Semiarid Unsaturated Zone Chloride Profiles: Archives of Past Land-Use Change Impacts on Water Resources in the Southern High Plains, USA

In press

Water Resources Research

Bridget R. Scanlon, Robert C. Reedy, J. Andrew Tachovsky

Bureau of Economic Geology, Jackson School of Geosciences,

The University of Texas at Austin, Texas, US

1.0 ABSTRACT

Unsaturated-zone (UZ) chloride profiles in semiarid regions provide a decadal to century scale record of past environmental changes, similar to climate-change records provided by tree rings and ice cores. Impacts of conversions from natural ecosystems to rainfed agriculture on water resources are recorded in chloride profiles in semiarid regions, as typified by the southern High Plains (SHP), Texas, southwestern US. Large chloride accumulations beneath natural grassland and shrubland ecosystems (3 profiles) reflect evapotranspirative enrichment of atmospherically derived chloride during the Holocene, indicating no recharge in interdrainage areas. Conversion to rainfed agriculture is recorded by downward displacement (10 profiles) or complete flushing (10 profiles) of chloride bulges, indicating increased recharge. Increased recharge associated with cultivation (median 24 mm/yr, 5% of precipitation, 19 profiles) was quantified using chloride mass balance calculations. The timing of land-use change was estimated using chloride data and results (43 to 89 yr) are consistent with aerial-photo records and land-owner surveys. New equilibrium volumetric recharge rates in the SHP (0.63 km^3/yr) will require decades to establish and represent 1 to 8 times recharge rates for baseline pre-cultivated conditions that is focused beneath ephemeral lake or playa drainages (0.08 to 0.83 km^3/yr). These chloride profiles generally represent decadal-scale monitoring of subsurface response to land-use change.

2.0 INTRODUCTION

Understanding impacts of past land-use changes on water flow and solute transport is critical for managing current and future water resources. The most extensive changes in land use have occurred as a result of expansion of rainfed cropland by 460% and pastureland by 560% globally during the past 300 yr, mostly at the expense of forests and to a lesser extent grasslands (Klein Goldewijk, 2001). Currently, rainfed agriculture represents 80% of cropland and accounts for 60% of global food production (Rockstrom and Falkenmark, 2000). Impacts of these land-use changes on water resources need to be understood to assess potential effects of future land-use changes, such as expansion of rainfed agriculture for production of food and bioenergy (Berndes, 2002; Bruinsma, 2003), land retirement programs (Hellerstein, 2006), and afforestation related to carbon sequestration in many regions (Scanlon et al., 2007).

How can we determine the direction and quantify the magnitude of impacts of land-use changes on subsurface water resources? The most direct evidence of impacts of land-use change on subsurface flow is provided by changes in groundwater levels (Favreau et al., 2002; Scanlon et al., 2005). However, availability of long-term groundwater-level monitoring records is extremely limited, much more than surface-water-monitoring records. In addition, groundwater levels are often dominated by effects of pumping that can mask any impact of land-use changes. Unsaturated zone monitoring of water content and matric potential can provide useful information on effects of land-use changes. Such records are generally limited to recent times; however, monitoring current conditions under different land-use settings and substituting space for time can provide information on potential impacts of land-use change (Finch, 2000). Unsaturated zone chloride profiles in semiarid regions provide an archive of past environmental changes, including climate variability and/or land-use change (Jolly et al., 1989; Walker et al., 1991; Cook et al., 1992). Chloride profiles beneath different land-use settings can provide information on impacts of land-use change by substituting space for time. In many cases, individual profiles provide an historical record of land-use change.

The purpose of this study was to evaluate the use of chloride profiles in semiarid regions as archives of impacts of past land-use changes on water resources. The southern High Plains (Texas) was used as a case study, and land use was converted from natural grasslands and shrublands to rainfed agriculture, mostly in the early to mid-1900s. Previous studies in this region have concentrated primarily on focused recharge beneath playas (endorheic ponds) in natural ecosystems (Wood and Sanford, 1995; Scanlon and Goldsmith, 1997) and recharge in areas of irrigated agricultural ecosystems (Sophocleous, 2005; McMahon et al., 2006). Reconnaissance studies evaluating impacts of rainfed agriculture in this region showed that conversion of natural ecosystems to cropland increased recharge from 0 to a median value of 24 mm/yr (5% of mean annual precipitation), based on regional groundwater-level trends and limited (4 boreholes) UZ chloride profiles in the shallow subsurface (2.9 to 5.0 m) in a 3,400 km^2 area (Fig. 1) (Scanlon et al., 2005). This study represents an expansion of the original work that includes a greater network of boreholes (19 additional boreholes) extending over a much larger area (~ ⅔ of SHP, 50,000 km^2), primarily in the northwest and southeast, greater depth intervals (2.9 to 11.3 m), measurements of chloride to calculate changes in groundwater recharge associated with land-use change and to estimate timing of land-use change, measurement and monitoring of matric potential head to estimate flow directions, and aerial-photo analyses to document the beginning of cultivation. Results of this study clearly demonstrate the use of UZ chloride profiles for quantifying the timing of land-use change and quantifying the associated increase in water fluxes that is very important for water resources in semiarid regions.

3.0 MATERIALS AND METHODS

3.1 Site Characteristics and History

The Southern High Plains (SHP) (75,500 km^2) is located in Texas and New Mexico (Fig. 1). The topographically flat area is internally drained by 16,400 playas. Previous studies showed that the primary source of recharge in the SHP is ephemeral lakes or playas (Wood and Sanford, 1995; Scanlon and Goldsmith, 1997). The natural ecosystem consisted of short grass prairie grazed by bison (Fahlquist, 2003), and agricultural development began in the late 1800s and early 1900s. Land use includes natural grasslands (33%) and shrublands (12%) used for grazing, rainfed agriculture (42%) dominated by cotton production, irrigated agriculture (11%), and other (2%) (1992 National Land Cover Data, Vogelmann et al., 2001; irrigated areas, Qi et al., 2002). Thickness of the UZ ranges from 0 to 135 m in the SHP, with a median value of 36 m. Median thickness of the UZ beneath rainfed agriculture is 30 m. Clay content in surface soils (upper 1.5 to 2 m) ranges from 1 to 68% (mean 28%) based on Soil Survey Geographic (SSURGO) database (USDA, 1995). Mean clay content generally decreases from about 36% in the northern part of the SHP to about 23% in the southern part.

Long-term mean annual precipitation ranges from 376 mm in the south (Midland, 1931–2005) to 501 mm in the north (Amarillo, 1947–2005) (Fig. 2). Most precipitation (76%) occurs from May through October, resulting from convective thunderstorms. There is no systematic variation in precipitation over time (Fig. 3). The dominant crop in the region is upland cotton (mean 58% of cultivated area; range 45 to 69%, 1971–2005) (USDA, 1995). Cotton in the region is generally grown as a continuous monoculture in the summer with no crop in the winter (fallow), although it is occasionally rotated with either grain sorghum (mean 18% of cultivated area, range 6 to 43%) or winter wheat (mean 18%, range 2 to 30%), which are the remaining primary regional crops. Cotton is planted in early June and harvested in November, which generally corresponds to the period of highest precipitation.

3.2 Chemical and Physical Measurements

The primary approach used to evaluate impacts of conversion of natural to agricultural ecosystems is UZ chloride profiles. Soil samples from these profiles were also analyzed for matric potential to determine direction of water movement. Twenty boreholes were drilled and sampled for chloride, bromide, and matric potential in areas of rainfed agriculture. Three additional boreholes were drilled in natural grassland and shrubland ecosystems to provide baseline data for comparison with profiles in rainfed agriculture. Matric potentials were monitored in natural grassland and shrubland sites (two) and at a rainfed agricultural site (Fig. 1).

Continuous soil cores were obtained using a direct-push drill rig (Model 6620DT, Geoprobe, Salina, KS). Boreholes in cropland areas were located a minimum of 20 m from field edges to ensure representative samples. Borehole depths ranged 2.9 to 11.3 m. Core samples were collected over 1.2 m depth intervals in plastic sample sleeves, and the recovered core length was cut in half, capped, and cold-stored in the field. The cores were sectioned in the laboratory for analyses of soil water content, chloride, bromide, and matric potential measurements at varying depth intervals from 0.3 to 1.5 m.

Chemical parameters analyzed included chloride and bromide in water leached from 416 UZ soil samples. Soils were air dried initially. Approximately 40 mL of double deionized water was added to about 25 g of soil. The mixture was placed in a reciprocal shaker for 4 hr, centrifuged at 7,000 rpm for 20 minutes, and the supernatant was filtered (0.2 μm). Soil samples were then oven dried at 105°C for 48 hr to determine gravimetric water content. Water-extractable chloride and bromide concentrations were measured using ion chromatography. The mean difference between replicate chloride measurements run on sample splits (31 samples) is 9% (standard deviation, 8%). Water-extractable concentrations were converted to concentrations in soil pore water by dividing by gravimetric water content and multiplying by water density. Soil texture analyses were conducted using sieve and hydrometer methods.

Natural ecosystems in semiarid regions are generally characterized by large chloride accumulations or bulges, which result from little or no recharge (Allison et al., 1990; Scanlon et al., 2005). Increased recharge associated with conversion of natural to agricultural ecosystems results in downward displacement of the chloride bulge. The term *deep drainage* refers to water fluxes below the root zone that have not yet reached the water table, whereas the term *recharge* is used once deep drainage has reached the water table. Drainage beneath cleared areas has been estimated using the chloride front displacement (CFD) method, where the chloride front depth represents the depth at which chloride concentrations increase sharply from low chloride in the flushed zone, representing high recharge beneath cleared areas, to high chloride at depth, representing low recharge beneath natural ecosystems (Walker et al., 1991). Drainage rates (D) are estimated by multiplying the apparent displacement velocity of the chloride front (v_{cf}) by the average volumetric water content in the flushed zone ($\overline{\theta}_w$; w, wet) (Walker et al., 1991):

$$D = v_{cf}\overline{\theta}_w = \frac{(z_2 - z_1)}{(t_2 - t_1)}\overline{\theta}_w \qquad (1)$$

where the depth of the chloride front under native vegetation is z_1 and under cultivation is z_2 and the time interval (t_2 - t_1) represents the time since cultivation began. Information on timing of the beginning of cultivation is required, which is sometimes not available from landowners or predates the earliest aerial photos. The CFD method provides only a lower bound on recharge rates for profiles where chloride is completely flushed.

Drainage or recharge rates can also be estimated using the chloride mass balance (CMB) approach (Allison and Hughes 1983), in which chloride input (precipitation (P) times chloride concentration in precipitation and dry fallout, C_P) balances chloride output (drainage rate, *D*, or recharge rate, *R* times chloride concentration in UZ pore water, C_{uz},):

$$PC_p = DC_{uz} \; or \; RC_{uz}, \quad D = R = \frac{P \times C_p}{C_{uz}} \qquad (2)$$

Studies in Australia have shown that the CMB approach, applied to the flushed portion of the chloride profile, results in recharge rates that are much less than those calculated using the CFD method because of high residual chloride in the UZ (Walker et al., 1991). However, good correspondence was found between CFD and CMB approaches beneath irrigated sites in Nevada (USA) (Stonestrom et al., 2003), suggesting that either approach may be applicable in some regions. The advantage of the CMB approach is that information on timing of land-use change is not required for drainage calculations and recharge rates can be calculated for profiles that are completely flushed.

Increases in drainage caused by cultivation are not immediately transmitted to the water table. Time is required to satisfy the water deficit in the UZ. An upper bound on the time lag can be provided by the time required for the chloride front to reach the depth of the water table (WT_d), given the velocity of the chloride front (v_{cf}) or the drainage rate from the CMB approach (D):

$$v_{cf} = D / \bar{\theta}_w \qquad t_L = WT_d / v_{cf} \tag{3}$$

where $\bar{\theta}_w$ is the average volumetric water content under wet conditions (*w*) in the flushed portion of the profile. However, recharge occurs when the wetting front reaches the water table. The wetting front is generally deeper than the chloride front by an amount equal to the displaced water in the leached zone (Jolly et al., 1989). The velocity of the wetting front is equal to the water-table depth divided by the time lag for the wetting front to reach the water table and is greater than that of the chloride front. The velocity of the wetting front can also be calculated from the drainage rate (D) estimated using CFD or CMB and the difference in mean volumetric water content between post-cultivation ($\bar{\theta}_w$; w, wet, flushed portion of profile) and pre-cultivation ($\bar{\theta}_d$; d, dry) below the wetting front in profiles in cultivated areas or in natural profiles:

$$v_{wf} = WT_d / t_L = D / (\bar{\theta}_w - \bar{\theta}_d) \tag{4}$$

The ratio of the velocities of the wetting front and chloride front can be determined by rearranging equations 3 and 4:

$$\frac{v_{wf}}{v_{cf}} = \frac{\bar{\theta}_w}{\bar{\theta}_w - \bar{\theta}_d} \tag{5}$$

Velocities of the wetting front and chloride front are equal when the initial volumetric water content in the profile is zero ($\bar{\theta}_d$ = 0), and the ratio of the two velocities increases as $\bar{\theta}_d$ increases.

Actual time lags between drainage at a particular depth and recharge at the water table can be calculated by using v_{wf} instead of v_{cf} in equation 3. This approach provides time lags for individual profiles. However, recharge is generally log-normally distributed (Cook et al., 1989), and regional recharge rates (R) can be described using the spatial distribution of deep drainage as follows:

$$R(t) = \int_{WT_d(\bar{\theta}_w - \bar{\theta}_d)/t}^{\infty} y f(y) dy \tag{6}$$

where the lower limit of integration is the drainage rate at time t (equation 4) and f(y) is the probability density function of the log-normal distribution of deep drainage:

$$f(y) = \frac{1}{y\sigma\sqrt{2\pi}} \exp[-(\ln y - \mu)^2 / 2\sigma^2] \tag{7}$$

where μ is mean and σ^2 is variance of the log-transformed recharge values (y).

The median chloride concentration in precipitation from 1980 through 2005 is 0.12 mg/L (range: 0.06 to 0.18 mg/L) (Muleshoe National Wildlife Refuge, MNWR; National Atmospheric Deposition program [NADP; http://nadp.sws.uiuc.edu/]) (Fig. 1). Concentrations increase toward the southeast across the SHP. Precipitation collectors in the NADP program open automatically during precipitation and close when precipitation ceases; therefore, they provide data only on wet fallout. Chloride concentrations in precipitation were doubled to account for dry fallout, which is consistent with total chloride fallout based on prebomb $^{36}Cl/Cl$ ratios at Amarillo (Scanlon and Goldsmith, 1997). Uncertainties in dry deposition cannot readily be determined because it is not quantified separately. The time required to accumulate chloride in the UZ was calculated by dividing the cumulative total mass of chloride from the land surface or the base of the root zone to the depth of interest by the chloride input:

$$t = \int_0^z \theta Cl_{uz} dz / (P \times Cl_p) \qquad (8)$$

This equation assumes that chloride input has remained uniform over time, which in some cases extends to thousands of years. This assumption is corroborated by chloride deposition calculated from pre-bomb $^{36}Cl/Cl$ ratios and from correspondence of accumulation times using chloride with those based on radioactive decay of ^{36}Cl at other sites in the southwestern US (Scanlon, 2000). The time required to accumulate chloride in the flushed zone (base of root zone, 1 m, to depth of chloride front) was used to estimate the timing of land-use change. This approach will apply following a change in land use only if there is no residual chloride in the profile (e.g., complete flushing).

Information on potential gradients is important for determining direction of water movement and monitoring of potentials provides information on depth of wetting and evapotranspiration. Soil water moves from regions of high to low total potential (sum of matric, gravitational, and osmotic potentials). Osmotic potential is generally ≤10% of total potential and is usually ignored (Scanlon et al., 2003). Matric potentials were measured in the laboratory on soil samples collected in the field using a chilled-mirror psychrometer in the dry range (≤-8 m; Model WP4T, Decagon Devices, Pullman, WA) and using tensiometers in the wet range (-8 to 0 m) (Model T5, UMS, Munich, Germany). Heat dissipation sensors (Model 229, Campbell Scientific, Inc., Logan, Utah) were installed at a natural site (MNWR; 15 depths between 0.15 and 12 m) and a rainfed site near Lamesa, about 10 m from the edge of the field (12 depths between 0.5 and 10 m) to monitor temporal variations in matric potential (Fig. 1). Heat dissipation sensors were calibrated in the laboratory from 0 m (saturated) to about -30,000 m (air dry ~10% relative humidity), using pressure plate extractors (0 to -50 m) and saturated salt solutions (-300 to -30,000 m). Calibration equations and temperature corrections were based on Flint et al. (2002).

Aerial photographs were examined to determine the beginning of cultivation at borehole locations by visual interpretation. The earliest aerial photos available that depict the borehole locations were produced between 1936 and 1941. Availability of newer photography varied for different areas, with coverage available at different times between the mid-1940's through the early 1970's. The images used were hardcopy photomosaics covering approximately 25 × 25 km areas (scale of 1:64,000), which provided sufficient detail to identify cultivated area outlines, transportation networks, etc., for comparison with current digital orthophotographic images. Landowners also reported the timing of the beginning of cultivation, to the best of their knowledge, for most sites.

4.0 RESULTS AND DISCUSSION

The general impact of natural and agricultural ecosystems on UZ flow can be seen in a plot of mean matric potential head versus mean chloride concentration for sampled profiles (Fig. 4). Profiles in natural ecosystems are characterized by high chloride concentrations (three profiles in this study: mean 477; 1,072; 2,514 mg/L) and low matric potentials (mean -144, -145, -278 m), both of which indicate low rates of water movement and dry conditions (Table 1). Values in natural ecosystems are similar to those found previously in the Southern High Plains (McMahon et al., 2006) and in the Central High Plains (Scanlon and Goldsmith, 1997) (Fig. 4). In contrast, profiles beneath rainfed agriculture have low chloride concentrations (19 profiles; median 8 mg/L; range 2 to 41 mg/L) and high matric potentials (19 profiles: median -9 m; range -1 to -76 m) in the flushed portion of the profiles, which indicate wetter conditions. One profile was not flushed below the root zone and was omitted (HO5-02; Table 1). Differences in UZ chloride and matric potentials between the two settings are statistically significant ($p < 0.001$). Data from these profiles suggest that conversion of natural to agricultural ecosystems should decrease chloride concentrations and increase matric potentials.

4.1 Natural Ecosystem Profiles

Chloride profiles beneath natural ecosystems in this region are similar to those reported for other natural settings in the High Plains and southwestern US in previous studies (Phillips, 1994; Scanlon et al., 2003; McMahon et al., 2006). Chloride profiles in this study did not extend deeply enough for the entire chloride bulge to be sampled, and maximum chloride concentrations were recorded at the base of the profile (1,331; 2,344; 4,887 mg/L) (Fig. 5, Table 1). The times required to accumulate chloride in the three profiles are 3,000; 4,200; 17,300 yr (equation 8, Table 1). Chloride and bromide concentrations were measured in two of the three profiles and are highly correlated (r^2 = 0.84, 0.99), with average Cl/Br ratios of 158 and 225 below 2 m depth, generally consistent with an atmospheric source of chloride (Davis et al., 1998). High chloride concentrations in these profiles are attributed to evapoconcentration of atmospherically deposited chloride from precipitation and dry fallout and indicate that no recharge has occurred in these settings during this time.

Matric potentials are low in profiles beneath natural ecosystems, indicating that these sediments are dry. Lowest matric potentials were generally measured near the surface (-726, -552 m), with the exception of one of the profiles, which was probably not sampled at a sufficiently shallow depth (Fig. 5). Matric potentials generally increase with depth to maximum values of -245, -193, and -142 m. In some profiles a shallow zone of high matric potentials near the surface probably reflects recent infiltration and redistribution of water. Matric potentials generally approximate total potentials because gravitational potentials are mostly ≤10% of total potential; therefore, the general upward decrease in matric potentials in the shallow subsurface in each profile indicates that water is moving upward and the profiles are drying. These results are similar to those measured in many other natural ecosystem settings throughout the southwestern US, and modeling analyses of these other sites indicate that profiles in these settings have generally been drying since the last glacial period (10,000 to 15,000 yr ago) (Scanlon et al., 2003).

Long-term monitoring of matric potential at the MNWR shows similar trends to the laboratory-measured profiles (Fig. 6). Deep infiltration and redistribution occurred to at least 3 m depth during a period of enhanced precipitation (541 mm) from September through November 2004, which represents 455% of the long-term period average (119 mm) and 129% of the long-term annual average (419 mm) precipitation at MNWR. Elevated precipitation during this period was regionally distributed, with 40% of the SHP area exceeding 300% of normal precipitation (Fig. 7). The soil profile remained wet throughout the winter, and drying began in late March, progressing sequentially from 0.3 to 2 m depth. Increases in matric potential at sequentially greater depths suggest predominantly piston flow. Matric potentials remained relatively stable throughout the winter of 2005–2006. In addition to showing deep infiltration related to excess precipitation, the most important process shown by the matric potential monitoring is the depth of active root activity. Rapid reductions in matric potentials to 2 m depth during spring and summer 2005 and to 3 m depth in spring 2006 indicate that roots to 3 m depth are actively transpiring and dried out the profile to near pre-infiltration conditions. Many studies have previously shown how shrubs can readily remove water at these low matric potentials (Smith et al., 1997). Bare soil evaporation is unlikely to contribute to this deep drying as shown by monitoring of nonvegetated systems in the Mojave Desert, Nevada (Scanlon et al., 2005). These matric-potential monitoring data explain how natural vegetation is extremely opportunistic in removing all infiltrated water and precluding groundwater recharge. Shrub removal and brush control is being conducted in many areas of Texas to increase recharge and runoff; however, these monitoring data indicate that grasses with very low shrub density, as at this monitoring site (Fig. 8), can be very effective in precluding recharge.

4.2 Rainfed Agricultural Profiles

Conversion of natural ecosystems to rainfed agriculture increased deep drainage and/or recharge, flushing chloride partly or completely through the UZ (Figs. 1 and 9). Chloride is completely flushed in 10 out of the 20 sampled profiles beneath rainfed agriculture (Table 1). The sampled depths of the flushed profiles range from 2.9 to 9.2 m (Table 1, Fig. 9a). Flushing in many of these profiles is consistent with groundwater-level rises (average 7 m) recorded over a 3,400-km^2 area of rainfed agriculture in the southeast part of the study area during the past few decades (average 43 yr) (Fig. 1; Scanlon et al., 2005). The time period represented by chloride in the flushed zone below the root zone (~1 m) ranges from 16 to 62 yr (equation 8; Table 1). With the exception of one profile (D03-03), these times are all less than the time when this land was converted to rainfed agriculture based on aerial photographic evidence and/or landowner reports. Higher chloride in the D03-03 profile may reflect some residual chloride and preferential flow.

The remaining 10 profiles are incompletely flushed and record an increase in chloride concentrations at depths ranging from 0.7 to 8.9 m (Table 1, Figs. 1 and 9b, c). Peak chloride concentrations at the base of these profiles are up to 2,103 mg/L (L05-01), which are similar to peak concentrations measured beneath natural ecosystems (1,331 to 4,887 mg/L). Increases in chloride concentrations at depth represent the transition from natural to cultivated ecosystems. The chloride-concentration increases are generally sharp, with gradients ranging from 360 to 1,000 mg/L/m, indicating piston-like displacement of chloride previously accumulated under natural conditions. The time period represented by chloride in the flushed zone below the root zone (~1 m depth) ranges from 44 to 89 yr (median 64 yr) (equation 8; Table 1). Aerial photographic records for different borehole locations indicate that cultivation began before the first images were available (1936–1941) at 5 of the 10 sites and generally corresponds with CMB ages of 64 to 89 yr for the various profiles. An example of the use of aerial photos to show the initiation of cultivation is provided in Fig. 10. Cultivation began from 1945 to 1974 in the remaining five profiles, according to landowner records and bracketed by aerial photo records, which correspond to CMB ages from 43 to 59 yr. Where flushing was restricted to the shallow subsurface (≤1 to 2 m), sampling depth resolution was insufficient to accurately estimate the transition time from natural to cultivated conditions.

Water fluxes (drainage or recharge) calculated for all profiles (completely and incompletely flushed) using the CMB approach for the flushed zone below the root zone (≥1 m) range from 5 to 92 mm/yr (median 24 mm/yr) (Table 1). Although the number of data points is low, the distribution of recharge values is generally log normal, with a mean of 33 mm/yr (standard deviation, 31 mm/yr) (Fig. 11). Recharge rates are generally high in the southeast part of the study area, where groundwater levels have been rising for the last few decades. Spatial variability in recharge rates is not correlated with average clay content in the upper 2 m from SSURGO data (r^2 = 0.04) or from profile texture data (r^2 = 0.01). Local variability in recharge rates is high and cannot readily be explained by variations in soil texture. Dominant soil textures in cropland profiles include loamy sand, sandy loam, and sandy clay loam—similar to soil textures in profiles beneath natural vegetation. Variability in recharge rates may be related to layering of sediment that is not evident from low-resolution textural analyses in profiles.

Profiles that are completely flushed of chloride generally have uniformly high matric potentials throughout (mean -1.0 to -26 m), indicating wet conditions (Fig. 9a, Table 1). Total potential gradients are downward in all flushed profiles, indicating downward water movement. Matric potentials in profiles that are incompletely flushed of chloride are variable. Some incompletely flushed profiles have high matric potentials throughout the profile (G05-02, mean -4.2 m) (Table 1). Others have high matric potentials in the flushed zone and low matric potentials at depth (e.g., B05-01, -13 m in flushed zone, -63 m below flushed zone; B05-02, -6.6 m, -30 m; L05-01, -5.9 m, -56 m) (Fig. 9b, c; Table 1). These low matric potentials are higher than mean matric potentials from similar depth intervals (~3 to 8 m) beneath natural ecosystems (A05-02, -170 m; D06-02, -130 m), suggesting that these matric-potential decreases toward the base of many profiles under cultivated areas may not represent true wetting fronts. Water contents would also be expected to be lower beneath wetting fronts, as seen in Australian sites (Jolly et al., 1989; Walker et al., 1991); however, water content in many profiles in the SHP does not vary systematically below these matric potential decreases (Fig. 9b, c). Theoretically, matric-potential fronts should be deeper than chloride fronts by an amount equal to the displaced water in the chloride-leached portion of the profile (Jolly et al., 1989).

The primary source of uncertainty in calculated water fluxes and ages results from spatial variability (Table 1). Incomplete flushing of chloride would result in underestimation of water fluxes and overestimation of ages in the flushed portion of profiles; however, calculated ages are consistent with the timing of land-use changes from aerial photo data and landowner surveys, suggesting that there is little or no residual chloride in the flushed portion of profiles. Uncertainty in rooting depth (assumed 1 m) has minimal impact on calculated timing of land-use change because soil-water chloride concentrations near the root zone are negligible. Variations in chloride deposition are linearly related to calculated fluxes and ages. Higher deposition rates would increase calculated water fluxes and decrease ages, which would not be consistent with other records of the timing of land-use change.

Matric potential was monitored beneath rainfed agriculture for a limited time (4/13/06 to 10/25/06). After an initial equilibration period of about 1 week, matric potentials remained stable over time and ranged from about -3 m at the surface to a minimum of -9.6 m (2.5 m depth) and to a maximum of -0.3 m at the base of the profile (10 m depth) (Fig. 12). Monitored matric potentials are similar to values measured in the laboratory on soil samples collected during instrument installation (Fig. 12). The total potential gradient is downward, indicating downward water movement.

4.2.1 Time Lags between Increased Drainage and Groundwater Recharge

Recharge has occurred beneath rainfed agriculture in the southeast part of the study area, as shown by completely flushed chloride profiles and by groundwater-level rises (Fig. 1, Scanlon et al., 2005). Incompletely flushed chloride profiles in other areas indicate that more time will be required before recharge will occur in these areas. Estimated depths to groundwater in incompletely flushed profiles range from 9 to 52 m (Table 1). Linear extrapolation of water velocities calculated using the CMB approach or from the chloride front to the water table (equation 3) indicates that it would take an estimated 120 to 490 yr (median 265 yr) for the chloride front to reach the water table for individual profiles (Table 1). These times represent an upper bound on the time lag for recharge to occur because recharge occurs when the wetting front reaches the water table. Wetting fronts should precede chloride fronts by an amount equal to the amount of water displaced in the leached portion of the profile. The profiles drilled in the SHP may not extend deep enough to show true wetting fronts. It should be possible to use water content information to estimate the depth of the wetting front from the depth of the chloride front; however, data on water contents prior to cultivation are not available in cultivated areas. However, water contents in profiles in natural ecosystems (median 0.13 m^3/m^3) can be used as an estimate for pre-cultivation water content (θ_d) because soil texture is similar in natural and cultivated sites. The velocity of the wetting front for the median drainage/recharge rate of 24 mm/yr is 0.34 m/yr based on a median water content of 0.20 m^3/m^3 (θ_w) beneath the flushed portion of profiles in cultivated areas and a median water content of 0.13 m^3/m^3 (θ_d) under natural vegetation (equation 4). This velocity is 2.9 times greater than that of the chloride front; therefore, time lags for drainage to recharge the water table would be reduced by a factor of about 3 if the wetting front is used rather than the chloride front. These estimates assume that soil texture remains uniform with depth. These relative rates of wetting and chloride front movement are similar to those reported by Jolly et al. (1989) and Walker et al. (1991).

Time lags between drainage and regional recharge to the water table that are based on the log-normal distribution of drainage or recharge with a log mean of 3.18 (33 mm/yr) and a log standard deviation of 0.8 (31 mm/yr) are shown for a water-table depth of 30 m, which is the median water-table depth under rainfed agriculture (Fig. 13). The equilibrium time was approximated by the time required to achieve an e-fold change in recharge ($1-e^{-1}$)) or 0.63 times the equilibrium mean recharge, 0.63 x 33 = 21 mm/yr) to reach the 30-m deep water table and indicated 58 yr, based on solution of equation 7. These time lags are generally consistent with water–level rises recorded in individual wells in the southeastern part of the study area, where increases occurred in the 1950s and 1960s in wells with water tables at 20 m depth and in the 1970s and 1980s in wells with water tables at 30 m depth (Scanlon et al., 2005). Time lags vary linearly with water-table depth. Time lags for the chloride front to reach the regional water table are 2.9 times that of the wetting front (168 yr versus 58 yr for a 30 m water table, Fig. 13), given the relative rates of movement of the two fronts. These results indicate that it will take decades for the impact of land-use change on water resources to be fully realized.

4.3 Mechanisms of Increasing Recharge Related to Cultivation

Mechanisms of increasing recharge (R) related to conversion of natural ecosystems to rainfed agriculture can be evaluated by examining a simple water balance equation:

$$R = P - ET - R_o \qquad (9)$$

Recharge can be increased by increasing precipitation (P) and/or decreasing evapotranspiration (ET) and/or runoff (R_0). Recharge can also be increased as a result of increased runoff and focused recharge beneath playas. Long-term precipitation records from the SHP do not reveal any systematic trend in precipitation over time (Fig. 3). Increases in diffuse, areally distributed recharge based on unsaturated zone data indicate that decreases in ET and/or runoff contribute to increased recharge beneath rainfed agriculture; however, there are no long-term monitoring records of ET or runoff for the SHP to assess the relative importance of these two factors in accounting for increased recharge. One of the primary differences between natural and agricultural ecosystems is perennial natural vegetation versus annual crops which may result in differences in ET between the two systems. Although cotton, the dominant crop in the SHP, is perennial, agriculture forces it to behave as an annual. Cotton cropland is generally fallow from November to early June. Precipitation during this winter fallow period averages 37% of annual precipitation (Fig. 2) and could account for increased recharge in the cropland systems. Stubble is generally left on the cotton fields during the winter, which should further reduce ET and/or runoff accounting for greater increases in recharge beneath cropland. Ploughing may also reduce runoff and partly account for increased recharge. Future studies should monitor ET and R_0 in natural and agricultural ecosystems to assess the importance of these factors in accounting for increased recharge in agricultural areas.

Another related factor important in controlling the difference in recharge between natural and agricultural ecosystems is rooting depth. Perennial vegetation in natural ecosystems has established deeper root systems than those of annual crops that can respond to episodic pulses of water that move deeper in soils. Precipitation in this semiarid region is episodic, with ~ 10% of monthly precipitation accounting for ~ 33% of total precipitation (precipitation Lubbock, 1931-2005). Deep infiltration and redistribution as a result of episodic high precipitation events can readily overwhelm the ability of crop roots to withdraw water. However, natural vegetation is extremely opportunistic, and their roots can access deep water and remove it, as shown by matric potential monitoring after deep infiltration in September 2004 (Fig. 6). Studies in Australia support the importance of episodic recharge, where 10% of monthly precipitation accounts for an estimated 80% of recharge based on modeling analyses (Zhang et al., 1999). Attempts to decrease recharge in these regions to reduce dryland salinity problems using winter cover crops will only be partly successful because such crops will generally not be able to eliminate episodic recharge.

4.4 Implications for Water Resources

Prior to cultivation, the primary source of recharge in the SHP was focused recharge beneath playas. Playa density is variable and ranges from 0.1% throughout the southwest to 2% in the northeast of the SHP, corresponding to an increase in clay content from 22 to 32% (SSURGO data). Estimates of playa-focused recharge from the literature range from 77 mm/yr to 120 mm/yr, based on tritium analyses (Wood and Sanford, 1995; Scanlon and Goldsmith, 1997). Assuming an average value of 100 mm/yr for playa recharge and multiplying by the area covered by playas throughout the SHP (1.4%, equivalent to 1,057 km^2 based on 75,500 km^2 area of SHP) results in an average recharge rate of 1.4 mm/yr and corresponds to a volumetric recharge rate of 0.11 km^3/yr for the SHP. Another baseline estimate of recharge is provided by groundwater chloride concentrations in the Central High Plains (Wood and Sanford, 1995). Large groundwater depths in this region indicate that any impacts of land-use change would not have reached the water table; therefore, the regional groundwater recharge estimate of 11 mm/yr may represent baseline recharge rates primarily from playas and corresponds to a volumetric recharge rate of 0.83 km^3/yr for the SHP. Regional groundwater models have used recharge rates of 1 mm/yr (0.08 km^3/yr) to 3 mm/yr (0.23 km^3/yr) for predevelopment conditions in the SHP (Luckey et al., 1986; Blanford et al., 2003).

Recharge estimates beneath rainfed cropland in the current study would have significant impacts on recharge in the region. Using a conservative estimate of 20 mm/yr would result in a volumetric recharge rate of 0.63 km^3/yr representing the integrated annual flux over the 42% of the SHP regional area (75,500 km^2) that has been converted to rainfed agriculture. Regional groundwater modeling of transient conditions since predevelopment required increased recharge rates of similar magnitude to simulate groundwater level changes (Luckey et al., 1986; Blanford et al., 2003). Therefore, under new equilibrium conditions, increased recharge from rainfed croplands could increase volumetric recharge rates by a factor of 1 to 8 relative to estimates for baseline predevelopment conditions (Table 2). Increased recharge in a 3,400-km^2 area in the southeast part of the study area (Fig. 1) has increased groundwater storage by an average of 1 m (7-m water-table rise × specific yield of 0.15) (Scanlon et al., 2005). This water-storage increase is equivalent to 3.4 km^3 of new water and represents a 56% increase in aquifer saturated thickness in this region. Time lags for increased recharge in other areas of the SHP range from decades to centuries and would require land use to remain constant during that time (Fig. 13). Land retirement programs, such as the Conservation Reserve Program (CRP) have resulted in about 15% of the land area being taken out of production in the study area since 1986 (http://www.nrcs.usda.gov/programs/crp/). Vegetation in CRP land reverts to that typical of natural ecosystems and would eliminate recharge as seen in natural ecosystems. Other agricultural practices, such as establishing winter cover crops to sequester nitrogen and reduce groundwater nitrate contamination, are being evaluated in the SHP and would also decrease recharge beneath croplands.

Rising water tables associated with increased recharge in rainfed agricultural areas, as seen in the southeast part of the study area (Scanlon et al., 2005), may ultimately result in water tables reaching the land surface and associated waterlogging and salinization, as seen in Australia (Allison et al., 1990). However, there is evidence that rising water tables may be partly responsible for increased irrigation that may be attributed in part to increased water availability and reduced energy costs for pumping. A comparison of digital orthophotographic images of Dawson County, located in the southeast part of the study area, where groundwater levels have risen during the past several decades, indicates that cropland irrigated by center-pivot systems expanded by 78% between 1995 (171 km^2) and 2005 (305 km^2) (Fig. 14). During the same period, National Agricultural Statistics Service data indicate that the total irrigated cropland area (including areas not using center-pivot technology) increased by 71% (185 km^2 to 316 km^2; U.S. Dept. of Agriculture, 2006).

4.5 Comparison with Other Rainfed Areas

Deep drainage rates have been estimated for approximately 100 profiles in 25 different study areas in Australia using the chloride front displacement method (Cook et al., 2001). Deep drainage rates for crop/fallow conditions range from ≤0.1 to >50 mm/yr in southern Australia. A representative average value for southern Australia would be about 10 mm/yr (Cook et al., 2001), slightly lower than the estimates for the SHP (median 24 mm/yr). Chloride profiles beneath crops and pastures in Australia differ from those in the SHP in that residual chloride concentrations are high (range 1,000 to 3,000 mg/L); therefore, the CMB approach could not be used to estimate drainage or recharge rates in these regions (Jolly et al., 1989). No explanation is provided for the high residual chloride levels; however, they may be related to preferential flow. Variations in deep drainage beneath cleared areas in Australia are related to differences in precipitation, surface soil texture, and land use (Allison et al., 1990; Kennett-Smith et al., 1994). Clay content in the upper 2 m of the soil profile was used as a surrogate to regionalize deep drainage rates in the 300- to 400-mm/yr precipitation zone beneath cleared areas on the basis of high correlations between deep drainage and clay content (Kennett-Smith et al., 1994). No such relationships were found in the SHP. In Australian sites with shallow water tables (≤10 to 20 m) and sandy soils, increased drainage related to rainfed agriculture has reached the water table, raising groundwater levels; however, estimated time lags for many areas range from decades to centuries (Cook et al., 2001).

Studies of the Continental Terminal aquifer (Eocene–Pliocene; silty sandstone) in southwest Niger indicate that cultivation increased recharge by about an order of magnitude from values of 1 to 5 mm/yr beneath natural savanna ecosystems to 10 to 47 mm/yr beneath millet fields with fallow periods (Leduc et al., 2001, Favreau et al. 2002). Cultivation results in soil crusting, increased runoff, and focused recharge in endorheic ponds (Leduc et al., 2001). Long-term groundwater-level rises (0.1 to 0.47 m/yr; 1963 to 1999) in cleared areas result from increased recharge rates of 10 to 47 mm/yr (porosity = 0.10) (Leduc et al., 2001). Larger groundwater-level rises from 1992 to 1999 (median 0.2 m/yr) resulted from increased average recharge rates of 20 mm/yr. These rises occurred despite severe droughts during the 1970s and 1980s. The recharge rates related to cultivation in Niger are similar to those in the SHP; however, the recharge mechanism in Niger is different from that in the SHP—focused recharge beneath endorheic ponds in Niger versus diffuse, areally distributed recharge in the SHP as shown by UZ matric potential and chloride data. However, some increased recharge may occur beneath playas in the SHP, although it is difficult to document because water levels in playas have not been monitored.

5.0 CONCLUSIONS

Chloride profiles provide an excellent record of the impacts of conversion of natural ecosystems to rainfed agriculture in the southern High Plains. Large chloride reservoirs beneath natural ecosystems indicate no recharge and reflect evapotranspirative concentration of atmospherically derived chloride during the Holocene (10,000 to 15,000 yr). Cultivation increased recharge, displacing chloride reservoirs downward. Complete displacement of preexisting chloride suggests predominantly piston-type flow. The chloride mass balance (CMB) approach was used to estimate recharge in the flushed portion of profiles. Recharge rates (median 24 mm/yr, 19 profiles) are similar to estimates from groundwater-level rises (median 24 mm/yr). The CMB approach was also used to estimate the timing of land-use changes in partly flushed profiles (43 to 89 yr), and these estimates are consistent with records from landowners and aerial-photo analyses. High matric potentials and downward hydraulic gradients associated with partly or fully flushed chloride profiles provide additional data to support recharge beneath croplands. Under new equilibrium conditions, volumetric recharge rates in the SHP (0.63 km^3/yr) represent 1 to 8 times recharge rates for baseline precultivated conditions (0.08 to 0.83 km^3/yr) based on environmental-tracer data and regional groundwater models. Increased recharge has already occurred in the southeast part of the study area; however, decadal time lags were estimated before the full impacts of cultivation on recharge throughout the SHP will be realized. Likely mechanisms responsible for increased recharge associated with cultivation include winter fallowing and inability of shallow-rooted crops to remove episodic deep infiltration characteristic of semiarid regions. Estimated recharge rates beneath rainfed croplands in this region are similar to those found in other regions in Australia (~10 mm/yr) and Africa (~20 mm/yr). These findings underscore the significance of land-use change in controlling water resources and the historical record of system response to land-use change provided by chloride profiles in semiarid regions.

6.0 ACKNOWLEDGMENTS

We would like to acknowledge funding from the Bureau of Reclamation and the Jackson School of Geosciences for this study. Reviews by Peter Cook and Ian Jolly improved the manuscript.

7.0 REFERENCES

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. Journal of Hydrology **119**:1-20.

Allison, G. B., and M. W. Hughes. 1983. The use of natural tracers as indicators of soil-water movement in a temperate semi-arid region. Journal of Hydrology **60**:157-173.

Berndes, G. 2002. Bioenergy and water—the implications of large-scale bioenergy production for water use and supply. Global Environmental Change-Human and Policy Dimensions **12**:253-271.

Blanford, T. N., D. J. Blazer, K. C. Calhoun, A. R. Dutton, T. Naing, R. C. Reedy, and B. R. Scanlon. 2003. Groundwater availability of the southern Ogallala aquifer in Texas and New Mexico: Numerical simulations through 2050. Texas Water Development Board.

Bruinsma, J., editor. 2003. World Agriculture: Towards 2015/2030, an FAO Study. Earthscan Publications.

Cook, P. G., W. M. Edmunds, and C. B. Gaye. 1992. Estimating paleorecharge and paleoclimate from unsaturated zone profiles. Water Resources Research **28**:2721-2731.

Cook, P. G., F. W. Leaney, and I. D. Jolly. 2001. Groundwater recharge in the Mallee Region and salinity implications for the Murray River. CSIRO Land and Water Technical Report 45/01, 133 p.

Cook, P. G., G. R. Walker, and I. D. Jolly. 1989. Spatial variability of groundwater recharge in a semiarid region. Journal of Hydrology **111**:195-212.

Davis, S. N., D. O. Whittemore, and J. Fabryka-Martin. 1998. Uses of chloride/bromide ratios in studies of potable water. Ground Water **36**:338-350.

Fahlquist, L. 2003. Ground-Water Quality of the Southern High Plains Aquifer, Texas and New Mexico, 2001. OFR 03-345, USGS, Austin, TX.

Favreau, G., C. Leduc, C. Marlin, M. Dray, J. D. Taupin, M. Massault, C. L. La Salle, and M. Babic. 2002. Estimate of recharge of a rising water table in semiarid Niger from H-3 and C-14 modeling. Ground Water **40**:144-151.

Finch, J. W. 2000. Modelling the soil moisture deficits developed under grass and deciduous woodland: The implications for water resources. Journal of the Chartered Institution of Water and Environmental Management **14**:371-376.

Flint, A. L., G. S. Campbell, K. M. Ellet, and C. Calissendorff. 2002. Calibration and temperature correction of heat dissipation matric potential sensors. Soil Sci. Am. J. **66**:1439-1445.

Hellerstein, D. 2006. USDA Land retirement programs. Pages 175-183 *in* K. Wiebe and N. Gollehon, editors. Agricultural Resources and Environmental Indicators. USDA.

Jolly, I. D., P. G. Cook, G. B. Allison, and M. W. Hughes. 1989. Simultaneous water and solute movement through unsaturated soil following an increase in recharge. Journal of Hydrology **111**:391-396.

Kennett-Smith, A., P. G. Cook, and G. R. Walker. 1994. Factors affecting groundwater recharge following clearing in the south western Murray Basin. Journal of Hydrology **154**:85-105.

Klein Goldewijk, K. 2001. Estimating global land use change over the past 300 years: The HYDE Database. Global Biogeochemical Cycles **15**:417-433.

Leduc, C., G. Favreau, and P. Schroeter. 2001. Long-term rise in a Sahelian water-table: the continental terminal in south-west Niger. Journal of Hydrology **243**:43-54.

Luckey, R. R., E. D. Gutentag, F. J. Heimes, and J. B. Weeks. 1986. Digital simulation of ground-water flow in the High Plains aquifer in parts of Colorado, Kansas, Nebraska, New Mexico, Oklahoma, South Dakota, Texas, and Wyoming. U.S. Geological Survey Professional Paper 1400-D, 57 p.

McMahon, P. B., K. F. Dennehy, B. W. Bruce, J. K. Bohlke, R. L. Michel, J. J. Gurdak, and D. B. Hurlbut. 2006. Storage and transit time of chemicals in thick unsaturated zones under

rangeland and irrigated cropland, High Plains, United States. Water Resources Research **42**:doi:10.1029/2005WR004417.
Phillips, F. M. 1994. Environmental tracers for water movement in desert soils of the American Southwest. Soil Science Society of America Journal **58**:14-24.
Qi, S. L., A. Konduris, D. W. Litke, and J. Dupree. 2002. Classification of irrigated land using satellite imagery, the High Plains aquifer, nominal data 1992. U.S. Geological Survey Water Resources Investigations Report 02-4236, 31 p.
Rockstrom, J., and M. Falkenmark. 2000. Semiarid crop production from a hydrological perspective: Gap between potential and actual yields. Critical Reviews in Plant Sciences **19**:319-346.
Scanlon, B. R., and R. S. Goldsmith. 1997. Field study of spatial variability in unsaturated flow beneath and adjacent to playas. Water Resources Research **33**:2239-2252.
Scanlon, B. R., D. G. Levitt, K. E. Keese, R. C. Reedy, and M. J. Sully. 2005. Ecological controls on water-cycle response to climate variability in deserts. Proceedings of the National Acadamy of Sciences of the United States of America **102**:6033-6038.
Scanlon, B. R., I. D. Jolly, M. Sophocleous, and L. Zhang. 2006. Global impacts of agricultural land-use changes on water resources: quantity versus quality. Water Resources Research.
Scanlon, B. R., K. Keese, R. C. Reedy, J. Simunek, and B. J. Andraski. 2003. Variations in flow and transport in thick desert vadose zones in response to paleoclimatic forcing (0-90 kyr): field measurements, modeling, and uncertainties. Water Resources Research **39**:1179; doi:1110.1029/2002WR001604.
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 USA. Global Change Biology **11**:1577-1593.
Smith, S. D., R. K. Monson, and J. E. Anderson. 1997. Physiological Ecology of North American Desert Plants. Springer-Verlag, Berlin.
Sophocleous, M. A. 2005. Groundwater recharge and sustainability in the High Plains aquifer in Kansas, USA. Hydrogeology Journal **13**:351-365.
Stonestrom, D., D. E. Prudic, R. J. Laczniak, K. C. Akstin, R. A. Boyd, and K. K. Henkelman. 2003. Estimates of deep percolation beneath native vegetation, irrigated fields, and the Amargosa-River channel, Amargosa Desert, Nye County, Nevada. USGS Open File Rept. 03-104.
U.S. Department of Agriculture 1995. Soil Survey Geographic Data Base, SSURGO. Natural Resour. Cons. Svc., USDA, Misc. Pub. 1527, variably paginated.
U.S. Department of Agriculture. 2003. National Agricultural Statistics Service database. Accessed October 2006 at URL http://www.nass.usda.gov/
Vogelmann, J. E., S. M. Howard, L. Yang, C. R. Larson, B. K. Wylie, and N. van Driel. 2001. Completion of the 1990s National Land Cover Data set for the conterminous United States from Landsat Thematic Mapper data and ancillary data sources. Photogrammetric Engin. and Remote Sensing **67**:650-662.
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. Journal of Hydrology **128**:49-67.
Wood, W. W., and W. E. Sanford. 1995. Chemical and isotopic methods for quantifying ground-water recharge in a regional, semiarid environment. Ground Water **33**:458-468.
Zhang, L., I. H. Hume, M. G. O'Connell, D. C. Mitchell, P. L. Milthorpe, M. Yee, W. R. Dawes, and T. J. Hatton. 1999. Estimating episodic recharge under different crop/pasture rotations in the Mallee region. Part 1. Experiments and model calibration. Agricultural Water Management **42**:219-235.