Vadose zone transport in dry forests of central Argentina: Role of land use

C. S. Santoni,[1] E. G. Jobbágy,[1] and S. Contreras[1,2]

Received 17 October 2009; revised 29 June 2010; accepted 2 August 2010; published 29 October 2010.

[1] Most sedimentary plains occupied by semiarid woody ecosystems have low groundwater recharge rates and high vadose zone salt accumulation. Their cultivation has often led to drainage of water below root zone, displacement of solutes, and rising water tables, affecting, in most extreme cases, long-term viability of agriculture. To explore this possibility in semiarid plains of South America, we characterized vadose flow using chloride data in dry forests of central Argentina, in an area that has been subject to intense deforestation and agricultural expansion during the last century. We selected five paired stands under natural dry forests and dryland agriculture (sites deforested ≥30 years ago) and sampled sediments (n = 3 boreholes) down to 6 m depth. Profiles were consistently dry and salty in forest stands with chloride inventories (0–6 m) of 150 g/m^2 to 9 × 10^3 g/m^2. Under cultivation 78% to 99% of the chloride stock was leached, and total water storage was ≥30% higher than in the dry forest, with soil water content close to field capacity. Estimates of groundwater recharge rates based on residual moisture flux approach (cumulative chloride versus cumulative water curves) suggested maximum values of 0.33 to 128.4 mm/yr for dry forest and agriculture, respectively. At agricultural stands recharge was also estimated using chloride front displacement, yielding minimum values ≥5.3 mm/yr. While the long-term impact of cultivation on regional groundwater hydrology is still unclear in the region, our findings suggest that land salinization processes are possible and need careful monitoring in areas with high agricultural expansion.

Citation: Santoni, C. S., E. G. Jobbágy, and S. Contreras (2010), Vadose zone transport in dry forests of central Argentina: Role of land use, *Water Resour. Res.*, *46*, W10541, doi:10.1029/2009WR008784.

1. Introduction

[2] Human activities can leave a strong imprint on the hydrological cycle of dry regions, not only through direct consumption and diversion of water resources [*Postel et al.*, 1996; *Alley et al.*, 2002], but also through their widespread impact on vegetation structure and functioning [*Vörösmarty et al.*, 2000; *Alley et al.*, 2002; *Baron et al.*, 2002; *Eberbach*, 2003; *Scanlon et al.*, 2007a]. Vegetation regulates the water balance of terrestrial ecosystems, influencing the partitioning of precipitation inputs into vapor (interception, soil evaporation, and plant transpiration) and liquid outputs (runoff and deep drainage below the root zone) [*Wainwright et al.*, 2002; *Seyfried et al.*, 2005]. Vegetation changes can have a strong effect on groundwater resources through their impact on recharge rates and vertical solute transport, affecting both the quantity and quality of water resources [*Seyfried et al.*, 2005]. In this article we explore how replacement of native dry forests by dryland annual crops in semiarid plains of South America affects the hydrological cycle, focusing on deep water and salt storage and transport.

[3] In arid and semiarid regions where water is the main limiting resource for vegetation development, ecosystems tend to maximize net productivity and plant water use, reducing liquid water outputs [*Specht*, 1972; *Hatton et al.*, 1997; *Eagleson*, 2002]. Under these conditions, particularly where deep rooted woody plants are dominant and soils have no barriers for root development, evapotranspiration matches rainfall inputs, leading to negligible deep drainage (i.e., losses of water beneath the root zone). Groundwater recharge in landscapes subject to these conditions is often restricted to intense rainfall and runoff events that result in focused recharge in low areas such as playa lakes (recharge features) [*Wood and Sanford*, 1995; *Wilcox et al.*, 2003; *Seyfried et al.*, 2005; *Small*, 2005]. Extremely low deep drainage and recharge rates have been reported in several dry forest and grassland ecosystems of Africa, Australia, Asia and North America. These studies were based on deep soil/sediment observations [*Rambal*, 1995; *Eberbach*, 2003; *Scanlon et al.*, 2005; *Seyfried et al.*, 2005; *Scanlon et al.*, 2006], saturated zone analysis [*Leaney and Allison*, 1986; *Salama et al.*, 1993], modeling techniques, and isotopic tracers such as ^{3}H, ^{14}C and ^{36}Cl [*Favreau et al.*, 2002; *Harrington et al.*, 2002]. In these systems, salt and moisture profiles suggest no leaching below the root zone and continuous buildup of atmospheric salts for long periods [*Cook et al.*, 1989; *Allison et al.*, 1990; *Edmunds and Gaye*, 1994; *Phillips*, 1994; *Walvoord et al.*, 2003; *Scanlon et al.*, 2006].

[1]Grupo de Estudios Ambientales, IMASL, Universidad Nacional de San Luis/CONICET, San Luis, Argentina.
[2]Bureau of Economic Geology, Jackson School of Geosciences, University of Texas at Austin, Austin, Texas, USA.

0043-1397/10/2009WR008784

[4] The onset of recharge and associated changes in water levels and water quality may take place when dry forests are replaced by nonirrigated crops [*Scanlon et al.*, 2006, 2007a]. A widely observed pattern of deep drainage initiation and gradual, but steady, rise of water levels and extensive salinization of soils has been described for areas of expanding dryland agriculture in southeast and southwest of Australia [*Cook et al.*, 1989; *George et al.*, 1997], and in the Sahel plains (but here without a clear process of salinization) [*Leduc et al.*, 2001; *Leblanc et al.*, 2008]. Initial stages of this process could be taking place also in the Great Plains of North America [*Scanlon et al.*, 2005, 2006, 2007a, 2007b]. Shallower root systems, poorer plant diversity, shifts in phenology, and/or repeated disturbance cycles following crop establishment and harvest are some of the most likely causes of this general pattern [*Hatton and Nulsen*, 1999; *Schenk and Jackson*, 2002; *Jobbágy et al.*, 2008]. Slight evapotranspiration declines can be sufficient to trigger this hydrological switch and the "nondrainage" assumption of dry forests can be reversed with a few sporadic episodes of deep drainage that represent a relatively small percentage of long-term rainfall inputs [*Cook and Walker*, 1990; *Kennett-Smith et al.*, 1994]. In the cases reported above, increases of groundwater recharge occurred under conditions without increases in precipitation or shifts in the climatic water balance, pointing to land use as the central cause [*George et al.*, 1997; *Leduc et al.*, 2001].

[5] Soil texture can be an important modulating factor of the hydrological response of dry forests to cultivation thorough its influence on water retention, uptake, and vertical transport rates [*Hillel*, 1998; *Fernandez-Illescas et al.*, 2001; *Sperry et al.*, 2002]. To exhaustively consume all precipitation water inputs in soils of increasingly coarser texture, ecosystems require deeper and denser root systems. Under a similar climatic setting, increasing sand content results in deeper wetting fronts and a less conductive matrix under unsaturated conditions, where deeper and denser root systems would be required for an exhaustive use of precipitation inputs [*Jackson et al.*, 2000; *Schenk and Jackson*, 2002; *Sperry et al.*, 2002; *English et al.*, 2005]. The behavior of deep drainage and salt accumulation/leaching under dry forests and crops across soil texture gradients has been explored [*Kennett-Smith et al.*, 1994; *Petheram et al.*, 2000] and it is a key requirement to assess groundwater degradation risks following agriculture expansion.

[6] Multiple approaches have been used to assess vegetation impacts on deep drainage and groundwater recharge [*Walker et al.*, 2002; *Scanlon et al.*, 2002]. Characterization of vadose chloride/moisture profiles in pairs of natural and disturbed stands has yielded some of the most valuable insights on dry regions [*Walker*, 1998; *Scanlon et al.*, 2007b, 2008]. Chloride is an environmental tracer with an atmospheric origin supplied to ecosystems by atmospheric (wet and dry) deposition. Its widespread absence in soil minerals and its relatively low uptake by plants make it valuable for water flow tracing. Once in the soil, chloride concentrations tend to increase with depth in response to the coupled effects of water uptake and chloride exclusion by roots. According to this process chloride concentration peaks give an indication of the fraction of precipitation that becomes deep drainage [*Allison and Hughes*, 1978].

[7] Agricultural production is currently expanding at very rapid rates in South America, as a result of strong global commodity markets, technological changes, land use policies, and rising precipitation levels. Dry forests are one of the most important biomes hosting this expansion [*Programa de las Naciones Unidas para el Medio Ambiente*, 2003; *Grau et al.*, 2005] and central Argentina is a clear example of that, where semiarid woody ecosystems known as "Chaco" and "Espinal" are increasingly replaced by nonirrigated annual crops [*Solbrig*, 1999; *Zak et al.*, 2004; *Paruelo et al.*, 2005; *Grau et al.*, 2008]. According to *Grau et al.* [2008] in the most cultivated areas, the region has lost ~20% of its forest cover in the last 30 years. Agricultural expansion in central Argentina has accompanied (and likely responded to) a period of rising precipitation levels that reached as much as 30% during the last century, particularly between the seventies and late nineties [*Barros et al.*, 2008]. In addition, land use changes have occurred across gradients of soil texture that could likely shape their ultimate influence on deep drainage. Understanding how the rapid expansion of cultivation in these dry forests alters groundwater recharge rates across soil types will not only support the long-term protection of local water and soil resources, but open the possibility to explore whether the nondrainage assumption of dry forests takes place and persists under increasing rainfall rates. This study aims (1) to test the "nondrainage" assumption and evaluate the magnitude of vadose salt storage in the Espinal dry forests and (2) to quantify the impact of dry forest replacement by annual crops on deep drainage and salt transport to aquifers. Our work is based on the description of the vertical distribution of moisture and solutes throughout deep soil/sediment profiles across a network of paired systems occupied by dry forests and nonirrigated annual crops. With the aid of atmospheric deposition estimates for the region, we use the natural abundance of chloride in soils as a tracer to estimate deep drainage/groundwater recharge rates.

2. Materials and Methods

2.1. Regional Setting

[8] We focused our study in the dry forests of central Argentina in the province of San Luis (Figure 1). The region hosts the western edge of a vast late Pleistocene to early Holocene loessic system with some alluvial deposits. Sediments are composed by fine to very fine sand of yellow to yellowish brown color. Feldspars, volcanic glass, and quartz are the dominant minerals [*Iriondo*, 1997; *Zárate*, 2003]. *Markgraf* [1989] revised the palaeoclimatic characteristics of this latitudinal belt (30°–34°) suggesting cooler conditions and winter-concentrated precipitation between 18 and 12 ka, when substantial temperature increases and a shift to summer-concentrated precipitation took place. A reduction of effective moisture suggested by the dominance of xeric tree taxa occurred at 9 ka and the modern vegetation began to dominate after 3 ka.

[9] In our study area, these plains host several alluvial fans (age ≥25 ka) that originate in the hills of San Luis and which are partially interbedded and covered by more recent loessic deposits [*Zárate*, 2003; *Tripaldi and Forman*, 2007]. Climate is semiarid with the warmest areas toward the west. Temperature average ranges between 22.5°C and 25.5°C during the warmest month (January), and between 6.5°C and 9.5°C for the coldest month (July) [*Echeverría and Bertón*, 2006]. Mean annual precipitation ranged from W

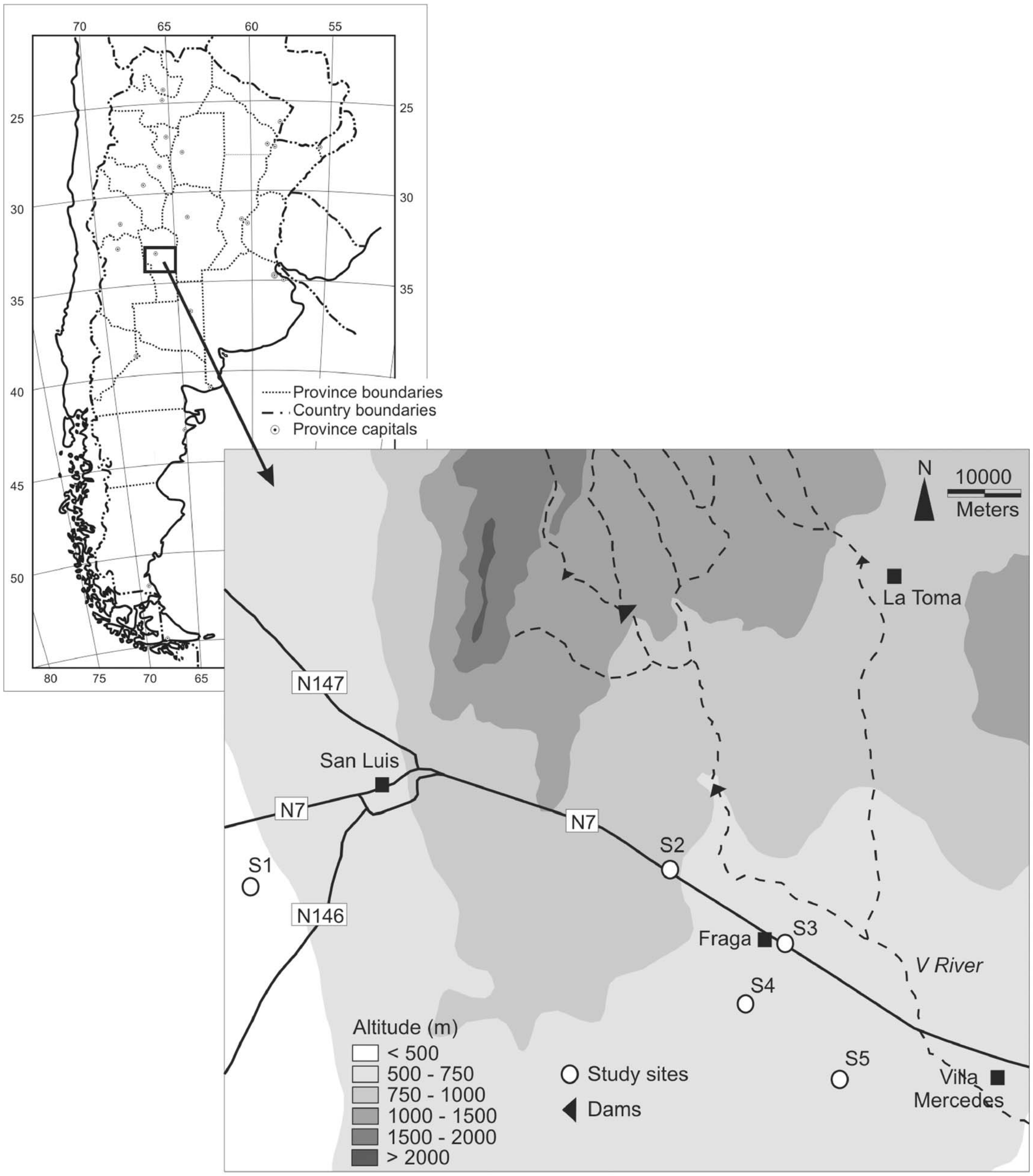

Figure 1. Study region. Towns (squares), main roads (solid lines) and rivers (dashed lines) are indicated.

to E between 350 and 600 mm/yr for the 1960–1999 period (data collected from the precipitation database managed by the Instituto Nacional de Tecnología Agropecuaria at San Luis). Most of the rainfall events (80%) take place during the summer season, occurring between November and March. High-volume events (≥60 mm) are unusual for the region, constituting only a 0.17% of rain events along period studied. During the last century rainfall increased by 30% in the region, with the highest increase observed between 1970 and 2000 [*Giulietti et al.*, 2003]. These more humid conditions in the region have probably favored dry forest clearing and dryland cultivation [*Viglizzo et al.*, 1995].

[10] The study area corresponds to the Espinal phytogeographical province [*Cabrera*, 1976] characterized by the dominance of short leguminous trees (≤8 m tall) of *Prosopis caldenia*, *Prosopis flexuosa*, and *Geoffroea decorticans*. Depending on grazing and fire history, among other possible causes, these species can form open savannas or closed woodlands [*Dussart et al.*, 1998]. A lower shrub stratum is composed by *Condalia microphylla*, *Capparis atamisquea* and *Larrea divaricata* among other species, with the last specie (*L. divaricata*) more dominant in disturbed areas [*Prose et al.*, 1987; *Baez and Collins*, 2008]. Grasses and forbs occupy the understory.

[11] Currently the Espinal forest in San Luis represents the westernmost and driest edge of nonirrigated agriculture in central Argentina. Since the beginning of the twentieth century, these forests have been gradually replaced by crops with rapid rates of transformation taking place during the last three decades, in part responding to sustained increases in precipitation [*Viglizzo et al.*, 1997]. Nowadays, approximately 30% of the original area of the Espinal dry forest is

Table 1. Study Sites[a]

Site	Precipitation (mm)	Cl Deposition (g/m^2yr)	Location	Profile (m)	Texture Sand (%)	Silt (%)	Clay (%)	Gravel (%)
S1	447	0.22	−33.439°, −66.538°	0–6	63	25	12	–
S2	538	0.26	−33.419°, −65.940°	0–6[b]	68	25	7	–
				6–7.5	17.5	7.5	–	75
S3	502	0.24	−33.505°, −65.772°	0–3	66	24	10	–
				3–6[b]	74	17	9	–
				6–7.5	22.5	7.5	–	70
S4	518	0.25	−33.584°, −65.839°	0–4[b]	68	32	–	–
				4–6	73	26	1	–
				0–4[b]	75	18	7	–
S5	542	0.26	−33.670°, −65.693°	4–6	79	14	7	–

[a]Mean annual precipitation corresponds to the 1962–1997 period. Chloride atmospheric deposition (g/m^2 yr) was calculated based on 2 years of measurement at a site located 35–80 km away from the study sites and adjusted with additional measurements from other sites [*Piñeiro et al.*, 2007]. Soil texture and gravel content in the sample are indicated for each distinctive sediment layer sampled at each site.

[b]Approximate contact between different sediment types.

cultivated [*Secretaría de Ambiente y Desarrollo Sustentable de la Nación*, 2007], with soybeans, corn, sunflower, wheat, and rye as the main crops, in the region. Areas planted with perennial forage crops such as alfalfa (*Medicago sativa*) and the tropical grass *Eragrostis curvula*, which were common until the early eighties, are gradually being replaced by annual crops [*Viglizzo et al.*, 1995]. Although groundwater-based irrigated agriculture is expanding in some areas, it still represents <1% of the cultivated area [*Instituto Nacional de Estadísticas y Censos*, 2002].

2.2. Study Sites

[12] Five paired sites (Figure 1) with neighboring plots occupied by dryland agriculture (A) and dry forest (DF) were selected across a NW-SE soil textural gradient typical of the region (i.e., from loamy sand to sandy loam soils). Paired plots at each site were at the same topographic position and with the same soil type. Topographic microdepressions, slopes, interfluvial areas, roads or borders with other land coverage were avoided. At all sites, water tables were deeper than 15 m (from 15 m to 45 m depth). Agricultural stands were cultivated for 30–90 years, depending on the site. An overgrazed dry forest (ODF) stand was included in the comparisons at one of the sites.

2.3. Measurements and Data Analysis

[13] Soil/sediment profiles were obtained at three positions in each stand. We established paired coring positions along transects running parallel to the contact line of both land cover types, and 50–75 m toward their interior. With this spacing we looked for a compromise between environmental similarity within each pair, favored by proximity; and spacing from the contact line to avoid edge effects. Bulk sediment samples were collected at intervals of 50 cm, with a 10 cm diameter hand auger, reaching 6 m depth below the land surface. Samples were homogeneized over the sampled interval and immediately sealed in polyethylene bags (double bag for each sample). A subsample was air dried and used for physical and chemical analysis; and the another one reserved for a gravimetric moisture measurement performed <48 h after sampling.

[14] A total of 390 soil samples were collected. Before physical and chemical analysis soil samples were hand sieved to break up soil aggregates and remove soil particles and plant fragments larger than 2 mm. Gravel and pebble-size particles were found only at depths ≥4 m (second sediment layer) at some sites (Table 1). Gravimetric water content was measured by weighing subsamples before and after oven drying at 105°C to constant weight. Soil texture was measured by using the Bouyoucos method [*Elliot et al.*, 1999]. Chloride content was measured in soil water extracts (1:2; soil-water ratio) with a solid-state ion-selective electrode [*Frankenberger et al.*, 1996] and a five point calibration scheme that included additional references and spikes. Standards calibrated with more precise ion chromatography equipment suggested a detection limit of 0.3 ppm and typical precision ranging from approximately 5% to 2% of the readings as concentration became higher. Nitrates and carbonates were measured at some depth intervals (0.5 – 1 – 3 – 5 m). A spectrophotometer was used to measure nitrate concentration [*Marban and Ratto*, 2005] while carbonates were estimated by volumetric differences in a Scheibler calcimeter [*Allison and Moodie*, 1965]. Because of the difficulty of collecting undisturbed soil samples at depth, bulk density was measured at different depths in an open wall profile with exposed sediments. Soil samples were taken carefully at one meter intervals down to five meters using a special auger to avoid disturbance of soil structure. The average value (±SD) of 1.2 g/cm^3 (±0.027) was obtained as the reference value required to estimate the volumetric water content and field capacity of the different soils under study [*Saxton and Rawls*, 2006].

[15] To better constrain our chloride budgets we estimated the age of the top sedimentary layer found at most sites through luminescence dating [*Forman*, 1989]. On an open wall close to the town of Fraga (Figure 1) we obtained three samples from the top homogenous sediment stratum (0–4 m), corresponding to that found down to 2 to 4 m in sites S2, S3, S4 and S5 (see Table 1). We used light tight 5 cm diameter and 15 cm long sections of black ABS pipe which were hammered into the desired sampling level (see *Forman* [1989] and *Tripaldi and Forman* [2007] for method details). The age of this sedimentary layer at its deepest edge was 9890 (±1125) years, coincident with the onset of drier conditions in the region suggested by *Markgraf* [1989] that may have favored aeolian deposition over an older, water-dissected sedimentary surface [*Prado et al.*, 1998].

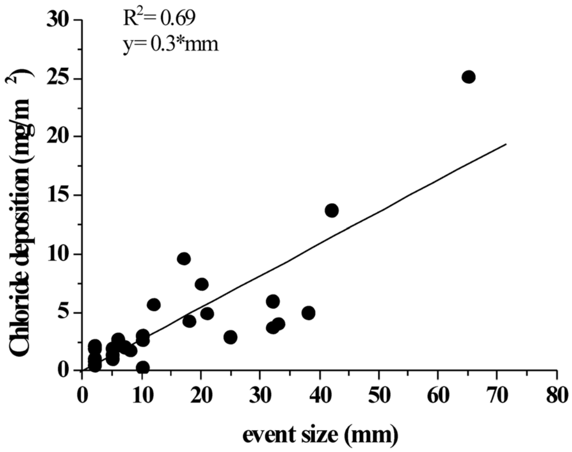

Figure 2. Chloride wet deposition (mg/m^2) of single events measured at La Punta (−33.18°, −69.32°) throughout a 2 year period (October 2005 to September 2007). The linear fitting equation (y = 0.3x, r^2 = 0.69) was used to calculate annual deposition rates on the basis of long-term precipitation records.

2.4. Chloride Budgets and Recharge Estimates

[16] In order to estimate chloride deposition rates at our sites we combined direct wet deposition measurements with adjusting and scaling procedures based on meteorological records and total (wet and dry) deposition measurements in another site. Mean annual precipitation for each site was calculated based on data for the 1960–1999 at Villa Mercedes (Instituto Nacional de Tecnología Agropecuaria at Villa Mercedes, −33.65° −65.43°). Wet chloride deposition was measured in San Luis during a 2 year period using a set of three funnel collectors placed when the precipitation event starts, at 1 m above the ground (to avoid any soil contribution). Samples were collected after each individual rain event and the funnels replaced. A total of 25 rainfall events ranging between 1 and 65 mm and involving 26% of all the precipitation inputs during a 2 year period were captured. Chloride concentration in rainfall water was analyzed using an ion chromatograph. Chloride concentration in rainfall ranged 0.03 mg/L to 1.14 mg/L, with an average (±SD) of 0.392 mg/L ± 0.292. Although we did not measured dry deposition, a year of wet plus dry and wet deposition measurements at the town of Pergamino in the Pampas (−33.89° −60.60°) indicated that a ratio of total to wet deposition of 1.6 [*Piñeiro et al.*, 2007], that we applied to our measurements. Based on the linear positive relation found between deposition and event size at San Luis (Figure 2), long-term deposition rates were estimated based on daily precipitation records for 1960–1999 at Villa Mercedes (Figure 1). Long-term deposition estimates for the study sites were finally adjusted to the ratio of their mean annual precipitation and that measured at Villa Mercedes (Table 1). This approach assumed that deposition was directly proportional to precipitation (i.e., same average chloride concentration), which is partially supported by the fact that the closest deposition records that we have for comparison, obtained in Pergamino with a mean annual precipitation of 900 mm/yr, showed reasonably similar chloride concentration values in precipitation (0.27 mg/L [*Piñeiro et al.*, 2007]).

[17] Recharge estimates based on chloride data were based on the assumption that (1) chloride transport is well approximated by piston flow, (2) all inputs to the ecosystem originate from atmospheric deposition, with rock weathering supply being negligible, and (3) plant uptake and storage in biomass and organic matter are negligible components of the chloride balance [*Allison et al.*, 1985]. This conservative behavior enables the calculation of the residual moisture flux [*Phillips*, 1994]:

$$R = \frac{P \times Cl_p}{Cl_{sw}} \times 1000 \qquad (1)$$

where R is the net downward residual flux at the depth of measurement (mm/yr), P is the precipitation (mm/yr), Cl_p is the chloride rain concentration (mg/L) (Cl^- deposition rate: g/m^{2}yr, see measurement methods above and values at Table 1), and Cl_{sw} is the measured Cl^- concentration in the soil water (mg/L). The value of Cl_{sw} was determined by plotting cumulative Cl^- content (mass Cl^- per unit volume of soil) with depth against cumulative water content (volume water per unit volume soil) at the same depths. In all cases straight/line segments were found once the rooting zone was discarded (0–2.5 and 0–1 m in dry forests and agriculture, respectively). The slope of these plots was used as the Cl_{sw} value of equation (1) [*Phillips*, 1994]. This method was used to estimate recharge at both dry forests and agriculture stands.

[18] At sites that were converted from natural dry forests to dryland agroecosystems, recharge was also estimated using the tracer front displacement method [*Walker et al.*, 1991], from the velocity of the tracer front (v) as follows:

$$R = v\theta = \theta\frac{z_1 - z_2}{t_1 - t_2} \qquad (2)$$

where R is the recharge (mm/yr), θ is the average water content over this depth interval, and z_1 and z_2 are the depths of the chloride fronts corresponding to times t_1 and t_2 related to the new (t_1: dryland agriculture) and old (t_2: rangeland) land uses (see Table 2 for agriculture times). Values of 1 m and 6 m were used for z_1 and z_2, respectively, in

Table 2. Vadose Zone Water Accumulation, Chloride Storage, and Recharge Estimates[a]

Site	Vegetation/ Use	Water Storage (mm)	Chloride Storage (g/m^2)	Recharge (mm/yr) RMF	Recharge (mm/yr) CFD
S1	DF	756 ± 137	9294 ± 3700	0.02	
S2	DF	431 ± 92	2044 ± 1556	0.04	
	A (~40)	898 ± 235	441 ± 696	13.2	10.4
S3	DF	400 ± 42	1560 ± 786	0.05	
	A (~40)	447 ± 62	17.9 ± 5	6.9	≥7.9
S4	DF	233 ± 28	203 ± 42	0.14	
	ODF	298 ± 105	198 ± 144	0.21	
	A (~90)	570 ± 19	30 ± 30	10.8	≥5.3
S5	DF	312 ± 78	154 ± 204	0.33	
	A (~35)	464 ± 49	0.8 ± 0.2	128.4	≥9.6

[a]Vegetation is characterized as typical dry forest (DF), overgrazed dry forest (ODF), and agriculture (A). Time under cultivation is indicated in parentheses (in years). Water and chloride storage values correspond to the total pools hosted by the top 6 m of the profiles (plus or minus standard deviation). Recharge was estimated based on two methods: Residual Moisture Flux (RMF) [see *Phillips*, 1994] and Chloride Front Displacement (CFD) as minimum recharge when the peak does not appear in the profile for dryland agriculture.

equation (2) when no chloride peak was present (yielding minimum estimate).

3. Results

3.1. Chloride and Water Profiles

[19] A consistently dry and salty zone was found below 2 m depth in all dry forest stands, suggesting that they experienced very low deep drainage (Table 2 and Figure 3). In dry forests stands, chloride pore water displayed peak concentrations of 1080 to 19400 mg/L, with peak values at depths (in average) of 1.5 m (S1), 2.5 m (S3), 4 m (S5), 5.5 m (S4) or at the base of the profile of the cores (S2) (Figure 3). Maximum chloride storage to 6 m depth reached ~9×10^3 g/m^2 at S1 (Table 2).

[20] In dry forests, high chloride accumulation zones coincided with low water content depths (Figures 3 and 4). Gravimetric moisture in dry forests had low values, ranging from 2.4% to 15.5% (average) (Figure 4). In sites under dry forest cover, soil water accumulation down to 6 m was between 233 and 756 mm (Table 2). Values of gravimetric water content in dry forests were below 50% of field capacity in most of the situations (Figure 4).

[21] The overgrazed dry forest stand at S4 showed lower pore water chloride concentrations than its nonovergrazed counterpart, with a peak at 5 m (average: 2040 mg/L), but differences were not statistically significant ($p > 0.05$). Chloride concentrations were lower in this profile down to 3.5 m (between 50 and 400 mg/L) and increased below 3.5 m depth approaching 5000 mg/L at base of the profile. Chloride storage was more variable and water storage higher in the overgrazed dry forest compared to its nonovergrazed counterpart (Table 2).

[22] In contrast to dry forests, agriculture stands had wetter profiles and lower chloride pore water concentrations ($p < 0.05$) with peaks of 2.7 to 2510 mg/L (averages for S5 and S2, respectively). Concentrations differed by at least 1 order of magnitude from those in dry forests, except for S2, that showed high chloride values below 4.5 m of depth (Figure 3). Agricultural profiles contained 1% to 22% (for S5 and S2, respectively) of the accumulated chloride mass (0–6 m depth) found in dry forests, suggesting intense salt leaching with increased recharge (Table 2). A couple of individual boreholes at dryland agriculture stands S2 and S4, showed high chloride accumulation at the bottom of the profiles (Table 2 and Figure 3), and were presumably not fully leached since the land use change occurred. Soil moisture content under agricultural sites was higher ($p < 0.05$) than that in dry forests and was close to field capacity at most sites (Figure 4). Water storage under agricultural areas down to 6 m depth duplicated that of dry forests (S2 and S4, Table 2). In the case of S3, the presence of an alfalfa pasture (a deep rooted, perennial species) may explain a drier soil profiles under cultivation, contrasting with those found under annual crops in the other sites. Water content exceeded 50% of field capacity in all situations at some depth intervals: between 3 m to 5.5 m (S2), 0.5 m to 1.5 (S3), 5.5 m (S4) and 6 m (S5).

3.2. Nitrate and Carbonate Salts

[23] Low nitrate-N concentrations were found in the vadose zone of both dry forest and cultivated sites, with average values that ranged from 0.34 mg/L to 2.15 mg/L, respectively. In dry forest stands, nitrate-N concentrations ranged from 0.39 mg/L at 5 m in S3 to 2.09 mg/L in the top 0.5 m depth in S5. In agricultural stands, values ranged from 0.34 mg/L at 3 ms depth in S3 to 2.15 mg/L in the top 0.5 m depth in S4. Carbonates were present in all profiles below 1 m and their concentration did not vary between dry forest and agricultural stands. Mean values (±SD) for dry forest and agricultural stands were 1.31% (±0.91) and 1.70% (±0.95), respectively. Highest carbonate concentrations were found between 2 and 3 m depth reaching 2.5%.

3.3. Recharge Estimates

[24] Annual recharge rates are extremely low in dry forests (<1 mm/yr), and only slightly altered by overgrazing, while at dryland agriculture this situation changes considerably (Table 2). Conversion of natural dry forest ecosystems to dryland agriculture, increased deep drainage, with recharge rates of 1 or 2 orders of magnitude higher, suggesting substantial flushing of salts throughout the measured sections of the vadose zone (Table 2 and Figures 3 and 4).

[25] Recharge estimates with the residual moisture flux approach showed characteristic slopes for the dry forest and agricultural stands. In the case of dry forests, cumulative chloride versus cumulative water curves had increasing slopes down to the rooting zone at approximately 2.5 m depth, and remained stable for the rest of the profile. Under agriculture, slopes were lower than those found in dry forests, and stable below the top meter. This approach showed differences with the front displacement method at agricultural stands, which offered a minimum boundary since in most cases the displaced peak was likely below the studied depth. At site S2-A, where the displaced peak was found, recharge estimate with this method were closer to those derived from the residual moisture flux model (Table 2).

[26] According to our estimates of atmospheric inputs and age of the top sedimentary layer we calculated the fraction of the historical chloride deposit that is still found in dry forest soils at all sites except S1, were the top sedimentary layer is an older one. We found that 94.7, 55.2, 5.1 and 3.6% of the presumable historical chloride inputs are preserved in the profiles of sites S2, S3, S5, and S4, respectively. Our recharge estimates have been based on the assumption of stable precipitation inputs, yet precipitation raises of ~30% during the last four decades illustrate how variable this input may have been in the past. We recalculated recharge estimates assuming a 30% cut in rainfall and, as a result, on chloride inputs, obtaining values from 0.13 to 0.37 mm/yr (sites S1 and S5, respectively) for dry forests and 4.9 (S3) to 89.9 mm/yr (S5) for agriculture (see Table 2).

4. Discussion

[27] Dry forests on sedimentary plains of central Argentina showed characteristic low recharge rates and high salt accumulation patterns displayed by other semiarid woody ecosystems in other continents [*Cook et al.*, 1989; *Allison et al.*, 1990; *Edmunds and Gaye*, 1994; *Phillips*, 1994; *Walvoord et al.*, 2003; *Scanlon et al.*, 2006]. Chloride accumulations at our dry forest sites are lower than those found for semiarid woodlands in Australia [*Cook et al.*, 1989], but similar to those reported in grasslands and shrub lands of Southern High Plains in the U.S. [*Scanlon et al.*, 2005, 2007b]. Yet all

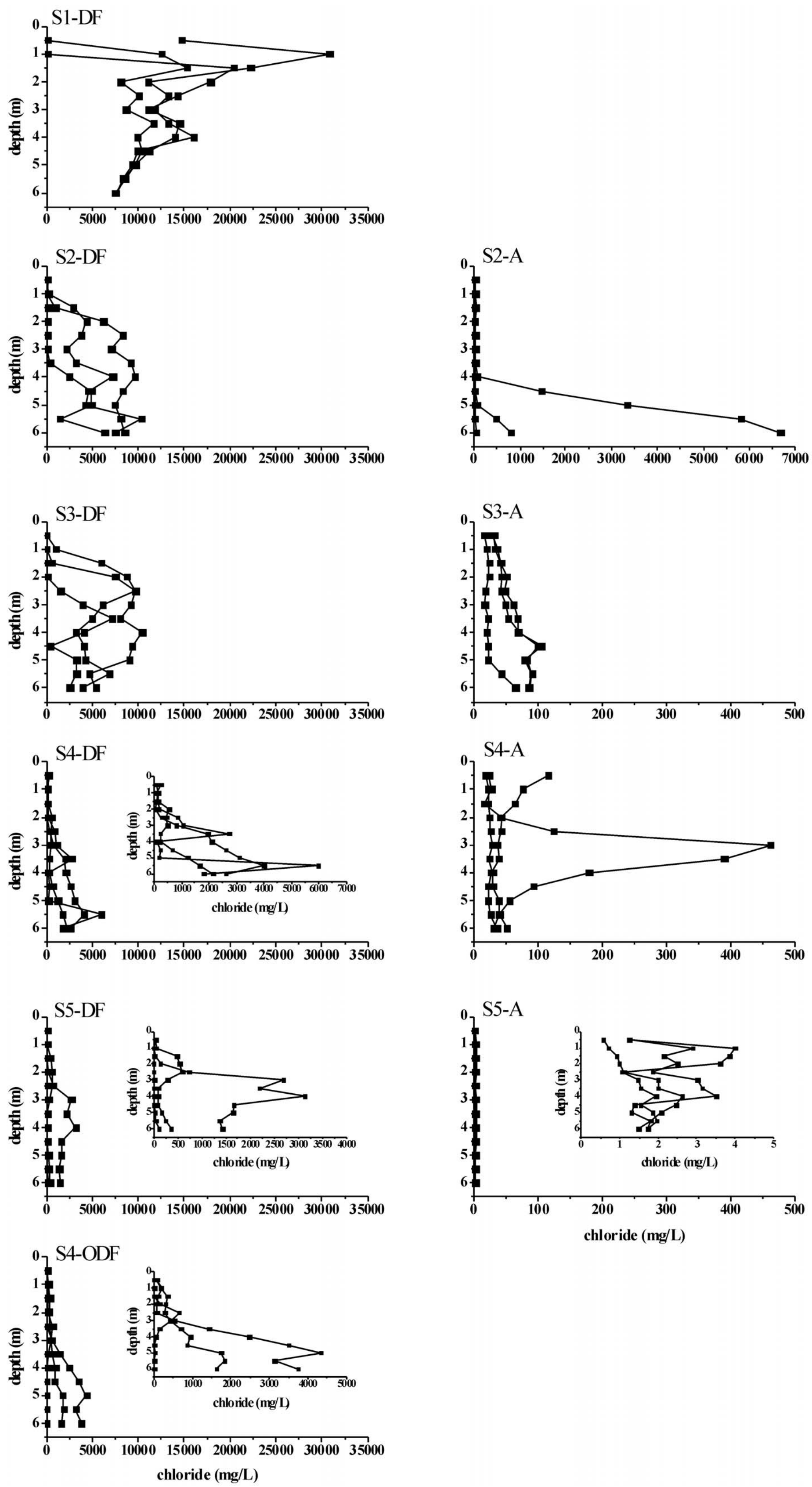

Figure 3. Pore water chloride concentration (mg/L) profiles in typical dry forest (DF), agriculture (A), and overgrazed dry forest (ODF) stands for all the boreholes (n = 3). Chloride concentration scales differ between DF and A stands. Insets show the same data using detailed scales for DF and A. Profile for S2-A has different scale.

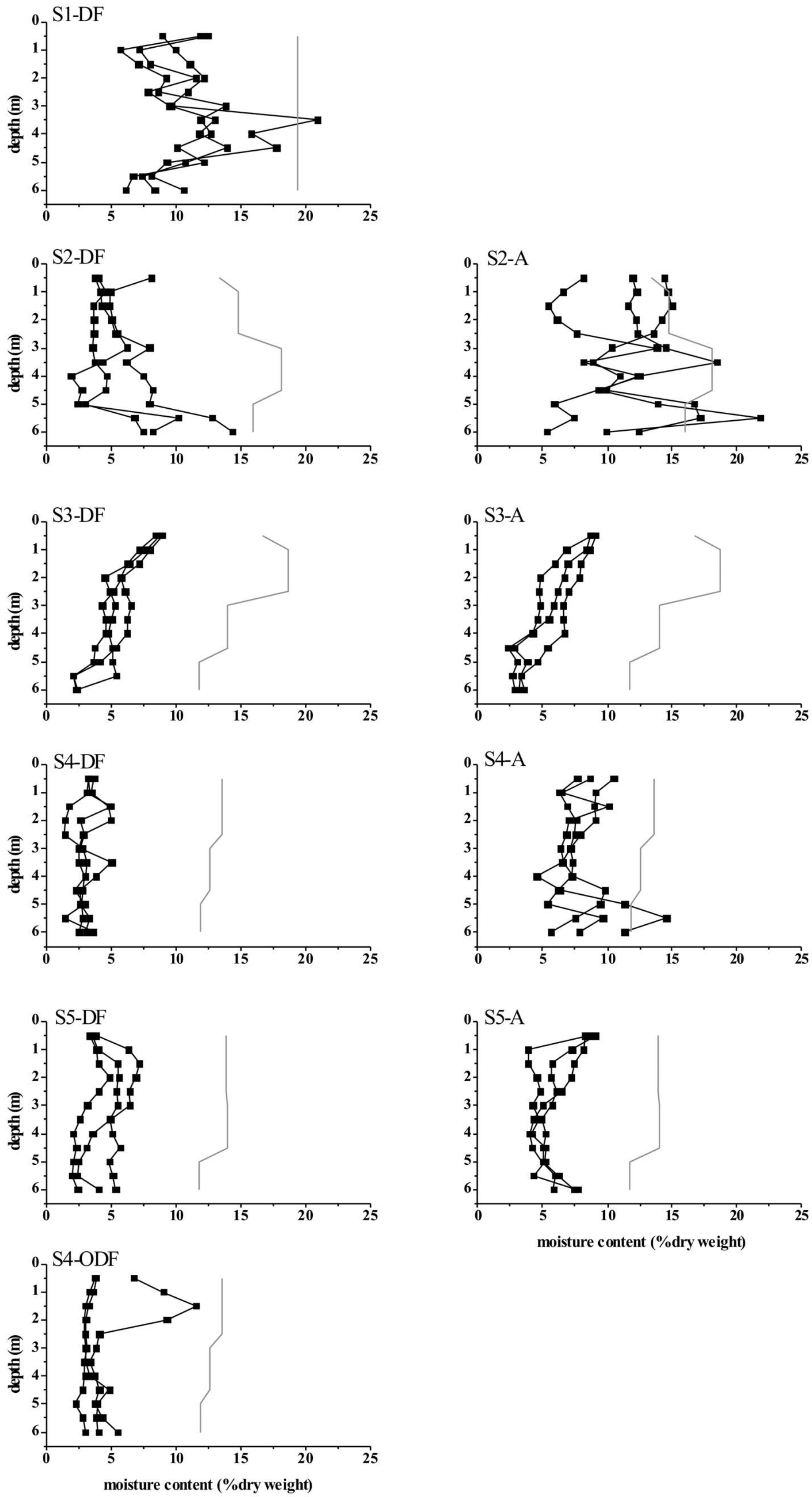

Figure 4. Gravimetric water content (%) profiles in typical dry forest (DF), agriculture (A), and overgrazed dry forest (ODF) stands for all the boreholes (n = 3). Grey lines indicate field capacity.

these regions displayed low recharge under their natural vegetation cover, suggesting that woody and grassland ecosystems in semiarid sedimentary terrains are very effective in maximizing precipitation water use [*Specht*, 1972; *Hatton et al.*, 1997; *Eagleson*, 2002]. The lower chloride accumulation found in Argentina and the U.S., compared to Australia, may result from lower atmospheric chloride inputs [*Allison and Hughes*, 1978; *Scanlon et al.*, 2005; *Scanlon et al.*, 2009].

[28] Agriculture introduced a clear change in chloride and water storage patterns across our study sites. Agricultural profiles lost 78%–99% of the chloride originally found in dry forests. This, together with soil water contents close to field capacity, indicates the onset of deep drainage and salt flushing following cultivation. Similar situations have been described in other dry forests converted into agriculture. In Australia, more than a century of dryland agriculture increased recharge, raising water tables close to the surface, concentrating salts in evaporative discharge zones and causing loss of 60,000 km^2 of agricultural land in the last century, affecting native vegetation in some cases [*George et al.*, 1997; *National Land and Water Resources Audit*, 2001]. In the High Plains of the U.S. the replacement of grasslands and shrub lands by dryland agriculture has triggered drainage, water table rises, and slight salinity increases in groundwater, but to a lesser extent than in Australia [*Scanlon et al.*, 2005, 2007b, 2009]. In many sectors of the High Plains, the large extent and intensity of groundwater based irrigation may have prevented some of the water table rises expected for dryland agriculture. Dry forest clearing for millet cultivation in the Sahel, has triggered a decline in evapotranspiration that, in this case, seems to have resulted in increased surface runoff rather than higher drainage, but have boosted groundwater recharge in the lower parts of the landscape [*Leduc et al.*, 2001; *Favreau et al.*, 2009]. At these sites, water levels have risen by 0.4 m/yr for the last 50 years [*Leduc et al.*, 2001; *Favreau et al.*, 2009]. The replacement of natural dry forests and grasslands by dryland agriculture represents a threat for the long-term productivity of agricultural land and natural vegetation as suggested by the Australian case. Overgrazing at one of our dry forest sites showed small effects on drainage and salt leaching, particularly compared to cultivation. Although further exploration is required to clarify which disturbances and ecological attributes need to be affected in dry forests to generate deep drainage, these preliminary observations suggest that continuous plant level pressures introduced by year-round grazing did not release as much water as the repeated annual disturbance of the whole ecosystem associated with cultivation. Plant traits, such as rooting depth, plant phenology and longevity and their integration and diversity across all the components are additional aspects that need attention.

[29] Soil texture could influence deep drainage and salt accumulation patterns and their response to cultivation. In sandy soils, declining water holding capacity and/or unsaturated hydraulic conductivity could limit the exhaustive use of precipitation by natural ecosystems. The depth of a soil column that can store a given amount of moisture increases in a more or less linear way with sand content, creating the need of a parallel increase of rooting depth for its exhaustive consumption. In contrast, unsaturated conductivity of soils increases exponentially with sand content, creating a dramatic constraint on the possibility of an exhaustive water use that can only be compensated by higher root density [*Sperry et al.*, 1998]. Biological limits on the allocation of plant energy to denser root systems could eventually limit the exhaustive use of water in very sandy soils as it has been shown for natural woody ecosystems on sand dune landscapes close to our study region [*Jobbágy et al.*, 2010]. Although sediment texture becomes coarser in sites receiving higher precipitation, this climatic difference is relatively small (between a 6% from the minimum and a 12% from the maximum) particularly in the context of past precipitation changes [*Giulietti et al.*, 2003]. We suggest that differences in recharge rates under dry forests are more likely responding to textural contrasts.

[30] Our study region offered a novel climatic context that has not been explored in other vadose moisture and salt transport studies in dry forests. While other studies have shown the onset or increase of groundwater recharge under steady or even declining rainfall trends [e.g., *Favreau et al.*, 2009], central Argentina has experienced a 30% increase in average annual precipitation over the last 100 years at our study region (538 mm/yr in 1900–1950 to 649 mm/yr in 1950–2000) [*Giulietti et al.*, 2003; *Barros et al.*, 2008]. Although this climatic shift could be a priori considered sufficient to cause a region-wide increase in recharge and salt leaching, deep drainage has remained very low under the dry forests that we examined, as found in chloride versus water cumulative-cumulative plots based on the *Phillips* [1994] method. These natural ecosystems have effectively consumed this additional water supply such that significant precipitation rises "per se" were unable to trigger the onset of deep drainage. On the other hand, the observed climatic shift, however, could have favored recharge and salt leaching indirectly, by creating better conditions for the expansion of dryland cultivation [*Viglizzo et al.*, 1995]. Market signals such as raising grain to livestock prices, technological improvements like no-till sowing, have likely converged with higher precipitation inputs facilitating expansion of agriculture over dry forest ecosystems [*Viglizzo et al.*, 1995]. Our findings highlight the importance of isolating land use versus climate effects on hydrological systems before straight biophysical connections like increasing precipitation and rising recharge rates, are assumed.

[31] The widespread effects of dry forest cultivation on groundwater levels and soil salinity found in Australia have not been described for central Argentina so far. Isolated cases of soil salinization, however, have been described for dry forests in central Paraguay [*Nitsch*, 1995; *Nitsch et al.*, 1998; *Jobbágy et al.*, 2008], and local to regional increases on water yields have been reported for heavily cultivated areas [*Piovano et al.*, 2004; *Jobbágy et al.*, 2008], yet assumed to be caused by secular precipitation increases by most scientists. Our study confirms how the same ecosystem properties of dry forests versus annual dryland crops that are responsible of dryland salinity in Australia, apply to central Argentina, yet the rates at which they can drive water table rises are strongly dependent on the local hydrogeological context. Groundwater salinity is relatively high in the area (500 to 3000 mg/L of dissolved solids [*San Luis Groundwater Resources Project*, 2002]), suggesting that on one hand the salt load leached from the vadose zone may not have a very significant impact on water quality, yet

rising water tables that approach the surface could still cause intense soil salinization under the semiarid conditions of the region. A better understanding of the role of sediment texture affecting recharge rates and vadose salt accumulation will help assess the differential risk of dryland salinity across sediment types.

[32] Dry and salty vadose zones of arid ecosystems in the U.S. have been shown to host large amounts of nitrate, presumably lost from natural ecosystems [*Edmunds*, 1999; *Walvoord et al.*, 2003] (but see *Jackson et al.* [2004]). Although dry forests studied by us are dominated by N-fixing leguminous species, profiles revealed nitrate concentration less than one thousand times lower than those reported by *Walvoord et al.* [2003], suggesting that these ecosystems have relatively small deep N outputs and that the flushing of their soil profiles following the onset of agriculture would not result in a large load of soluble N to groundwater, as it has been suggested for arid ecosystems in the U.S. [*Walvoord et al.*, 2003] and north Africa [*Edmunds*, 1999]. Probably, cultivated plots at our study region had gone through a process described in the Southern High Plains [*Scanlon et al.*, 2009], where soils with initial low nitrate concentrations have shown a buildup of deep nitrate pools resulting from mineralization and nitrification of soil organic nitrogen [see also *Portela et al.*, 2006].

[33] Our results indicate that vegetation is a critical control on long-term water transport and salt mobilization across the ecosystem–vadose zone–groundwater continuum. In spite of a sharp increase on precipitation in the last century, the dry forests that we studied maintained the characteristic low recharge rates and high salt accumulation patterns displayed by semiarid woody vegetation in other regions. Replacement of native vegetation by agriculture began a century ago and is still rapidly expanding. This land use change could modify groundwater hydrology and salt transport in sedimentary plains of South America. Networks of monitoring wells capable of recording long-term trends in groundwater levels and salinity across cultivated areas in plains of central Argentina are currently missing. They are, however, crucial to detect the early onset of regional dryland salinity processes that could cause great damage to production and natural resources.

[34] **Acknowledgments.** Very helpful field and laboratory collaboration was provided by Victoria Marchesini, Silvina Ballesteros, Ana Acosta, Esteban Crespo, Gervasio Piñeiro, Jimena Juri Ayub, María Beatríz Núñez, Mónica Bacchetta, John Kim, Germán Baldi, Marcelo Nosetto, Alfonsina Tripaldi, Steve Forman, and Marisa Puente. Jimena Juri Ayub and Bridget Scanlon provided valuable comments and guidelines to improve the manuscript. We thank the anonymous reviewers and associate editor for their insightful review. We appreciate the support by local farmers facilitating sampling work in their properties. This work was funded by grants from SECyT (Argentina), Fundación Antorchas, the Inter-American Institute for Global Change Research (IAI, CRN II 2031), which is supported by the U.S. National Science Foundation (grant GEO-0452325), and the U.S. National Science Foundation (BIO-0717191). C. S. Santoni was supported by CONICET (Beca Doctoral Interna) and S. Contreras by a postdoctoral fellowship awarded by the Spanish Ministry of Science and Technology (2008-0486).

References

Alley, W. M., R. W. Healy, J. W. LaBaugh, and T. E. Reilly (2002), Flow and storage in groundwater systems, *Science*, *296*, 1985–1990, doi:10.1126/science.1067123.

Allison, G. B., and M. W. Hughes (1978), The use of environmental chloride and tritium to estimate total recharge to an unconfined aquifer, *Aust. J. Soil Res.*, *16*, 181–195, doi:10.1071/SR9780181.

Allison, L. E., and C. D. Moodie (1965), Carbonate volumetric calcimeter method, in *Methods of Analysis*, *Agron. Monogr.*, vol. 9, part 2, edited by C. A. Black, pp. 1389–1392, Am. Soc. of Agronomy, Madison, Wis.

Allison, G. B., W. J. Stone, and M. W. Hughes (1985), Recharge in karst and dune elements of a semi-arid landscape as indicated by natural isotopes and chloride, *J. Hydrol.*, *76*, 1–25.

Allison, G. B., P. G. Cook, S. R. Barnett, G. R. Walker, I. D. Jolly, and M. W. Hughes (1990), Land clearance and river salinisation in the western Murray Basin, Australia, *J. Hydrol.*, *119*, 1–20, doi:10.1016/0022-1694(90)90030-2.

Baez, S., and S. L. Collins (2008), Shrub invasion decreases diversity and alters community stability in northern Chihuahuan Desert plant communities, *PLoS ONE*, *3*, e2332, doi:10.1371.

Baron, J. S., N. L. Poff, P. L. Angermeier, C. N. Dahm, P. H. Gleick, N. G. Hairston Jr., R. B. Jackson, C. A. Johnston, B. D. Richter, and A. D. Steinman (2002), Meeting ecological and societal needs for freshwater, *Ecol. Appl.*, *12*, 1247–1260.

Barros, V. R., M. E. Doyle, and I. A. Camilloni (2008), Precipitation trends in southeastern South America: Relationship with ENSO phases and with low-level circulation, *Theor. Appl. Climatol.*, doi:10.1007/s00704-007-0329-x.

Cabrera, A. L. (1976), Regiones fitogeográficas de la República Argentina, in *Enciclopedia de Agricultura, Jardinería y Fruticultura*, vol. 2, 2nd ed., pp. 1–85, ACME S.A. C. I., Buenos Aires.

Cook, P. G., and G. R. Walker (1990), The effect of soil type on groundwater recharge in the Mallee region, *Rep. 28*, Cent. for Groundwater Stud., Glen Osmond, South Aust., Australia.

Cook, P. G., G. R. Walker, and I. D. Jolly (1989), Spatial variability of groundwater recharge in a semiarid region, *J. Hydrol.*, *111*, 195–212, doi:10.1016/0022-1694(89)90260-6.

Dussart, E., P. Lerner, and R. Peinetti (1998), Long-term dynamics of two populations of *Prosopis caldenia* Burkart, *J. Range Manage.*, *51*, 685–691, doi:10.2307/4003613.

Eagleson, P. S. (2002), *Ecohydrology: Darwinian Expression of Vegetation Form and Function*, doi:10.1017/CBO9780511535680, Cambridge Univ. Press, Cambridge, U. K.

Eberbach, P. L. (2003), The eco-hydrology of partly cleared, native ecosystems in southern Australia: A review, *Plant Soil*, *257*, 357–369, doi:10.1023/A:1027392703312.

Echeverría, J. C., and J. A. Bertón (2006), Temperatura del aire, in *Aptitud Forestal de la Provincia de San Luis*, edited by J. C. Echeverría, E. G. Jobbágy, and A. D. Collado, pp. 13–23, Inst. Nac. de Tecnol. Agropecuaria San Luis, San Luis, Argentina.

Edmunds, W. M. (1999), Groundwater nitrate as a palaeoenvironmental indicator, in *Geochemistry of the Earth's Surface*, edited by H. Ármannsson, pp. 35–39, Int. Assoc. of Geochem. and Cosmochem., Balkema, Rotterdam, Netherlands.

Edmunds, W. M., and C. B. Gaye (1994), Estimating spatial variability of groundwater recharge in the Sahel using chloride, *J. Hydrol.*, *156*, 47–59, doi:10.1016/0022-1694(94)90070-1.

Elliot, E. T., J. W. Heil, E. F. Kelly, and H. Curtis Monger (1999), Soil structural and other physical properties, in *Standard Soil Methods for Long-Term Ecological Research*, edited by G. P. Robertson et al., pp. 74–85, Oxford Univ. Press, New York.

English, N. B., J. F. Weltzin, A. Fravolini, L. Thomas, and D. G. Williams (2005), The influence of soil texture and vegetation on soil moisture under rainout shelters in a semi-desert grassland, *J. Arid Environ.*, *63*, 324–343, doi:10.1016/j.jaridenv.2005.03.013.

Favreau, G., C. Leduc, C. Marlin, M. Dray, J.-D. Taupin, M. Massault, C. Le Gal La Salle, and M. Babic (2002), Estimate of recharge of a rising water table in semiarid Niger from ^3H and ^{14}C modeling, *Ground Water*, *40*, 144–151, doi:10.1111/j.1745-6584.2002.tb02499.x.

Favreau, G., B. Cappelaere, S. Massuel, M. Leblanc, M. Boucher, N. Boulain, and C. Leduc (2009), Land clearing, climate variability, and water resources increase in semiarid southwest Niger: A review, *Water Resour. Res.*, *45*, W00A16, doi:10.1029/2007WR006785.

Fernandez-Illescas, C. P., A. Porporato, F. Laio, and I. Rodriguez-Iturbe (2001), The ecohydrological role of soil texture in a water-limited ecosystem, *Water Resour. Res.*, *37*, 2863–2872, doi:10.1029/2000WR000121.

Forman, S. (1989), Applications and limitations of thermoluminescence to date quaternary sediments, *Quat. Int.*, *1*, 47–59, doi:10.1016/1040-6182(89)90008-6.

Frankenberger, W. T., M. A. Tabaitabai, D. C. Adriano, and H. E. Doner (1996), Bromine, chlorine and fluorine, in *Methods of Soil Analysis Part 3*, edited by S. L. Sparks, pp. 833–868, Am. Soc. of Agron., Madison, Wis.

George, R. J., D. J. McFarlane, and R. A. Nulsen (1997), Salinity threatens the viability of agriculture and ecosystems in *Western Australia, Hydrogeol. J.*, *5*, 6–21, doi:10.1007/s100400050103.

Giulietti, J. D., J. C. Echeverría, and A. D. Collado (2003), Condicionantes históricos en la desertificación en San Luis, in *Con las Metas Claras. La Estación Experimental Agropecuaria San Luis: 40 Años a Favor del Desarrollo Sustentable*, edited by M. O. Aguilera and J. L. Panigatti, pp. 11–24, Inst. Nac. de Tecnol. Agropecuaria, San Luis, Argentina.

Grau, H. R., T. M. Aide, and N. I. Gasparri (2005), Globalization and soybean expansion into semiarid ecosystems of Argentina, *Ambio*, *34*, 265–266.

Grau, H. R., N. I. Gasparri, and T. M. Aide (2008), Balancing food production and nature conservation in the neotropical dry forests of northern Argentina, *Global Change Biol.*, *14*, 985–997, doi:10.1111/j.1365-2486.2008.01554.x.

Harrington, G. A., P. G. Cook, and A. L. Herczeg (2002), Spatial and temporal variability of ground water recharge in Central Australia: A tracer approach, *Ground Water*, *40*, 518–527, doi:10.1111/j.1745-6584.2002.tb02536.x.

Hatton, T. J., and R. A. Nulsen (1999), Towards achieving functional ecosystem mimicry with respect to water cycling in southern Australian agriculture, *Agrofor. Syst.*, *45*, 203–214, doi:10.1023/A:1006215620243.

Hatton, T. J., G. D. Salvucci, and H. I. Wu (1997), Eagleson's optimality theory of an ecohydrological equilibrium: Quo vadis?, *Funct. Ecol.*, *11*, 665–674, doi:10.1046/j.1365-2435.1997.00159.x.

Hillel, D. (1998), *Environmental Soil Physics*, 771 pp., Academic, San Diego, Calif.

Instituto Nacional de Estadísticas y Censos (2002), *Censo Nacional Agropecuario del Año 2002* [CD-ROM], Buenos Aires.

Iriondo, M. H. (1997), Models of deposition of loess and loessoids in the upper Quaternary of South America, *J. South Am. Earth Sci.*, *10*, 71–79, doi:10.1016/S0895-9811(97)00006-0.

Jackson, R. B., et al. (2000), Belowground consequences of vegetation change and its treatment in models, *Ecol. Appl.*, *10*, 470–483, doi:10.1890/1051-0761(2000)010[0470:BCOVCA]2.0.CO;2.

Jackson, R. B., S. T. Berthrong, C. W. Cook, E. G. Jobbágy, and R. L. McCulley (2004), Comment on "A reservoir of nitrate beneath desert soils," *Science*, *304*, 51, doi:10.1126/science.1094294.

Jobbágy, E. G., M. D. Nosetto, C. S. Santoni, and G. Baldi (2008), El desafío ecohidrológico de las transiciones entre sistemas leñosos y herbáceos en la llanura Chaco-Pampeana, *Ecol. Austral*, *18*(3), 305–322.

Jobbágy, E. G., M. Nosetto, P. Villagra, and R. Jackson (2010), Water subsidies from mountains to deserts: Their role sustaining groundwater-fed oases in a sandy landscape, *Ecol. Appl.*, doi:10.1890/09-1427.1, in press.

Kennett-Smith, A., P. G. Cook, and G. R. Walker (1994), Factors affecting groundwater recharge following clearing in the south western Murray Basin, *J. Hydrol.*, *154*, 85–105, doi:10.1016/0022-1694(94)90213-5.

Leaney, F. W., and G. B. Allison (1986), Carbon-14 and stable isotope data for an area in the Murray Basin: Its use in estimating recharge, *J. Hydrol.*, *88*, 129–145, doi:10.1016/0022-1694(86)90201-5.

Leblanc, M. J., G. Favreau, S. Massuel, S. O. Tweed, M. Loireau, and B. Cappelaere (2008), Land clearance and hydrological change in the Sahel: SW Niger, *Global Planet. Change*, *61*, 135–150, doi:10.1016/j.gloplacha.2007.08.011.

Leduc, C., G. Favreau, and P. Schroeter (2001), Long-term rise in a Sahelian water-table: The Continental Terminal in south-west Niger, *J. Hydrol.*, *243*, 43–54, doi:10.1016/S0022-1694(00)00403-0.

Marban, L., and S. E. Ratto (2005), Métodos de extracción y determinación de nitratos en suelo (II), in *Tecnologías en Análisis de Suelos: Alcance a Laboratorios Agropecuarios*, edited by L. Marbán and S. E. Ratto, pp. 49–52, Asoc. Argent. de la Cienc. del Suelo, Buenos Aires.

Markgraf, V. (1989), Paleoclimate in central and South America since 18,000 BP based on pollen and lake level records, *Quat. Sci. Rev.*, *8*, 1–24, doi:10.1016/0277-3791(89)90018-8.

National Land and Water Resources Audit (2001), *National Land and Water Resources Audit Australian dryland salinity assessment 2000: Extent, impacts, processes, monitoring and management options*, 771 pp., Canberra.

Nitsch, M. (1995), El desmonte en el chaco central del Paraguay: Influencia sobre el agua subterránea y la salinización de los suelos paper presented at 2nd Simposio Sobre Aguas Subterráneas y Perforación de Pozos en el Paraguay, Dep. de Abastecimiento de Agua para el Chaco-Paraguay, San Lorenzo, Paraguay.

Nitsch, M., R. Hoffmann, J. Utermann, and L. Portillo (1998), Soil salinization in the central Chaco of Paraguay: A consequence of logging, *Adv. Geoecol.*, *3*, 495–502.

Paruelo, J. M., J. P. Guerschman, and S. R. Verón (2005), Expansión agrícola y cambios en el uso del suelo, *Cienc. Hoy*, *15*, 14–23.

Petheram, C., L. Zhang, G. Walker, and R. Grayson (2000), Towards a framework for predicting impacts of land-use on recharge: A review of recharge studies in Australia, *Tech. Rep. 28/00*, CSIRO Land and Water, Melbourne, Canberra, ACT, Australia.

Phillips, F. M. (1994), Environmental tracers for water movement in desert soils of the American Southwest, *Soil Sci. Soc. Am. J.*, *58*, 15–24, doi:10.2136/sssaj1994.03615995005800010003x.

Piñeiro, G., E. G. Jobbágy, R. B. Jackson, C. S. Santoni, S. I. Portela, and C. Di Bella (2007), RP-RainNet: The Rio de la Plata atmospheric deposition network. Set up and preliminary results, *Eos Trans. AGU*, *88*(23), Jt. Assem. Suppl., Abstract B33A-07.

Piovano, E. L., D. Ariztegui, S. M. Bernasconi, and J. A. McKenzie (2004), Stable isotopic record of hydrological changes in subtropical Laguna Mar Chiquita (Argentina) over the last 230 years, *Holocene*, *14*, 525–535, doi:10.1191/0959683604hl729rp.

Portela, S. I., A. E. Andriulo, M. C. Sasal, B. Mary, and E. G. Jobbágy (2006), Fertilizer vs. organic matter contributions to nitrogen leaching in cropping systems of the Pampas: ^{15}N application in field lysimeters, *Plant Soil*, *289*, 265–277, doi:10.1007/s11104-006-9134-z.

Postel, S. L., G. C. Daily, and P. R. Ehrlich (1996), Human appropriation of renewable fresh water, *Science*, *271*, 785–788, doi:10.1126/science.271.5250.785.

Prado, J. L., J. Chiesa, G. Tognelli, E. Cerdeño, and E. Strasser (1998), Los mamíferos de la formación Río Quinto (Plioceno), provincia de San Luis (Argentina). Aspectos bioestratigráficos, zoogeográficos y peleoambientales, *Estud. Geol.*, *54*, 153–160.

Programa de las Naciones Unidas para el Medio Ambiente (2003), GEO América Latina y el Caribe. Perspectivas del medio ambiente, Of. Reg. para Am. Lat. y el Caribe, Mexico City.

Prose, D. V., S. K. Metzger, and H. G. Wilshire (1987), Effects of substrate disturbance on secondary plant succession: Mojave Desert, California, *J. Appl. Ecol.*, *24*, 305–313, doi:10.2307/2403806.

Rambal, S. (1995), From daily transpiration to seasonal water balance: An optimal use of water?, in *Time Scales of Biological Responses to Water Constraints*, edited by J. Roy, J. Aronson, and F. di Castri, pp. 37–51, SPB Acad., Amsterdam.

Salama, R. A., P. Farrington, G. A. Bartle, and G. D. Watson (1993), Salinity trends in the wheatbelt of Western Australia: Results of water and salt balance studies from Cuballing catchment, *J. Hydrol.*, *145*, 41–63, doi:10.1016/0022-1694(93)90219-Y.

San Luis Groundwater Resources Project (2002), Evaluación de posibilidades físicas y económicas de riego con aguas subterráneas en la provincia de San Luis: Hidrogeología de la zona Central, in *Los Recursos Hidrológicos Subterráneos de la Provincia de San Luis. Un Proyecto de Cooperación Técnica Argentino-Australiano*, edited by S. M. Lunter, Map 1, Bur. of Rural Sci. Agric., Fish. and For. Aust., San Luis, Argentina.

Saxton, K. E., and W. J. Rawls (2006), Soil water characteristics estimates by texture and organic matter for hydrologic solutions, *Soil Sci. Soc. Am. J.*, *70*, 1569–1578, doi:10.2136/sssaj2005.0117.

Scanlon, B. R., R. W. Healy, and P. G. Cook (2002), Choosing appropriate techniques for quantifying groundwater recharge, *Hydrogeol. J.*, *10*, 18–39, doi:10.1007/s10040-001-0176-2.

Scanlon, B. R., R. C. Reedy, D. A. Stonestrom, D. E. Prudic, and K. F. Dennehy (2005), Impact of land use and land cover change on groundwater recharge and quality in the southwestern US, *Global Change Biol.*, *11*, 1577–1593, doi:10.1111/j.1365-2486.2005.01026.x.

Scanlon, B. R., K. E. Keese, A. L. Flint, L. E. Flint, C. B. Gaye, W. M. Edmunds, and I. Simmers (2006), Global synthesis of groundwater recharge in semiarid and arid regions, *Hydrol. Processes*, *20*, 3335–3370, doi:10.1002/hyp.6335.

Scanlon, B. R., I. Jolly, M. Sophocleous, and L. Zhang (2007a), Global impacts of conversions from natural to agricultural ecosystems on water resources: Quantity versus quality, *Water Resour. Res.*, *43*, W03437, doi:10.1029/2006WR005486.

Scanlon, B. R., R. C. Reedy, and J. A. Tachovsky (2007b), Semiarid unsaturated zone chloride profiles: Archives of past land use change impacts on water resources in the southern High Plains, United States, *Water Resour. Res.*, *43*, W06423, doi:10.1029/2006WR005769.

Scanlon, B. R., R. C. Reedy, R. L. Baumhardt, and G. Strassberg (2008), Impact of deep plowing on groundwater recharge in a semiarid region: Case study, High Plains, Texas, *Water Resour. Res.*, *44*, W00A10, doi:10.1029/2008WR006991.

Scanlon, B. R., D. A. Stonestrom, R. C. Reedy, F. W. Leaney, J. Gates, and R. G. Cresswell (2009), Inventories and mobilization of unsaturated zone sulfate, fluoride, and chloride related to land use change in semiarid regions, southwestern United States and Australia, *Water Resour. Res.*, *45*, W00A18, doi:10.1029/2008WR006963.

Schenk, H. J., and R. B. Jackson (2002), The global biogeography of roots, *Ecol. Monogr.*, *72*, 311–328, doi:10.1890/0012-9615(2002)072[0311:TGBOR]2.0.CO;2.

Secretaría de Ambiente y Desarrollo Sustentable de la Nación (2007), *Primer Inventario Nacional de Bosques Nativos: Informe Regional Espinal Segunda Parte*, 1st ed., 154 pp., Secr. de Ambiente y Desarrollo Sustentable de la Nacion, Buenos Aires.

Seyfried, M. S., S. Schwinning, M. A. Walvoord, W. T. Pockman, B. D. Newman, R. B. Jackson, and F. M. Phillips (2005), Ecohydrological control of deep drainage in arid and semiarid regions, *Ecology*, *86*, 277–287, doi:10.1890/03-0568.

Small, E. E. (2005), Climatic controls on diffuse groundwater recharge in semiarid environments of the southwestern United States, *Water Resour. Res.*, *41*, W04012, doi:10.1029/2004WR003193.

Solbrig, O. T. (1999), Observaciones sobre biodiversidad y desarrollo agrícola, in *Biodiversidad y Uso de la Tierra. Conceptos y Ejemplos de Latinomérica*, edited by S. D. Mateucci et al., pp. 29–40, Cen. de Estud. Avanzados, Univ. de Buenos Aires, Buenos Aires.

Specht, R. L. (1972), Water use by perennial evergreen plant communities in Australia and Papua New Guinea, *Aust. J. Bot.*, *20*, 273–299, doi:10.1071/BT9720273.

Sperry, J. S., F. R. Adler, G. S. Campbell, and J. P. Comstock (1998), Limitation of plant water use by rhizosphere and xylem conductance: Results from a model, *Plant Cell Environ.*, *21*, 347–359, doi:10.1046/j.1365-3040.1998.00287.x.

Sperry, J. S., U. G. Hacke, R. Oren, and J. P. Comstock (2002), Water deficits and hydraulic limits to leaf water supply, *Plant Cell Environ.*, *25*, 251–263, doi:10.1046/j.0016-8025.2001.00799.x.

Tripaldi, A., and S. L. Forman (2007), Geomorphology and chronology of Late Quaternary dune fields of western Argentina, *Palaeogeogr. Palaeoclimatol. Palaeoecol.*, *251*, 300–320, doi:10.1016/j.palaeo.2007.04.007.

Viglizzo, E. F., Z. E. Roberto, M. C. Filippin, and A. J. Pordomingo (1995), Climate variability and agroecological change in the central Pampas of Argentina, *Agric. Ecosyst. Environ.*, *55*, 7–16, doi:10.1016/0167-8809(95)00608-U.

Viglizzo, E. F., Z. E. Roberto, F. Lértora, E. López Gay, and J. Bernardos (1997), Climate and land-use change in field-crop ecosystems of Argentina, *Agric. Ecosyst. Environ.*, *66*, 61–70, doi:10.1016/S0167-8809(97)00079-0.

Vörösmarty, C. J., P. Green, J. Salisbury, and R. B. Lammers (2000), Global water resources vulnerability from climate change and population growth, *Science*, *289*, 284–288, doi:10.1126/science.289.5477.284.

Wainwright, J., A. J. Parsons, W. H. Schlesinger, and A. D. Abrahams (2002), Hydrology-vegetation interactions in areas of discontinuous flow on a semi-arid bajada, southern New Mexico, *J. Arid Environ.*, *51*, 319–338, doi:10.1006/jare.2002.0970.

Walker, G. R. (1998), Using soil water tracers to estimate recharge, in *The Basics of Recharge and Discharge, Part 7*, edited by L. Zhang and G. R. Walker, CSIRO, Collingwood, Victoria, Australia.

Walker, G. R., I. D. Jolly, and P. G. Cook (1991), A new chloride leaching approach to the estimation of diffuse recharge following a change in land use, *J. Hydrol.*, *128*, 49–67.

Walker, G. R., L. Zhang, T. W. Ellis, T. J. Hatton, and C. Petheram (2002), Estimating impacts of changed land use on recharge: Review of modeling and other approaches appropriate for management of dryland salinity, *Hydrogeol. J.*, *10*, 68–90, doi:10.1007/s10040-001-0181-5.

Walvoord, M. A., F. M. Phillips, D. A. Stonestrom, R. D. Evans, P. C. Hartsough, B. D. Newman, and R. G. Striegl (2003), A reservoir of nitrate beneath desert soils, *Science*, *302*, 1021–1024, doi:10.1126/science.1086435.

Wilcox, B. P., M. S. Seyfried, and D. D. Breshears (2003), Water balance on rangelands, in *Encyclopedia of Water Science*, edited by B. A. Stewart and T. A. Howell, pp. 791–794, Marcel Dekker, New York.

Wood, W. W., and W. E. Sanford (1995), Eolian transport, saline lake basins, and groundwater solutes, *Water Resour. Res.*, *31*, 3121–3129, doi:10.1029/95WR02572.

Zak, M. R., M. Cabido, and J. G. Hodgson (2004), Do subtropical seasonal forests in the Gran Chaco, Argentina, have a future?, *Biol. Conserv.*, *120*, 589–598, doi:10.1016/j.biocon.2004.03.034.

Zárate, M. A. (2003), Loess of southern South America, *Quat. Sci. Rev.*, *22*, 1987–2006, doi:10.1016/S0277-3791(03)00165-3.

S. Contreras, E. G. Jobbágy, and C. S. Santoni, Grupo de Estudios Ambientales, IMASL, Universidad Nacional de San Luis/CONICET, Ej de los Andes 950, San Luis 5700, Argentina. (jobbagy@unsl.edu.ar)