soil texture to estimate matric potential (Eq. 1), and EC to estimate osmotic potential (Eq. 2) (Campbell 1985). We also performed field measurements of soil water potential and predawn leaf water potential.

Matric potential was estimated from Campbell (1985):

$$\Psi_m = \Psi_e \left(\left(\frac{\theta}{\theta_s} \right)^{-b} \right) \quad (1)$$

where Ψ_m = matric potential (MPa), Ψ_e = entry water potential, estimated from the particle size distribution and its geometric mean diameter, both of which vary with soil texture (Campbell 1985), θ/θ_s = volumetric soil water content in partially saturated condition/volumetric soil content in saturated conditions, b = parameter estimated from Ψ_e and size particle geometric mean and standard deviation.

Osmotic potential was calculated as:

$$\Psi_{os} = (-36 \cdot EC) \quad (2)$$

where Ψ_{os} = osmotic potential (MPa) and EC = electrical conductivity (dS m⁻¹) of the soil solution. Assuming negligible gravitational effects, soil water potential (Ψ_s) was estimated as the sum of the matric and the osmotic potential.

Predawn leaf water potential was measured in two of the most representative species in the area, *L. divaricata* (shrub) and *P. flexuosa* (tree). We used the pressure chamber technique to estimate predawn leaf water potential as an index of soil water potential (Turner 1988). Predawn leaf water potential was measured in a total of 20 *L. divaricata* and 10 *P. flexuosa* individuals located in one pair of 2006-disturbed and undisturbed woodland plot. All sampled plants were located in a radius of 10 m from the position at which soil moisture measurements were performed. A small branch in good growing conditions was sampled from each individual. Measurements were done during two consecutive days in April 2010.

Soil water potential was also measured using psychrometric chambers (C-30; Wescor). We drilled five boreholes per site down to 3 m, near to the *L. divaricata* and the *P. flexuosa* plants. After soil extraction, 2-g subsamples were immediately placed inside the chambers allowing for stabilization for 2 h, and then measured using a HR-33T microvoltmeter (Wescor).

Statistical analysis

As most measurements were conducted on the same experimental units, they showed spatial and time correlation (different depths in the same borehole and/or different dates on the same site). Thus, we used a modified repeated measures analysis (Littell et al. 1996), implemented with SAS v.6.12. We used a one-way ANOVA to compare leaf water potential between patch types in *L. divaricata* and *P. flexuosa*. In all cases, we used a significance level of $P < 0.05$.

Results

Soil moisture and water dynamics

Disturbed areas, subject to shrub removal, had lower water storage than their undisturbed woodland controls when the whole profile (0–3 m) was considered. VSWC, as recorded by the tube-access TDR probe throughout two growing seasons, showed a difference between treatments that was consistent for shallow soil layers (Fig. 1a, b) and most noticeable at 1–2 m depth (Fig. 1c). This pattern reversed between 2 and 2.75 m for some sampling dates (Fig. 1d), but between 0.5–2 m depth soil water storage in disturbed areas was, on average, 12 mm lower than in undisturbed areas, with differences ranging from +1.2 to –24 mm. At shallow depths, undisturbed and disturbed treatments presented similar peak values during the rainy period (warm growing season), yet disturbed areas presented lower minimum moisture levels compared to the undisturbed ones (Fig. 1a) ($P < 0.05$). No statistically significant interactions were found between date and treatment for any depth. Pasture areas, where all woody vegetation had been removed, showed a similar trend to those subject to shrub removal, presenting lower soil moisture than the contiguous undisturbed areas: –14 mm on average down to 2.75 m of depth.

Continuous TDR records of VSWC (logger-connected probes) confirmed the temporal dynamics captured by the tube-access TDR probe. At 0.2 m depth, disturbed areas (2–3 years after disturbance) showed, on average, 19 % less water than undisturbed areas; at 0.6 m, the differences between sites were smaller but consistent through time, with 3 % lower water in disturbed areas. GSWC, measured

less frequently but throughout a longer period (2006–2009), showed the same general pattern observed using the other two methods: all disturbed plots ($n = 3$ sites, 3 boreholes per site) presented lower moisture contents than the undisturbed areas mainly between 1 and 1.5 m ($P < 0.002$). Although TDR data has to be considered with special caution under the contrasting soil salt concentrations that we found under disturbed and undisturbed plots, the pattern shown in Fig. 1 is in agreement with independent data collected with other two methods: (1) Ψ_m , estimated as the difference between Ψ_s and Ψ_{os} (see Fig. 4 below), and (2) gravimetric measurements performed on additional plots and additional dates (Table 1).

Chloride stock and electrical conductivity

Chloride stocks presented a typical bell shape (Scanlon 1991), showing a lower concentration near the surface (0.1–1 m), a higher concentration at intermediate depth (2–4 m) and again lower values below 5–6 m (Fig. 2). Chloride storage in the soil profile (0.5–6 m) decreased according to the age of the disturbance. The soil chloride pool was 6.8 (ranges from 9 to 4 kg Cl m⁻²), 5.5 (ranges from 8 to 3.5 Cl m⁻²) and 3.7 kg Cl m⁻² (ranges from 5.7 to 3.4) for undisturbed and disturbed areas of 1 and 3 years old, respectively (Fig. 2). On average, those areas cleared 2–3 years ago retained between 40 and 70 % of the chloride mass that they presumably had before being cleared.

Consistent with the changes in the chloride profiles, the EC in disturbed soil areas was on average 20–50 % lower, depending on depth, than the EC of undisturbed areas (Fig. 3a). Taken together, these results (Figs. 2, 3a) clearly show partial salt leaching from the top meters of the soil. Chloride accumulation mirrored soil moisture: those sites with lower EC generally had lower soil moisture, giving Fig. 3b a remarkable resemblance to Fig. 3a.

Leaf and soil water potential: the osmotic component

Soil water potential did not show statistical differences between disturbed and undisturbed areas ($P = 0.8$) for most of the analyzed depths (Fig. 4), with the exception of 0.5 m where it was lower in undisturbed areas. Predawn leaf water potential was different between species, but not statistically different between treatments, neither for *L. divaricata*. ($P = 0.13$) nor for *P. flexuosa* ($P = 0.14$) (Fig. 4, arrows). Soil water potential (Ψ_s), estimated as the sum of matric (Ψ_m) and osmotic potential (Ψ_{os}) (Eq. 1, 2) was also similar between disturbed and undisturbed areas, which agrees with leaf water potential data above. However, osmotic potential was on average twice as low in undisturbed plots than in disturbed ones. Since total soil water potential Ψ_s (both methods) did not change after

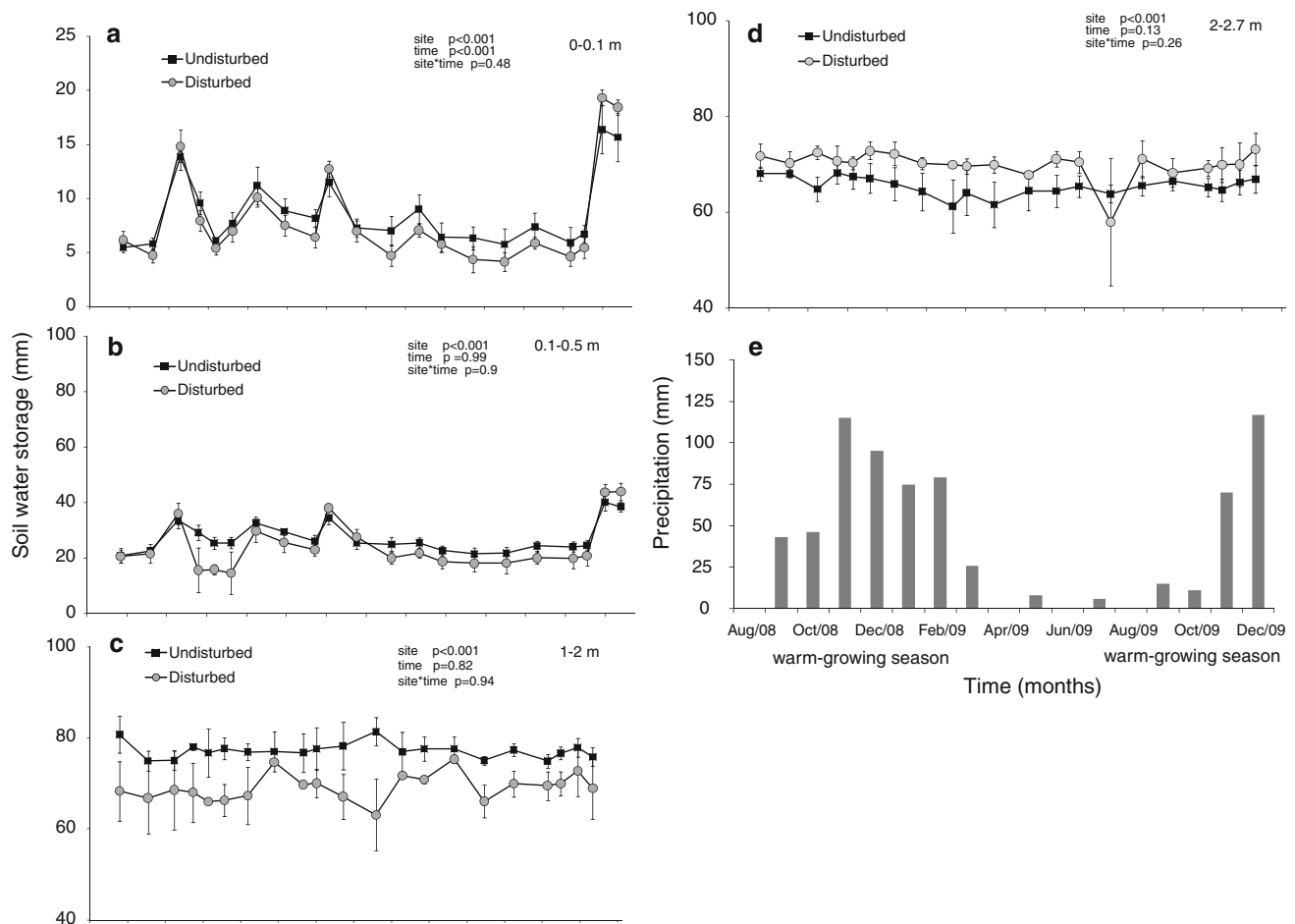


Fig. 1 a–d TDR probe data of volumetric soil water content measured in disturbed (filled circle) and undisturbed (filled square) plots: mean values and standard errors ($n = 3$ for disturbed, $n = 5$ for

undisturbed) at different depth for 21 consecutive measurements (2–3 years post disturbance). e Monthly precipitation in the area. Dates at the bottom of the graph indicate month/year

Table 1 Differences in gravimetric soil water content (GSWC, %) between plots disturbed in different years and their adjacent undisturbed plots

	Dec 2006	Jan 2007	Mar 2007	Jun 2007	Nov 2007	Feb 2008	Jun 2008	Jan 2009
Disturbed in 2004	−0.9**	−1.2	−1.9**	−1.3**	−1.4**	0.8**	−1.4**	−2.1**
Disturbed in 2005		0.1	0.4**	0.3	−1.1	0.6	−1.3	−1.9
Disturbed in 2006				0.1*	−0.1	1.7*	−0.6	−1.0

Mean values from 0 to 3 m depth from December 2006 to January 2009 ($n = 3$ boreholes per site); negative values mean lower GSWC for disturbed plots

* $P < 0.01$ and ** $P < 0.005$, respectively

shrub removal but its osmotic component was half of that in undisturbed plots, a compensating increase in the matric component in the forest areas is expected, consistent with results of Fig. 1a–c.

Discussion

Traditionally, studies relating ecosystem disturbances with their water dynamics have used soil water content as a key

state variable, paying little attention to the role that salt reservoirs can have in regulating them. In woodlands of central Argentina, we found that soil water storage appears to be modulated by the presence (under natural conditions) and partial leaching (following disturbance) of a large salt pool. Measurement of soil water content and calculation of Ψ_m consistently showed that disturbed areas presented less water accumulated in the profile compared to undisturbed woodlands. Furthermore, the pasture, considered as the most disturbed system in our study, showed the lowest

water content values. This unexpected result led us to reject the typical “fallow-effect” of disturbances and to consider more complex effects in which salt dynamics played an important role.

Comparable results were found in one study carried out in a juniper forest in western USA which showed that the

soil of deforested areas presented lower water storage (between 0 and 1 m) compared to control areas (Fernald and Garduno 2010). The authors explained their results by an increment in water uptake by grasses, the biomass of which increased considerably after deforestation. In our situation, even when grass biomass may have increased after woody plant removal (Marchesini 2011), the explanation does not seem to be so straightforward, for at least two reasons. First, the higher water content was not only restricted to surface layers but involved depths down to 2 m (Fig. 1a–c), and second, overall (woody + grass) aboveground primary productivity in disturbed areas was still half of that observed in undisturbed ones (Marchesini 2011). Furthermore, evapotranspiration (ET) data estimated using a remote sensing approach indicated that the ET of dry forest areas doubled the ET of the disturbed plots (Marchesini et al. 2009). Newman and colleagues (2010) showed a similar pattern in New Mexico, USA, with higher chloride content and evapotranspiration in vegetated patches but also lower water content in the non-vegetated patches profile. The authors explained their findings by accounting for the temporal scale at which processes occur, with chloride representing long-term accumulation processes and water content reflecting processes that have occurred more recently. Other studies, however, have shown the opposite trend, with more soil water in areas where woody vegetation has been removed (Mapping et al. 2003; Zou et al. 2008) or, as recently found for our study region, in deforested plots subject to annual cultivation for several decades (Santoni et al. 2010). These apparent contradictions may be explained by the initial salt storage of undisturbed systems. Water content increase seems to be more likely in regions with naturally low concentrations of

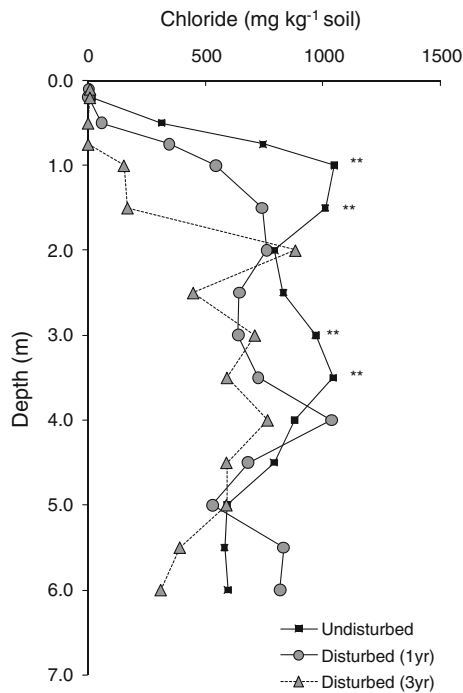
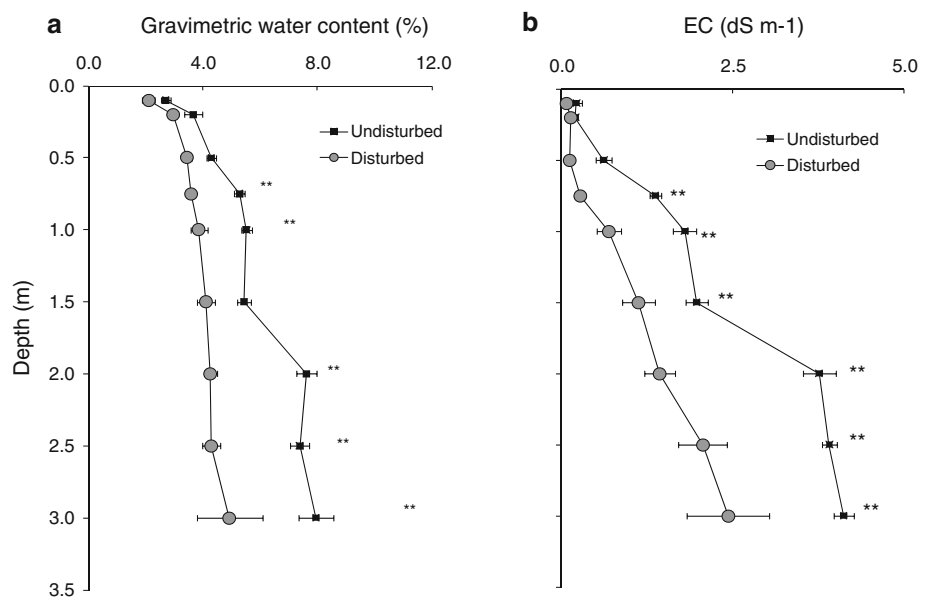


Fig. 2 Chloride concentration profiles in undisturbed ($n = 3$) and disturbed areas of 1 ($n = 3$) and 3 years post-disturbance ($n = 1$). Mean values for each situation for measurements taking during November 2007 and January 2009. Asterisks indicate level of statistical significance at each depth point ($P < 0.05$)

Fig. 3 Electrical conductivity and gravimetric water content for one representative pair of disturbed and undisturbed plots of 3 year ($n =$ five boreholes per site). Mean values and standard errors up to 3 m for March–April 2010 (end of growing season). Asterisks indicate level of statistical significance at each depth point ($P < 0.005$)



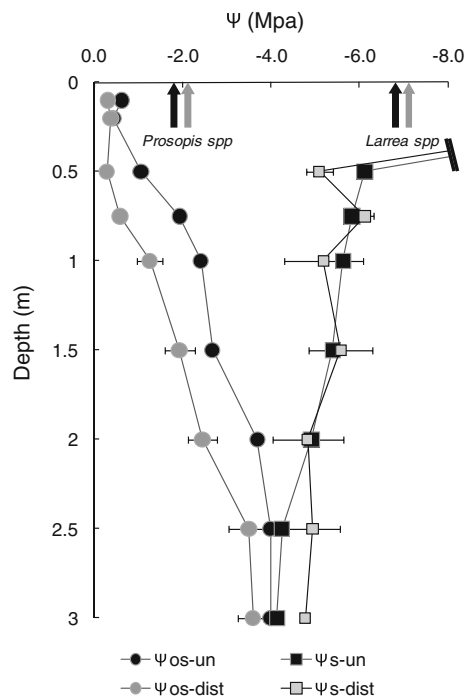


Fig. 4 Soil water potential (Ψ_s , filled square) and soil osmotic potential (Ψ_{os} , filled circle) profiles in an undisturbed and 3 years disturbed pair ($n = 5$ boreholes per site). Ψ_s measured with psychrometric chambers in April 2010; Ψ_{os} estimated through EC [Eq. 2]. The horizontal distance between Ψ_s and Ψ_{os} provides a graphic estimation of Ψ_m . Arrows represent average values leaf water potential for the indicated species in undisturbed (black arrows) and disturbed areas (gray arrows) respectively. Truncation of Ψ values at 0.5 m indicates soil water potential lower than -20 MPa for both disturbed and undisturbed plots

salts, whereas where these are initially higher this apparent “drying” may take place.

Consistent with our first prediction, chloride stocks in the chronosequence of disturbed areas revealed considerable salt leaching over time. Chloride values in our study agree in range and maximum values with those reported for semiarid woodlands in western USA (Scanlon 1991; Walvoord et al. 2003). In contrast, the estimated values in our study were an order of magnitude higher than those found for nearby dry forests with coarser-textured soil ($\sim 75\%$ sand) and twice as much rainfall, and therefore with higher deep percolation (Santoni et al. 2010). As mentioned in the “Introduction”, the large pool of chlorides in our woodland profiles, apparently in steady state under no disturbance, suggests a strict control of water loss by woody vegetation, favoring evapotranspiration and minimizing deep percolation.

Despite no significant differences being found in leaf or soil water potential between disturbed and undisturbed sites, soil water content was almost always lower in disturbed plots than in undisturbed ones. This apparent paradox can be explained by considering the osmotic

component of water potential, indicated by the EC. The fact that the lower the soil EC, the lower the soil moisture (Fig. 3), suggests that leaching of salts in the profile (which seems to have occurred 2 or 3 years before we started the measurements) can enhance the supply of available water by increasing the osmotic potential, and thus allowing plants to generate a lower matrix potential. The overall decreasing soil salinity (Fig. 4) and increasing moisture divergence with time after disturbance (Table 1) is consistent with this explanation. Further support is provided by the alignment of data on the Ψ_{os} versus Ψ_m space between the -4 and -7 MPa isolines (Fig. 5): the lower the matrix component, the higher the osmotic potential. Additionally, water availability control by the osmotic component can be the cause of the different falling rates of water content in the soil after a recharge event (Fig. 1a): disturbed sites generally reached lower minimum water content than undisturbed ones, indicated by the absence of salts and consequent relaxation of the osmotic limitation.

It is interesting to compare predawn leaf water potential between the two species studied. While values for *Larrea* coincide with those estimated using Campbell’s (1985) equation and soil measurements (~ -7 MPa), *P. flexuosa* values were much higher (~ -2 MPa) suggesting that this species could be using an additional source of water below 3 m depth. A stable isotope study ($\delta^{18}\text{O}$ and $\delta^2\text{H}$) and water table level fluctuation in an area located 200 km from our study site showed groundwater uptake for *Prosopis flexuosa*, especially at the end of the dry season (Jobbágy et al. 2011), and these findings are consistent with

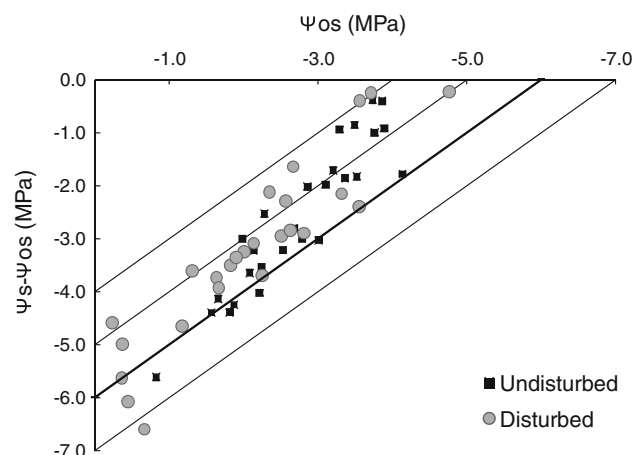


Fig. 5 Relationship between osmotic potential against matrix potential (calculated as the soil water potential minus the osmotic component) for both treatments. Linear regression ($r^2 = 0.88$) between osmotic potential (Ψ_{os}) and matrix potential calculated as soil water potential minus osmotic potential ($\Psi_s - \Psi_{os}$) for one undisturbed and its 3-year paired disturbed (five boreholes per site \times five depths; April 2010 data, depths between 0.5 and 3 m). Solid lines indicate constant Ψ_s (-4 , -5 , -6 and -7 MPa) and were drawn for reference

other studied that indicate a deep root profile for this genus in other areas of its distribution range (Jackson et al. 1999; McElrone et al. 2004).

Hydraulic redistribution, common in dry woody ecosystems, is an additional mechanism that could explain the higher water content observed in undisturbed forest (Caldwell et al. 1998; Hultine et al. 2004; Fernández et al. 2008; Licata et al. 2008). By transferring water from deeper layers up to the zone of our observations, the original populations of trees could have sustained wetter soils while the large-scale removal of trees would have interrupted this process. While our data suggest an important role of natural salt pools in the soil dictating water dynamics and its response to vegetation changes, new studies offering continuous soil and plant water potential, and stable isotope tracing of moisture sources along soil profiles, are clearly needed before our conceptual model is used for basing management decisions.

In a recent review, Moore and Heilman (2011) highlighted that ecohydrological studies usually assume that large changes in vegetation will inevitably drive changes in transpiration—which may or may not occur. They suggest the inclusion of other factors as determinants of water balance, such as soil texture or rooting depth and spatial distribution of vegetation. Based on our results, we propose to take customarily into account soil salt stocks, because they could eventually restrict water availability for the plant community, and deep soil stocks in arid and semiarid sedimentary plains may be more common than previously thought (Scanlon et al. 2006; Jobbágy et al. 2008). Changes in the salt stocks could lead to the development of a new ecosystem state with a lower hydrological osmotic limitation, an aspect that has so far apparently been neglected in the ecology of semiarid ecosystems.

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