Ecological Applications, 21(7), 2011, pp. 2367–2379

Changes in hydrology and salinity accompanying a century of agricultural conversion in Argentina

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Abstract. Conversions of natural woodlands to agriculture can alter the hydrologic balance, aquifer recharge, and salinity of soils and groundwater in ways that influence productivity and sustainable land use. Using a land-use change chronosequence in semiarid woodlands of Argentina's Espinal province, we examined the distribution of moisture and solutes and estimated recharge rates on adjacent plots of native woodlands and rain-fed agriculture converted 6–90 years previously. Soil coring and geoelectrical profiling confirmed the presence of spatially extensive salt accumulations in dry woodlands and pervasive salt losses in areas converted to agriculture. A 1.1-km-long electrical resistivity transect traversing woodland, 70-year-old agriculture, and woodland, for instance, revealed a low-resistivity (high-salinity) horizon between ~3 m and 13 m depth in the woodlands that was virtually absent in the agricultural site because of leaching. Nine-meter-deep soil profiles indicated a 53% increase in soil water storage after 30 or more years of cultivation. Conservative groundwater-recharge estimates based on chloride tracer methods in agricultural plots ranged from ~12 to 45 mm/yr, a substantial increase from the <1 mm/yr recharge in dry woodlands. The onset of deep soil moisture drainage and increased recharge led to >95% loss of sulfate and chloride ions from the shallow vadose zone in most agriculture plots. These losses correspond to over 100 Mg of sulfate and chloride salts potentially released to the region's groundwater aquifers through time with each hectare of deforestation, including a capacity to increase groundwater salinity to >4000 mg/L from these ions alone. Similarities between our findings and those of the dryland salinity problems of deforested woodlands in Australia suggest an important warning about the potential ecohydrological risks brought by the current wave of deforestation in the Espinal and other regions of South America and the world.

Key words: agriculture; deforestation; ecohydrology; Espinal province, Argentina; groundwater recharge; land-use change; salinity; salt leaching; semiarid regions; soil resistivity imaging; South America.

Introduction

Population growth, demand for food, and the drive for green energy are accelerating the conversion of natural lands to agriculture. Globally, croplands have increased in area six-fold in the last three centuries and are expected to expand another 18% by 2050 (e.g., Goldewijk 2001, Tilman et al. 2001). Since the most productive lands of the planet are generally under cultivation already (Ramankutty et al. 2002), future agricultural expansions will increasingly occur in more environmentally sensitive and economically risky marginal lands. The rapid and expansive land-use changes that are occurring in semiarid regions of Latin America are a prime example of this process (Grau et al. 2005, Gasparri and Grau 2009). Similar patterns of change are occurring on other continents as well.

A greater concern beyond agricultural productivity and profitability of these newly established croplands is their potential impacts on the surrounding environments (Jackson et al. 2009). Changing the existing groundwater recharge–discharge relationships, for example, may affect the quality and quantity of surface and groundwater resources (e.g., Jackson et al. 2005, Wilcox and Thurow 2006, Jobbágy and Jackson 2007). The magnitude of such environmental impacts often take decades to materialize, by which time they can sometimes be unmanageable (e.g., Bekle et al. 2004).

Large-scale replacement of natural vegetation with annual agricultural crops can alter the hydrologic balance of a region (Scanlon et al. 2005). The magnitude of such changes depends on a number of factors, including evaporative demand and precipitation patterns, canopy interception, growing-season length, agricultural land management practices, soil water storage and infiltration characteristics (e.g., Brauman et al. 2010). In arid and semiarid regions where vegetation productivity tends to be water limited, plants

Manuscript received 29 October 2010; accepted 2 January 2011; final version received 24 February 2011. Corresponding editor: B. P. Wilcox.
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and ecosystems have adopted strategies to access and use nearly all available water in the surrounding environment (Hillel and Tadmor 1962, Schenk and Jackson 2002, Weltzin et al. 2003, Seyfried et al. 2005). As a result, deep soil moisture drainage that could become groundwater recharge tends to be small. One common implication of this exhaustive water-use behavior is the retention and accumulation of significant quantities of salts in the plant-accessible regions of the vadose zone.

Annual agricultural crops are often shallowly rooted and, unlike native species, may not be able to extract soil water at low matric potentials (Canadell et al. 1996, Jackson et al. 1996). Shorter growing seasons and other differences in crops compared with natural vegetation can all leave a larger volume of moisture unused in the soil (Savage et al. 1996). Land-use changes from natural vegetation to agriculture in semiarid settings can therefore lead to the onset of deep soil water drainage where it was negligible before (Oconnell et al. 1995, Seyfried and Wilcox 2006, Scanlon et al. 2007). A potential consequence of this increased drainage is salt mobilization and migration in the vadose zone, which can pose a threat to water quality through salt leaching to surface and groundwater systems. Over longer time scales, however, the land may become unproductive if water tables rise sufficiently to affect plant growth and if salts accumulate at the surface through capillary rise and evaporative concentration (Williams 1987). The present-day salinity crisis of Australia is a good example of the long-term implications of poorly managed land-use change and agricultural expansion at the expense of native vegetation. Waterlogging and soil salinization from land clearing over a century ago have resulted in the loss of thousands of hectares of agricultural land and continue to cost the Australian economy millions of dollars each year through lost productivity and land-value depreciation (Australian Bureau of Statistics 2002, Clarke et al. 2002).

The purpose of this study was to evaluate how vadose-zone soil water and salt dynamics are altered when natural woodland ecosystems are replaced by large-scale dryland agriculture. We focused on semiarid regions of central Argentina, hypothesizing that expanding dryland agriculture there would cause rapid and large-scale mobilization of salts stored beneath the natural woodlands. Previous observations showed important changes in surface water and shallow chloride concentrations attributable to agricultural conversion (Santoni et al. 2010). Here, we used a unique deep-soil data set to quantify soil water storage, groundwater recharge, and salt inventories using soil cores and geophysical measurements to determine how woodland-to-agriculture conversions alter water and salt dynamics in the deep vadose zone. We used a combination of soil moisture, soil-water chemistry, and geophysical soil electrical-resistivity measurements along a chronosequence of 6-yr-old to ~90-yr-old agriculture and paired woodland plots. The proximity and hydrological links between our study region in the Espinal and the adjacent Pampas, one of the most productive agricultural provinces in the world (FAO 2007), amplify the long-term implications of current land-use changes and the future of sustainable land use there.

Methods

Regional setting

Our study region in the Espinal province is a 120-km^2 area located ~60 km southeast of the city of San Luis, Argentina (Fig. 1). This section of the Espinal (originally covered by forests) is located in the western highland edge of the sedimentary plain that covers central Argentina and stretches to the lower Pampas (originally covered by grasslands) and the Atlantic Ocean to the east (Fig. 1). The fine, unconsolidated sandy materials consisting of quartz, feldspars, and volcanic glass that underlie the region are loessic sediments derived from the Andes to the west and Sierra Pampeanas and Paraná basins to the north beginning about 10 million years ago (Iriondo 1997, Zárate 2003). The proximity to the original sediment sources makes the loess in our study sites sandier than is typical for loess-derived soils (sand fraction >70%). The thickness of these loessic sediments in our region is >60 m (San Luis Groundwater Resources Project 2002).

The regional climate is semiarid with a mean annual rainfall of ~525 mm that occurs mostly (~80%) in the warm season (November to March). Mean atmospheric temperatures in the austral winter and summer range from ~6° to 10°C and ~22°–26°C, respectively. Despite seasonally concentrated and episodic rainfall, surface water-transport features are absent across the landscape, suggesting that runoff has been minimal during the ~9000-yr lifetime of the surface sediments (Santoni et al. 2010).

Prior to the introduction of cattle in the 1600s the vegetation of the Espinal province was characterized by dry woodland ecosystems dominated by *Prosopis caldenia*, *P. flexuosa*, and *Geoffroea decorticans* trees, which formed a fairly dense woody system when undisturbed by grazing, fire, or other disturbances (Dussart et al. 1998, Lewis et al. 2009). European settlements in the 1880s to 1920s led to the establishment and subsequent expansion of agriculture and intensive grazing. Since the 1980s the rate of agricultural expansion has accelerated through deforestation and the conversion of grasslands and pastures to crop production fueled by the rising global demand for soybean, corn, and other grains (Viglizzo and Frank 2006, Zak et al. 2008). To date, nearly 65% of the Espinal phytogeographical province has been replaced by crops and pastures (Informe Regional Espinal 2007).

Site selection

We selected a total of 13 sites (5 dry woodland and 8 adjacent agriculture sites), with the agriculture sites

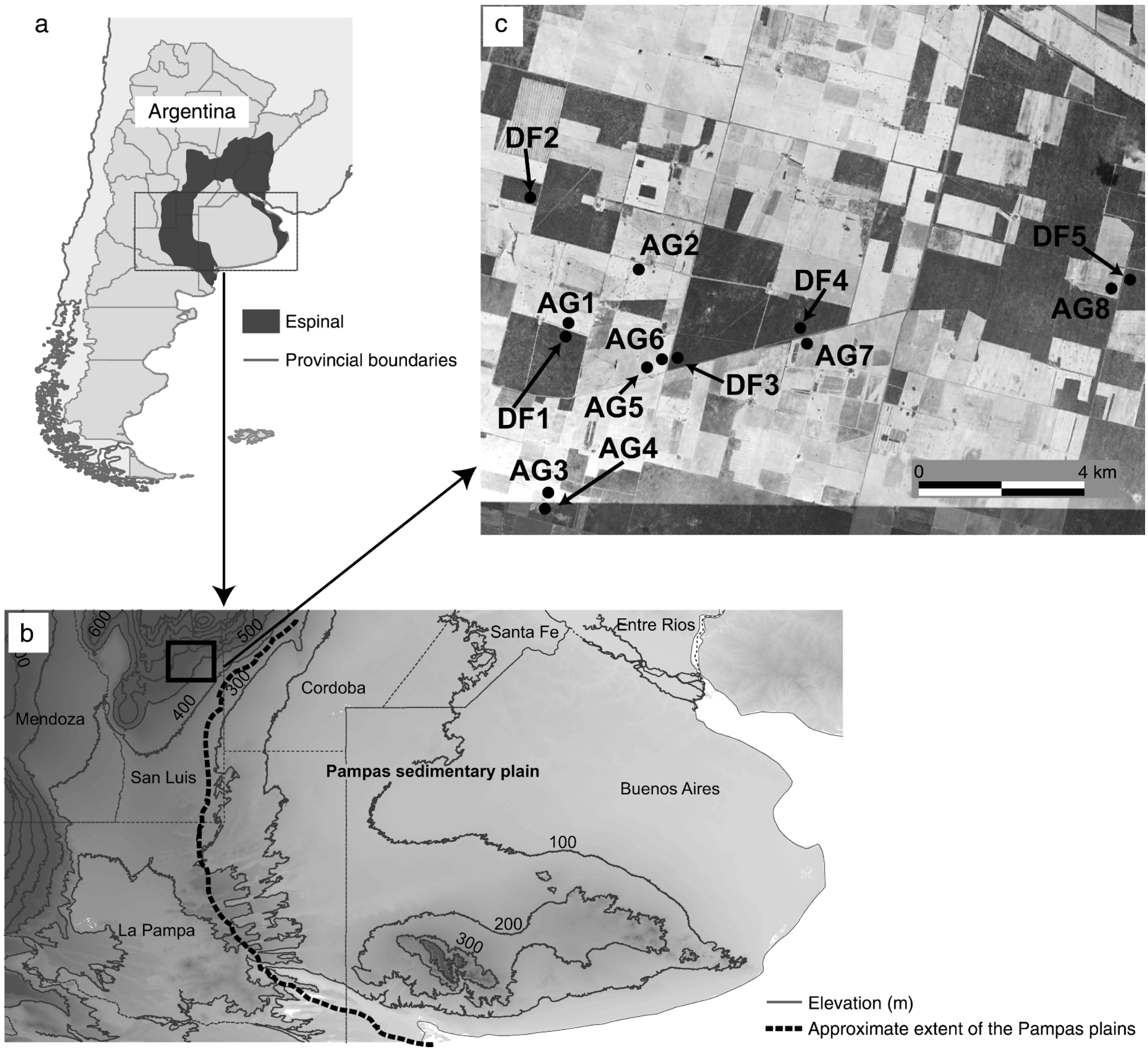

FIG. 1. (a) Sites investigated in this study, located within the Espinal phytogeographical province of central Argentina. (b) The vast Pampas sedimentary plain, one of the most agriculturally productive regions in the world, located east of San Luis, receives surface water and potentially ground water draining from (c) the highland rim where the study sites are located. Eight agriculture sites with ages 6 (AG 1), 15 (AG 2), 30 (AG 3, AG 4), 40 (AG 5, AG 6), 90 (AG 7), and 70 (AG 8) years, and five dry woodland sites (DF1–DF5), are all located within a 15 × 8 km area near the city of San Luis. [Image (c) is from Google Earth].

having been continuously cultivated for the last 6 to ~90 years. The dry woodland sites (DF1–DF5) were similar to each other in terms of tree density, canopy cover, and biomass characteristics and were typical of the woodland ecosystems in the region (Dussart et al. 1998, Lewis et al. 2009). Agriculture sites were of 6 (AG1), 15 (AG2), 30 (AG3, AG4), ~40 (AG5, AG6), ~90 (AG7), and ~70 (AG8) years of age. The ages of the agriculture sites were determined from Landsat imagery for the youngest site (2000–2002 images), from historic aerial photographs for the 15- to 40-yr-old sites, and from aerial photographs and landowner histories for the 70- and 90-yr-old sites. Paired woodland and agriculture sites (DF1–AG1, DF3–AG5/6, DF4–AG7, and DF5–AG8) with similar geological and geomorphological traits were used for adjacent comparisons. All selected agricultural sites were unirrigated and were subject to typical crop rotations of the region, primarily a combination of annual crops such as corn, sunflower, wheat, and soybean since the 1990s and alfalfa pastures and annual forage grasses during the 1980s and before (Viglizzo and Frank 2006). The only fertilizer applied in crop fields is nitrogen, mostly in the form of urea and typically at rates <100 $kg \cdot ha^{-1} \cdot yr^{-1}$. Woodland and agriculture plots selected for the study were at least 0.5 km^2 in area and had no significant topographic undulations, slopes, roads, or other cultural or natural disturbances.

Soil sampling and analysis

Soil samples were collected from eight agriculture and five dry woodland sites on three occasions between February 2009 and March 2010. Paired sites were always sampled either on the same day or within 2–3 days. At each site one or more boreholes was made using a bucket auger to a maximum depth of 10 m for soil sampling. The occurrence of caliche, which was encountered occasionally in the landscape, limited the augering to 6 m at the AG2 site and 7 m at the DF1 and DF5 sites. Soil samples were collected in 0.5-m increments in each borehole. Approximately 150–200 g of soil from each 0.5-m interval were subsampled after homogenizing the extracted soils. To minimize soil moisture losses, samples were immediately placed in air-tight double polyethylene bags.

Gravimetric moisture content of soils was obtained by oven-drying a portion of each collected sample at 105°C to constant mass. The hydrometer method was used to determine the texture characteristics of soil samples (∼6 samples per borehole) from each study site (Gee and Bauder 1986). The concentrations of Cl^-, Br^-, SO_4^{-2}, NO_3^{-2}, and PO_4^{-3} in 1:1 mixtures of oven-dried soil and deionized water were measured in the laboratory using an ion chromatograph (Dionex IC-2000; Dionex Corporation, Sunnyvale, California, USA). Bulk density (ρ_b) of soils was estimated (1.6 g/cm^3) based on the average soil texture characteristics (sand 73%, silt 24%, clay 3%; $n = 83$ core samples) (Saxton and Rawls 2006).

Geophysical characterization of salt distributions

The abundances of salt and water influence the electrical conductivity of soils. In consequence, electrical-resistivity imaging (ERI) can be used to map soil-conductivity patterns in the subsurface. Resistivity imaging is also useful for accessing deeper parts of the subsurface that are beyond the reach of augers. ERI is a well-established method used for commercial water and mineral prospecting. Its adoption in environmental hydrology (Slater et al. 2000) and ecohydrological and biological research (Jackson et al. 2005, Jayawickreme et al. 2008, Robinson et al. 2008), however, is relatively recent.

To obtain soil resistivity at the study sites, a multi-electrode resistivity unit was used. Electrodes were installed in a straight line with either 2 m or 4 m (sites AG8–DF5) separation distances between electrodes to attain the desired imaging depths and resolutions. Standard electrode-pairing methods (Wenner, Dipole-Dipole, or Schlumberger; Loke 2000) were used to capture the optimal two-dimensional (2D) distribution of ground resistivity. The 2-m electrode separation allowed for an imaging depth of ∼20 m, whereas the 4-m electrode separation extended the imaging depth to ∼40 m. At most sites, resistivity data were collected along 142-m-long transects. At the AG8–DF5 sites we collected resistivity data along a 1.1-km transect traversing the woodland (DF5)–agriculture (AG8)–woodland (south of AG8) transition to capture potential soil conductivity and land-use relationships. The field data collected were inverted with commercially available RES2DINV software (Loke 2009) to obtain estimates of soil conductivity at each site. A simple conventional least-squares scheme was used for inverting all data sets.

To differentiate the relative influence of water and salt contents on measured soil resistivity, we supplemented the field measurements with detailed resistivity measurements in the laboratory using soil samples from the field sites. Using soils with saturation-extract salt concentrations of 5060, 1666, 292, and 241 μS/cm, we manipulated soil water contents to determine relationships of soil moisture and salt concentrations to soil electrical resistivity. These measurements were made in a Plexiglas box with terminals for current injections and electrical potential readings. A soil volume replicating the bulk density of soils under field conditions was carefully packed in the Plexiglas box multiple times with different soil water contents for measuring the resistivity–soil moisture relationship. According to these laboratory manipulations, the soil water saturation (S) to soil resistivity (ρ) relationship for all samples could be described by the following power-law function:

$$S = m\rho^{-n} \tag{1}$$

where $n = 0.803 \pm 0.04$. The coefficient m is related to salt concentration of the soil as

$$m = aC_s^{-b} \tag{2}$$

where a and b are fitting parameters with values 4286.8 and 0.98 ($R^2 = 0.9$), respectively; C_s is measured salt concentration in soil (mg/kg).

Groundwater recharge and salinity assessment approach

Groundwater recharge at the study sites was estimated using the chloride mass balance (CMB) and tracer front displacement (TFD) methods (Allison and Hughes 1978, Walker et al. 1991, Allison et al. 1994). According to the CMB method, recharge (R) can be computed as

$$R = C_p/C_s \tag{3}$$

where C_p is the Cl^- deposition rate (g·m^{-2}·yr^{-1}), and C_s is measured Cl^- concentration in soil water (g/m^3). C_p for the region is $\sim 246 \pm 16$ mg·m^{-2}·yr^{-1} (mean ± SD) (Santoni et al. 2010). C_s was computed based on the method described in Phillips (1994).

The TFD method is useful for estimating R where the steady-state assumption required for the CMB method may not hold because of a recent land-use change. The method requires fewer assumptions than the CMB method (see Allison et al. 1994, Wood and Sanford 1995). R according to TFD is expressed as

$$R = \theta v = \theta[(Z_2 - Z_1)/(t_2 - t_1)] \tag{4}$$

where v is the tracer front displacement velocity, $(t_2 – t_1)$ is the time since land-use change, Z_1 and Z_2 are

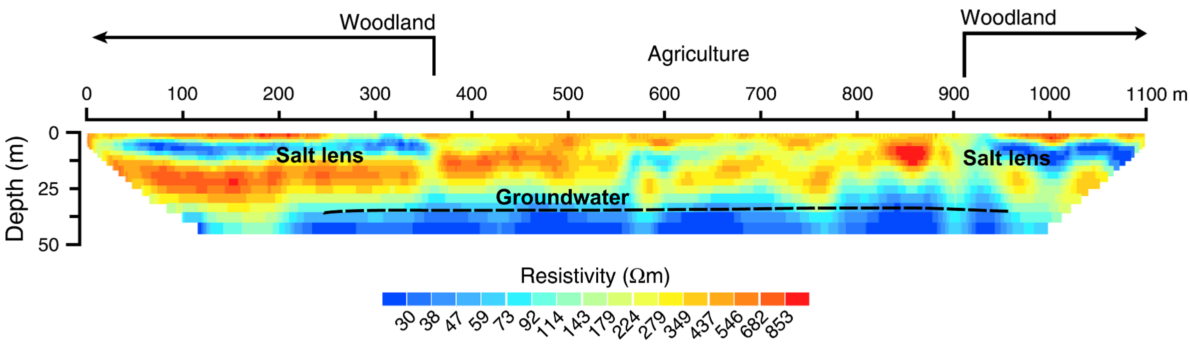

FIG. 2. Two-dimensional soil electrical-resistivity (ρ) profile along the woodland–agriculture–woodland transect at sites AG 8 and DF 5 (see Fig. 1 for location). The plot in the center (from ~360 m to 900 m) has been in agriculture for nearly 70 years. The groundwater table is ~35–40 m below surface. Topographic undulations were minimal along the transect, with <3 m elevation difference from the beginning to the end. The thin, but spatially continuous, low-soil-resistivity (high-soil-conductivity) zone (~3–13 m) beneath woodlands (0–360 m, 900–1100 m) illustrates areas with high salt concentrations that are conspicuously absent under agriculture.

positions of the Cl^- front corresponding to times t_1 (dry woodland) and t_2 (agriculture), and θ is the average moisture content over the Z_2,Z_1 interval.

An increase in soil water infiltration would first result in an increase in soil moisture, assuming the soil is below field capacity initially. As a result, any change at the water table will be delayed by time T_{gw}, which is a function of the vadose-zone thickness (Z_{vz}), soil moisture deficit (Δθ), and the recharge rate (R) (Cook et al. 1993, Leaney et al. 2003). T_{gw} can be expressed as

$$T_{gw} = Z_{vz}\Delta\theta/R. \quad (5)$$

The value of Δθ is computed as the difference in averaged moisture content between agriculture and dry woodland soil profiles below 2.5 m.

The onset of recharge at the groundwater table would raise the water table at a rate of ΔGW_z, which can be expressed as

$$\Delta GW_z = (R - D)/\Phi \quad (6)$$

where Φ is formation porosity and R and D are rates of groundwater recharge and discharge, respectively. While the discharge rate (D) would likely change slowly if the water table began to rise, we assume this term to be negligible for the timescale of our analysis; Φ is assumed to be constant with depth.

A downward-migrating moisture front would dissolve and transport salts previously stored in the vadose zone. For this transient process, and assuming nearly all the vadose salts would ultimately be released to the water table, the resulting groundwater salinity (C_{gw}) can be computed as;

$$C_{gw} = (M_{vz} + M_{gw})/(Rt + \Phi Z_{ml}) \quad (7)$$

where M_{vz} is the mass of salt added from the vadose to saturated zone, M_{gw} is the mass of salt in groundwater initially (500 mg/L; San Luis Groundwater Resources Project 2002), t is the time for transfer of salts from vadose to the saturated zone, and Z_{ml} is mixing zone thickness (assumed to be 5 m) where incoming and existing groundwater would mix primarily due to diffusive processes (Leaney et al. 2003). This estimate assumes the groundwater salinity change is only due to salt loading from the recharging water and no salt is lost from groundwater during time t.

RESULTS

Salt and water distributions in woodlands and agriculture

Geophysical and soil coring data provide a clear picture of changes in the subsurface, highlighting the impact of agriculture on soil salinity, water availability, and recharge. The ~1.1-km-long electrical resistivity transect traversing the woodland, agriculture, and woodland succession at the DF5 and 70-yr-old AG8 study sites revealed significant spatial differences in soil electrical resistivity between the two land uses (Fig. 2). The low-resistivity (high-conductivity) horizon between ~3 m and 13 m depth in the woodlands (at lateral distances of 0–360 m and 900–1100 m in Fig. 2), for example, is clearly absent in the ~540-m-wide crop field (360–900 m). The well-defined boundaries of agriculture and woodland units and their clear spatial correlation to measured soil-resistivity patterns suggest a strong coupling between the above- and belowground systems. The contrast in soil moisture profiles (Fig. 3) between the woodlands and the agriculture plot further illustrates the coupling between land use and the vadose zone.

Soil chemical analysis from deep coring in all of the 6-yr-old to 90-yr-old woodland sites confirmed the presence of large water-soluble salt inventories in the vadose zone of the region. Extractable chloride and sulfate were the most abundant anions (Table 1). Generally all woodland sites had low Cl^- concentrations in the top 1–3 m (Fig. 4a). Below that depth however, a sharp increase in Cl^- concentrations was observed, which remained elevated in all of the ~10-m-deep soil profiles. Compared to the woodlands, where the averaged Cl^- concentration was ~100 mg/kg, the paired

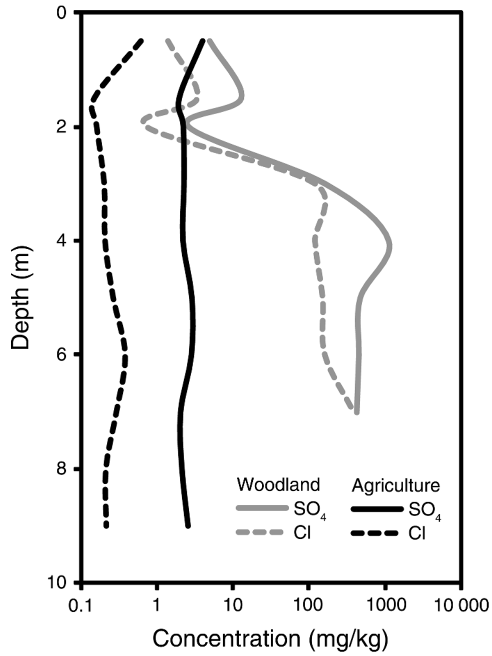

FIG. 3. Chloride and sulfate concentrations in soil cores at the DF 5 and AG 8 study sites. Agriculture soils show clear evidence of near-complete salt leaching compared to the nearby woodland. The spatial dimensions of this contrast are clearly captured by the electrical-resistivity transect at the site (Fig. 2). Note the log scale of the *x*-axis.

agriculture sites with deep soil cores (AG5, AG6, AG7, and AG8) were nearly devoid of Cl^- (<5 mg/kg) throughout the ~9-m-deep profiles (Fig. 4b). The 30-yr-old AG3 and AG4 sites also had completely leached Cl^- profiles in the sampled 9.5–10 m depth. The exceptions were the young agriculture sites, 6-yr-old AG1 and 15-yr-old AG2. Both of these sites showed Cl^- profiles resembling that of the nearby woodlands (DF1, DF2), presumably because of insufficient time for complete leaching to have taken place.

Sulfate concentrations in soil cores were consistent with the soil Cl^- data but showed more spatial variability and some variation with respect to agriculture age and leaching (Fig. 4c and d). All woodland sites on average had over 250 mg/kg of SO_4^{-2} in the 2–9 m depth. Comparatively, sulfate was nearly absent (2.8 ± 3.2 mg/kg) in older (>30 yr) agriculture sites (AG3–AG6) except for the 90-yr-old AG7 site, where the SO_4^{-2} concentrations were similar to values in its paired dry woodland (DF4); this difference is likely the result of gypsum or other sulfate-bearing minerals that accumulated locally in the sediments (San Luis Groundwater Resources Project 2002). Albeit minor, AG3 and AG4, which are 30-yr-old plots, also showed lower SO_4^{-2} concentrations than the older AG5 and AG6 plots. Compared to older sites, the younger agriculture sites had substantially greater SO_4^{-2} concentrations. The 15-yr-old AG2 site had over 3000 mg/kg of SO_4^{-2} in the 4–6 m depth interval, the highest concentration observed at any site. While soil coring at AG2 was limited to the top 6 m of soil, geoelectrical profiles indicated the presence of a much thicker (>10–15 m) conductive horizon with significant salt concentrations. Soil coring at the nearby DF2 dry woodland also confirmed the presence of high SO_4^{-2} levels to at least 10 m depth (Fig. 4c).

Salt losses in agriculture plots were accompanied by consistent increase in soil moisture across depths. Average gravimetric soil moisture profiles in older agriculture sites (AG3–AG8) were consistently higher than in woodland plots (DF2–DF5; Fig. 5). Comparing values for these categories, the average relative moisture difference was ~291 mm, a 53% increase in vadose water storage for conversions of dry woodland to agriculture in the 0–9 m depth. Soil moisture increases were also evident in the 6-yr-old and 15-yr-old agriculture sites, where the average moisture increase with agriculture was

TABLE 1. Solute concentrations (mean ± SD) in 6–10 m deep soil profiles at the various dry woodland (DF) and adjacent agricultural (AG) study sites in Espinal province, Argentina.

Site†	Solute concentration (mg/kg)				
	Cl^-	SO_4^{-2}	Br^-	NO_3^{-2}	PO_4^{-3}
AG1 (6 yr)	85 ± 107	176 ± 325	0.9 ± 0.8	2.6 ± 1.6	- -
DF1	54 ± 65	147 ± 189	0.8 ± 0.7	0.1 ± 0.1	- -
AG2 (15 yr)	64 ± 113	338 ± 495	0.4 ± 0.3	0.5 ± 0.3	0.1 ± 0.1
DF2	99 ± 85	926 ± 602	0.4 ± 0.2	0.2 ± 0.2	0.1 ± 0
AG3 (30 yr)	1.1 ± 1	1.4 ± 0.2	0.1 ± 0	0.3 ± 0.5	0.2 ± 0.1
AG4 (30 yr)	0.8 ± 0.5	1.3 ± 0.3	0 ± 0.1	0.3 ± 0.4	0.2 ± 0.1
AG5 (~40 yr)	1.1 ± 0.8	5.3 ± 4.8	0.2 ± 0	0.1 ± 0.1	0.1 ± 0
AG6 (~40 yr)	0.7 ± 0.2	3.3 ± 3	0.2 ± 0	0.1 ± 0.2	0.1 ± 0
DF3	139 ± 66	241 ± 147	0.7 ± 0.2	0.6 ± 0.7	0.1 ± 0
AG8 (70 yr)	0.2 ± 0.2	2.5 ± 0.6	0.1 ± 0.1	1.1 ± 1.4	- -
DF	141 ± 138	355 ± 363	0.3 ± 0.3	0.7 ± 0.7	- -
AG7 (~90 yr)	2.7 ± 2.1	295 ± 259	0.2 ± 0.1	0.8 ± 0.4	0.1 ± 0
DF4	145 ± 0.7	235 ± 171	0.7 ± 0.3	0.9 ± 0.6	0.1 ± 0

Notes: Bromide, nitrate, and phosphates were generally low at all sites; "- -" signifies no measurable phosphate. Chloride and sulfate were the most abundant anions, with significant differences in their distribution between agriculture and woodland areas.
† All study sites were at least 0.5 m^2 in area. All selected agricultural sites were unirrigated, and the length of time the site had been in agriculture is given in parentheses.

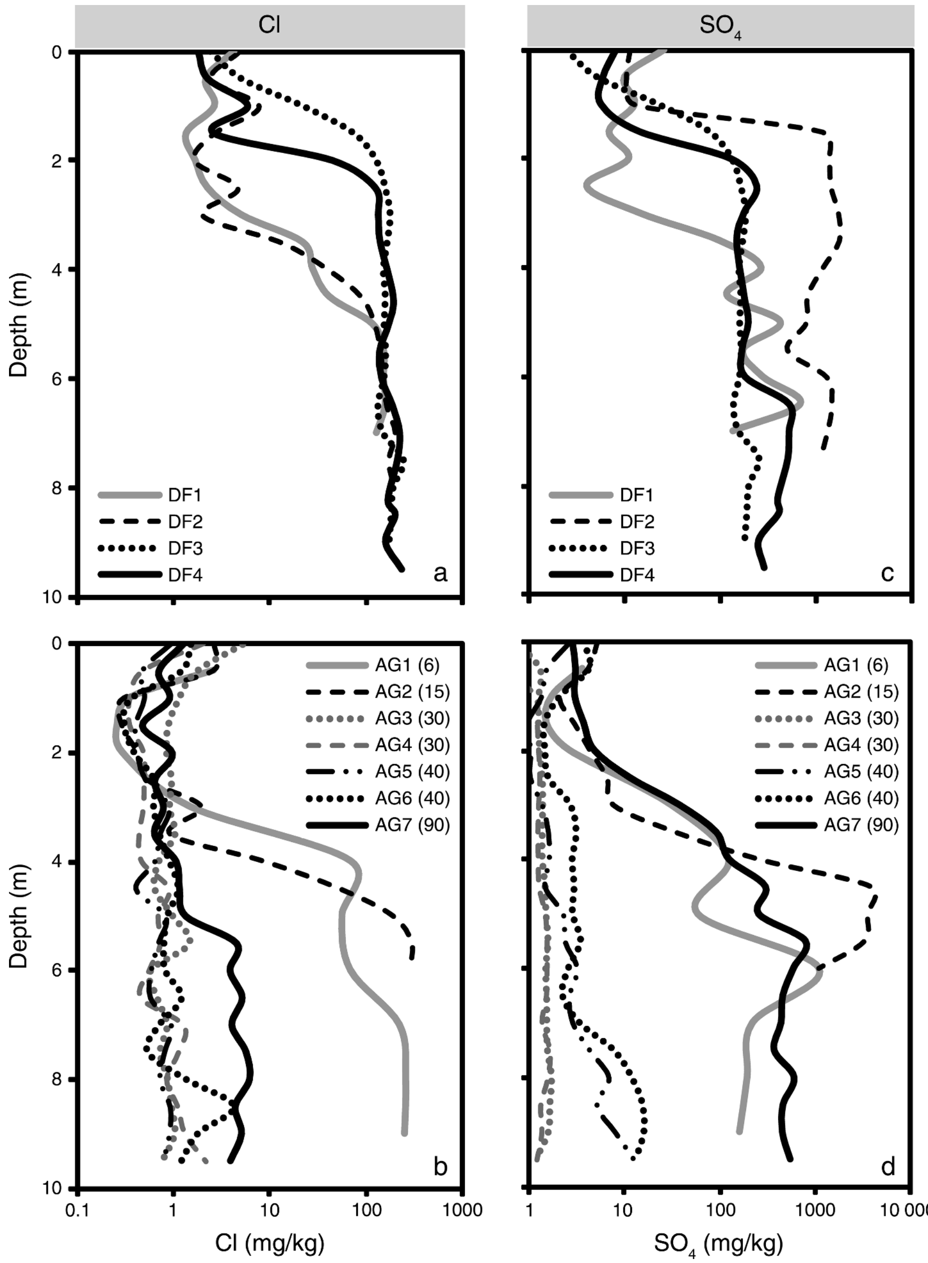

FIG. 4. Chloride and sulfate profiles at the study sites. Paired woodland–agriculture (AG1–DF1, AG6–DF3, AG7–DF4) sites are identified with thicker lines and the same colors in both woodland and agriculture plots. Chloride and sulfate concentrations in (a, c) woodland sites are distinctly higher than at (b, d) older agriculture sites. Peak concentration of Cl is reached at slightly different depths at different woodland sites, but the concentrations are similar below 5 m. Sulfate, however, is substantially different among woodlands. Sulfate at the 90-yr-old AG 7 site is potentially affected by the presence of gypsum or other sulfate-bearing minerals.

~31% in the top 7 m (0.056 ± 0.008 g water/g soil for agriculture vs. 0.043 ± 0.003 g water/g soil for woodland). In native woodlands, moisture variability was minimal across different sites in the top 1 to 3 m of soils. Additionally, all woodland sites had significantly less moisture in the 1.5–3 m interval than any of the agriculture sites had. This difference is almost certainly attributable to deeper and more persistent root water uptake by woodland trees compared to agricultural plants, similar to processes observed in nearby regions (Marchesini et al. 2009, Nosetto et al. 2011). The average moisture differences between woodlands and older agriculture were also noticeably large at greater (>8 m) depth (Fig. 5).

The spatiotemporal progression of vadose-zone changes following the introduction of agriculture was

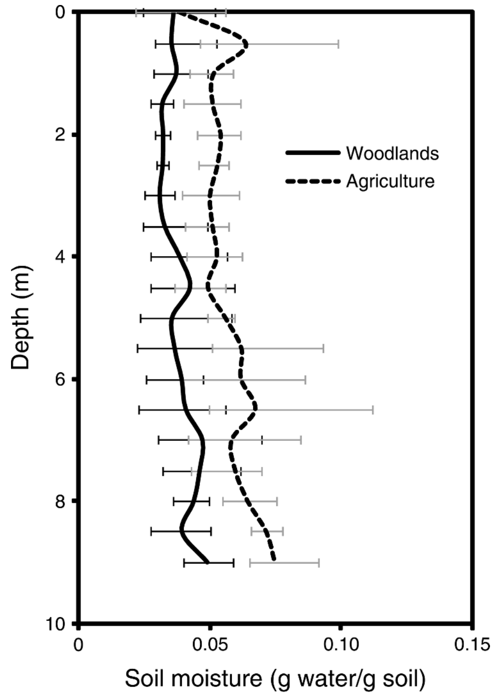

FIG. 5. Average gravimetric soil moisture distributions at old agriculture (AG3–AG8, 30–90 yr-old) and dry woodland (DF1–DF5) sites. Error bars show the range of observations between sites for each sampled depth. Moisture contents are consistently higher in agricultural soils than in woodlands. Data were collected between February 2009 and March 2010. The range of moisture observed at shallow depths may be influenced by water use and storage differences between measurements.

evident from the electrical-resistivity differences between 6-yr (AG1), 40-yr (AG6), and 70-yr (AG8) agriculture sites and their paired woodland plots (DF1, DF3, and DF5, respectively). Differences between AG1 and DF1 suggest that changes reaching depths of ~5–7 m may occur in less than a decade after introducing agriculture (Fig. 6a). These relatively rapid changes appeared to be spatially uniform with little evidence of preferential flows or concentrations. An increase in soil moisture (~31% g/g in the top 7 m of the AG1 crop field compared to its paired forest) appeared to be the dominant hydrologic change during early phases of vadose zone change, visible as the increased soil conductivity in the resistivity image (Fig. 6a). At more intermediate time scales, the vadose-zone differences appeared to propagate deeper (>15 m), comparing for instance the 40-yr-old agriculture site (AG6) and its paired woodland (DF3) (Fig. 6b). The higher electrical resistivity of agricultural soils at AG6 compared to its paired woodland site shows that vadose-zone salt losses have propagated deeper than the augured 9 m depth. At the 70-yr-old agriculture site (AG8), a complete transformation of the vadose zone resulting from these salt losses is evident, with much of the vadose zone showing increased soil resistivity compared to the adjacent woodland (Fig. 6c).

Solute inventories and sources

Differences in salt concentrations between agriculture and woodland soils translated to substantial vadose-zone salt losses with woodland-to-agriculture conversions. Based on average solute concentrations in the soil profiles, total chloride and sulfate inventories among four older (≥40 yr) agriculture sites ranged from ~60 to 145 $kg \cdot ha^{-1} \cdot m^{-1}$ compared to their paired woodlands where those inventories were ~6000–8000 $kg \cdot ha^{-1} \cdot m^{-1}$ (Table 2). Furthermore, electrical-resistivity imaging (ERI)-derived average salt concentrations below the augured 9 m depth showed the presence of large salt inventories at most woodland sites (Table 3). At the younger 6- and 15-yr-old agriculture sites, salt inventories were substantially larger than those at older agriculture sites, but were not greatly different from the adjacent woodlands (Table 2).

Chloride and sulfate in soil cores appeared to have originated from different sources, which may have important implications for spatial distribution of vadose salts in the region. Evaporative concentration of salts from precipitation appeared to be the main cause of Cl^- accumulation based on the $Cl^- : Br^-$ ratio (252–323) in woodland soil profiles (Carpenter 1978, McCaffrey et al. 1987). However, a similar conclusion could not be reached for sulfate, which was ~1.6 to >8.0 times that of Cl^- concentrations. Available evidence suggests that the loess is rich in S-bearing volcanic material derived from both local and distant sources (Zárate 2003). Depending on how these materials are distributed and incorporated into the loess locally, and how the loess has evolved over time, a degree of spatial variability is expected, as we observed across our study sites.

Groundwater recharge

Mean annual groundwater recharge estimated with the chloride mass balance (CMB) method was <1 mm/yr in all dry woodland sites, extremely small relative to the 525 mm/yr rainfall in the region (Table 2; Eqs. 3 and 4). In contrast, the estimated recharge across all old-agriculture (≥30-yr-old) sites was 29.7 ± 8.8 mm/yr (mean ± SD) , approximately 6% of annual rainfall (~525 mm). The estimated groundwater recharge was >31 mm/yr for the same agriculture sites using the tracer front displacement (TFD) method and the approximate salt front displacements determined with the electrical-resistivity transects (Table 2).

DISCUSSION

Land-use conversions from natural semiarid woodland ecosystems to agriculture resulted in significant increases in vadose-zone water as well as salt leaching to deeper layers. The onset of these changes appeared to be quite rapid and consistent based on soil cores and geophysical images of the deep vadose zone. Lower annual water use by agricultural crops and sandy unconsolidated soils that readily facilitated deep soil

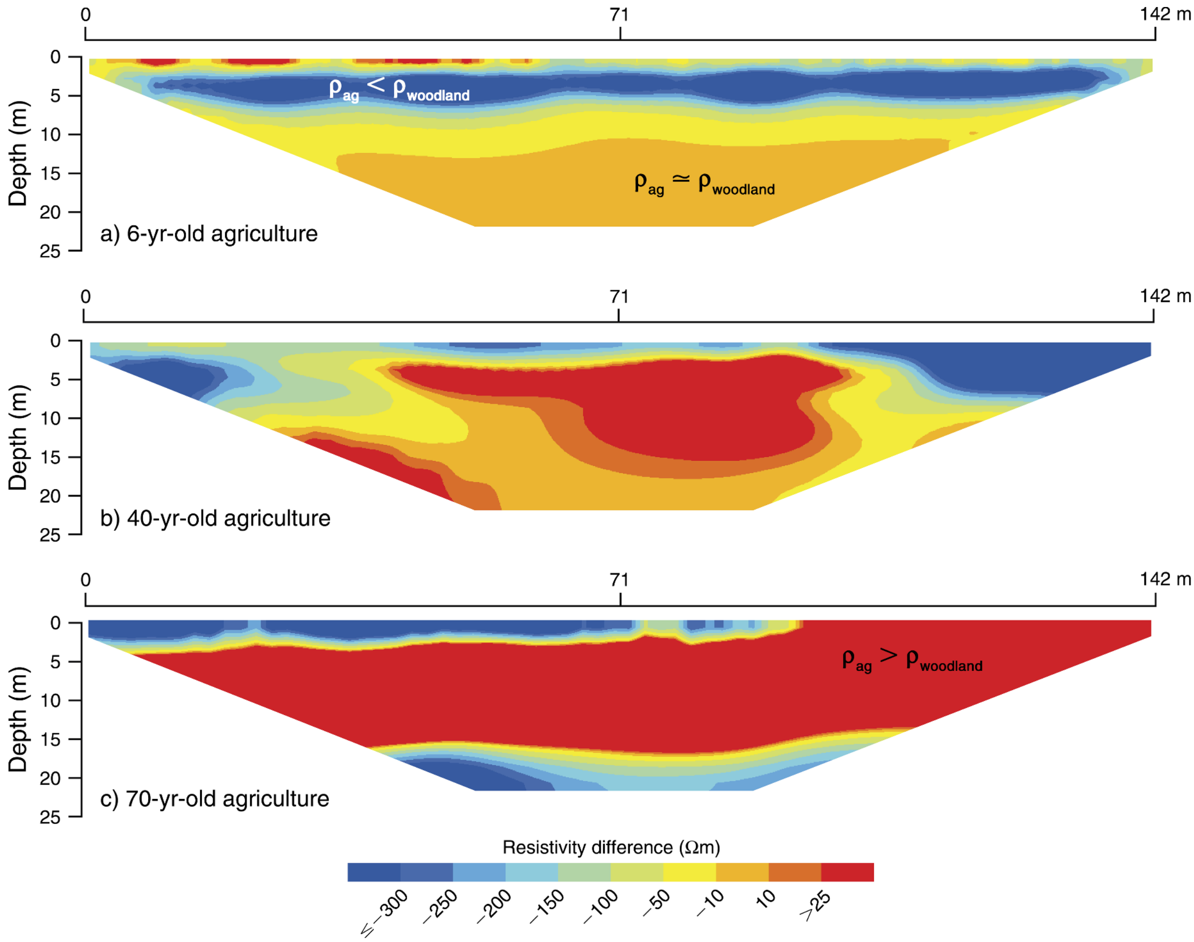

FIG. 6. Soil electrical-resistivity differences ($\rho_{ag} - \rho_{wood}$) between parallel geoelectrical transects in DF1–AG1 and DF3–AG6 paired sites and nearby sections from 150–292 m and 400–542 m at DF5–AG8 sites (Fig. 2). Positive differences indicate areas where soil electrical conductivity has decreased in agricultural soils relative to the conductivity of the woodland soils. All data sets were corrected for temperature effects prior to differencing (Sen and Goode 1992). (a) Uniform and layered resistivity difference between the 6-yr-old agriculture–woodland pair is driven by a ~31% (0–7 m) increase in soil moisture in the agriculture plot. At greater depths (> ~7 m) resistivity differences are negligible, indicating similar salt-water characteristics. (b) At the 40-yr-old crop field (AG 6) vadose-zone transformations have propagated deeper, with positive resistivity difference (center) revealing the effect of salt leaching in agriculture on soil electrical conductivity despite a substantial increase in soil moisture (53% in 0–9 m depth). (c) Salt leaching has progressed even further at the 70-yr-old site based on the pervasive soil resistivity increase in the ~5–20 m depth range. Resistivity decrease below ~20 m depth there could be suggestive of increasing water contents and salt concentrations above the water table located ~35–40 m below the surface. At the near surface, woodland soils tend to exhibit higher soil resistivities and resistivity variations compared to agricultural soils. This result is related to tree distribution along the measured transects and is a consequence of potentially very low soil moisture contents in the vicinity of trees. The negative resistivity differences toward the edges in panel (b) and near the surface in panel (c) are the results of such higher and more variable woodland soil resistivities.

water drainage appeared to foster these observed changes in the vadose zone of this system.

The woodlands investigated in this study (Espinal province, Argentina; see Fig. 1) exhibited groundwater recharge rates (<1mm/yr) typical of semiarid natural ecosystems, but changed substantially following land-use conversions (Noy-Meir 1973, Cook et al. 1989, Scanlon et al. 2009). According to our recharge estimates, each hectare of woodlands converted to agriculture may contribute ~300 m^3 of additional water into the region's aquifers every year. However, the actual impact of this increased recharge may not become evident immediately following the conversion to agriculture, but will likely be lagged by ~40–100 years depending on the thickness of the vadose zone, which ranged from ~35 to 80 m at our study sites (Eq. 5). Following the above lag period, the increased recharge at the water table will cause the groundwater table to rise, which we estimate to be a conservative ~77 mm/yr. This estimated rate however is a lower bound since porosity is likely to decline and moisture to rise in the deeper vadose zone below the depth at which their direct measurements were made (Eq. 6).

TABLE 2. Soil sampling depths, vadose-zone salt storage, and groundwater-recharge estimates at the eight agricultural (AG) and five paired dry woodland (DF) study sites.

Site	Borehole depth (m)	Salt storage (kg/ha)		Recharge (mm/yr)†	
		$Cl^- + SO_4^{-2}$	ERI‡§	CMB	CFD
AG1 (6 yr)	9	28 290§	49 000	- -	12
DF1	7	23 033	38 000	0.1	- -
AG2 (15 yr)	6	100 715	143 000	14.1	≥12.5
DF2	10	87 109§	128 000	0.1	- -
AG3 (30 yr)	10	385	- -	22.7	≥17.5
AG4 (30 yr)	9.5	316	- -	34	≥17
AG5 (∼40 yr)	9.5	1110	- -	23.4	≥19.7
AG6 (∼40 yr)	9.5	567	29 000	29.5	≥16 (∼32)
DF3	9	55 966	78 000	0.07	- -
AG8 (70 yr)	9	1018§	21 000	45.1	9.2 (≥39)
DF5	7	57 904	103 000	0.08	–
AG7 (∼90 yr)	10	47 535	- -	23.3	≥9.7 (≥24)
DF4	10	57 760	- -	0.06	- -

Note: An "- -" entry means that no data were computed.

† Recharge estimates are based on chloride mass balance (CMB) and tracer front displacement (TFD) methods. TFD represents a potential minimum-recharge estimate, particularly at older agriculture sites where a Cl^- peak was not readily visible in soil profiles. A conservative 10-m displacement was assumed in those situations. TFD estimates in parentheses for AG6, AG7, and AG8 are based on tracer displacement distances estimated with geoelectrical profiles (13m, >20m, and >35m, for AG5, AG7, and AG8, respectively).

‡ ERI is the electrical resistivity-based, spatially averaged salt concentration along a ∼100-m section of the resistivity transect for the same cored depth (values to the nearest thousand kg/ha).

§ Computed budgets are for the same depth as the paired site.

Increased soil moisture and deep drainage following the cultivation of woodlands is displacing salts from the vadose zone. This process is clearly evident from our soil cores and geophysical images (Figs. 2 and 4). The absence of the shallow conductive horizon in the ∼540-m-wide AG8 cropland (Fig. 2), for example, shows that salt leaching is pervasive within agricultural land uses. We estimate that salt leaching could account for a gradual one-time release of >100 Mg/ha of chloride and sulfate from soils to the regional aquifers. This estimate would be larger if total salt concentrations based on ERI (electrical-resistivity imaging) were considered. Based on an approximate salt front displacement of ∼36 cm/yr (estimated at AG3; AG4, ≥10 m; AG6, ≥13 m; and AG 8, ≥30 m), the leaching Cl^- and SO_4^{-2} salts alone can potentially raise the region's groundwater salinity to over 4000 mg/L from the present ∼500–1000 mg/L, as described by Eq. 7. This estimate, which assumes no secondary transport of salts from the saturated zone via lateral groundwater flow, is much larger than our geophysics-based estimate of groundwater salinity (∼1600 mg/L) at AG8–DF5, which showed evidence of vadose salts already reaching the saturated zone. However, the resolution decay and other limitations with geophysical methods reduce the reliability of ERI-derived salinities at greater depths. Nonetheless, collective evidence so far suggests that deep soil water drainage and salt leaching from the vadose zone is already occurring and will continue to occur in the region because of land-use conversions.

Pervasive increases in soil salinity or groundwater levels have not yet been documented in central Argentina. This may be because large-scale deforestation for agriculture is a relatively recent phenomenon in the region (Mapa forestal provincial de San Luis 2002). The absence of systematic groundwater/surface water monitoring and reporting may also be a reason. Our observations nonetheless suggest that such hydrological impacts are likely to emerge in the near future. Furthermore, the unfolding land-use-change scenario in central Argentina bears many similarities to the situation in Australia, where thousands of hectares of agricultural lands have been lost in the recent decades, attributable to salinization and waterlogging (Pierce et al. 1993, George et al. 1997). Widespread deforestation there for dryland agriculture over a century ago has shifted the water balance in favor of higher groundwater recharge (George et al. 1997). As a result, rising water tables have waterlogged significant areas of the landscape. Naturally saline groundwater that reached the surface was further concentrated by evaporation, making the salinity levels toxic for many plants. Even though the salinity implications of woodland removal in

TABLE 3. Electrical-resistivity imaging (ERI)-based salt-concentration estimates below the augured depths at woodland sites (Eqs. 1 and 2).

Site	Depth interval (m)	Salt concentration† (mg/kg)
DF1	7–20	2900–1400
DF2	10–20	3200–1500
DF3	9–20	1600–800
DF5	7–20	800–400

† Estimated concentration range is based on 11% and 23% soil-water saturations from woodland and agriculture soil profiles, respectively.

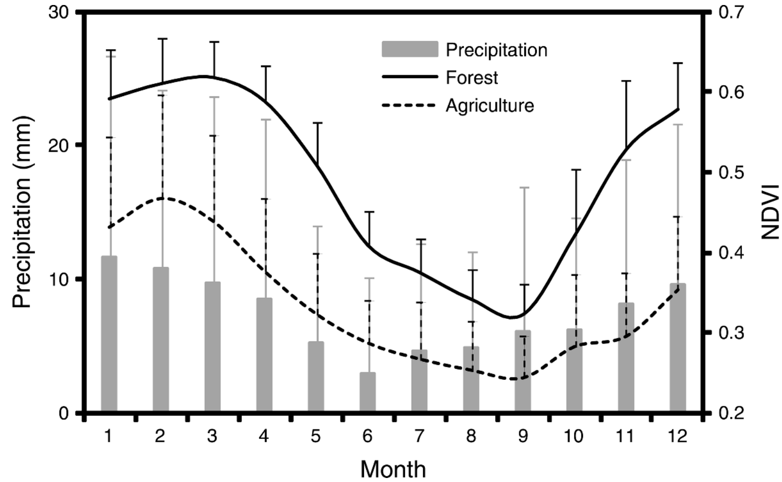

FIG. 7. Mean monthly precipitation (from 40 years of daily data) and 16-day MODIS (moderate-resolution imaging spectroradiometer) normalized-difference vegetation index (NDVI) from January 2004 to December 2007 at four dry woodland and four agriculture plots in the vicinity of the main study sites. Replacing dry woodlands with agriculture has led to lower and more-variable leaf cover in the region, potentially translating to reduced annual water use by agricultural crops. Error bars show the standard deviation of the data.

Australia were recognized almost a century ago (Wood 1924), salinization and waterlogging have continued despite significant efforts to restore the original water balance (Clarke et al. 2002). Through social costs, financial losses from lost agriculture productivity, depreciated land value, and mitigation costs, these changes continue to cost the Australian economy billions of dollars each year (Short and McConnell 2000). Whether a similar scenario will unfold in Argentina is uncertain at the moment, but our findings suggest that monitoring and caution is warranted, especially because of the surface and groundwater connectivity between the semiarid regions where deforestation is expanding and the lowland plains of Pampas where agriculture is already well established. A few surface-water samples collected in 2009 along Rio Quinto, the main outlet for water discharges from the study region to the lowland Pampas, showed a >2000 mg/L increase in dissolved solids along a ~100-km stretch downstream. These and other observations suggest further salt and water loading may seriously affect ecosystem sustainability in the lowlands where groundwater discharges are mainly plant mediated (Archibald et al. 2006, Nosetto et al. 2009, Portela et al. 2009).

Restoring the water balance of forests under agricultural fields is a necessary first step to mitigate current hydrological shifts. Woodland conversions to cultivation can substantially reduce the vegetation cover and decrease the evapotranspirative flux significantly (Fig. 7). In wetter areas of the pampas, soybean cultivations have been shown to reduce the evapotranspirative flux by nearly one third from ~1100 mm/yr in plantations to ~700mm/yr, demonstrating also that woodlands are large water users (Nosetto et al. 2011). Evapotranspirative reductions of the magnitude identified by Nosetto et al. (2011) are perhaps less likely in semiarid areas, but are sufficiently large to cause deep soil water drainage, as we have observed in this study. Management strategies that try to optimize plant water consumption in agricultural areas, including double cropping (two crops per year), deep-rooted perennial species (e.g., alfalfa), or the inclusion of tree plantations in the landscape are some agronomic options that can help to restore the water balance and maximize biomass production at the same time (Dunin et al. 1999, George et al. 1999, Boletta et al. 2006).

Our findings on land-use conversions in central Argentina highlight the water-resource implications of agriculture in a semiarid setting. With an anticipated rise in demand for arable lands for crops and other uses globally, the current wave of deforestation in central Argentina is unlikely to cease in the foreseeable future. As a result, broader hydrological impacts in the region can be expected. Drawing important lessons from costly experiences in Australia and elsewhere, stakeholders need to identify feasible solutions to impede land clearing and mitigate the hydrological impacts of vast agricultural operations in the region. Concurrently, the scientific community should seek additional data and increase efforts for a more accurate accounting of the salinity and hydrological changes that result from land-use transformations here and elsewhere around the world.

ACKNOWLEDGMENTS

We thank Silvina Ballesteros, Ricardo Paez, Esteban Crespo, Marcelo Nosetto, Javier Houspanoussian, Germán Baldi, Kaiguang Zhao, Nancy Scott, and Chinling Chen for their assistance with data collection, sample preparations, and analyses. We acknowledge and greatly appreciate the support of local farmers for allowing us to carry out our research on their properties. This work was supported by the National Science Foundation (DEB number 0717191) and the Inter-American Institute for Global Change Research (IAI, CRN II 2031, supported with funds from GEO-0452325 by the NSF). C. S. Santoni was supported by CONICET (Beca Doctoral Interna). We thank an anonymous reviewer and members of the Jackson laboratory for comments on earlier drafts of the manuscript.

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