Appendix A

Groundwater Recharge in the Southern High Plains

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Introduction

This appendix provides a brief overview of previous estimates of groundwater recharge on the Southern High Plains, and presents the results of additional field work and modeling analyses conducted in conjunction with development of the Southern Ogallala GAM model. The field work, conducted in collaboration with the U.S. Geological Survey (USGS) National Water Quality Assessment (NAWQA) program, was included as part of the GAM study to provide additional information concerning irrigation return flow.

Review of Recharge Estimates from Previous Studies

The primary sources of recharge to the Ogallala aquifer in the Southern High Plains are playas, headwater creeks, and irrigation return flow. Previous studies indicate that recharge in interplaya settings from precipitation is negligible, as shown by high chloride concentrations in the unsaturated zone (Aronovici and Schneider 1972; Scanlon et al. 1997). If recharge rates were high in interplaya settings, chloride would be flushed out of the soil profile. Evidence of high recharge rates beneath playas is provided by low levels of calcium carbonate, low chloride concentrations, and deep penetration of bomb pulse tritium from nuclear testing in the 1950s and 1960s (Scanlon et al., 1997; Wood and Sanford, 1995). Recharge rates estimated from tritium concentrations in the unsaturated zone beneath individual playas range from 3 to 4.7 inches per year (in/yr) (Scanlon et al., 1997; Wood et al., 1997). Because of the spatial focusing of recharge beneath playas, it is difficult to calculate the average recharge rate to the aquifer from unsaturated zone data. Numerical modeling studies by Mullican et al. (1997) indicated that accurate representation of spatial focusing of recharge is not important for the Ogallala aquifer; similar modeling results were obtained whether recharge was focused beneath playas or was applied uniformly. Regional estimates of recharge were provided by groundwater tracers such as chloride and tritium. The average groundwater chloride concentration of 25 mg/L in the northern half of the Southern High Plains resulted in a regional recharge rate of 0.4 in/yr (Wood and Sanford, 1995). High tritium concentrations in groundwater in the southeastern part of the Ogallala aquifer in the vicinity of Lubbock, Lynn, and Dawson counties resulted in recharge estimates from 0.5 to 3.2 in/yr (mean 1.6 in/yr) (Nativ, 1988).

Irrigation-return flow may also contribute significant amounts of recharge to the aquifer. Many areas of the aquifer have been irrigated since the 1940s. Irrigation inefficiency was high during early decades, but decreased over time, particularly during the 1980s and 1990s. Luckey et al. (1986) estimated irrigation return flow to be 50% of applied irrigation water or net withdrawal in 1940 to 1960, decreasing to 37 to 46% in the 1960 to 1980 period. Field studies in Nevada in loam to clay-loam soil indicated that flood irrigation on alfalfa resulted in 20% return flow (Roark and Healy, 1998).

Recharge rates applied in previous groundwater modeling studies of the Southern High Plains are variable. Recharge rates in the Knowles et al. (1984) model ranged from 0.06 to 0.83 in/yr. These estimates were based on a study of water content monitoring at irrigated and non-irrigated sites in each county conducted by Klemt (1981). Luckey et al. (1986) applied an average recharge rate of 0.13 in/yr in the Southern High Plains during the predevelopment period. A more recent modeling study conducted by Stovall et al. (2000) applied an average recharge rate of 2.8 in/yr based on automated inverse modeling. In addition to irrigation return flow applied during aquifer development Luckey et al. (1986) applied an additional 2 in/yr to irrigated and dryland areas during the 1960 to 1980 period.

Recharge Estimation for Current Groundwater Availability Modeling Project

Additional studies were conducted to evaluate recharge in the Southern High Plains. The previous regional estimate of recharge based on groundwater chloride concentrations in Wood and Sanford (1995) was reexamined to evaluate any potential impact of irrigation return flow. Field studies were conducted in collaboration with the U.S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) program. The NAWQA program involved drilling and sampling of boreholes in irrigated and non-irrigated sites. Samples were collected for tritium analysis to evaluate recharge rates. Pressure monitoring devices were also installed to evaluate infiltration beneath irrigated and non-irrigated sites.

Recharge Estimates Based on Groundwater Chloride Concentrations

Recharge was previously estimated by Wood and Sanford (1995) using an average groundwater chloride concentration of 25 mg/L in the northern half of the Southern High Plains because this region is not affected by saline lakes. Areas of known contamination were excluded in the estimate. The recharge rate was calculated using the chloride mass balance approach, which equates the chloride input to the system (precipitation rate [19.1 in/yr] times chloride concentration in precipitation and dry fallout [0.58 mg/L]) to the chloride output (recharge rate times chloride concentration in groundwater [25.2 mg/L]).

$$R = \frac{P \, x \, Cl_p}{Cl_{gw}} = \frac{19.1 in/yr \; x \; 0.58 mg/L}{25.2 mg/L} = 0.4 in/yr \qquad (1)$$

These data resulted in a recharge rate of 0.4 in/yr. We reevaluated this recharge estimate by examining the chloride concentration in precipitation, the precipitation rate, and the groundwater chloride concentration. Chloride concentrations in precipitation (0.13 ± 0.02 mg/L), based on data from the National Atmospheric Deposition (NADP) program (http://nadp.sws.uiuc.edu) from 1985 – 2000 at the Muleshoe National Wildlife Reserve (MNWR) and at a site near the Texas-Oklahoma border (OK29; 0.12 ± 0.02 mg/L), are much lower than the previous estimate of 0.5 mg/L, which was based on one year of data from Amarillo (Wood and Sanford; 1995). The differences in chloride input may be partly related to the fact that the NADP values represent wet deposition only, whereas the data from Amarillo represent wet and dry fallout. Studies by Izbicki (USGS, pers. comm., 2001) indicate that values based on wet deposition should be multiplied by a factor of ~2 to approximate wet and dry deposition. Chloride input was also estimated for the Amarillo region using pre-bomb 36Cl/Cl ratios (~0.3 mg/L). The value of 0.3 mg/L is slightly greater than 2 times the NADP value but is more appropriate than 0.58 mg/L. The average precipitation for the region, based on data from the National Climatic Data Center (http://lwf.ncdc.noaa.gov) for the period 1931 – 2001, was 17.8 in/yr.

Groundwater chloride concentrations were also reevaluated. The previous estimate of 25.2 mg/L (Wood and Sanford, 1995) did not evaluate the impact of irrigation return flow. Groundwater chloride concentrations in irrigated and non-irrigated regions and the distribution of irrigated regions were obtained from the Texas Water Development Board (http://www.twdb.state.tx.us) (fig. 1). High chloride concentrations related to saline lakes and contaminated sites were omitted from average values by excluding data in the predominantly red-colored zone shown in Figure 1 and other concentrations greater than 2 standard deviations above the mean log of the remaining values. Average chloride concentrations in irrigated (15.9 mg/L) and non-irrigated (17.3 mg/L) sites were remarkably similar to and are slightly lower than the average groundwater chloride concentration

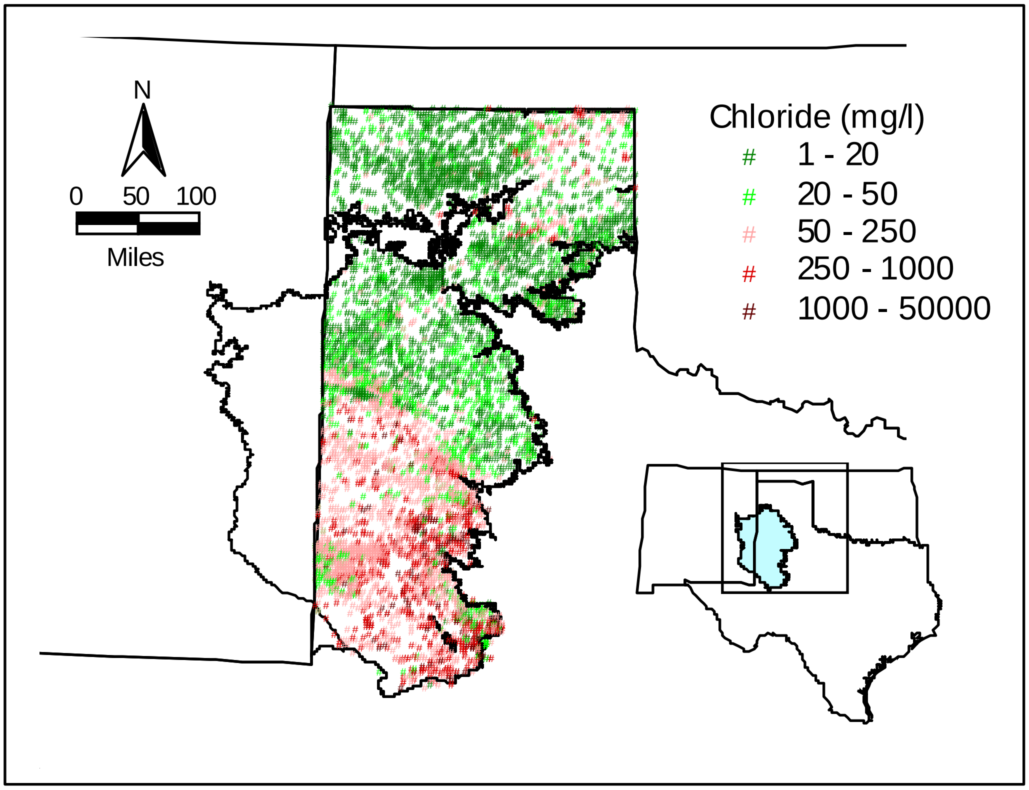

Figure 1. Groundwater chloride concentrations in the High Plains Aquifer in Texas.

previously estimated by Wood and Sanford (1995). The recharge rate for non-irrigated regions was calculated as follows:

$$R = \frac{P\,x\,Cl_p}{Cl_{gw}} = \frac{17.8in/yr\ x\ 0.3mg/L}{16.7mg/L} = 0.31\,in/yr \quad (2)$$

This estimate of 0.31 in/yr is slightly lower than that estimated by Wood and Sanford (1995). Recharge for irrigated regions can be estimated using the following equation:

$$R = \frac{P \times Cl_p + Irr \times Cl_{irr}}{Cl_{gw}} \quad (3)$$

where *Irr* is the irrigation application and Cl_{irr} is the chloride concentration in the irrigated water.

Estimating recharge rates in irrigated areas requires information on the irrigation application amount, the chloride concentration in the irrigation water, and the chloride concentration in groundwater in irrigated regions. Because groundwater is used for irrigation, one would assume that the chloride concentrations in groundwater in nonirrigated regions and the irrigation water are similar. The similarity in groundwater chloride concentrations in irrigated and nonirrigated regions suggests that either (1) the database and data mining approach are inadequate, (2) there is mixing and dilution of irrigation return flow with regional Ogallala groundwater, (3) irrigation water had not reached the groundwater, or (4) the irrigation system was 100% inefficient. The first two reasons seem most plausible because the field studies described in this report indicate that irrigation water has reached the water table in the areas examined and the maximum inefficiencies of irrigation systems is generally considered to be 50% during flood irrigation. The database includes chloride concentrations that represent a long time period and there may be temporal trends in chloride concentrations that are not being evaluated in this approach. The Ogallala aquifer represents a huge reservoir of water, and mixing and dilution may mask the input from irrigation return flow. Irrigated regions would not be expected to represent steady state conditions, which add an additional complexity to the use of chloride to estimate groundwater recharge. Tritium concentrations provide a much better indicator of irrigation return flow than chloride and are discussed in a later section.

Recharge Estimates Based on Tritium

Concentrations of tritium in the unsaturated zone and groundwater were used to estimate recharge rates in irrigated areas. Tritium is used to trace water movement because it is part of the water molecule. Tritium is a radioactive isotope of hydrogen with a half life of 12.32 years. Tritium occurs naturally in the atmosphere and enters the subsurface primarily through precipitation. Tritium fallout increased as a result of atmospheric nuclear testing that began in the early 1950s and peaked in 1963 (fig. 2). The distribution of tritium in the subsurface can be used to determine the average velocity of the water. The recharge rate is calculated by multiplying the velocity by the average water content in the unsaturated zone.

Two boreholes were drilled in areas that had been irrigated since 1958 (Roberts and Maple sites) and one borehole was drilled in a non-irrigated site in the MNWR for comparison with irrigated sites (fig. 3). The boreholes in the irrigated sites were located about 5 feet (ft) distant from the edge of cultivated fields. The drilling, sampling, and analyses were conducted by the USGS as part of the NAWQA program. The methods used in this study are similar to those described in McMahon et al. (2002). The boreholes were drilled using an ODEX air-hammer drilling method (Hammermeister et al., 1986). Core samples were collected in an aluminum-lined core barrel for measurement of porosity, water content, and tritium analyses in the unsaturated zone. Groundwater samples at the irrigated sites were also collected for tritium analysis. The porosity of the samples was measured by completely saturating the cores under a vacuum and calculating the volume of pore space as the difference in mass between the saturated and oven-dried sample (McMahon et al., 2002). Gravimetric water content was measured by oven drying the sample at 105ºC for 24 hours and calculating the difference in mass between the initial (field) sample weight and the oven dried sample weight. Unsaturated zone tritium analyses were conducted on water extracted from 1 kilogram (kg) of core by vacuum distillation at 80ºC. Tritium analyses of unsaturated zone pore water and groundwater were conducted at the USGS Tritium Laboratory in Menlo Park, CA using liquid scintillation with electrolytic enrichment (Thatcher et al., 1977).

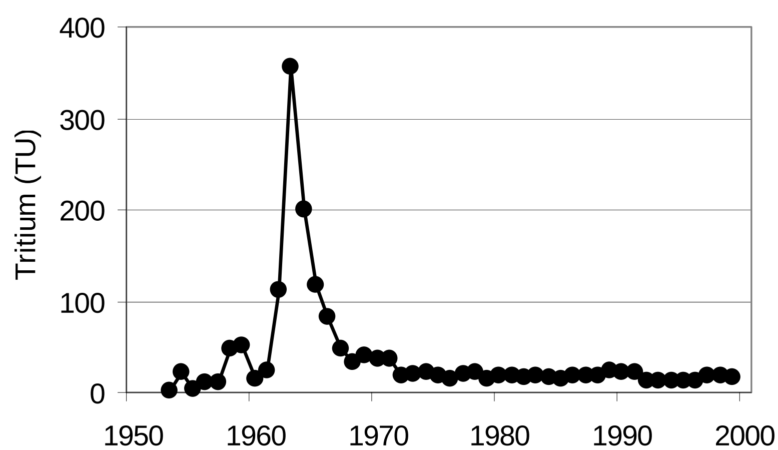

Figure 2. Average annual atmospheric tritium fallout for Ottawa, Ontario.

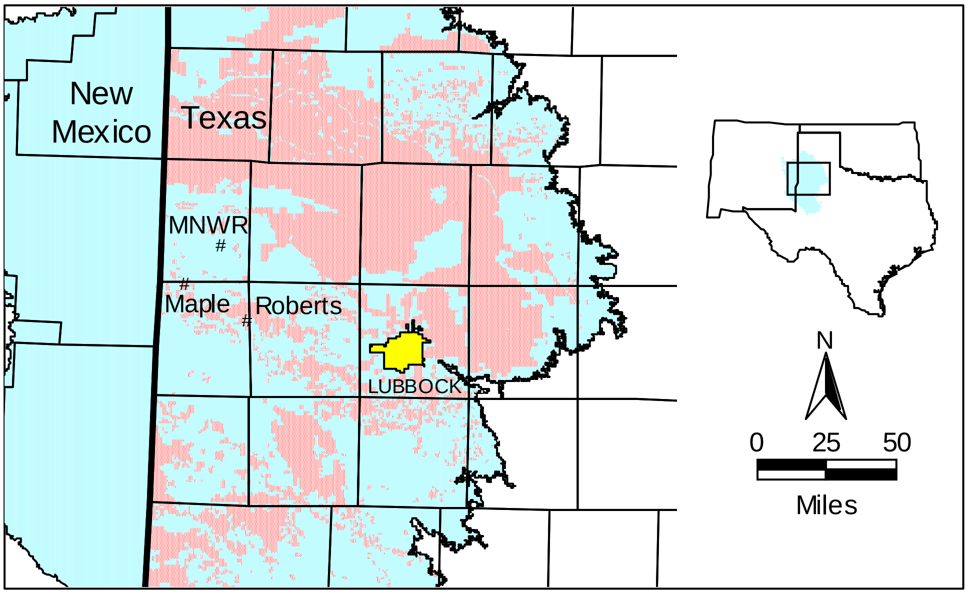

Figure 3. Map of borehole locations with Texas and New Mexico county lines. Blue areas represent extent of Southern High Plains Aquifer. Pink areas represent 1994 irrigated lands in the Texas portion of the aquifer.

Water content at the non-irrigated site was highest at the surface (about 25%) as a result of a recent precipitation and decreased to 10 to 13% at depth (fig. 4). A similar range in water contents was measured at the irrigated sites (8 to 21%). The porosity of the sediments at each site was quite variable, indicating layered sediments of different textures. Porosities ranged from approximately 25 to 48%.

Post-bomb tritium was generally restricted to the root zone at the non-irrigated MNWR site and indicates negligible recharge (fig. 4). The highest tritium concentration at this site was 6.5 tritium units (TU) at a depth of 3.75 ft. In contrast, post-bomb tritium was found throughout much of the unsaturated zone at the two irrigated sites and in the groundwater at the Roberts irrigated site. At depths greater than 20 to 30 ft, some depth intervals containing tritium concentrations below the detection limit are underlain by post-bomb pulse tritium, which suggests non-piston or preferential flow. A range of recharge rates was estimated from the tritium data in the irrigated sites. Pre-bomb tritium activity was estimated to have been about 8 TU (Thatcher, 1962). The activity of pre-bomb tritium in core samples at the time of sampling, A, was distinguished from post-bomb values by using the radioactive decay equation:

$$A = A_0 e^{-\lambda t} \tag{4}$$

where A_0 is the pre-bomb tritium activity, λ is the half-life, and t is elapsed time. The tritium center of mass is calculated as follows:

$$T_{1/2} = \int_0^z \theta T dz \tag{5}$$

where θ is the volumetric water content (ft^3/ft^3), T is the tritium activity (TU) and dz is the depth interval. The velocity of the water was calculated using two methods. In the first approach, the depth of the tritium activity center of mass in the subsurface was calculated and assumed to correspond in time with the peak of atmospheric activity (i.e., 1963). In the second approach, the depth of the deepest occurrence of post-bomb tritium activity was determined and assumed to correspond in time with the onset of atmospheric testing (i.e., 1953). Both approaches assume piston-type flow conditions whereas penetration of tritium below the center of mass at both irrigated sited may have occurred as a result of referential flow. For comparison and to limit the effects of preferential flow in the calculations, the centers of mass were additionally approximated by extrapolating the tritium peaks to pre-bomb values, thus removing the "tails" of the tritium distributions at depth (fig. 4).

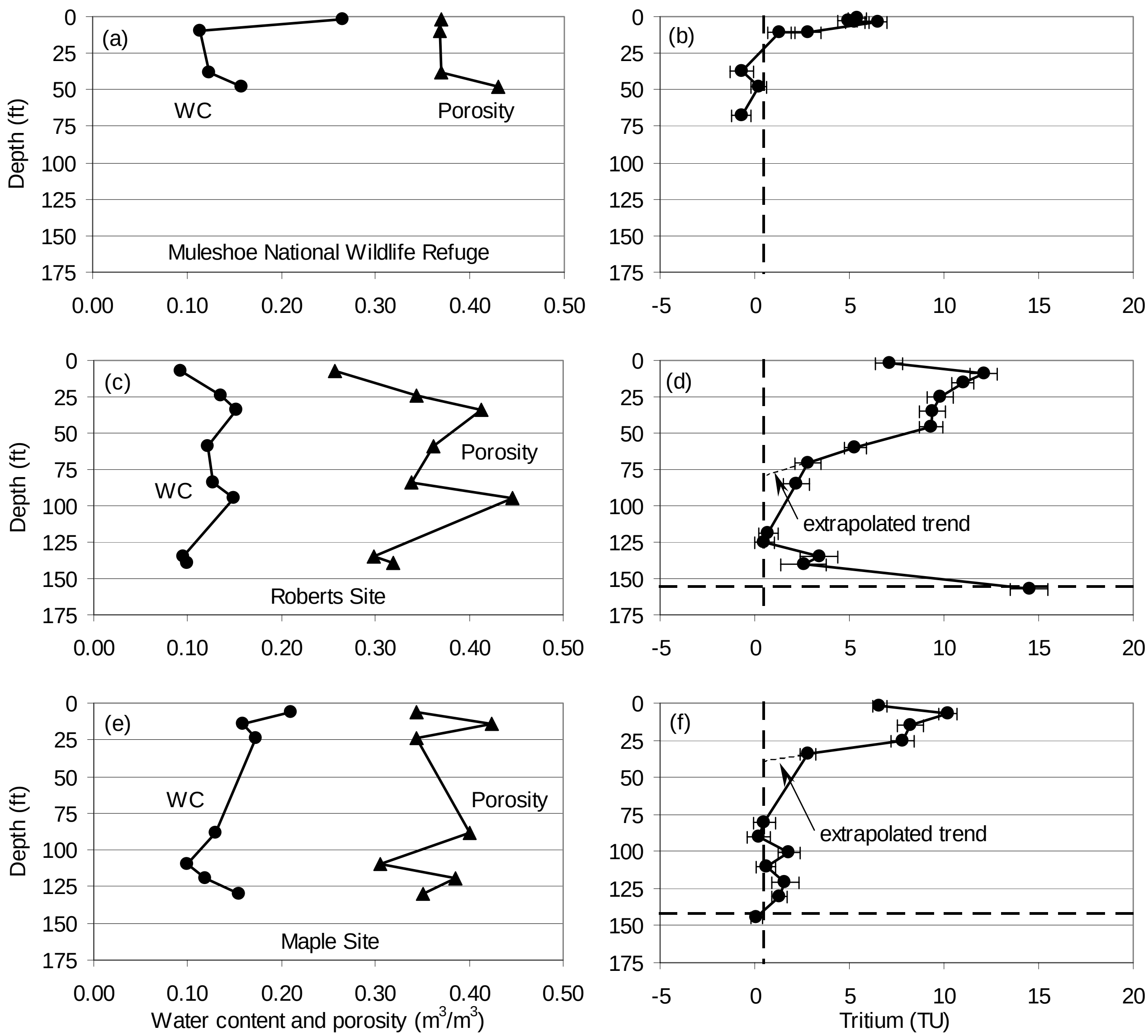

Figure 4. Water content, porosity, and tritium activity with depth at Muleshoe NWR (a, b), Roberts Site (c, d) and Maple Site (e, f). Tritium error bars represent uncertainty (1σ) in measured tritium activity. Vertical dashed lines represent calculated present-day activity (0.5 TU) of pre-bomb tritium (~8 TU) after ~50 years of decay. Horizontal dashed lines represent approximate depth of water table and deepest tritium values represent groundwater activity for the two irrigated sites.

The results of the calculations indicate that the center of mass approach to calculating recharge rates is not sensitive to preferential flow at these sites, as most of the mass is in the near-surface bulge and removal of the tails at depth resulted in only a negligible decrease in calculated recharge rates (table 1). The depth of the center of mass of tritium ranged from 13 to 35 ft. The resultant velocities ranged from 0.34 to 0.91 feet per year (ft/yr) based on the time since peak fallout (1963) and the sampling time (2001) (38 years). The recharge rates, calculated by multiplying the velocity by the average water content, ranged from 0.7 to 1.3 in/yr.

The calculations based on the deepest occurrence approach were very sensitive to the presumed preferential flow at both sites. The deepest occurrence of post-bomb tritium ranged from 136 to 148 ft with average velocities ranging from 2.8 to 3.1 ft/yr and recharge rates ranging from 4.6 to 5.0 in/yr. These recharge rates are approximately 2 to 3 times greater than the rates calculated by removing the tails of the tritium distributions at depth (2.4 and 1.7 in/yr, respectively).

These estimates could be considered bounding values for recharge. The tritium data provide average recharge estimates for the time period considered (38 to 48 years); however, recharge rates probably changed over time as irrigation practices varied. Irrigation began in 1958 at both sites. The plots were furrow irrigated initially, followed by sprinkler irrigation. Cotton was the main crop at both sites. The efficiency of irrigation systems improved over time and the amount of irrigation return flow probably decreased substantially with time. However, the current tritium profiles cannot provide any information on the temporal variability in irrigation return flow.

Matric Potential Monitoring

Heat dissipation sensors were installed to monitor the negative pressures in the unsaturated zone to determine the direction of water movement and to evaluate drainage beneath the irrigated plots. Heat dissipation sensors consist of a heater and thermocouple wire in a cylindrical porous ceramic material. A heat pulse is applied for 30 seconds(s) and the amount of heat dissipation is determined by measuring the temperature change of the instrument. Heat dissipation increases with increasing water content in the soil and is related to the negative pressures in the unsaturated zone through laboratory calibration. The heat dissipation sensors (Model 229, Campbell Scientific Inc., Logan, UT) were calibrated in a pressure plate apparatus in the laboratory using procedures outlined in Scanlon and Andraski (2002) and Flint et al. (2002). Initially, the temperatures were calibrated for temperature changes after 20 seconds of heating. Later, calibrations were changed to use 30 seconds of heating.

Table 1: Recharge estimates from tritium distributions beneath the irrigated sites.

Site	Method	Data used	Depth (ft)	Time (yr)	Average Velocity (ft/yr)	Average Water Content (ft^3/ft^3)	Recharge Rate (in/yr)
Roberts	Center of mass	All	35	38	0.91	0.123	1.3
		(-) Tails	28	38	0.73	0.126	1.1
	Deepest occurrence	All	148	48	3.08	0.123	4.6
		(-) Tails	75	48	1.57	0.126	2.4
Maple	Center of mass	All	16	38	0.42	0.147	0.7
		(-) Tails	13	38	0.34	0.176	0.7
	Deepest occurrence	All	136	48	2.84	0.147	5.0
		(-) Tails	39	48	0.82	0.176	1.7

Note: Recharge rates based on depth of post-bomb 3H center of mass and time since peak atmospheric 3H levels (38 years) and on deepest occurrence of post-bomb 3H levels and time since onset of atmospheric testing (48 years).

Heat dissipation sensors were installed in the deep boreholes drilled by the USGS. The instruments were placed at different depths and were surrounded by a sand and silica flour mixture. A bentonite plug was used to separate the different heat dissipation sensors. In addition, shallow boreholes were drilled to approximately 10 ft using a trailer-mounted drilling rig (Giddings Machine Co., Inc, Giddings, TX) and were located within the swing of the center pivot to monitor water movement directly beneath the irrigated area. The shallowest depth that could be monitored in the irrigated fields was approximately 1.6 ft (0.5 m) because of the approximated 1.3-ft plough depth. The instruments are connected to a data logger and powered by a solar panel. The instruments are logged daily and data are telemetered to the Bureau of Economic Geology using a cell phone system.

Matric potential profiles in the non-irrigated site were generally much lower (more negative) than those in the irrigated sites in the upper 10 ft in the spring and summer, indicating generally drier conditions in the non-irrigated site (fig. 5). The vertical matric potential profile in the non-irrigated site indicates matric potentials as low as –20 to –25 bars in the shallow subsurface, increasing to matric potentials close to zero at a depth of approximately 38 ft. The increase in matric potentials with depth indicates an upward driving force for water movement and suggests upward flow. These monitoring results are similar to results from interplaya settings at the Department of Energy Pantex Plant near Amarillo, Texas (Scanlon et al., 1997). The vertical matric potential profiles in the irrigated plots are close to zero throughout the profile indicating fairly wet conditions as a result of irrigation.

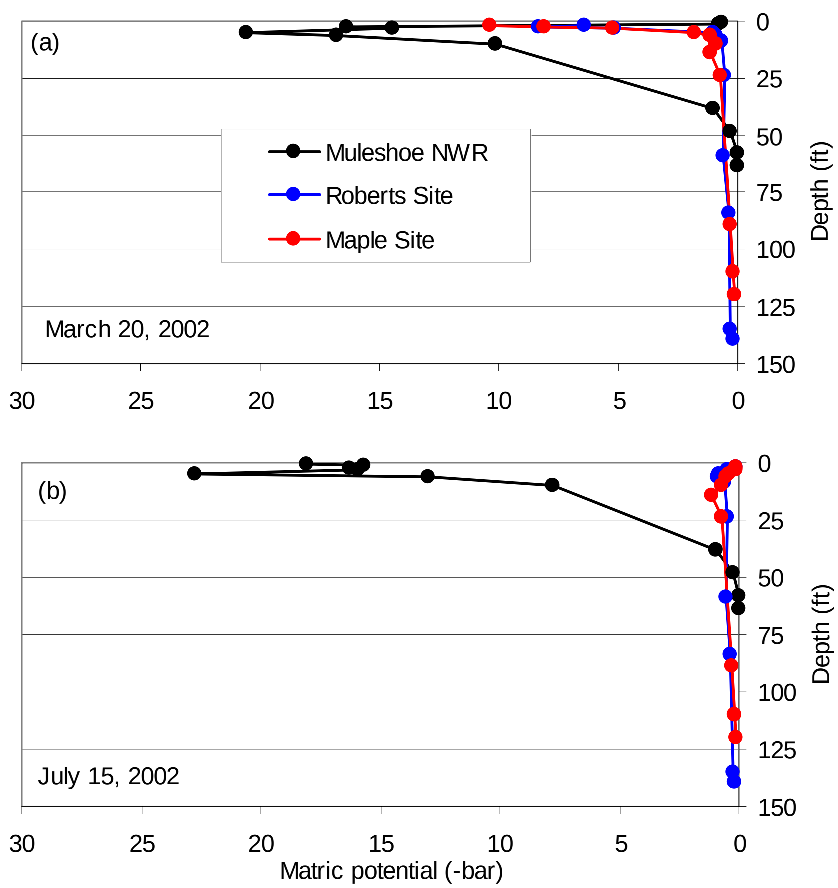

Figure 5. Monitored matric potential depth profiles measured during spring (a) and summer (b) of 2002 at the natural and irrigated sites.

The time series plots of matric potentials provide information on infiltration of water as a result of precipitation and irrigation (fig. 6). Precipitation data are only available from MNWR but are repeated in the other plots to represent widespread rainfall events. Monitoring of the heat dissipation sensors installed in the deep boreholes began in July and August 2001, whereas monitoring of those installed in the shallow boreholes began in November 2001. Monitoring results from the heat dissipation sensors indicate infiltration and deeper penetration of water to a maximum depth of 2.5 ft in response to precipitation in the spring of 2002 at the nonirrigated site (MNWR). However, matric potentials decreased in May and June and remained low throughout the rest of the summer. Matric potentials in the upper 10 ft were generally in the range of –10 to –15 bars, indicating extremely dry conditions. Matric potentials increased to values close to zero at 64 ft depth because there is a perched water table close to this zone.

In contrast, matric potentials throughout much of the unsaturated zone beneath the irrigated plots ranged from –0.5 to –1 bar, which is much wetter than the non-irrigated site. The irrigated sites also show infiltration in response to both precipitation and irrigation. As reported by the landowners, the center pivot irrigation systems at both sites require approximately one week for a single pass around the field. The application depth per pass varies from 0.8 inches at the Maple site to 1.0 inches at the Roberts site. Matric potentials increased to a depth 3.3 and 6.6 ft at both irrigated sites in April. Irrigations at the Roberts site in June and early July resulted in infiltration and redistribution to the 2.5 ft depth. Drying occurred in the top 3.3 ft from late July through August despite continued irrigation at the Roberts site. The Maple site was irrigated beginning in late May. Matric potentials showed no response until early July and again in mid-August when infiltration occurred to the 6.6 ft depth. The monitoring data at both irrigated sites indicate that, in the absence of precipitation, current center pivot irrigation methods generally result in infiltration and redistribution to depths of less than 3.3 ft. When irrigations occur in conjunction with larger precipitation events (≥~1.0 inches), infiltration and redistribution occur to depths between 6.6 and 9.8 ft. Therefore, the monitoring data indicate that the soil profile is much wetter beneath the irrigated fields than at the non-irrigated site.

Time Lag for Irrigation Return Flow

An important factor for transient ground-water simulations in the Southern High Plains is the time lag between drainage below the root zone from irrigation and recharge at the water table. We evaluated this time lag using an analytical approach and numerical modeling. The analytical approach required the following assumptions. We assumed uniform texture and initial matric potential conditions and piston flow. We also assumed unit gradient conditions, i.e. pressure head gradient dh/dz = 0; therefore, the only driving force was gravity. Under these conditions, the flux is equal to the unsaturated hydraulic conductivity at the prevailing water content, $ku(\theta_f)$, and the time required for a wetting front induced by a constant flux rate to arrive at a given depth, z, is:

$$t = \frac{z \times \left(\theta_f - \theta_i\right)}{k_u\left(\theta_f\right)} \qquad (6)$$

where θ_f is the final water content and θ_i is the initial water content in the profile.

From our monitoring of interplaya settings at the Department of Energy Pantex Plant near Amarillo, matric potentials at depths below the root zone are approximately –10 bars. This matric potential was applied to calculate initial water contents for each of the sediment textures. Van Genuchten parameters published by Schaap and Leij (1998) were applied for a range of sediment textures derived from multiple parameter databases (table 2).

The retention functions were solved for water contents corresponding to flux rates of 6, 9, and 12 in/yr and determined the number of years required for irrigation return flow to move down

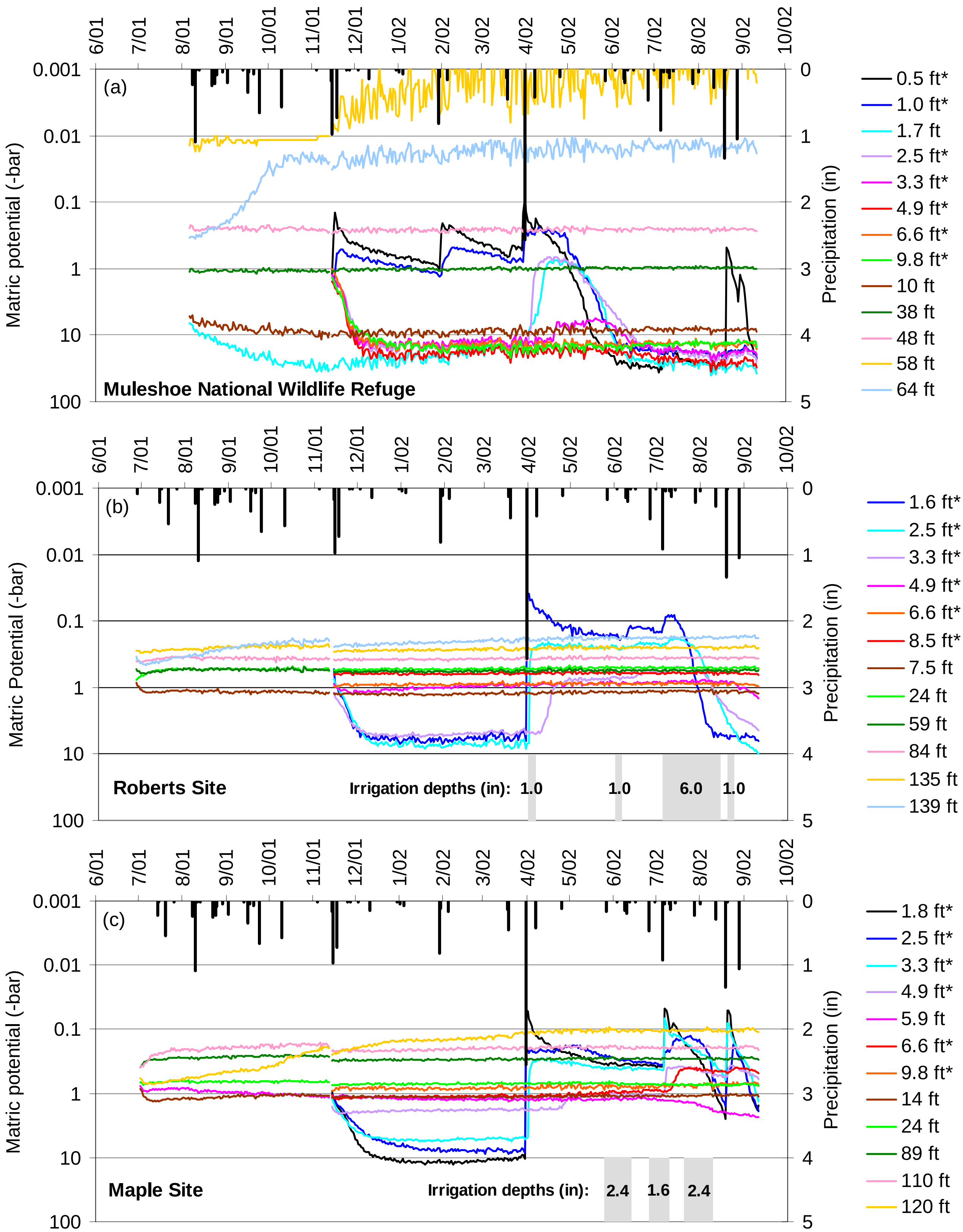

Figure 6. Time series of monitored matric potential with depth at natural (a) and irrigated (b, c) locations. Precipitation values shown for all locations were monitored at Muleshoe NWR. Depths marked with an asterisk (*) indicate sensors installed in shallow boreholes; at the irrigated sites, these sensors are within the irrigated area. Vertical gray bars behind irrigation depth values represent periods of active center pivot irrigation.

1 ft (i.e., the time required for storage to increase from θ_i to θ_f at a flux of $ku(\theta_f)$). For a given texture and flux rate, the wetting front rate of advance is linear and the results were calculated for an unsaturated zone thickness of 100 ft (fig. 7). Sensitivity of arrival time to variations in initial potentials was also examined by varying the initial matric potentials from –10 to –0.5 bars. For a thickness, z, other than 100 ft, the wetting front rate of advance can be calculated by multiplying the results by the ratio of (z/100). The lag time for a sand is less than 5 years; this decreases with increasing applied flux from 6 to 12 in/yr and is insensitive to initial water potential conditions. The lag time for a sandy clay loam was much greater because of the increased storage capacity relative to sand. The lag time for a sandy clay loam varied from 33 years for an applied irrigation of 6 in/yr to 19 years for an applied irrigation of 12 in/yr.

Numerical Simulations

We also conducted numerical simulations of unsaturated flow to evaluate the irrigation return flow lag time. We used the code UNSAT-H (Fayer, 2000) to perform vertical one-dimensional flow simulations. The upper boundary condition was based on 50 yrs of climatic history generated for Lubbock using the computer code GEM (Johnson et al., 2000). Daily weather data were input to the simulations, including precipitation, temperature, relative humidity, solar radiation, and wind speed. A 50-year time period was simulated. To evaluate the impact of irrigation, the weather data were modified by increasing the applied precipitation. Irrigation amounts of 15 inches and 30 inches were evaluated for the 1950 to 1960 time period. The irrigation was distributed in 5 separate applications each lasting 3 days at bimonthly intervals in the summer. The irrigation amount was reduced over time from the designated amount in 1951 to 1960 to 75% in the next 10 years (1961 through 1970), 50% in the following 10 years (1971 through 1980), and 25% in the remaining 20 years (1981 through 1999). Plant transpiration was included in the simulations and parameters for cotton were incorporated (Dugas et al., 1985). A unit gradient lower boundary condition was used to evaluate drainage from irrigation return flow. Profile depths of 15 and 50 ft were used to evaluate variations in irrigation return flow over time at those depths.

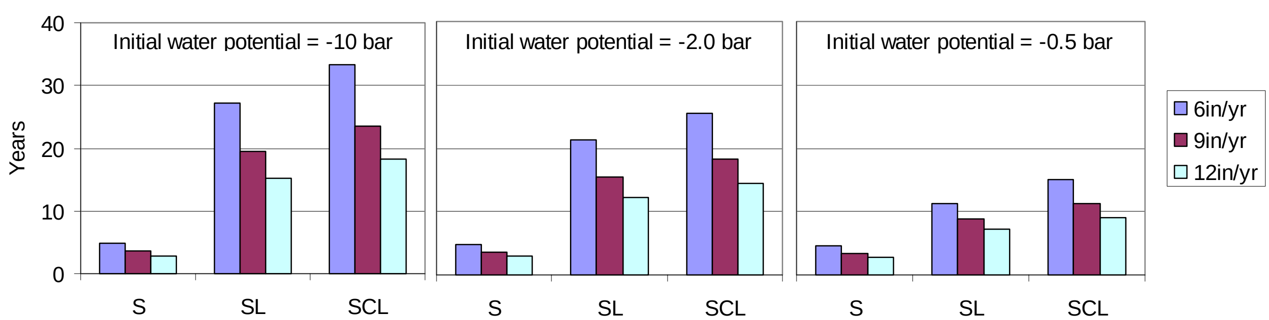

Figure 7. Calculated times required for a piston-flow wetting front to arrive at the base of a 100 ft-thick soil profile of uniform texture. Results are shown for uniform initial water potentials of –10 bar, -2 bar, and –0.5 bar for sand (S), sandy loam (SL), and sandy clay loam (SCL) textures and various imposed recharge rates.

Table 2: van Genuchten parameters used for irrigation return flow time lag estimates.

Texture	θ_s (cm^3/cm^3)	θ_r (cm^3/cm^3)	α (1/cm)	n	Ks (cm/day)
S	0.375	0.053	0.0355	3.16	646
SL	0.387	0.039	0.0269	1.45	76
SCL	0.384	0.063	0.0209	1.32	13

The simulation results indicated that the lag time for irrigation return flow for a sand texture was 0 (15-ft profile) to 3 years (50-ft profile) (table 3). The 3-year lag time is similar to that predicted by the previously described analytical calculations (fig. 7). The simulated drainage ranged from an average of 5 to 14 inches depending on the applied irrigation (15 to 30 inches). These values correspond to about 60% of the applied irrigation. Temporal variations in drainage were much greater at 15 ft than at 50 ft (fig. 8). Drainage rates were similar at 15 and 50 ft depths when the applied irrigation was 30 inches. The lag time for irrigation return flow increased as the sediment texture became finer, particularly when the applied irrigation was low (15 inches). Drainage rates were lower for the sandy clay loam and the lag times ranged from 4 to more than 50 years. These results indicate that various factors are important in controlling the lag time, including applied irrigation amount, sediment texture, and profile depth.

Additional simulations were conducted to determine the impact of a low-permeability layer, such as caliche, on irrigation return flow and drainage. Hydraulic properties were not available for caliche, but were approximated with those of a clay from the Schaap and Leij database (1998). The hydraulic conductivity for a clay in this database is high, and further simulations were conducted with a hydraulic conductivity of 4.4×10^{-8} in/s (equivalent to 10^{-7} cm/s). Results of these simulations indicated little difference in drainage or lag time when the higher conductivity was used to represent the caliche, but showed zero drainage for all cases where the lower hydraulic conductivity was used. The tritium results from the irrigated field plots described earlier indicate that caliche does not preclude drainage from irrigated plots and suggests that caliche is not very effective in reducing drainage.

Table 3: Simulated drainage lag times

USDA Texture Class	Profile Depth (ft)	Initial Annual Irrigation (in)	Simulated Drainage (in)	Lag time (yr)
Sand	15	15	9.8	0
		30	16.7	0
	50	15	5.3	3
		30	16.5	1
Sandy loam	15	15	1.65	4
		30	7	1
	50	15	1.1	21
		30	6.46	5
Sandy clay loam	15	15	0.06	19
		30	1.42	4
	50	15	0	>50
		30	0.94	23

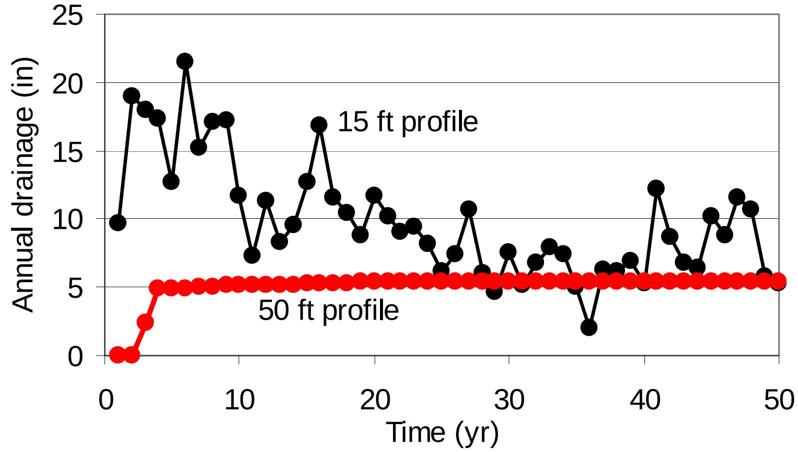

Figure 8. Comparison of simulated annual drainage from the bases of two irrigated sand profiles with different thickness.

References

Aronovici, V. S., Schneider, A. D., and Jones, O. R., 1972, Basin recharge of the Ogallala aquifer: J. Irrig. Drain. Div. ASCE, p. 65-75.

Dugas, W. A., Upchurch, D. R., and Ritchie, J. T., 1985, A weighing lysimeter for evapotranspiration and root measurements: Agronomy Journal, v. 77, p. 821-825.

Fayer, M. J., 2000, UNSAT-H Version 3.0: Unsaturated Soil Water and Heat Flow Model, Theory, User Manual, and Examples: Pacific Northwest National Laboratory 13249, p. variously paginated.

Flint, A. L., Campbell, G. S., and Calissendorff, C., 2002, Calibration and temperature correction of heat dissipation matric potential sensors: Soil Sci. Am. J.

Hammermeister, D. P., Blout, D. O., and McDaniel, J. C., 1986, Drilling and coring methods that minimize the disturbance of cuttings, core, and rock formations in the unsaturated zone, Yucca Mountain, Nevada, in Proceedings of the National Water Well Association Conference on Characterization and Monitoring of the Vadose (Unsaturated Zone), Worthington, Ohio, p. 507-541.

Klemt, W. B., 1981, Evaluating the ground-water resources of the High Plains of Texas, neutron probe measurements of deep soil moisture as an indication of aquifer recharge rates.

Johnson, G. L., Daly, C., Taylor, G. H., and Hanson, C. L., 2000, Spatial variability and interpolation of stochastic weather simulation model parameters: J. Appl. Climatology, v. 39, p. 778-796.

Knowles, T. R., Nordstrom, P., and Klemt, W. B., 1984, Evaluating the ground-water resources of the High Plains of Texas: Texas Department of Water Resources Report 288, 119 p.

Luckey, R. R., Gutentag, E. D., Heimes, F. J., and Weeks, J. B., 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.

Mullican, W. F. I., Johns, N. D., and Fryar, A. E., 1997, Playas and recharge of the Ogallala aquifer on the Southern High Plains of Texas–An examination using numerical techniques: The University of Texas at Austin, Bureau of Economic Geology, Report of Investigations No. 242, 72 p.

McMahon, P. B., Dennehy, K. F., Ellett, K., Sophocleous, M. A., Paschke, S. S., Michel, R. L., and Hurlbut, D., 2002, Evaluation of Recharge Fluxes to the Central High Plains Aquifer, Southwestern Kansas, 2000-01: Water Resour. Inv. Rept.

Nativ, R., 1988, Hydrogeology and hydrochemistry of the Ogallala Aquifer, Southern High Plains, Texas Panhandle and Eastern New Mexico.

Roark, D. M., and Healy, D. F., 1998, Quantification of deep percolation from two flood- irrigated alfalfa fields, Roswell Basin, New Mexico: USGS Water Resour. Inv. Rept. 98-4096, 32 p.

Scanlon, B. R., Andraski, B. J., and J., B., 2002, Miscellaneous methods for measuring matric or water potential, Methods of Soil Analysis.

Scanlon, B. R., and Goldsmith, R. S., 1997, Field study of spatial variability in unsaturated flow beneath and adjacent to playas: Water Resour. Res., v. 33, p. 2239-2252.

Schaap, M. G., and Leij, F. J., 1998, Database-related accuracy and uncertainty of pedotransfer functions: Soil Science, v. 163, p. 765-779.

Stovall, J., Rainwater, K., and Frailey, S., 2000, Groundwater modeling for the Southern High Plains: Final Report Submitted to Llano Estacado Regional Water Planning Group.

Thatcher, L. L., 1962, The distribution of tritium fallout in precipitation over North America: Bulletin of the International Association of Scientific Hydrology, v. 7, p. 48-58.

Thatcher, L. L., Janzer, V. J., and Edwards, K. W., 1977, Methods for determination of radioactive substances in water and fluvial sediments: U.S. Geological Survey Techniques of Water Resources Investigations, Chapter A-5, 95 p.

Wood, W. W., and Sanford, W. E., 1995, Chemical and isotopic methods for quantifying ground-water recharge in a regional, semiarid environment: Ground Water, v. 33, p. 458-468.

Wood, W. W., Rainwater, K. A., and Thompson, D. B., 1997, Quantifying macropore recharge: examples from a semi-arid area: Ground Water, v. 35, no. 6, p. 1097-1106.