Impact of deep plowing on groundwater recharge in a semiarid region: Case study, High Plains, Texas

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[1] Groundwater recharge is critical in semiarid regions where aquifers are currently being mined for intensive irrigation. Land use management related to agriculture can be used to control partitioning of water near the land surface and to potentially manage water resources. The purpose of this study was to quantify impacts of deep plowing in rainfed (nonirrigated) agriculture in a semiarid region on groundwater recharge, which had not been previously evaluated. Deep (0.7 m) plowing was conducted once in 1971 to remove low-permeability soil layers (0.15–0.70 m deep) in Pullman clay loam (20,000 km^2 area) in a bench terrace system in the semiarid High Plains in Texas (USA). Deep plowing was followed by conventional tillage. Boreholes were drilled in deep plowed cropland (three boreholes) and also beneath conventionally tilled cropland (four) and natural ecosystems (three) to provide baseline controls. Soil samples were analyzed for water content, chloride concentrations, and matric potentials to quantify subsurface water movement. Bulges of chloride that accumulated beneath natural ecosystems during the past 9,000–14,000 years (Holocene period) provided a marker to quantify time-integrated response of subsurface drainage to hydrologic forcing in deep-plowed cropland. Displacement of chloride bulges to depths of 10.7, 12.3, and 13.7 m beneath deep-plowed cropland indicate minimum drainage rates of 58, 60, and 81 mm/a, whereas drainage beneath conventionally tilled cropland ranged from 0 (nonterraced) to 9 and 14 mm/a (bench terraced). Deep plowing slightly increased crop yield during wet years by reducing waterlogging. If deep plowing were applied to 10% of the Pullman soils, it could increase regional volumetric recharge by 0.1 km^3/a, which is similar to the current regional volumetric recharge. Low-permeability soil layers are widespread in cropland areas globally, and deep plowing could greatly enhance groundwater recharge in such areas, which is critical in semiarid regions where recharge is negligible.

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1. Introduction

[2] Water and food scarcity are significant issues in semiarid regions because of limited supplies and increasing demand from population growth [*Vorosmarty et al.*, 2000]. These issues may become even more critical in the future with projected climate change and related increases in duration and intensity of droughts [*Intergovernmental Panel on Climate Change* (*IPCC*), 2007]. Groundwater resources are a critical issue in semiarid regions where intensive irrigation is practiced. Many of these aquifers have not been effectively recharged for millennia and groundwater abstractions constitute mining of an essentially nonrenewable resource [*Foster and Chilton*, 2003]. *Calder* [2005] emphasized the importance of linking land use management and water resources. Land use change can have a much greater impact than climate change on water resources in semiarid regions. Field studies in Niger showed that changes from natural savannahs to millet crops enhanced water resources through increased recharge by a factor of 10, in spite of decadal droughts since the 1970s [*Favreau et al.*, 2002]. Land use management, mostly associated with agriculture, can have large-scale impacts on partitioning of water among the various components of the water budget:

$$D = P + I + R_{on} - E - T - R_{off} - \Delta S \qquad (1)$$

where D is deep drainage below the root zone, P is precipitation, I is irrigation, R_{on} is run on, E is evaporation, T is transpiration, R_{off} is runoff, and ΔS is change in soil water storage, all terms expressed in depth units (volume/unit area). In nonirrigated areas, I is zero. Deep drainage becomes recharge when it reaches the water table. By changing water partitioning near the land surface, land use changes can impact crop production and/or water resources.

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[3] It is important to acknowledge tradeoffs between green water (precipitation that infiltrates into the soil and is used in crop production) and blue water (water in streams and aquifers available for multiple uses, including human consumption or irrigation) when assessing impacts of land use management on water resources [*Falkenmark and Lannerstad*, 2005]. Programs focused on green water emphasize "more crop per drop" where all water should be routed to crops and partitioning of water into other components of the water cycle, e.g., E, R_{off}, or D, is considered a loss to the system [*Falkenmark and Lannerstad*, 2005; *Molden*, 2007]. Programs concentrating on blue water resources generally consider changes from green water to blue water, e.g., R_{off} or D, as beneficial for water resources, assuming water quality is good and that these transfers do not result from inefficient irrigation. Converting natural ecosystems to rainfed (nonirrigated) agriculture changed green water to blue water through increased recharge by one to 2 orders of magnitude in semiarid regions globally [*Allison et al.*, 1990; *Favreau et al.*, 2002; *Scanlon et al.*, 2007b]. Rainfed agriculture is often referred to as dryland agriculture. The increases in recharge are attributed to reductions in rooting depth from perennial natural vegetation to annual crops, long fallow periods associated with crops that result in reduced ET, and soil crusting and changes in runoff in some regions (e.g., Niger).

[4] Many different agricultural practices, such as crop rotations, tillage practices, and irrigation application methods, can markedly alter the water cycle with implications for green and blue water and associated crop production. Tillage practices used over the past several decades have shifted from conventional inversion type tillage, which buries residues, to reduced tillage and, more recently, no tillage which retains progressively more residue on the soil surface for conservation of soil water through increased infiltration and decreased evaporation [*Unger*, 1992]. This shift in tillage practices has also been driven by increased carbon sequestration associated with no tillage [*Post and Kwon*, 2000; *West and Post*, 2002]. Many agricultural soils feature low-permeability layers formed from mechanical cultivation (hardpans), from clay illuviation (clay pans), or cementation by iron oxides, calcium carbonate, or silica [*Brady and Weil*, 2002]. Examples of these low-permeability layers include duplex soils in Western Australia where sands overlie clays in the B horizon at depths of 0.1 to 0.4 m [*Chittleborough*, 1992] and hardpans in southeastern United States [*Raper et al.*, 2005]. These low-permeability layers result in waterlogging in some areas [*McFarlane and Williamson*, 2002] and increased soil penetration resistance that decreases root growth and access to soil water. With the measured and projected increase of heavy precipitation events for the 21st century in most regions attributed to increases in atmospheric water vapor and global warming [*IPCC*, 2007], the frequency of waterlogging in areas of low-permeability soils is likely to increase. Such waterlogging could possibly be mitigated by deep plowing. Deep plowing has been proposed in some areas with low-permeability layers, such as the North China Plain, with the goal of promoting deep root growth that would be expected to increase crop yield and reduce water losses to deep drainage and ultimately groundwater recharge (reduce green to blue water) [*Zhang et al.*, 2004]. However, deep plowing could also enhance water losses from the shallow root zone, increasing green to blue water resources through recharge. There is very little quantitative information on impacts of deep plowing on the water cycle.

[5] Deep plowing in areas of low-permeability soils could enhance recharge in these regions and ultimately result in more sustainable water development. The more traditional approach for increasing recharge in semiarid regions is through artificial recharge in surface basins, also termed spreading basins [*Bouwer*, 2002]. Such basins are generally recharged using surface water, which is often limited in semiarid regions, except in areas where it is transported from more humid settings, such as from humid northern California to semiarid southern California by the California Aqueduct and from the Colorado River to semiarid central Arizona by the Central Arizona Project [*Izbicki et al.*, 2008].

[6] The objective of this study was to address the following question: What is the impact of deep plowing on groundwater recharge in areas of low-permeability soils in semiarid regions?

[7] The authors are not aware of any previous study on this topic. Unlike artificial recharge structures, deep plowing could be applied over large areas and could significantly increase recharge. The impact of deep plowing on subsurface water fluxes was addressed in this study using soil physics and environmental tracer data from profiles in deep-plowed cropland relative to profiles in conventionally tilled cropland and natural ecosystems, which provide baseline information. Although we recognize that varying water fluxes related to different tillage practices may impact water quality, this topic is outside the scope of this study. Previous research in this study area showed that deep plowing to 0.7 m in 1971 reduced bulk density and penetration resistance and increased infiltration during subsequent tests conducted in 1975 and 2002 [*Baumhardt et al.*, 2008]. Although conventional tillage after deep plowing consolidated and possibly compacted the upper 0.3 m zone, it did not impact the deeper zone. They showed that crop yield increased with deep plowing; however, the yield differences during 1984, 1999, 2003, and 2004 accounted for 84% of the total yield increase. Comparison of precipitation records over these years indicated very high precipitation (60–100 mm) over short times ($\leq$5 days) during the early part of the growing season that resulted in waterlogging in the conventionally tilled areas but drained in the deep-plowed areas. *Baumhardt et al.* [2008] concluded that increased drainage in deep-plowed areas reduced flood injury of crops and that the impacts of deep plowing remained effective until at least 2002–2003. This study builds on that research by quantifying impacts of deep plowing on subsurface water fluxes and discussing implications for water resources.

2. Materials and Methods

2.1. Site Characteristics and History

[8] The field sites are located in the U.S. Department of Agriculture (USDA) Agricultural Research Service (ARS), Bushland, Texas, hereafter referred to as the Bushland site (35°11′N, 102°5′W) (Figure 1). The Agricultural Research Station at Bushland was established in 1936. The site is located on Pullman clay loam which extends along the northern part of the southern High Plains and the central

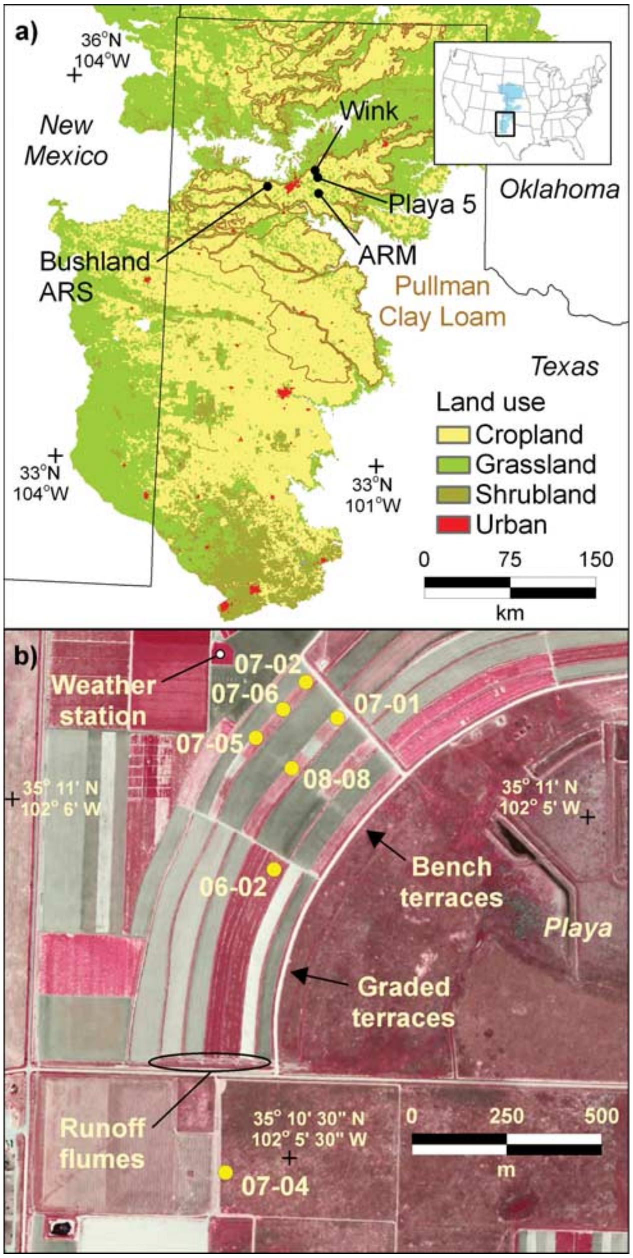

Figure 1. (a) Extent of the Pullman clay loam and hydrologically equivalent soil series (Sherm) in the southern and central High Plains from SSURGO data [*USDA*, 1995]. Inset shows the High Plains in blue with the map area outlined. Boreholes in natural grassland/shrubland settings were drilled adjacent to Wink playa and Playa 5, and a borehole in conventionally tilled cropland was drilled in Armstrong County (ARM). Generalized land use is based on 1992 National Land Cover Data (NLCD) [*Vogelmann et al.*, 2001]. Cropland includes the following NLCD classifications: pasture/hay, row crops, small grains, and fallow. (b) Location of sampled profiles at the Bushland Agricultural Research Station on a 2006 infrared image (National Agriculture Imagery Program) showing the playa and surrounding terraces. Borehole 07-04 is in a natural grassland/shrubland setting; borehole 06-02 is in conventional tillage graded terrace; boreholes 07-01 and 08-08 are in conventional tillage bench terrace; and boreholes 07-02, 07-05, and 07-06 are in deep-plowed bench terrace. The Bushland site is in Potter County.

High Plains (20,000 km^2 area), and represents 30% of the area of the Texas High Plains (Figure 1). Clay content in the upper 2 m throughout the Pullman clay loam ranges from 39 to 46% (area weighted mean 43%) (SSURGO data [*U.S. Department of Agriculture* (*USDA*), 1995]. Soil texture at the Bushland site consists of about 30% clay, 53% silt, 17% sand, and 2% organic matter [*Jones et al.*, 1995]. The Pullman clay loam soil consists of multiple buried soils that have undergone long-term pedogenic processes of downward clay movement (illuviation) to form dense soil layers, enriched with clay, that reduce hydraulic conductivity and limit infiltration and root proliferation. Land use on the Pullman clay loam includes 14% native grasslands/shrublands, 60% rainfed agriculture, 25% irrigated agriculture, and 1% other (Figure 1) (1992 National Land Cover Data (NLCD)) [*Vogelmann et al.*, 2001; *Qi et al.*, 2002]. The topographically flat area is internally drained by ephemeral lakes or playas ($\sim$10,000 playas) that represent $\sim$5% of the land surface. The number of playas in the Pullman clay loam represents about 50% of the total number of playas in the Texas High Plains (94,000 km^2 area), indicating a higher playa density in this region. Previous studies showed that the primary source of recharge under native grassland/shrubland ecosystems in this region is ephemeral lakes or playas with a regional recharge rate of 11 mm/a, based on groundwater chloride data [*Wood and Sanford*, 1995; *Scanlon and Goldsmith*, 1997]. The median thickness of the unsaturated zone is 73 m (44 m (10th percentile) to 110 m (90th percentile)).

[9] Long-term mean annual precipitation at the Bushland site ranges from 240 to 827 mm (mean 472 mm, 68 year record, 1939–2006). Within the 1960–2007 time, 19% of total precipitation resulted from small precipitation events ($\leq$6 mm per individual event) whereas most precipitation (68% of total) resulted from $\leq$25 mm events. Only 9% of total precipitation resulted from large ($\geq$50 mm) events. The dominant crops for rainfed production (no irrigation) include winter wheat (*Triticum aestivum*) and sorghum (*Sorghum bicolor* (L.) Moench) because they are more tolerant of water deficit stress from limited precipitation than corn or soybeans [*Baumhardt and Anderson*, 2006]. In the central High Plains, rainfed winter wheat and sorghum are typically grown in a 3-year rotation producing 2 crops in 3 years (wheat, sorghum, fallow (WSF rotation)) with intervening 11-month fallow periods between crops that promote storage of precipitation as soil water and generally stabilize crop yields [*Jones and Popham*, 1997].

[10] Precipitation in the study area during an average year provides about 25% of the potential ET required for crop production. To offset the resulting crop water use deficit under rainfed conditions, graded terrace structures (Figures 1 and 2a), built in 1949 with uniform 2% slope to reduce erosion, were modified in 1958 to reduce storm water runoff and increase infiltration. About 50% of the graded terraces were modified by constructing a level bench terrace system (Figures 1 and 2b), which features a gently sloping terrace watershed, two times wider than the smaller conservation terrace bench [*Hauser*, 1968]. Runoff from the terrace watersheds was contained within contours of the level terrace benches. The benches range from 24 to 36 m wide by 410 to 460 m long. This conservation bench terrace system was designed to harvest excess rainwater from

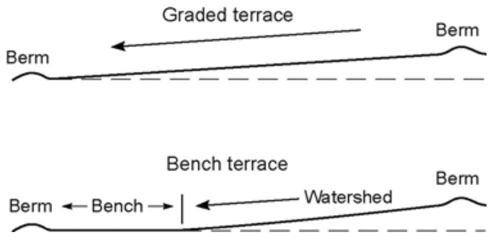

Figure 2. Schematic of graded terrace and bench terrace systems. The graded terrace has a uniform 2% slope between the berms (0.4 m high). The bench terrace system consists of a contributing terrace watershed and level terrace bench. The watershed is two times wider than the bench.

cropped terrace watersheds for use by crops planted on level terrace benches.

[11] In September of 1971, level terrace benches were either deep plowed to 0.7 m depth using a 1.0-m single blade moldboard adjusted to retain topsoil in the upper part of the profile or untreated as a control and subsequently maintained with conventional sweep tillage stubble mulch residue management. Deep plowing was performed to break up a low-permeability, montmorillonitic, silty clay illuvial Bt layer between 0.15 and 0.70 m depth. Conventional tillage involves plowing to 0.1 m depth and is used with stubble mulch residue management. This technique does not invert the surface residue but mixes the upper 0.1 m of soil and was introduced after droughts in the 1930s to reduce erosion. The crop rotation on the terraced watersheds and graded terraces consisted of wheat, sorghum, fallow rotation. The level terrace benches received sufficient runoff from the adjacent terrace watersheds to support various annual summer cropping systems, such as sorghum or sunflowers, with an intervening winter fallow period. An additional profile was drilled in a conventionally tilled, rainfed agricultural site in a nearby region in Armstrong County that was in cropland since the 1920s (Figure 1).

[12] Various components of the water balance were monitored during the 36-year experiment, including daily precipitation from a nearby standard rain gauge, event-based watershed runoff contributions estimated using gauged flumes installed in 1958 on adjacent watersheds of identically managed graded terraces [*Hauser*, 1968] (Figure 1), and soil water content at planting and at harvest time. Soil water contents were measured gravimetrically by oven drying soil samples collected at 0.3 m depth intervals to 1.8 m depth and corrected for plant available soil water (water held between −3 and −150 m matric potential) using the measured retention functions for this soil.

2.2. Chemical and Physical Measurements

[13] The primary approach used to evaluate impacts of deep plowing on subsurface water movement was measurement and analysis of unsaturated zone chloride profiles [*Scanlon et al.*, 2007b]. The chloride profiles provide a time-integrated response of subsurface drainage to hydrologic forcing in deep-plowed cropland. Soil samples from these boreholes were also analyzed for matric potential to determine direction of water movement [*Scanlon et al.*, 2007b]. Ten boreholes were drilled and sampled for chloride, bromide, and matric potential (Table 1 and Figure 1). Although borehole drilling was conducted at different times (Table 1), the timing should only impact the surficial

Table 1. Measured Chloride, Water Content, and Matric Potential Data and Estimated Water Velocities and Fluxes for Drilled Boreholes[a]

Setting Borehole	Date	Flushed Depth (m)	Chloride ≥ 1 m: Mean (mg/L)	Chloride ≥ 1 m: Range (mg/L)	WC Mean (m^3/m^3)	MP Mean (m)	Chloride Front Displacement: Velocity (mm/a)	Chloride Front Displacement: Flux (mm/a)	Chloride Front Displacement: Time to WT	Wetting Front: Vel (mm/a)	Wetting Front: Depth (m)	Wetting Front: Time to WT
						Natural						
07-04	03-07	14.6	672	179–3042	0.16	−213						
Playa 5	10-94	17.6	582	118–2515	0.16	−265						
Wink	09-92	25.9	601	33–4171	0.21	−183						
						Deep Plow (Bench Terrace)						
07-02	03-07	18.0 (12.3)	3.9	2.1–10	0.24	−7.4	250–314	60–75	203	938	34	45
07-05	08-07	21.9 (13.7)	7.4	4.3–12	0.28	−7.3	289–353	81–99	177	825	30	56
07-06	08-07	15.8 (10.7)	6.3	4.5–8.9	0.28	−7.9	206–269	58–75	242	625	23	85
						Conventional Tillage (Bench Terrace)						
07-01	03-07	14.5 (2.8)	7.5	3.1–17	0.24	−31	36–50	9–12	1464			
08-08	08-08	18.3 (3.7)	18	14–26	0.27	−24	54–75	14–20	964			
						Conventional Tillage (Graded Terrace)						
06-02	12-06	13.4 (1.1)	461	88–600	0.18	−79						
						Conventional Tillage (No Terrace)						
ARM	05-06	8.5 (1.0)	881	436–1225	0.21	−101						

[a]Borehole setting includes natural grassland and shrubland at Bushland (07-04) and at adjacent sites (Playa 5 and Wink), deep plow in a bench terrace at Bushland, conventional tillage in a bench terrace and graded terrace at Bushland, and conventional tillage with no terracing at a site in Armstrong County (ARM). Borehole numbering for the Bushland profiles is based on the year of drilling (e.g., 07, 2007) and the drilling sequence. Borehole locations can be found in Figure 1. Date refers to the month and year the boreholes were drilled. Borehole depth; chloride flushed depth (in parentheses); depth-weighted mean, minimum, and maximum chloride concentrations below the root zone (≥1 m); mean volumetric water content (WC) calculated by multiplying gravimetric water content by bulk density of 1,600 kg/m^3; and mean matric potential (MP) below the root zone (1 m depth). Water velocity and drainage flux are based on the chloride front displacement (equation (2)) and time to water table (equation (3)). Water velocity from wetting front is based on equation (4), and estimated wetting front depth is calculated by multiplying the velocity by 36 years and time to water table (WT), similar to equation (3) but with v_{wf} rather than v_{cf}.

samples. Three boreholes were located in native grassland/shrubland ecosystems to provide baseline data for comparison with profiles beneath cropland ecosystems. One of the boreholes in a native grassland/shrubland setting was at the Bushland site (Figure 1) and the other two were in nearby areas adjacent to Wink playa and Playa 5, which were drilled as part of another study [*Scanlon and Goldsmith*, 1997] (Figure 1). One borehole was drilled in a conventional tillage, graded terrace system; two boreholes in a conventional tillage, bench terrace system; and three boreholes in a deep-plowed, bench terrace system (Figure 1). An additional borehole was drilled in conventionally tilled rainfed agriculture in nearby Armstrong County (Figure 1).

[14] Continuous soil cores were obtained using a direct push drill rig (model 6620DT, Geoprobe, Salina, Kansas) in this study. Borehole depths ranged from 8.5 to 25.9 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.

[15] Analyzed chemical parameters included chloride and bromide in water leached from 231 unsaturated zone 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 h, centrifuged at 7,000 rpm for 20 min, and the supernatant was filtered (0.2 μm). Soil samples were then oven dried at 105°C for 48 h to determine gravimetric water content. Water-extractable chloride and bromide concentrations were measured using ion chromatography. Water-extractable concentrations (mg/L) were converted to concentrations in soil pore water (mg Cl-/L of pore water) by multiplying by the extraction ratio (g water/g dry soil) and dividing by gravimetric water content (g/g).

[16] Information on potential gradients is important for determining the direction of subsurface water movement. Soil water moves from regions of high to low total potential (sum of matric, gravitational, and osmotic potentials). Osmotic potential is generally $\leq$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, Washington) and using tensiometers in the wet range (-8 to 0.0 m) (model T5, UMS, Munich, Germany). Total potential gradients were calculated by adding the gravitational potential, which was set to 0 at the predevelopment (preirrigation) water table depth of about 50 m at the Bushland site.

2.3. Drainage/Recharge Calculations

[17] 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.*, 2003]. Increased drainage below the root zone associated with land use change often displaces the chloride bulge downward through the profile. Drainage becomes recharge when it reaches the water table. Drainage can be 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 drainage beneath cropland, to high chloride at depth, representing low drainage beneath natural ecosystems [*Walker et al.*, 1991]. Drainage rates (D) were estimated by multiplying the apparent displacement velocity of the chloride front (v_{cf}) by the average volumetric water content ($\bar{\theta}_w$; w, wet) under wet conditions (w) in the chloride flushed portion of the profile [*Walker et al.*, 1991]:

$$D = v_{cf}\bar{\theta}_w = \frac{(z_2 - z_1)}{(t_2 - t_1)}\bar{\theta}_w \quad (2)$$

where the depth of the chloride front under native vegetation is z_1 and under cropland is z_2 and the time interval ($t_2 - t_1$) represents the time between the beginning of cultivation and soil sample collection. When evaluating the impact of deep plowing, the time interval can represent the time since deep plowing occurred. An upper bound on the time lag (t_L) between increased drainage below the root zone and groundwater recharge at the water table can be estimated from the velocity of the chloride front (v_{cf}):

$$t_L = (WT_d - z_2)/v_{cf} \quad (3)$$

where WT_d is the water table depth. However, recharge occurs when the wetting front (zone where matric potential decreases sharply, becomes more negative) reaches the water table [*Jolly et al.*, 1989]. The wetting front is generally deeper than the chloride front by an amount equal to the displaced water in the chloride flushed zone [*Jolly et al.*, 1989]. The velocity of the wetting front can also be calculated from the drainage rate estimated using the CFD (equation (2)) and the difference in mean volumetric water content between postcultivation ($\bar{\theta}_w$) in the chloride flushed portion of the profile and precultivation ($\bar{\theta}_d$; d, dry) below the wetting front in profiles in cultivated areas or estimated from natural profiles, respectively:

$$v_{wf} = D/\left(\bar{\theta}_w - \bar{\theta}_d\right) \quad (4)$$

The ratio of velocities of the wetting front and chloride front can be determined by rearranging equations (2) through (4):

$$\frac{v_{wf}}{v_{cf}} = \frac{\bar{\theta}_w}{\bar{\theta}_w - \bar{\theta}_d} \quad (5)$$

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 (equation (3)) between drainage at a particular depth and recharge at the water table can be calculated by using v_{wf} (equation (4)) instead of v_{cf} in equation (3) and unsaturated zone thickness between the wetting front and the water table.

3. Results and Discussion

[18] The general impact of natural and agricultural ecosystems on unsaturated flow can be seen in a plot of depth-weighted mean chloride concentration versus mean matric potential head for sampled profiles (Figure 3 and Table 1). Only one profile was drilled beneath native grassland/shrubland at the Bushland site; however, the high

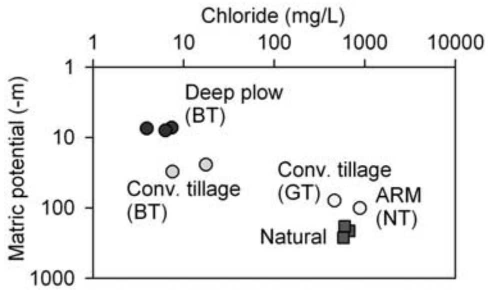

Figure 3. Relationship between depth-weighted mean matric potential and mean soil water chloride concentrations (mg Cl/L of pore water) for profiles in different land use management settings: profiles in natural ecosystems have low matric potentials (more negative) and high chloride concentrations; profiles in conventional tillage with no terracing (Armstrong (ARM) County profile) or in graded terraces (GT) have low matric potential and high chloride concentrations; the profile in conventional tillage bench terrace (BT) has moderate matric potential and low chloride concentrations; and all profiles in deep-plowed bench terraces have high matric potentials and low chloride concentrations.

mean chloride concentration (672 mg/L) and low mean matric potential (−213 m) are similar to those in nearby profiles in natural ecosystems adjacent to playas (Wink and Playa 5) (Figure 1 and Table 1). High chloride concentrations and low matric potentials indicate low rates of water movement and dry conditions.

[19] Profiles in deep-plowed, nonirrigated cropland have low mean chloride concentrations and high mean matric potentials in the upper 10.7, 12.3, and 13.7 m depth zones (Table 1), indicating much higher downward water fluxes than profiles beneath natural ecosystems. The conventionally tilled profiles in the bench terrace system (07-1, 08-8) were only flushed of chloride to depths of 2.8 and 3.7 m depth, which is much shallower than the chloride flushing depths in profiles in deep-plowed cropland. The conventionally tilled profile in the graded terrace (06-2) was not flushed of chloride below the root zone (∼1 m), indicating no deep drainage and is similar to the conventionally tilled profile in nearby Armstrong County. These data indicate that deep plowing has a large effect on partitioning of water near the land surface and markedly increases drainage in deep-plowed profiles relative to profiles in conventionally tilled cropland and natural ecosystems.

3.1. Profiles Beneath the Natural Ecosystem

[20] The chloride profile sampled beneath the natural ecosystem at Bushland is similar to those beneath natural ecosystems in nearby areas (Table 1) and in other regions of the High Plains and southwestern U.S. in previous studies [*Phillips*, 1994; *Scanlon et al.*, 2003; *McMahon et al.*, 2006]. The chloride profiles are bulge shaped with peak concentrations of 2,515, 3,042, and 4,171 mg/L at depths of 1.0, 1.0, and 1.7 m (Figure 4c and Table 1). The time required to accumulate chloride in this profile at the Bushland site (10,600 years) was estimated by dividing the total mass of chloride in the profile by the chloride deposition rate (precipitation 472 mm/a; chloride in precipitation and dry fallout 0.31 mg/L [*Scanlon and Goldsmith*, 1997]). This chloride accumulation time is similar to that for the other two chloride profiles beneath natural ecosystems in nearby areas (Playa 5: 8,500 years, Wink playa: 9,900 years). Concentrations of chloride and bromide are highly correlated (r^2 = 0.98) in the Bushland profile, with a mean Cl/Br ratio of 251, consistent with an atmospheric source of chloride [*Davis et al.*, 1998]. High chloride concentrations in profiles beneath natural ecosystems are attributed to evapoconcentration of atmospherically deposited chloride from precipitation and dry fallout and indicate that no recharge has occurred in this setting during the Holocene.

[21] Matric potentials represent the dominant component of total potential in natural ecosystems and are low at all three natural sites (−265, −213, and −183 m), indicating that these sediments are dry (Table 1 and Figure 4b). The lowest matric potential in the Bushland profile (−530 m) is near the land surface. Matric potentials increase with depth to a maximum value of −141 m at this site. Total potential increases with depth, indicating an upward driving force for water movement. Mean matric potential in this profile is similar to those in nearby profiles in natural settings (Table 1). The results from this region are similar to those in many other natural ecosystems throughout the southwestern U.S. and modeling analysis of these other sites indicates that profiles in these settings have generally been drying since the last glacial period (10,000–15,000 years ago) [*Scanlon et al.*, 2003].

3.2. Profiles Beneath Deep-Plowed and Conventionally Tilled Cropland

[22] Chloride profiles from deep-plowed cropland indicate that deep plowing disrupted the flow-limiting subsurface layer, leading to increased drainage below the root zone. Increased drainage is shown by downward displacement of chloride bulges (flushing) to depths of 10.7, 12.3, and 13.7 m (Figure 4f and Table 1). The depth of the chloride flushed zone is much greater beneath deep-plowed cropland than beneath conventionally tilled cropland (2.8 and 3.7 m) in the bench terrace system (Figure 4i and Table 1). The displacement of the chloride front beneath the conventionally tilled bench terrace from the root zone (1 m) to 2.8 and 3.7 m depth resulted in minimum water velocities of 36 and 54 mm/a (1.8 and 2.7 m divided by 50 years) and corresponding minimum water fluxes of 9 and 14 mm/a. The potential for increasing groundwater recharge through deep plowing of soils containing embedded flow-limiting layers was calculated as the difference in chloride front displacement for the deep-plowed and conventionally tilled bench terraces during the 36-year experiment. The 36-year time extends from 1972 to 2007. Although deep plowing was done in September 1971, it should not impact the water cycle until summer 1972 when seasonal precipitation occurs. The 2007 time is based on the majority of borehole drilling and sampling (Table 1). The lower bounds on water velocities caused by deep plowing were calculated using the chloride front displacement (equation (2)) where z_1 is 3.3 m (average of 2.8 and 3.7 m flushing depths in conventionally tilled profiles), z_2 is maximum flushed depths in the deep-plowed profiles (10.7, 12.3, and 13.7 m), and t_1 is time between deep plowing and sample collection (36 years) (velocities: 206, 250, and 289 mm/a; Table 1). The corresponding water fluxes or drainage rates (58, 60, and

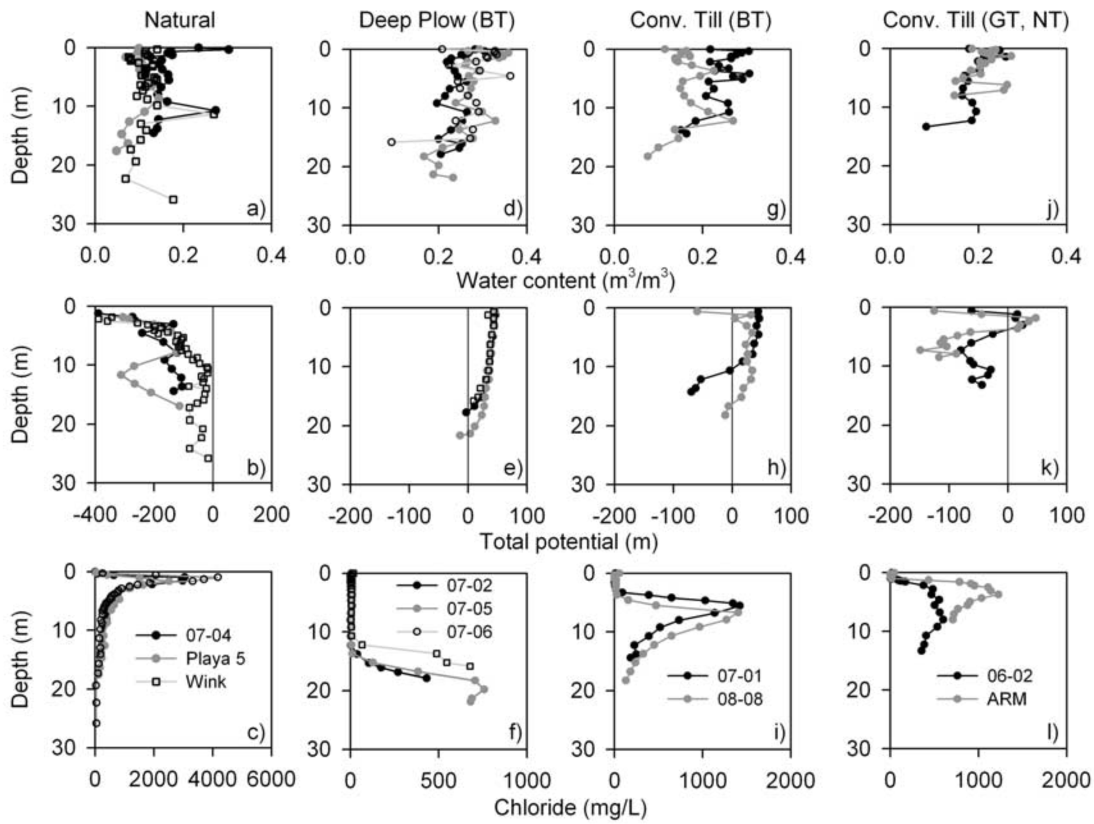

Figure 4. Volumetric water content, total potential (matric plus gravitational (50 m at land surface) potential) and chloride concentrations in soil water (a, b, c) in profiles in natural ecosystems, (d, e, f) in deep-plowed bench terraces (BT), (g, h, i) in conventional tillage bench terraces, and (j, k, l) in conventional tillage graded terraces (GT, 06-02) and no terraces (NT, ARM).

81 mm/a) were calculated by multiplying the velocities by the depth-weighted average water contents in these sections of the profiles (equation (2)) (Table 1). These drainage estimates are considered minimum values because flushing in the conventionally tilled soil occurred between 1949 and 2007, whereas deep plowing was not conducted until 1971. The drainage rates represent 12–17% of the mean annual precipitation (483 mm/a) over this time period. These drainage rates are 5 to 7 times greater than the regional recharge rate (11 mm/a) on the basis of groundwater chloride concentrations [*Wood and Sanford*, 1995]. However, the regional recharge estimate is based on a chloride wet and dry deposition value of 0.58 mm/a from 1 year of data (1984–1985) and a more reasonable chloride input value may be 0.31 mg/L on the basis of prebomb ^{36}Cl data [*Scanlon and Goldsmith*, 1997]. Reducing the chloride input by a factor of 1.9 would decrease the regional recharge rate by a similar amount from 11 to 6 mm/a. Drainage in deep-plowed areas would range from 10 to 14 times this lower regional recharge rate (6 mm/a). Upper bounds on drainage rates (75, 75, and 99 mm/a) from deep-plowed profiles were calculated by ignoring any flushing caused by conventional tillage (i.e., using z_1 of 1.0 m (root zone) rather than 3.3 m in equation (2)). Drainage rates could not be calculated using the chloride mass balance approach because information on chloride concentrations in runoff from adjacent terrace watersheds was not available.

[23] The deep-plowed profiles have generally high water contents throughout with slightly lower water contents toward the base of the 07-05 profile (Figure 4d). Total potential gradients are downward in deep-plowed profiles indicating downward water movement (Figure 4e), which is consistent with the low chloride concentrations. Total potentials are generally high with a slight decrease toward the base of the profiles. Reductions in total potentials toward the base of the profiles are not considered to represent the wetting front because minimum matric potentials in the deep-plowed profiles (−25, −36, and −42 m) are much higher than mean matric potentials under natural ecosystems (−183, −213, and −265 m) (Table 1).

[24] The deep-plowed and conventionally tilled profiles are in a bench adjacent to a terrace watershed. Runoff from the watershed provides additional water to the bench system. To evaluate the impact of this contributed runoff, a profile was drilled beneath conventionally tilled cropland in a graded terrace that has no benches (06-02, Figures 4j–4l). Flushing of chloride in this profile is restricted to the root zone (~1 m depth), indicating no deep drainage below the root zone (Table 1). Flushing of chloride to 2.8 and 3.7 m depth in the conventionally tilled profiles in the bench system is attributed to increased input of water from runoff from the adjacent terrace watershed.

[25] Similarity of chloride and matric potential profiles beneath conventionally tilled cropland in the graded terrace and another profile in nearby Armstrong County where no

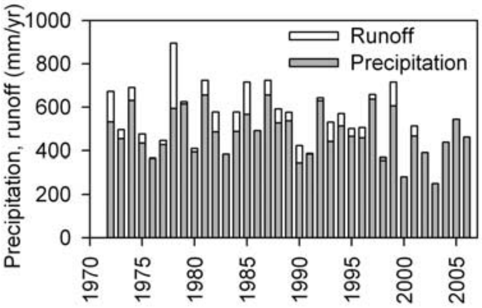

Figure 5. Precipitation and runoff from terrace watersheds onto terrace benches estimated from monitoring adjacent graded terraces.

terracing was practiced (Figure 4l) indicates that terracing alone does not seem to impact subsurface water fluxes and that the lack of drainage and recharge in these settings may apply regionally to the area covered by the Pullman clay loam soils (20,000 km^2).

3.2.1. Time Lag Between Increased Drainage and Groundwater Recharge

[26] Depth to groundwater beneath the deep-plowed profiles was 49 m prior to large-scale irrigation (1949) and has declined to 76 m (2006) on the basis of water-level data from well 756702, 2.5 km south of the Bushland site (http://www.twdb.state.tx.us). The aquifer saturated thickness decreased by $\sim$ 50% from 51 to 24 m over this time. The time lag between deep drainage below the root zone and groundwater recharge should be estimated on the basis of velocities of wetting fronts rather than chloride fronts, because recharge occurs when the wetting front reaches the water table [*Jolly et al.*, 1989]. Profiles in deep-plowed cropland do not extend deeply enough to show true wetting fronts. It should be possible to use water content data to estimate wetting front velocities (equation (4)) and depths from the CFD drainage rate; however, data on water contents prior to cultivation are not available in cropland areas. Water content in the profile in the natural ecosystem (mean 0.16 m^3/m^3 on the basis of a bulk density of 1600 kg/m^3) can be used as an estimate for precultivation water content (θ_d) because soil texture is similar in natural and cultivated sites. Estimated wetting front velocities are 625, 825, and 938 mm/a on the basis of $\bar{\theta}_w$ of 0.24, 0.28, and 0.28 m^3/m^3 and maximum CFD drainage rates (75, 75, and 99 mm/a) (Table 1) The corresponding wetting front depths are 23, 30, and 34 m ($v_{wf} \times$ 36 years). The current wetting front depths and velocities indicate that it would take an additional 45, 56, or 85 years to reach the water table at 76 m depth. Estimated time lags to reach the water table from the chloride front are 177, 203, and 242 years, which are 3 to 4 times those based on the wetting fronts and provide an upper bound on time lags between deep drainage and groundwater recharge. Time to recharge may be lower because initial water contents in the lower 25 m of the profiles are probably higher than the estimates based on natural ecosystems, because water tables have declined over this zone since predevelopment. The estimated times to recharge assume that soil texture remains uniform with depth. Previous studies of artificial recharge at the Bushland site indicate that there are no regional barriers to flow below the soil zone [*Aronovici et al.*, 1972; *Schneider and Jones*, 1988].

3.2.2. Water Balance

[27] The chloride and matric potential data provide information on net impact of deep plowing on subsurface water movement over the 36 year period; however, these data do not provide information on temporal variability in drainage rates. By calculating annually averaged drainage rates, we have assumed a uniform rate over time; however, drainage could occur in response to extreme events and be negligible most of the time. The bulk density, penetration, and infiltration tests conducted in 1975 and 2002 indicate that deep plowing is still impacting drainage [*Baumhardt et al.*, 2008]. Higher crop yields in deep-plowed benches until 2003 indicate that deep plowing is still effective in increasing drainage [*Baumhardt et al.*, 2008]. Monitoring different components of the water budget (precipitation, runoff, soil water) provides additional insights into flow processes beyond the time-integrated data provided by the chloride and matric potential profiles.

[28] Precipitation during the deep plowing experiment ranged from 250 to 656 mm (mean 483.3 ± 18.5 mm; 185 mm = 1 standard deviation) (Figure 5). Precipitation during the growing season (June through October) resulted mostly from convective thunderstorms and averaged 362 mm or 75% of mean annual precipitation.

[29] Annual runoff onto the benches from adjacent terraces ranged from 0 to 310 mm, representing 0–50% of the corresponding annual precipitation (Figure 5) and annual runoff was poorly correlated with annual precipitation (r = 0.40). Mean annual runoff (48.8 ± 10.2 mm) was only 10% of mean annual precipitation (483 mm). Approximately 70% of runoff occurs during the growing season (June through October). Runoff typically occurs after large summer rains during the sorghum growing season and the subsequent fallow period that replenishes the soil water profile [*Jones et al.*, 1985]. The exceptionally large annual runoff in 1978 (302 mm) was attributed to two separate and atypically large 1-day rainfall events totaling $\sim$210 mm.

[30] Cumulative precipitation and runoff delivered to benches was partitioned to soil water storage during fallow periods, summer crop transpiration, evaporation, and drainage. Precipitation and runoff during fallow periods (November through May) over the study period averaged 220 ± 15 mm and 15 ± 5 mm, respectively. The available soil water at planting averaged 169 mm (range: 51–291 mm) for deep-plowed plots, which was significantly higher than the average (152 mm; range: 31–263 mm) for conventionally tilled bench terrace plots (paired *t* test, n = 34, t = 2.034, p(two tailed) = 0.02) (Figure 6). Deep-plowed plots generally retained more soil water at planting except for some years when previous rotation crops may have depleted soil water unevenly. The increased water storage in deep-plowed plots relative to conventionally tilled plots is attributed to increased rain infiltration and redistribution of water within the soil where deep plowing had disrupted less permeable subsoil layers. Under rainfed cropping conditions, increased soil water at planting usually translates to increased growing season crop evapotranspiration.

[31] Future studies could consider detailed monitoring of subsurface water movement related to precipitation events to better understand flow processes and timing of deep

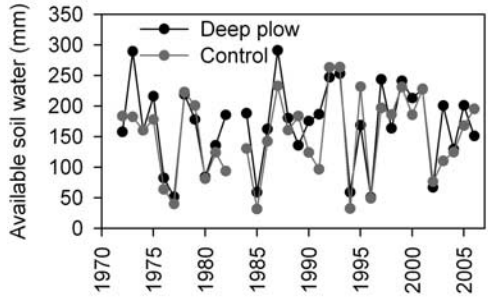

Figure 6. Variations in available soil water measured in deep plow and conventional tillage (control) areas in the bench terrace system at Bushland.

drainage. However, monitoring in these structured fine-grained soils is difficult because of preferential pathways that develop around the instruments and installation on the basis of our previous experience adjacent to playas (B. R. Scanlon, unpublished data, 2008) and monitoring results may simply be an artifact of installation. Drilling and sampling was limited in this study and should be expanded in future studies to increase the number of paired comparisons between deep plowing and other tillage practices. Studies should also be conducted in deep-plowed areas outside of the bench terrace systems adjacent to playas (e.g., Bushland site), to evaluate the potential for increasing drainage/recharge without the additional runoff from terrace structures.

3.3. Implications for Water Resources and Crop Production

[32] Previous studies in the southern High Plains showed that conversion from natural ecosystems to rainfed agriculture increased groundwater recharge from 0 mm/a to a median value of 24 mm/a, 5% of mean annual precipitation, similar to findings in other semiarid regions globally [*Scanlon et al.*, 2007a, 2007b]. In contrast, results from this study indicate that there is no recharge beneath rainfed cropland in conventionally tilled Pullman clay loam soils and that chloride flushing is restricted to the root zone (1 m depth). We attributed the lack of recharge through the Pullman clay loam in part to less conductive layers in finer textured soils compared with coarser textured soils farther to the south in the High Plains where mean clay content in upper 2 m is 23% [*Scanlon et al.*, 2007b].

[33] One-time deep plowing in the bench terrace system resulted in increased drainage rates (minimum 58, 60, and 81 mm/a) that are 4 to 9 times greater than drainage rates beneath conventionally tilled cropland (07-01: 9 mm/a; 08-08: 14 mm/a) in this bench terrace system (Table 1). Drainage rates beneath deep-plowed Pullman cropland are 2.6 to 3.6 times higher than the median drainage/recharge rate (24 mm/a) beneath rainfed agriculture in the southern High Plains from previous studies [*Scanlon et al.*, 2007b]. These drainage rates are also much higher than estimated drainage rates beneath two irrigated sites in the southern High Plains (17 and 32 mm/a [*McMahon et al.*, 2006]). The higher drainage rates beneath the deep-plowed system may also be attributed to increased runoff from adjacent terrace watersheds; however, monitored mean annual runoff was only 10% of mean annual precipitation (Figure 5). Therefore, deep plowing is an effective approach for enhancing drainage below the root zone and groundwater recharge in areas where low-permeability soils restrict subsurface flow. Improved subsurface drainage could also reduce waterlogging problems associated with low-permeability soils in many regions [*McFarlane and Williamson*, 2002].

[34] Rainfed agriculture relies heavily on stored soil water from fallow periods to supplement erratic rainfall during the crop growing season. Climate modeling indicates that the frequency of heavy precipitation events is very likely to increase during the 21st century [*IPCC*, 2007]. Therefore, waterlogging could increase in low-permeability soils and deep plowing could be used to mitigate this problem. With projected increases in the duration and intensity of droughts related to climate change [*IPCC*, 2007], sustainability of rainfed agriculture may decrease and reliance on irrigated agriculture may increase. Deep plowing provides a mechanism of increasing groundwater recharge that could be used for periodic irrigation in these regions.

[35] Previous researchers [*Zhang et al.*, 2004] suggested that increased root penetration associated with deep plowing would provide an opportunity for crops to access soil water from greater depths in the soil zone and reduce water losses below the root zone; however, the results from this study indicate that deep plowing increases water losses below the root zone. Root growth varies with crop type and extends to 1.5 m for sorghum; however, root density is greatest in the upper 0.5 m. Rooting depths were not regularly measured in this study; therefore, it is difficult to determine whether root depths increased in deep-plowed profiles. However, the results indicate that whether the root zone expanded or not, there is sufficient excess soil water to increase drainage below the root zone and ultimately recharge the aquifer. The inability of crops to use all the excess soil water in deep-plowed profiles may be related to limited water demand by seedlings during the early growing season when drainage of excess precipitation is likely and to restriction of root growth caused by limited crop growth period.

[36] Increased drainage associated with deep plowing could negatively impact crop production by reducing soil water availability in the crop root zone. However, data from this study indicate that soil water is greater in deep-plowed areas relative to that in conventionally tilled areas (Figure 6). Increased erosion could result from deep plowing; however, deep plowing has only to be conducted once and remains effective for decades as indicated by infiltration, penetration, and bulk density tests conducted in 1975 and 2002 and variations in crop yield up to 2003 [*Baumhardt et al.*, 2008].

[37] What impact would deep plowing have on volumetric groundwater resources in this region? Cropland currently represents 85% of the land use (60% rainfed and 25% irrigated) in the area of the Pullman clay loam soil series (20,000 km^2 area) (Figure 1). If we assume that deep plowing could be applied to 10% of the cropland (1,700 km^2 area) in the Pullman clay loam series, volumetric recharge could be increased by up to 0.12 km^3/a, assuming an average recharge rate of $\sim$70 mm/a based on limited data from this study (Table 1). This recharge rate is similar to the regional volumetric recharge estimated from playas using the chloride input in this study (6 mm/a $\times$ 20,000 km^2 area = 0.12 km^3/a). The estimated time lag between drainage below the root zone (1 m depth) and

recharge would range from 78 to 117 years on the basis of mean water table depth in the Pullman clay loam of 73 m and wetting front velocities at the Bushland site of 625 to 938 mm/a (Table 1).

[38] How would increased recharge beneath deep-plowed cropland compare with increased recharge from artificial recharge using spreading basins? There are advantages and disadvantages associated with each approach for enhancing recharge that require consideration in a detailed comparison. Deep plowing would mobilize more salt than spreading basins because the latter involves high water fluxes over small areas that would create flushed conduits to the aquifer. However, vulnerability to salt loading is not very high in the area of the Pullman clay loam soils because groundwater quality is high (median total dissolved solids based on well data is 375 mg/L) and the median saturated thickness of the aquifer of 25 m should dilute incoming salts.

[39] Previous artificial recharge studies at Bushland indicate that spreading basins could be installed adjacent to playas and should enhance groundwater recharge [*Aronovici et al.*, 1972; *Schneider and Jones*, 1988]. Studies at the Bushland site involved removal of the Pullman clay loam (1.3 m thick) in three different basins up to 0.4 ha in area and ponding water for 187 days over a 7 year experimental period [*Schneider and Jones*, 1988]. There are $\sim$10,000 playas in the Pullman clay loam soils that occupy 5% of the land surface. If we assume that recharge structures were installed adjacent to 100 playas with surface areas of 0.4 ha and assuming a recharge rate of 0.4 m/d over 100 d/a, this would result in a volumetric recharge rate of 0.016 km^3/a, which is about one seventh of that estimated from deep plowing. The time lag to recharge the water table would be much shorter in spreading basins, on the order of tens of days [*Schneider and Jones*, 1988]. Deep plowing could be deployed over a much larger area; however, recharge rates would be much lower and time lags greater. It is possible that both approaches could be used to increase groundwater recharge in this area because they complement each other in terms of area covered and timing of recharge.

4. Conclusions

[40] Deep plowing to 0.7 m depth in 1971 in the Pullman clay loam soil that covers a 20,000 km^2 area in the central and southern High Plains increased drainage below the root zone to depths of 10.7, 12.3, and 13.7 m over 36 years in a bench terrace system based on chloride front displacement. The chloride profile beneath the natural grassland ecosystem is bulge shaped (peak, 3042 mg/L, depth 1.0 m) and provides a marker to trace increased drainage beneath deep-plowed areas. Conventionally tilled profiles in the bench terrace system had chloride flushed to depths of 2.8 and 3.7 m, resulting in average water velocities below the root zone (1 m) of 36 and 54 mm/a and corresponding drainage rates of 9 and 14 mm/a. Runoff from adjacent terraces to the benches (mean 10% of precipitation) increased drainage beneath the conventionally tilled profile relative to a conventionally tilled profile in a graded terrace system (control, no runoff) where no drainage was recorded below the root zone (1 m depth). Minimum water velocities that could be attributed to the impact of deep plowing are 206, 250, and 289 mm/a and the corresponding drainage rates are 58, 60, and 81 mm/a. Minimum estimates of drainage beneath the deep-plowed areas are 4 to 9 times higher than those beneath the conventionally tilled profile in the bench terrace system (9 and 14 mm/a) and 10 to 14 times the regional recharge rate (6 mm/a) on the basis of groundwater chloride data. The chloride data provide a time-integrated response of subsurface drainage/recharge to hydrologic forcing over the 36 years since the site was deep plowed, but do not provide information on temporal variability in drainage rates. Enhanced flushing associated with deep plowing indicates that crop roots did not remove additional soil water, possibly because of early precipitation that exceeded the water needs of seedling crops and seasonally limited deep root growth. Minimum estimates of the time lag between deep drainage and groundwater recharge range from 45 to 85 years on the basis of wetting fronts. A preliminary analysis suggests that deep plowing could increase volumetric recharge much more than traditional spreading basins used for artificial recharge. Deep plowing reduced waterlogging associated with intense early growing season precipitation through improved drainage that, consequently, increased crop yield. Deep plowing has the potential to increase groundwater resources, converting green water to blue water by removing the effect of shallow, low-permeability soil layers in semiarid regions in areas where recharge is limited by such soils.

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