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Impact of land use and land cover change on groundwater recharge and quality in the southwestern US

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Abstract

Humans have exerted large-scale changes on the terrestrial biosphere, primarily through agriculture; however, the impacts of such changes on the hydrologic cycle are poorly understood. The purpose of this study was to test the hypothesis that the conversion of natural rangeland ecosystems to agricultural ecosystems impacts the subsurface portion of the hydrologic cycle by changing groundwater recharge and flushing salts to underlying aquifers. The hypothesis was examined through point and areal studies investigating the effects of land use/land cover (LU/LC) changes on groundwater recharge and solute transport in the Amargosa Desert (AD) in Nevada and in the High Plains (HP) in Texas, US. Studies use the fact that matric (pore-water-pressure) potential and environmental-tracer profiles in thick unsaturated zones archive past changes in recharging fluxes. Results show that recharge is related to LU/LC as follows: discharge through evapotranspiration (i.e., no recharge; upward fluxes $<0.1\,\mathrm{mm\,yr^{-1}}$) in natural rangeland ecosystems (low matric potentials; high chloride and nitrate concentrations); moderate-to-high recharge in irrigated agricultural ecosystems (high matric potentials; low-to-moderate chloride and nitrate concentrations) (AD recharge: ~ 130–$640\,\mathrm{mm\,yr^{-1}}$); and moderate recharge in nonirrigated (dryland) agricultural ecosystems (high matric potentials; low chloride and nitrate concentrations, and increasing groundwater levels) (HP recharge: ~ 9–$32\,\mathrm{mm\,yr^{-1}}$). Replacement of rangeland with agriculture changed flow directions from upward (discharge) to downward (recharge). Recent replacement of rangeland with irrigated ecosystems was documented through downward displacement of chloride and nitrate fronts. Thick unsaturated zones contain a reservoir of salts that are readily mobilized under increased recharge related to LU/LC changes, potentially degrading groundwater quality. Sustainable land use requires quantitative knowledge of the linkages between ecosystem change, recharge, and groundwater quality.

Key words: agriculture, dryland, ecohydrology, global change, groundwater contamination, groundwater recharge, irrigation, land cover, land use, nitrate, nitrogen, water resources

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Introduction

Understanding impacts of land use/land cover (LU/LC) change on the hydrologic cycle is needed for optimal management of natural resources. The global impact of LU/LC change on the hydrologic cycle may surpass that of recent climate change (Vorosmarty *et al*., 2004). Impacts of LU/LC change on atmospheric components of the hydrologic cycle (regional and global climate) are increasingly recognized (Bonan, 1997; Pielke *et al*., 1998; Pitman *et al*., 2004). Impacts of LU/LC change on subsurface components of the hydrologic cycle are less well recognized, particularly

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groundwater recharge. The potential scale of subsurface impacts is large. Groundwater is Earth's largest freshwater resource. Reduced reliability of surface water supplies in the western US with projected climate change during the next century (Service, 2004) may result in increased reliance on groundwater. Widespread changes in LU/LC have occurred as a result of agricultural expansion. In the past 300 years, cultivated cropland has increased by factors of ~ 70 in the US and ~ 5 globally (Richards, 1990). The projected global increase of agricultural lands is $\sim 20\%$ over the next 50 years (Tilman *et al*., 2001).

In this study, rangeland is defined as uncultivated lands (grasslands and shrublands), excluding urban areas, dominated by natural vegetation and generally used for grazing by livestock or herbivorous wildlife. Agricultural areas are classified as irrigated or nonirrigated (dryland or rainfed). Most recharge studies have been conducted in natural rangeland ecosystems (Cook *et al*., 1989; Phillips, 1994; Tyler *et al*., 1996; Izbicki, 2002); however, replacing rangeland with agricultural ecosystems alters many of the parameters controlling recharge, such as climate, soils, and vegetation. Increased evapotranspiration (ET) because of large-scale irrigation alters regional climate through precipitation recycling (Moore & Rojstaczer, 2002; Adegoke *et al*., 2003). Irrigation increases the amount of water applied to the system, generally enhancing groundwater recharge (Roark & Healy, 1998; McMahon *et al*., 2003). Tillage affects recharge by changing soil structure (Leduc *et al*., 2001). Agricultural conversion alters key vegetation parameters that affect recharge, including fractional vegetation coverage, wilting point, and rooting depth. Reducing fractional vegetation coverage to zero during fallow periods between crop rotations can increase recharge, as shown in the Northern Great Plains, US (Miller *et al*., 1981). Lysimeter studies indicate that devegetation can increase recharge even in desert environments (Gee *et al*., 1994; Scanlon *et al*., 2005). The wilting point represents the minimum matric (pore-water-pressure) potential at which plants take up water. Increasing the wilting point from that typical of natural rangeland vegetation (matric potential, expressed in meters of water ~ -600 to -800 m for creosote; Odening *et al*., 1974; Smith *et al*., 1997) to much higher values typical of agricultural crops (~ -150 m; Savage *et al*., 1996) should increase groundwater recharge. Replacement of deep-rooted perennial Eucalyptus in Australia by shallow-rooted annual crops increased recharge by up to two orders of magnitude (Allison *et al*., 1990; Petheram *et al*., 2000). The situation is reversed in Argentina where shallow-rooted grasses are replaced by deep-rooted trees, decreasing recharge (Jobbágy & Jackson, 2004). Linkages between vegetation and hydrology are central to the emerging field of ecohydrology (Rodriguez-Iturbe, 2000; Newman *et al*., 2003).

Variations in recharge associated with LU/LC changes can have negative impacts on groundwater quality because thick unsaturated zones in semiarid and arid regions contain a reservoir of salts that accumulated over thousands of years (Allison *et al*., 1990; Phillips, 1994; Walvoord *et al*., 2003) and can be flushed into underlying aquifers. Increased recharge associated with agricultural development in southeastern Australia mobilized salts that had accumulated beneath native mallee eucalyptus vegetation, degrading groundwater quality (Allison *et al*., 1990). Miller *et al*. (1981) attributed saline seep development in the northern Great Plains to fallow periods in dryland agriculture, flushing salts from marine sediments in the unsaturated zone. Studies have shown that artificial recharge in southern California (US) can mobilize naturally occurring arsenic, chromium, and other salts impairing groundwater quality (Aiken & Kuniansky, 2002). Sustainable resource management planning requires considering the impacts of LU/LC changes on both the quantity and quality of groundwater.

A variety of approaches can be used to assess the impact of LU/LC changes on subsurface hydrology. The most direct approach is relating LU/LC changes to water-table fluctuations. Unsaturated-zone profiles of matric potential and water-borne tracers (natural and anthropogenic) archive past variations in recharge from LU/LC change, particularly in thick unsaturated zones in arid and semiarid regions. Correlating current LU/LC practices with recharge can be used to assess the impact of changes in LU/LC on recharge by using space as a proxy for time. Variations in groundwater chemistry with time provide information on the impact of LU/LC changes on water quality.

The purpose of the current study was to evaluate the impact of LU/LC change on groundwater recharge and associated groundwater quality in the US South-west. The study considers a range of LU/LC settings that includes natural rangeland ecosystems and irrigated and nonirrigated (dryland) agricultural ecosystems. The study is also unique in combining analyses of: (1) unsaturated-zone profiles of matric potential and environmental tracers, (2) multiseasonal time series of matric potentials, and (3) long-term saturated zone regional water-table and groundwater chemistry data. The Amargosa site in Nevada is typical of irrigated agriculture in desert regions. Study sites in the High Plains (HP; Texas) represent one of the largest agricultural regions in North America. Building on past research on other continents (e.g., Allison *et al*., 1990; Jobbágy & Jackson, 2004), results of this study have global implications for the relations between LU/LC changes and groundwater resources.

Materials and methods

This study evaluated the impact of LU/LC changes on groundwater quantity and quality in four areas, the Amargosa Desert (AD) and three sites in the High Plains in Texas (Fig. 1). Long-term (30 years) mean annual precipitation varies among the sites: 113 mm (AD), 500 mm (HP1), 440 mm (HP2), and 457 mm (HP3) (Table 1). All sites in this study were located in interfluvial geomorphic settings (i.e., areas that are not subject to inundation from streams or ephemeral lakes (playas)). Various approaches were used to quantify the impacts of LU/LC changes on groundwater recharge and quality. The different approaches complement each other in providing a comprehensive evaluation of LU/LC changes on groundwater quantity and quality at varying space and time scales. Profile and time-series data on matric potential in the unsaturated zone provide information on the direction of water movement and system response to variations in precipitation and irrigation. Unsaturated-zone profiles of environmental tracers integrate system response at the local scale to variations in climate and LU/LC changes over decadal to millennial timescales. Water-table fluctuations and trends in groundwater solutes (total dissolved solids (TDS), chloride, nitrate, and sulfate) provide information on regional response to LU/LC changes at decadal timescales.

Site characteristics and history

The Nevada site is located in the Amargosa Farms area of the Amargosa Desert, which constitutes the north-central part of the Mojave Desert (Fig. 1). Soil and sediment textures, from continuous borehole samples, are predominantly sands and gravels. Boreholes were sited in areas of natural rangeland vegetation (primarily creosote bush – *Larrea tridentata* and saltbush – *Atriplex contertifolia*) (one borehole) and irrigated agriculture (mostly alfalfa, various species) (six boreholes, two each in three irrigated fields, Fig. 2). Rangeland in this region is not grazed by livestock. Prior to agricultural conversion, all fields were covered by natural rangeland vegetation. Field 1 was converted in 1993 to produce alfalfa continuously under center-pivot irrigation. Fields 2 and 3 were converted in the early 1960s. Field 2 produced alfalfa continuously under wheel-line irrigation through 1978 and under center-pivot irrigation until 1983. Field 2 stood fallow from 1983 to 1991, when alfalfa production resumed. Field 3 produced turfgrass in the 1960s through early-mid 1970s was used intermittently from 1970s to 1980; produced oats, barley, and vegetables from 1981 to 1987; and produced alfalfa after 1987. Field 3 has been irrigated since the early 1960s. Information on the irrigation method prior to 1987 is not available. A center-pivot system was installed in 1987. Irrigation applications ranged from 2 m yr^{-1} (fields 1 and 2) to 2.7 m yr^{-1} (field 3). Commercial nitrogen fertilizer (90 kg ha^{-1}) was applied during each spring and summer to fields 1 and 2. Liquid fertilizer was applied to field 2 in spring 1992, when the field was brought back into production. Compost was used to fertilize field 3. Annual application rates were about 20 kg N ha^{-1} for fields 1 and 2, and 34 kg N ha^{-1} for field 3 (Stonestrom *et al*., 2003).

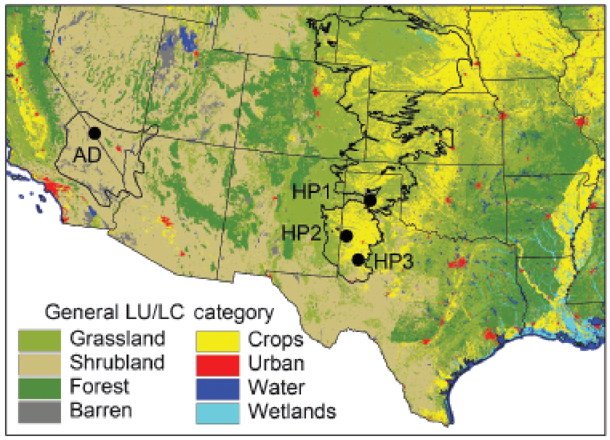

Fig. 1 Study area locations and generalized use/land cover (LU/LC) based on National Land Cover Data (NLCD) in the US Southwest. Rangeland ecosystem includes grassland and shrubland. The High Plains and the Mojave Desert are delineated. Several categories represent combinations of NLCD Classifications: Grassland (grassland/herbaceous); Forest (deciduous, evergreen, mixed forest); Crops (pasture/hay, row crops, small grains, fallow).

The US High Plains region (451 000 km^2) represents 27% of agricultural land and accounts for 30% of all irrigation groundwater usage in the US (Dennehy, 2000). The Southern High Plains (75 470 km^2) include the Panhandle region of Texas, generally south of the city of Amarillo and extends into eastern New Mexico. The area is topographically flat and drains internally to $\sim$ 20 000 playas. The rangeland ecosystem originally consisted of shortgrass prairie grazed by bison (Fahlquist, 2003). The HP aquifer was discovered in the late 1800s. Cattle ranchers began to settle at that time, followed by farmers practicing dryland agriculture. The discovery of oil in the 1920s heralded the beginning of widespread irrigated agriculture. Today the SHP region produces one-third of US beef cattle (Fahlquist, 2003).

Table 1 Mean annual precipitation, Cl concentration in precipitation (Cl_P), irrigation (Cl_I), fertilizer (Cl_F), total Cl input (Cl_{tot}), and irrigation application rate at the Amargosa Desert site fields 1–3 (AD1–AD3) and High Plains sites

Site	Precipitation ($mm\,yr^{-1}$)	Cl_P ($mg\,L^{-1}$)	Cl_P ($g\,m^{-2}\,yr^{-1}$)	Irrigation ($m\,yr^{-1}$)	Cl_I ($mg\,L^{-1}$)	Cl_I ($g\,m^{-2}\,yr^{-1}$)	Cl_F ($g\,m^{-2}\,yr^{-1}$)	Cl_{tot} ($g\,m^{-2}\,yr^{-1}$)
AD1	113	0.51	0.06	2.0	7.3	14.6	0.1	14.8
AD2	113	0.51	0.06	2.0	6.7	13.4	0.1	13.6
AD3	113	0.51	0.06	2.7	6.9	18.6	0.8	19.5
HP1	500	0.30	0.15	–	–	–	–	0.15
HP3	457	0.32	0.15	–	–	–	–	0.15

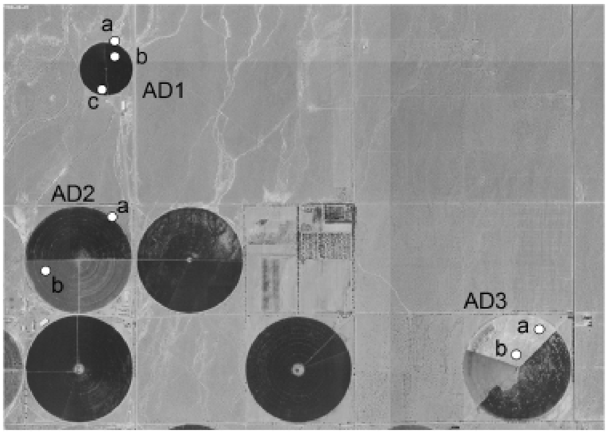

Fig. 2 Borehole locations in relation to center pivot irrigation systems at the Amargosa Desert, Nevada, site (6/1/98 photograph).

Cotton is the dominant crop (20% of US production), followed by corn and sorghum (Texas Water Development Board (TWDB), 2003; US Department of Agriculture, 2003). One-third of agricultural land is irrigated and accounts for 94% of total groundwater use within the SHP (Fahlquist, 2003).

Boreholes in the Texas High Plains 1 (HP1) site were drilled and sampled at three locations for matric potential and chloride, and matric potential was monitored at two of these locations in rangeland settings. Rangeland vegetation at this site generally consists of grasses, with some shrubs, and is used for grazing by livestock. Soil texture from borehole data ranges from clay to clay loam. Boreholes at the HP2 site were drilled as part of the USGS National Water Quality Assessment (NAWQA) program (Dennehy, 2000). LU/LC settings include rangeland (HP2a) and irrigated cotton (HP2b, HP2c). Soil texture from borehole data is predominantly gravelly sandy loam. The rangeland site is at the Muleshoe National Wildlife Refuge that was established in 1935 for migratory birds. Vegetation consists of short grasses with scattered mesquite, which is not grazed by livestock. Irrigation began in the late 1950s and early 1960s. Irrigation technology progressed from furrow (1950s) to hand-moved and side-roll sprinklers (1960–1980s), and finally to the center pivot. Reported irrigation amounts range from $0.3\,m\,yr^{-1}$ (HP2c) to $0.6\,m\,yr^{-1}$ (HP2b). Chemical and isotopic samplings of these sites are described in McMahon *et al.* (in press). Results of matric potential monitoring at these sites are described here. Sites in the HP3 region include one borehole in rangeland (sand dune area, bunch grasses (various species) and sparse mesquite – *Prosopis glandulosa*), one in irrigated cotton, and three in dryland cotton. Rangeland settings in this region are grazed by livestock. Soil texture from borehole data is predominantly sandy loam. Irrigation at this site began in 1955 with a side roll sprinkler system that was replaced by a center-pivot system in the late 1980s. Reported irrigation applications averaged about $0.45\,m\,yr^{-1}$.

Physical measurements

Soil water moves from regions of high to low total potential (sum of gravitational, matric, and osmotic potentials). Potential (energy) is reported herein per unit weight of water (i.e., as a head in meters of water). Gravitational potential is expressed as height relative to a common datum, defined herein for convenience as the water table. Matric (pressure) potential represents the interaction between the liquid and solid matrix of the soil and includes capillary and adsorptive forces and is always negative. Osmotic potential reflects energy resulting from solutes and is generally much smaller in magnitude ($\leq 10\%$) than matric potential in arid regions (Scanlon *et al.*, 2003). Water potential (sum of matric and osmotic potentials) is generally considered equivalent to matric potential because osmotic potential is small. Low (highly negative) matric potentials indicate dry conditions associated with negligible water flux, whereas high potentials (less negative, close to zero) indicate wet conditions associated with high water flux.

A variety of instruments are used to measure potentials in unsaturated soils and sediments. Instruments based on thermocouple psychrometers (Models SC-10X and CX2, Decagon Devices, Pullman, WA, USA) measure water potential, and heat dissipation sensors (HDS) and tensiometers measure matric potential (Andraski & Scanlon, 2002; Scanlon & Andraski, 2002). In this study, water potentials were measured in the laboratory using unsaturated-zone samples collected in moisture-tight containers from seven boreholes at the AD site, three boreholes at the HP1 site, and six boreholes at the HP3 site with a CX2 meter (AD samples) or with an SC-10X sample changer (HP1 and HP3 samples) (Decagon Instruments, Pullman, WA, USA; brand identification does not imply endorsement). These instruments measure the relative humidity of air brought into equilibrium with the sample, which is converted to water potential using the Kelvin equation (Jury *et al*., 1991). Measurement uncertainty is ± 20 m; therefore, these instruments are most accurate for drier samples (e.g., in rangeland sites). Matric potentials in the wet range (0 to -8 m; encountered at the HP3 site) were measured in the laboratory on unsaturated-zone soil samples from the field using tensiometers. Tensiometers measure the pressure of water inside a water-saturated ceramic cup equilibrated with the surrounding sample using a pressure transducer (Tensimeter; Soil Measurement Systems, Tucson, AZ, USA).

Matric potential was monitored in the field with HDS at the HP1 site in two rangeland settings and at the HP2 site in one rangeland and two irrigated settings. HDS (Model 229, Campbell Scientific Inc., Logan, UT, USA) measure matric potential via thermal properties of a calibrated porous element that is in hydraulic equilibrium with the surrounding soil (Flint *et al*., 2002; Scanlon & Andraski, 2002). The instruments were installed in shallow boreholes at depths ranging from 0.2 to 6 m (HP1 site), from 0.5 to 3 m (HP2 site), and in deep boreholes to 36.6 m depth (HP2 site) (Fig. 6). Shallow boreholes at the HP2 irrigated sites were drilled within the swing of the center pivot irrigation system, whereas deep boreholes were drilled at the edge of the irrigation system.

Water-table fluctuations can be used to estimate recharge rates in areas not impacted by large-scale pumping; e.g., in nonirrigated areas. The recharge rate (L/T) is

$$R = S_y \Delta h / \Delta t, \qquad (1)$$

where S_y is aquifer-specific yield (dimensionless = volume of water that drains by gravity flow per unit change in water-table height per unit area of aquifer), h is water-table height (L), and t is time (T) (Healy & Cook, 2002). The use of this equation assumes that all water-table changes result from recharge (i.e., groundwater pumpage, ET, and net lateral flow are negligible). Therefore, this approach should be applied only in nonirrigated areas. To the degree that water levels were locally affected by generally increasing pumpage for domestic supplies in nonirrigated settings, recharge estimates based on water-table fluctuations provide a lower bound on induced recharge resulting from LU change. Geographic information system (GIS) software was used to analyze long-term water-level changes in irrigated and nonirrigated areas in the High Plains in Texas on the basis of historical water-level data for the area; however, recharge rates were calculated only for nonirrigated areas (TWDB, 2003). The early water-table data were based on measurements between 1910 and 1980, with an average date of 1958 (5861 wells). A map approximating the pre-1980s water table was generated using 1 km^2 grid cells and compared with a similar map created using water levels ranging from 1981 to 2002, with an average date of 1999 (3837 wells). The spatial distribution of water-level changes, Δh, was mapped by subtraction. Maps were also made of the spatial distribution of dates of water-level measurements for both time frames (pre-1980s and post-1980s), and the spatial distribution of elapsed time between measurements, Δt, was calculated. Finally, the spatial distribution of recharge was estimated using Eqn (1) assuming a reasonable value of S_y (0.15; Knowles *et al*., 1984).

Unsaturated-zone core sample tracer analysis

Soil cores from AD (seven boreholes), HP1 (three boreholes), and HP3 (five boreholes) sites were analyzed in the laboratory for chloride concentrations in pore water. Nitrate was also analyzed in soil samples from the AD and HP3 sites. Gravimetric water content was measured by oven drying soil samples at 105 °C to constant weight. Double deionized water was added to the dry soils in a 1 : 1 weight ratio, and samples were either shaken periodically for 24 h (AD sites) or shaken continuously for 4 h (HP sites). Chloride and nitrate concentrations in supernatant liquid were measured by ion chromatography with $\pm 0.1\,\mathrm{mg\,L^{-1}}$ accuracy and converted to concentrations of soil pore water by dividing by gravimetric water content and multiplying by water density.

Recharge was quantified using tracer front displacement methods in sites that were converted from natural rangeland to irrigated agricultural ecosystems (Walker *et al*., 1991). Recharge was estimated from the velocity of the tracer front (v) as follows:

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

where θ is the average water content over this depth interval, and z_1 and z_2 are the depths of the chloride or nitrate fronts corresponding to times t_1 and t_2 related to the new (irrigated) and old (rangeland) land uses.

Recharge was also estimated using the chloride mass balance (CMB) approach (Allison & Hughes, 1983), which equates chloride inputs (precipitation and dry fallout, P, irrigation, I, and fertilizer, F, times the chloride concentration in precipitation and dry fallout, C_P, irrigation, C_I, and fertilizer, C_F) with chloride output (recharge rate, R, times chloride concentration in unsaturated-zone pore water, C_{uz}, or groundwater, C_{gw}):

$$PC_P + IC_I + FC_F = RC_{uz} = RC_{gw},$$
$$R = \frac{PC_P + IC_I + FC_F}{C_{uz}} = \frac{PC_P + IC_I + FC_F}{C_{gw}}. \tag{3}$$

Natural chloride deposition ranged from 0.06 to 0.15 $g\,m^{-2}\,yr^{-1}$ based on (1) bulk chloride concentrations in precipitation and dry fallout at the AD site in Nevada (5 years record; Stonestrom *et al*., 2003), (2) prebomb $^{36}Cl/Cl$ ratios at the HP1 site (Scanlon & Goldsmith, 1997), and (3) National Atmospheric Deposition program data on chloride concentrations in precipitation at the HP3 site from 1980 to 2002 (http://nadp.sws.uiuc.edu/), increased by a factor of two to account for dry fallout (Table 1). Chloride in irrigation water was based on samples from supply wells. Chloride from fertilizer was estimated from information provided by producers. The time (t) for a tracer to reach the water table from the land surface is calculated as follows:

$$t = \frac{L\theta}{R}, \tag{4}$$

where L is the travel distance from land surface to the water table, θ is average water content in the unsaturated zone over that distance, and R is the recharge rate. Additionally, the time required to accumulate chloride in the unsaturated zone was calculated by dividing the cumulative total mass of chloride from the surface to the depth of interest by the chloride input across the land surface.

Groundwater solutes

Groundwater quality changes in the High Plains in Texas were evaluated using GIS analysis to correct bias because of temporal and spatial clustering of well-water samples. Historical groundwater quality data were obtained from the Texas Water Development Board (2003). The early groundwater solute data were based on measurements between 1936 and 1980, with an average date of 1958 (2206 well-water samples). These data were compared with measurements from 1981 to 2000, with an average date of 1992 (1074 well-water samples). Solute distributions were skewed toward high values; therefore, analyses were based on $\log_{10}$ values. Data were grouped to reflect the predominant LU/LC category in the vicinity of each well according to the National Land Cover Data (NLCD; satellite imagery from $\sim$ 1992; Vogelmann *et al*., 2001). Urban areas were excluded. Irrigated areas were identified using the NLCD imagery classification results of Qi *et al*. (2002). Because of the checkerboard pattern of land use in much of the region, 1 km buffer zones around LU/LC categories were used, assigning the highest priority in the resulting overlapping cells to irrigated areas, assuming that irrigation would exert the greatest influence on local groundwater chemistry, secondary priority to dryland areas, and lowest priority to rangeland areas. Continuous 1 km^2 grid maps of solute concentrations were made for each time frame. Concentration distributions were determined for each time frame from the map grid cells that included groundwater sample information for either time frame (2530 cells). Groundwater quality changes were calculated by difference. Bruce & Oelsner (2001) showed that water quality varied with well type on the basis of a comparison of water quality between paired domestic and public water supply wells in the High Plains in Kansas. To assess the impact of well type, we compared changes in water quality over time by well type (domestic, irrigation, and public water supply wells).

Results and discussion

General relationships between LU/LC and groundwater recharge and quality

To summarize relationships between LU/LC and recharge among different sites, average matric potential was plotted against average chloride concentration for soil samples from the upper unsaturated zone beneath the bulk of active roots, 2–5 m depth zone (Fig. 3). The plot includes additional data from rangeland and irrigated sites in the AD, HP in Texas and Kansas (Prudic, 1994; Scanlon & Goldsmith, 1997; McMahon *et al*., 2003). Several additional boreholes in rangeland settings were sampled at the AD and HP1 sites but were omitted from Fig. 3 for clarity. Data for the Kansas site were linearly extrapolated from deeper measurements. The different LU/LC settings fall into three distinct groupings: (1) rangeland sites – low matric potentials and high chloride concentrations, (2) irrigated sites – intermediate matric potentials and chloride concentrations, and (3) dryland sites – high matric potentials and low chloride concentrations.

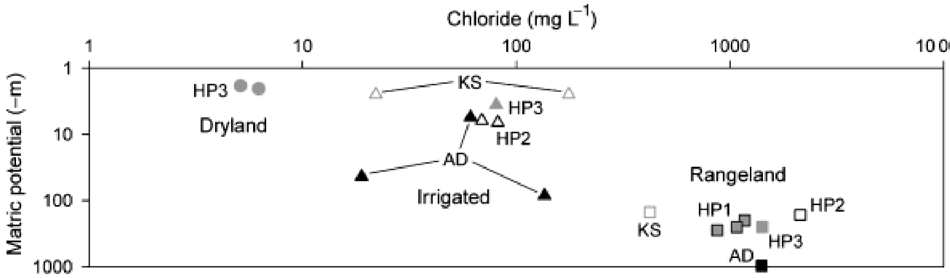

Fig. 3 Relationship between pore-water chloride concentration and matric potential (expressed as a head, in meters of water) for boreholes lacated in rangeland (squares), irrigated agricultural (triangles), and dryland agricultural (circles) ecosystems. Values shown are averages for measurements at 2–5 m depth.

Differences in mean chloride concentrations for all three LU/LC populations are significant to $P<0.01$ (two-tailed t-test). Differences in mean matric potential in rangeland and agriculture (dryland and irrigated) are significant to $P<0.01$. Some matric-potential scatter in the AD irrigated profiles may be attributed to use of a water activity meter, which has high uncertainties because of the logarithmic relationship between water potential and relative humidity (Gee *et al.*, 1992).

Rangeland ecosystems

Data for rangeland ecosystems at all sites have similar characteristics: low water potentials, upward potential gradients, and bulge-shaped chloride profiles. Representative rangeland profiles are shown for the AD and HP3 sites (Figs. 4a, b and 5a, b). Water potentials and total potentials are similar because gravitational potentials (referenced to the water table) were small relative to water potentials (Figs. 4a and 5a). Minimum water potentials near the root zone range from −1900 m (AD1a) to −300 m (HP3). Total potential increases with depth below the root zone, indicating upward water movement. Water content varies with soil texture in the different regions. Average water content (2–5 m depth) was lowest at AD1a (0.05 $m^3 m^{-3}$; sand and gravel), higher in the HP3 sites (0.09–0.15 $m^3 m^{-3}$; sandy loam) and highest at the HP1 site (0.16–0.17 $m^3 m^{-3}$; clay to clay loam). The unsaturated zone in rangeland settings contains a reservoir of solutes. Chloride profiles are bulge shaped, with peak concentrations from 2344 mg L^{-1} (HP3; Fig. 5b) to 4171 mg L^{-1} (HP1) near the root zone. The depth of the chloride peak ranges from 0.9 m (HP1) to 3.8 m (HP3; Fig. 5b). The total amount of chloride represents accumulation times of ~ 6000 years (HP1 site) to ~ 10 000–12 000 years (AD and HP1 sites). The profile at the HP3 site did not extend deep enough for the vertical extent of the chloride bulge or corresponding chloride accumulation time to be estimated. Chloride accumulation times at

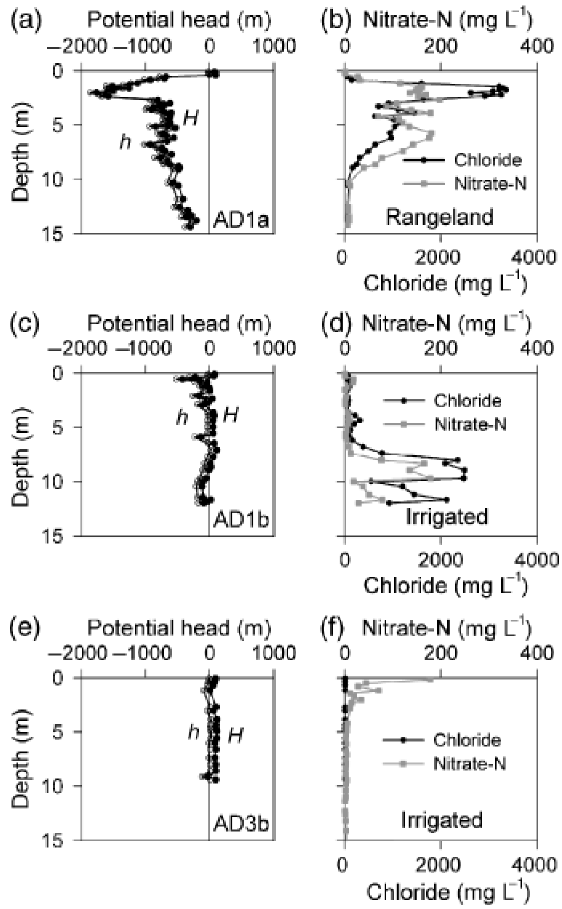

Fig. 4 Unsaturated zone potential and pore-water chloride and nitrate concentration profiles for (a, b) borehole AD1a in a rangeland setting and (c, d) borehole AD1b and (e, f) borehole AD3b in irrigated agricultural settings at the Amargosa Desert site. h, water potential, H: total potential.

the HP1 site are consistent with radiocarbon dates of paleosols overlain by sand dunes that record a shift from cooler, wetter conditions in the Pleistocene and early Holocene to warmer, drier conditions about 6000 years ago (Olson & Porter, 2002). Isotopic data ($\delta^{13}C$)

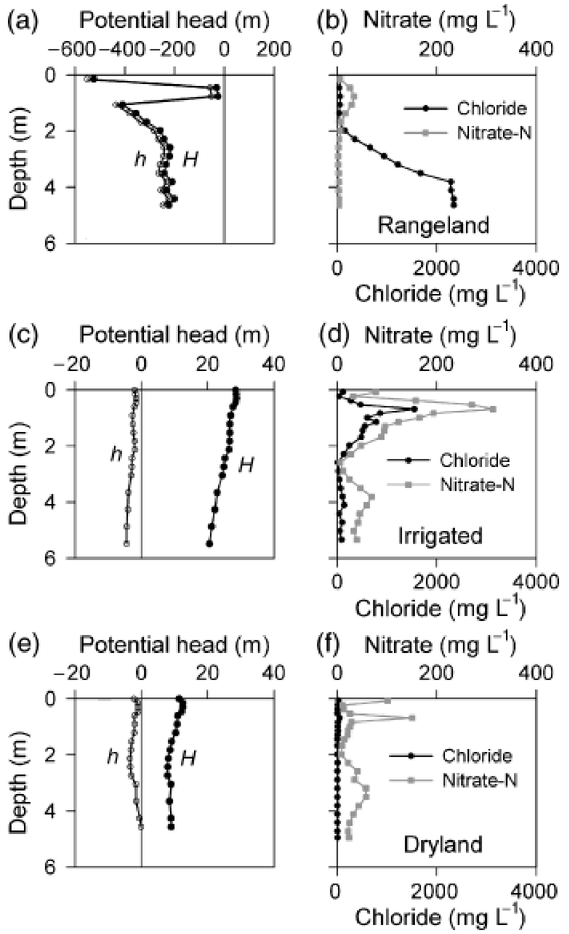

Fig. 5 Unsaturated zone potential and pore-water chloride and nitrate concentration profiles in (a, b) rangeland, (c, d) irrigated agricultural, and (e, f) dryland agricultural ecosystems at the HP3 site. *h*, matric potential, *H*: total potential.

also record a shift from C3 plants typical of cooler, wetter climates to C4 plants characteristic of warmer, drier climates at this time. The accumulation times for some of the HP1 and AD sites are similar to that of the Pleistocene/Holocene boundary about 10 000–15 000 years ago and are similar to bulge-shaped chloride profiles in interfluvial basins throughout the US Southwest (Scanlon, 1991; Phillips, 1994; Tyler *et al*., 1996). Lower chloride concentrations below this zone correspond to downward water fluxes during the Pleistocene of about 0.5 mm yr^{-1} (AD1a and HP1 site) to 2 mm yr^{-1} (HP1 site), assuming no change in the chloride deposition rate. The nitrate-N profile at the AD site is also bulge shaped, with peak concentrations of 179–198 mg L^{-1} at depths of 2.7–5.7 m (Fig. 4b). Nitrate-N concentrations in rangeland settings at the HP2 (McMahon *et al*., in press) and HP3 (Fig. 5b) sites are much lower than those at the AD site (Fig. 4b). The lack of high nitrate concentrations in the unsaturated zone beneath rangeland settings suggests more efficient nitrate extraction by plants in these settings, leading to a lack of nitrate buildup beneath the root zone. However, the low nitrate concentrations in these profiles do not necessarily represent the entire rangeland setting in the HP in Texas, given the high spatial variability of subsoil nitrate accumulations in rangeland settings throughout the southwestern US (Walvoord *et al*., 2003).

Matric potentials monitored in rangeland settings at the HP1 and HP2 sites are similar. Data from the HP2 site are described in this section. Matric potentials at this site were generally low, and total potential increases with depth, indicating upward water movement (Fig. 6b). Wetting fronts penetrated to a maximum depth of 1 m in March/April 2002 and 2004 and in October 2002 in response to high precipitation (Fig. 6a). Increases in matric potentials occur at progressively greater depths with time, indicating predominantly piston-type flow. Wetting during the summer generally only occurred to 0.3 m, and infiltrated water was removed rapidly by vegetation. The efficiency of rangeland vegetation in using available water is clearly seen in the data from summer 2004, when 354 mm of precipitation was recorded (mid-June–August); however, water infiltrated only to 0.3 m and was removed by mid-September. Sharp decreases in matric potential occur each year in spring (April–May to June) when vegetation becomes active and depletes soil moisture. Drying occurs at all depths to 1 m more or less in unison, indicating systematic and effective removal of water from the root zone by plants. The profile was driest during the latter half of 2003 because precipitation in 2003 (205 mm) was much lower than in 2002 (500 mm).

Irrigated agricultural ecosystems

The impact of replacing rangeland with irrigated agriculture is archived in the unsaturated zone of recently converted irrigated sites (1993, AD1b; Fig. 4c, d; AD1c; Table 2), whereas profiles that have been irrigated for longer times record only irrigated conditions (1960s; AD3b, Fig. 4e, f; HP3, Fig. 5c, d). For example, high chloride and nitrate-N concentrations near the base of the AD1b profile represent downward displacement of solutes that accumulated previously near the root zone when the site was undisturbed rangeland (Fig. 4d). In contrast, low chloride and nitrate-N concentrations in the upper 8 m depth represent the chemical composition of return flow from the 2 m yr^{-1} of irrigation that started in 1993. The vertical tracer displacement caused by irrigation indicates macroscopic water velocities of 0.8–0.9 m yr^{-1}

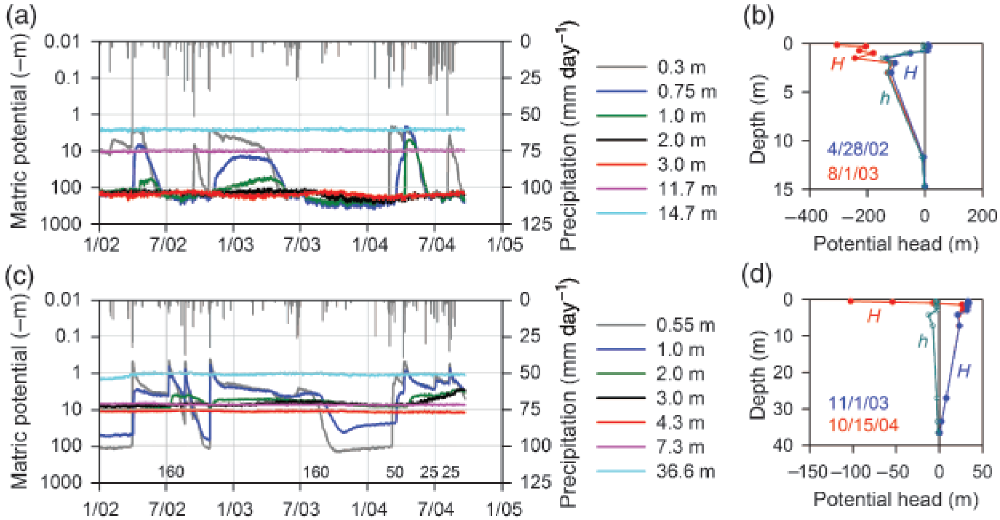

Fig. 6 Time series of matric potential and precipitation (a, c) and synoptic profiles of matric and total potential (b, d) at the HP2 site in rangeland (a, b) and irrigated (c, d) agricultural ecosystems. Total potential profiles (H, solid symbols) represent both wet and dry periods while matric potential head profiles (h, open symbols) represent only wet periods (for clarity). Irrigation application amounts (e.g., 160-mm) and times (shading) are shown for irrigated site (c).

Table 2 Estimated recharge rates using the tracer front displacement (TFD) method (Eqn (2)) for boreholes located at the Amargosa Desert (AD) site fields 1 and 2 (irrigated)

BH	Tracer	Z_{int} (m)	ΔZ (m)	T (years)	V (m yr^{-1})	θ_{int} (m^3 m^{-3})	R_{TFD} (mm yr^{-1})
AD1b	Cl	1.8–8.9	7.1	8	0.9	0.22	200
	NO_3-N	1.6–8.3	6.7	8	0.8	0.22	180
AD1c	Cl	1.8–13.3	11.5	8	1.4	0.20	280
	NO_3-N	1.6–12	10.4	8	1.3	0.19	250
AD2a	NO_3-N	0–9.2	9.2	9	1	0.15	150
AD2b	NO_3-N	0–7.7	7.7	9	0.9	0.17	150

BH, borehole; Z_{int}, displacement interval depths; ΔZ, displacement distance; T, displacement time; V, average downward displacement velocity; θ_{int}, depth-weighted average water content; R_{TFD}, recharge rate (Eqn (2)).

and recharge rates of 180–200 mm yr^{-1} (Eqn (2); Table 2). The similarity in recharge estimates based on chloride and nitrate-N increases confidence in these recharge estimates. Recharge was also estimated using the CMB approach applied to the irrigated portion of the profile (130 mm yr^{-1}; Table 3). Chloride was derived primarily from irrigation water; chloride input from commercial fertilizer (<2% Cl by weight; application rate 0.1–0.9 g m^{-2} yr^{-1}) was relatively low (Table 1). The CMB flux estimate for the AD1c profile is about two times higher than the estimate from the chloride and nitrate-N front displacements and reflects uncertainties in these recharge estimates (Tables 2 and 3).

Fields 2 and 3 at the AD site were irrigated for much longer (early 1960s) than field 1 (1993); thus, irrigation water had completely flushed the preirrigation chloride and nitrate-N bulges from three of the four measured profiles (Fig. 4e, f). One of the profiles (AD3b) is affected by a calcite-cemented impeding layer that causes lateral flow. The chloride deposition flux in field 2 is similar to that in field 1 (Table 1). Variable chloride concentrations in field 2 profiles (AD2a, AD2b) result in recharge rates ranging from 190 to 430 mm yr^{-1} (Table 3). Higher irrigation application in field 3 results in higher chloride input from irrigation (18.6 g m^{-2}; Table 1). Compost was used to fertilize field 3 (0.4–0.7 kg m^{-2}) once every 2 years and included 10–15 g N kg^{-1} of fertilizer and 3 g Cl kg^{-1} of fertilizer, resulting in a chloride input of 0.8 (0.6–1) g m^{-2} yr^{-1}. Recharge rates in this field range from 390 to 530 mm yr^{-1} (Table 3). Use of liquid fertilizer in field 2 (AD2a, AD2b) when it was put back into production in spring 1992 provided an additional tracer pulse to estimate recharge rates (Eqn (2)). Nitrate-N concentrations were five to eight times higher at the peak relative to the average nitrate-N concentrations above the peak (Fig. 7). Recharge

Table 3 Estimated recharge rates using the chloride mass balance (CMB) method at the AD and HP3 sites

Site	LU	BH	Cl_{dep} ($g\,m^{-2}\,yr^{-1}$)	Z_{int} (m)	θ_{int} ($m^3\,m^{-3}$)	Cl ($mg\,L^{-1}$)	R_{CMB} ($mm\,yr^{-1}$)	V ($m\,yr^{-1}$)
AD	Irrigated	1b	14.8	1.5–5.9	0.17	113	130	0.8
		1c	14.8	1.2–11.0	0.17	23	640	3.8
		2a	13.6	0.8–9.8	0.16	32	430	2.7
		2b	13.6	1.2–9.7	0.16	71	190	1.2
		3a	19.5	0.9–9.9	0.15	50	390	2.6
		3b	19.5	0.8–16.0	0.22	37	530	2.4
HP	Dryland	3b	0.15	1.5–4.7	0.15	4.7	32	0.2
		3c	0.15	2.0–4.3	0.09	13	12	0.1
		3d	0.15	2.2–2.9	0.09	16	9	0.1
		3e	0.15	1.7–4.4	0.15	6.1	25	0.2

LU, land use; BH, borehole; Cl_{dep}, chloride deposition rate (Table 1); Z_{int}, depth interval for CMB calculations; θ_{int}, average water content; Cl, average chloride concentration; R_{CMB}, recharge rate (Eqn (3)); V, average downward water velocity calculated from R_{CMB}; AD, Amargosa Desert; HP, High Plains.

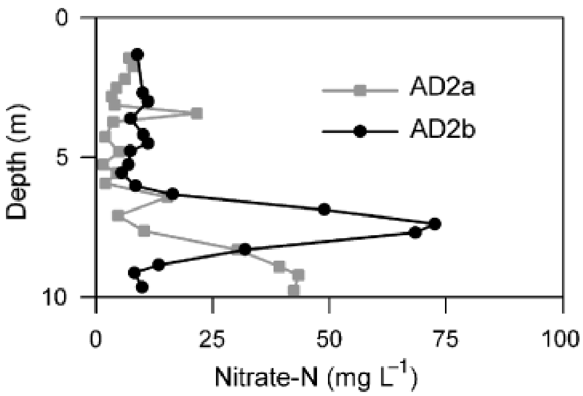

Fig. 7 Nitrate–N concentration profiles for boreholes AD2a and AD2b at the Amargosa Desert site showing download displacement from irrigation of nitrate pulse from liquid fertilizer application in 1992.

estimates based on the nitrate-N pulse (150 $mm\,yr^{-1}$) were less than those based on the CMB approach (190–430 $mm\,yr^{-1}$; Tables 2 and 3).

Recharge estimates beneath all three irrigated fields (130–640 $mm\,yr^{-1}$) represent 6–30% of the applied irrigation water; however, all except one value falls within the 7–32% range. The resultant travel times to the water table (35 m deep) range from 9 to 46 years (Eqn (4)). Average water contents in irrigated profiles in all three fields (0.15–0.22 $m^3\,m^{-3}$) are much higher than that in the rangeland setting (AD1a; 0.05 $m^3\,m^{-3}$). Measured water potentials are also higher beneath irrigated fields than beneath rangeland (Fig. 4a, c, e).

Much lower irrigation application rates (~ 0.3–0.6 $m\,yr^{-1}$) in the HP3 site result in lower drainage rates below the root zone. Bulge-shaped tracer profiles with peak concentrations of 1550 $mg\,L^{-1}$ chloride and 313 $mg\,L^{-1}$ nitrate-N are attributed to low irrigation rates and evapoconcentration of salts, both applied and naturally occurring (Fig. 5d). This process could ultimately result in salinization of soils. Measured matric potentials are relatively high (≥ -5 m). Total potential decreases with depth, indicating downward water flux (Fig. 5c). The CMB approach was not applied at the HP3 site because of lack of data on agricultural chloride inputs through time.

The regional impacts of irrigation can be seen in groundwater quality changes in the Southern High Plains (SHP). Increases in median solute concentrations ranged from 34% (TDS) to 221% (nitrate-N) beneath irrigated areas in area B in the SHP (Table 4, Figs 8 and 9). The number of cells estimated to contain groundwater contaminated by nitrate (drinking water standard 10 $mg\,L^{-1}$ nitrate-N) increased from 3% to 18%. Restricting comparison with irrigation wells resulted in a 38% increase in TDS and a 445% increase in nitrate, which is similar to the results obtained from comparing water quality beneath irrigated areas using the GIS analysis (Tables 4 and 5). McMahon *et al*. (2004) showed that water quality improves with depth below the water table in the HP in Kansas. Our analysis showed similar trends in the SHP. However, median water quality degraded over time despite an increase in median well depth (Table 5). Increases in solutes in groundwater are attributed to the leaching of salts that accumulated naturally in the soils over thousands of years prior to cultivation, application of fertilizers, and evapoconcentration of applied groundwater in irrigation. The magnitude of the impact of irrigation on groundwater quality in the SHP is attributable to the long history and high density of irrigation. That similar impacts are not evident in the AD region reflects the sparseness of irrigated fields, the relatively short time

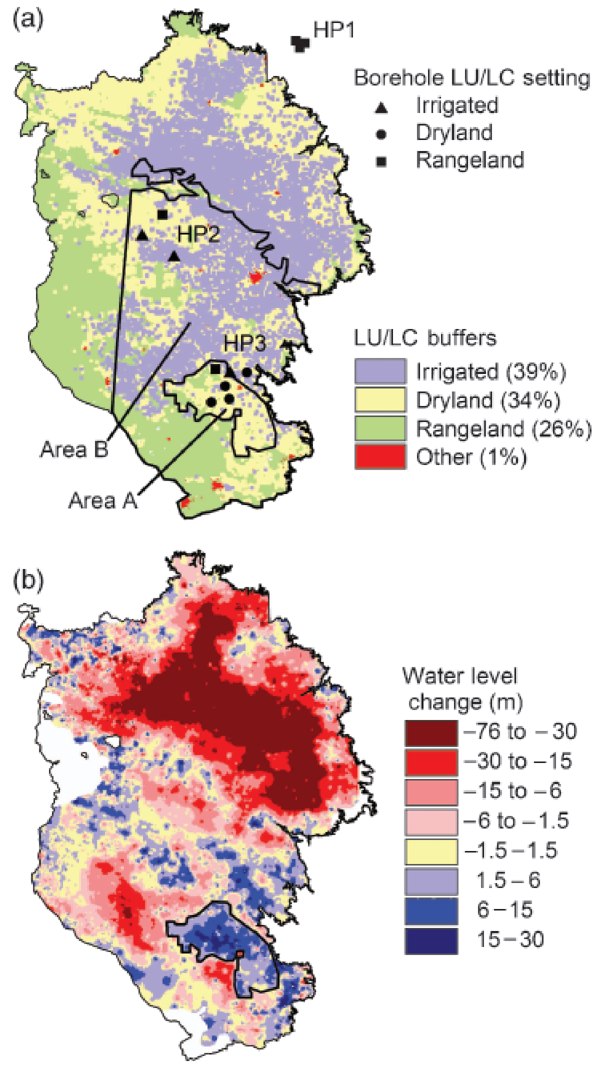

Fig. 8 Spatial distribution of (a) dominant land use/land cover (LU/LC) category buffer zones and (b) water level changes in the Southern High Plains in Texas and New Mexico. Borehole locations symbolized by LU/LC category are shown for the HP1, HP2, and HP3 sites. Area A is 3400 km^2 and dominated by dryland farming that was analyzed for recharge (Eqn (1)). Area B is 32 400 km^2 and includes irrigated and dryland agricultural areas that were analyzed for the impact of LU/LC on groundwater quality. Bank areas in the water-level-change map indicate areas with no data. Actual area percentages are irrigated, 11%, Dryland, 41%; Rangeland, 46%; and Other, 2%.

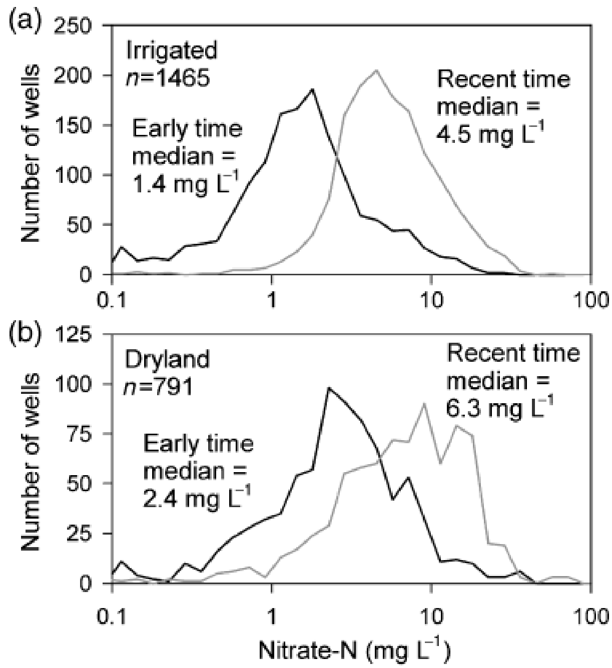

Fig. 9 Temporal changes in the distribution of ground water nitrate-N concentrations in areas dominated by (a) irrigated and (b) dryland agriculture in the Southern High Plains in Texas (Area B, Fig. 8). Median elapsed time is 29 years. $\log_{10}$ values for the four populations shown are normally distributed, with Shapiro-Wilk W-statistic values for ranging from 0.87 to 0.99 (all $P < 0.001$).

that the area has been under cultivation, and the screening of irrigation wells at large depths below the water table.

Matric potential monitoring at the HP2 site provides information on flow processes related to irrigation. Data from one of the irrigated sites (HP2b; Fig. 6c, d) are typical. Monitoring data indicate that infiltrated water regularly penetrates deeper in the irrigated site (2–3 m) than in the nearby rangeland site (≤ 1 m) (Fig. 6a, c). Furthermore, irrigation sites are more susceptible to deep percolation during naturally occurring wet periods. Although wetting begins at similar times each year in irrigated and rangeland sites, irrigated profiles remain wet for much longer because of irrigation applications in the summer. Drying in the rangeland profile began in April–May each year, whereas drying in the irrigated profiles began later, in mid-July to mid-August. Total potential gradients below the root zone at both irrigated sites have indicated consistently downward flow (Fig. 6d), whereas flow has consistently been upward at the rangeland site (Fig. 6b).

Dryland agricultural ecosystems

The impact of dryland agriculture on recharge was evaluated in the SHP using GIS techniques. Rising water tables in dryland-dominated areas could be related to climate variability instead of LU/LC changes. However, precipitation records for the period of LU/LC change show no statistically significant ($P = 0.01$) long-term trends (seasonal Mann–Kendall test; Hirsch & Slack, 1984). Moreover, restriction of water-table increases to dryland areas indicates that climate forcing

Table 4 Temporal changes (%) in groundwater quality by land use category for area B in the Southern High Plains (Fig. 8)

	Rangeland ($N = 274$)			Irrigated ($N = 1465$)			Dryland ($N = 791$)		
Analyte	Begin	End	Change (%)	Begin	End	Change (%)	Begin	End	Change (%)
TDS	1054	1020	−3	673	900	34	832	1088	31
Nitrate-N	2.1	3.2	52	1.4	4.5	221	2.4	6.3	163
Chloride	215	215	0	100	167	67	150	214	43
Sulfate	233	230	−1	153	218	42	190	234	23

Values (mg L^{-1}) represent median analyte concentrations at the beginning and end of an average 34-year period (range 10–60 years). *N*, number of grid cell locations used in analysis; TDS, total dissolved solids.

Table 5 Temporal changes (relative %) in groundwater quality by dominant land-use category and selected well uses (domestic, irrigation, and public supply) for wells located in area B in the Southern High Plains (Fig. 8)

		Domestic				Irrigation				Public			
LU/LC	Time frame	#	*D*	TDS	*N*	#	*D*	TDS	*N*	#	*D*	TDS	*N*
Rangeland	Begin	154	24 (151)	1074 (150)	3.2 (44)	76	47 (72)	674 (76)	1.4 (75)	26	57 (26)	811 (26)	1.4 (25)
	End	55	34 (44)	1162 (48)	5.7 (54)	18	41 (14)	772 (17)	3.4 (18)	61	57 (61)	890 (61)	2.0 (61)
	Change (%)			8	78			15	143			10	43
Dryland	Begin	177	30 (164)	820 (160)	3.3 (58)	229	46 (192)	652 (229)	1.2 (220)	10	46 (10)	727 (10)	1.8 (7)
	End	106	36 (70)	1155 (90)	7.7 (106)	65	48 (43)	1162 (57)	7.0 (64)	33	46 (31)	849 (33)	4.5 (33)
	Change (%)			41	133			78	483			17	150
Irrigated	Begin	148	31 (133)	693 (139)	3.2 (75)	700	49 (622)	626 (698)	1.1 (678)	14	41 (13)	806 (14)	2.7 (11)
	End	142	38 (95)	955 (101)	8.0 (142)	258	59 (162)	866 (211)	6.0 (258)	51	50 (48)	833 (50)	5.4 (51)
	Change (%)			38	150			38	445			3	100

Values for depth (*D*, m), total dissolved solids (TDS, mg L^{-1})) and NO_3-N (*N*, mg L^{-1}) represent median values at the beginning and end of an average 34-year period (range 10–60 years) for the number of wells shown in parentheses. Not all data were available for all wells. Numbers of wells within each category and time frame (#) are shown for reference. For example, for domestic-use wells located in rangeland areas, some data were available for a total 154 wells in the beginning time frame, for which the median depth of 151 wells was 24 m, median TDS for 150 wells was 1074 mg L^{-1}, and median NO_3-N for 44 wells was 3.2 mg L^{-1}. Note that most median well depths remained the same or increased with time but that all median solute concentrations increased with time. LU, land use; LC, land cover.

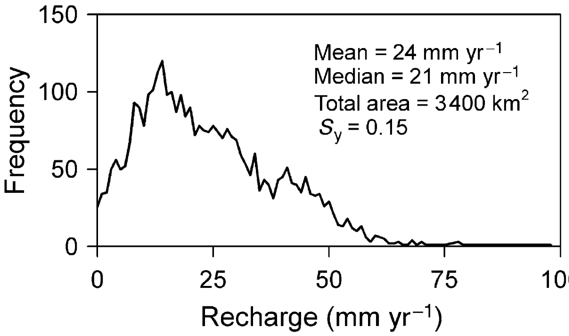

Fig. 10 Frequency distribution of recharge for Area A (Fig. 8).

alone cannot be the cause. Water-table increases ranged from 1.5 to 23 m and averaged 6.7 m in approximately 3400 km^2 of SHP dominated by dryland agriculture (area A, Fig. 8). The water-table increases occurred over periods ranging from 5 to 67 years and averaging 43 years and are in general agreement with a larger scale compilation of water-table changes for the entire HP (McGuire, 2001). The increases cannot be explained by water-table rebound resulting from reduction in irrigation pumpage because these areas have never been irrigated. Recharge estimates average 24 mm yr^{-1} and range from 4 to 57 mm yr^{-1} (Eqn (1)) and represent an increase of 3.4×10^9 m^3 in recoverable water (specific yield = 0.15; Knowles *et al.*, 1984) (Fig. 10). The average recharge rate (24 mm yr^{-1}) represents 5% of mean annual precipitation (457 mm yr^{-1}, Table 1) in this region. Water-table increases generally occurred earlier at shallower depths and propagated to greater depths over time (Fig. 11).

Increased recharge in the Southern High Plains could have resulted from either focused recharge beneath playas because of higher runoff from dryland areas or

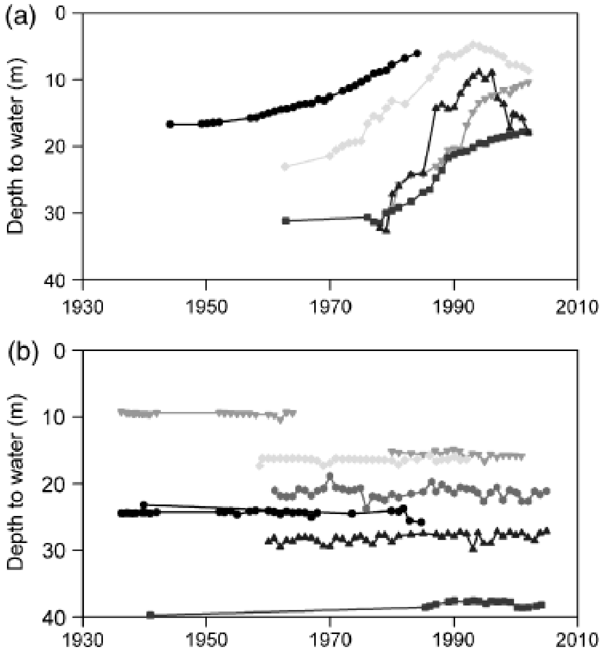

Fig. 11 Temporal changes in water-table depth for representative wells in predominantly (a) dryland (Area, Fig. 8) and (b) rangeland areas (Southwest of Area A).

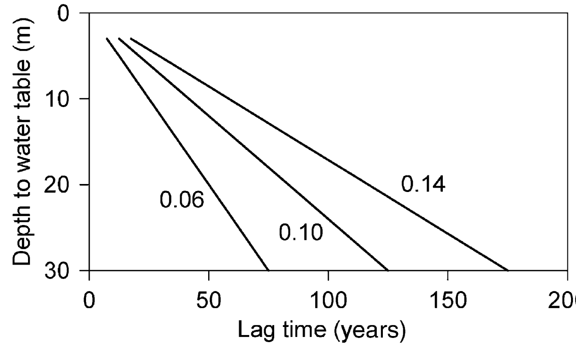

Fig. 12 Calculated lag time to the water table based on Eqn. 4 for a recharge rate of 24 mm yr^{-1} and unsaturated zone water contents ranging from 0.06 to 0.014 m^3m^{-3}.

diffuse (areally distributed) recharge beneath dryland areas or some combination of the two. Increased recharge beneath dryland agriculture in the Niger valley in Africa was attributed to increased runoff resulting in focused recharge because the region is internally drained (Leduc *et al.*, 2001). However, playa density is extremely low in some dryland areas in the SHP where water-table rises were large, showing that diffuse recharge also plays a role. Evidence of diffuse recharge in interplaya settings is provided by high matric potentials (≥ -4 m) (Fig. 5e). Total potential generally decreased with depth, indicating downward water movement and recharge. Low chloride concentrations in these profiles with average values below the root zone (1.5–2.2 m) of 4–17 mg L^{-1} correspond to recharge rates of 9–32 mm yr^{-1}, representing 2–7% of long-term precipitation (Table 3). These recharge rates represent time-averaged values for 50–100 years based on chloride accumulation times, in agreement with the rates estimated from water-table increases. The data suggest that recharge occurs beneath dryland agriculture, whereas it does not occur beneath rangeland ecosystems. Changes associated with cultivation thus best explain the observed increases in water-table elevations over the past few decades. The time lag between increased water flux below the root zone and water-table rise varies depending on the water flux, the average water content in the unsaturated zone, and the depth of the water table (Eqn (4); Fig. 12). Although it is difficult to estimate time lags precisely because of changing water-table depths, the travel times to 21 and 14 m depths (average of early and late water-table depths in area A) are 90 and 60 years, respectively, assuming an average water content of 10% and an average recharge rate of 24 mm yr^{-1}. This chronology is consistent with the widespread introduction of dryland farming following the Civil War.

Groundwater quality has degraded beneath dryland-dominated areas in the SHP (area B, Figs 8 and 9, Table 4). The greatest increase was in nitrate-N (163%), whereas increases in other ions ranged from 23% (sulfate) to 43% (chloride). Groundwater contamination by nitrate (≥ 10 mg L^{-1} nitrate-N) increased from 6% to 33% in dryland areas. Degradation of groundwater quality is attributed to leaching of salts that accumulated naturally in the soils and to application of fertilizers.

Comparison of recharge rates among LU/LC settings

Estimated recharge rates vary widely among and within LU/LC settings (Fig. 13). Discharge is occurring through ET in rangeland ecosystems, as shown by upward total-potential gradients and long-term chloride accumulations in the root zone (see also Prudic, 1994; Scanlon & Goldsmith, 1997; Stonestrom *et al.*, 2003). Discharge fluxes are generally low (<0.1 mm yr^{-1}), as shown by modeling analyses (Scanlon *et al.*, 2003; Walvoord *et al.*, 2004). These results are consistent with regional analyses throughout the US Southwest, which indicate that there has been virtually no recharge in interfluvial basin floor settings since the late Pleistocene about 10 000–15 000 years ago (Phillips, 1994; Tyler *et al.*, 1996; Scanlon *et al.*, 2003).

Recharge rates in irrigated settings vary over an order of magnitude based on data from this study and other studies in New Mexico and Kansas (19–

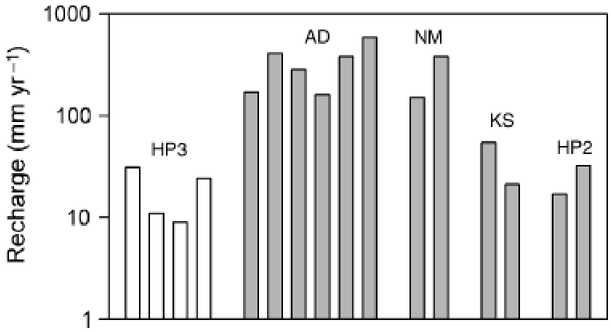

Fig. 13 Range of recharge rates for dryland (unshaded) and irrigated (shaded) agriculture in the SW U.S. HP3, High Plains in Texas; AD, Amargosa Desert; NM, New Mexico (Roark and Healy, 1998); KS, High Plains in Kansas (McMahon *et al.*, 2003) and HP2, High Plains in Texas (McMahon *et al.*, in press). Average recharge rates were calculated when more than one recharge estimate was available for a site.

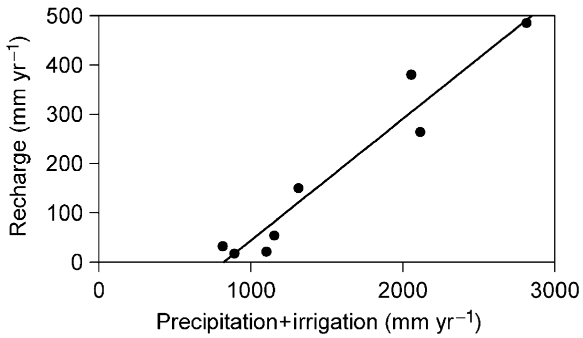

Fig. 14 Relationship between recharge and irrigation + precipitation ($r^2 = 0.94$) based on data from the SW US.

485 mm yr^{-1}, Fig. 13). Recharge rates correlate with mean annual irrigation and precipitation amounts ($r^2 = 0.94$; Fig. 14) and represent 2–19% of the irrigation plus precipitation amounts. Highest recharge rates are recorded in Nevada sites, intermediate in Kansas sites, and lowest in Texas sites. Irrigation rates are inversely related to mean annual precipitation ($r^2 = 0.87$). Irrigation rates were reported by farmers and not metered; therefore, they are somewhat uncertain. Estimated travel times to the water table (Eqn (4)) range from 9 to 46 years at the AD site ($\sim$ 35 m deep water table) to 132–373 years at the HP2 site (33–47 m deep water table; McMahon *et al.*, in press).

Recharge occurs in dryland agricultural regions, as evidenced by water-table increases, downward total-potential gradients, and low chloride concentrations in the unsaturated zone (Figs. 3, 5e, f and 8). Water-table increases were also recorded beneath dryland agriculture in the HP in Oklahoma ($\leq$ 6 m in 50 years; Luckey & Becker, 1999). Recharge rates equal to 4% of precipitation were deduced from groundwater modeling (Luckey & Becker, 1999), which is consistent with the 5% estimated in this study for the mean recharge in area A. The increased recharge in dryland areas may be attributed to fallow periods in combination with increased permeability of surficial soils because of plowing. The importance of fallow periods was shown in the Northern Great Plains by reduced recharge when crop-fallow rotations were replaced by perennial alfalfa in experimental plots (Halvorsen & Reule, 1980). Fallow periods were also shown to increase soil water storage and drainage compared with continuous cropping in Australia (O' Connell *et al.*, 2003). The fallow period in the cultivated HP3 site extends from about late November to early June, when about half of the annual precipitation occurs (52–56%). In contrast, native vegetation becomes active in the spring, drying out the soil beneath uncultivated, rangeland areas (Fig. 6a). LU and LC differences thus account for the large contrasts in recharge between dryland and rangeland vegetation. Agricultural terracing in dryland areas may further enhance recharge by reducing runoff and increasing soil water storage. Expanded monitoring of soil moisture and matric potential in dryland areas will provide additional insights into the controls and timing of recharge in these areas, as well as the role of fallow periods.

Impacts of potential future LU/LC changes on groundwater recharge and quality

Relations between LU/LC settings and groundwater recharge evident in this study allow a better assessment of impacts of future LU/LC changes on the quantity and quality of groundwater. Major drivers for LU/LC changes include economics, resource availability (e.g., especially groundwater for irrigation), and biological-resource management through State and Federal programs and policies. Decreasing groundwater availability in the SHP has resulted in increasingly efficient irrigation systems. Recharge from irrigation decreases as irrigation application decreases (Fig. 14). Conversely, soil and groundwater salinization increases with increased irrigation efficiency. For example, a 95% efficient irrigation system (5% of water drains below the root zone) should result in a 20-fold increase in chloride in recharge water (chloride is excluded from crop water uptake) as well as nitrate not used by plants. Studies in Kansas suggest that increasingly efficient irrigation systems are resulting in more areas being irrigated with no net benefit to groundwater quality or quantity (McMahon *et al.*, 2003). Degradation of groundwater quality caused by irrigation will not be readily reversed by changes from irrigated to dryland or rangeland settings.

Past LU/LC changes on the water cycle were not planned. As relationships between LU/LC and recharge become more widely recognized, further impacts of LU/LC changes on the hydrologic cycle can be better evaluated. In addition, LU/LC changes may be managed to modify specific components of the water cycle, such as recharge. Considerable interest already exists in increasing groundwater recharge to depleted aquifers and attempting to develop sustainable water-resource management plans. Approaches under consideration include removal of invasive vegetation (to reduce ET and thereby increase recharge). Engineered playa modifications have been proposed for the SHP, although playas are currently protected as wetlands. Results of the current study suggest that conversion of rangeland to dryland agriculture will increase recharge at the expense of rangeland species. Current LU/LC in the SHP consists of 46% rangeland, 41% dryland agriculture, and 11% irrigated agriculture. Large areas of rangeland could potentially be converted to dryland agriculture. Under the Federal Conservation Reserve Program, however, up to 25% of cultivated land has been removed from production to decrease soil erosion and provide habitat for wildlife. If rangeland is converted to dryland agriculture to increase groundwater recharge, the time lag for increased drainage below the root zone to reach the water table must be taken into account (Fig. 12). The potential for degrading groundwater quality should also be considered, requiring characterization of solute accumulations in the unsaturated zone prior to LU/LC conversion. Managed changes in LU/LC are already being considered to reduce dryland salinity problems in large areas of the Northern High Plains (US) and Murray Basin (Australia) (Halvorsen & Reule, 1980; Salama *et al*., 1999; Dawes *et al*., 2002). The ability to control recharge by modifying LU/LC may become a powerful tool for water resources management in the future.

Conclusions

The point to regional scale analysis and variety of approaches used in this study, including soil physics and environmental tracers, complemented each other in developing a conceptual understanding of the impact of LU/LC change on spatiotemporal variability in recharge. Good correspondence between recharge estimates based on CMB, chloride- and nitrate-tracer velocities, and water-table-elevation increases confidence in the recharge estimates.

The general result of this study is that groundwater recharge is related to LU/LC setting as follows:

(1) Recharge is negligible beneath semiarid and arid rangeland ecosystems, where total-potential gradients are upward, matric potentials and water contents are low, and chloride and other salts have been building up in the unsaturated zone for thousands of years.

(2) Recharge is moderate-to-high beneath irrigated agricultural ecosystems (e.g., 130–640 mm yr^{-1} at the AD site). Total-potential gradients are downward, matric potentials and water contents are high, and chloride levels in unsaturated-zone porewater are low to moderate.

(3) Recharge is low to moderate beneath dryland agricultural ecosystems (e.g., 9–32 mm yr^{-1} at the HP3 site). Total-potential gradients are downward, matric potentials and water contents are high, and chloride levels in unsaturated-zone porewater are low.

Unsaturated-zone chloride and nitrate profiles archive changes in recharge related to recent conversion of rangeland to agricultural ecosystems (e.g., in the AD). Increased recharge associated with dryland as well as irrigated agriculture can lead to degradation of groundwater quality because of leaching of salts that have been accumulating in the unsaturated zone for thousands of years prior to cultivation, because of application of fertilizers, and, in irrigated areas, because of evapoconcentration of applied groundwater. In the SHP, median groundwater nitrate-N concentrations increased by 221% beneath irrigated areas and 163% beneath dryland areas, reflecting LU/LC-induced contamination of groundwater.

Recharge rates in irrigated areas correlate with the magnitude of irrigation plus precipitation ($r^2 = 0.94$). Irrigation applications are inversely related to mean annual precipitation ($r^2 = 0.87$) and were highest at the AD site (2.0–2.7 m yr^{-1}) and lowest at the HP3 (0.3 m yr^{-1}) site. Salt accumulation occurs in areas of low irrigation applications (HP3, 0.3 m yr^{-1} irrigation).

The strong correlation between recharge and LU/LC shown in this study suggests that managed changes in LU/LC can be used to control groundwater recharge and groundwater quality.

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References

Adegoke JO, Pielke RA, Eastman J *et al*. (2003) Impact of irrigation on midsummer surface fluxes and temperature under dry synoptic conditions: a regional atmospheric model study of the US High Plains. *Monthly Weather Review*, **131**, 556–564.

Aiken GR, Kuniansky EL (2002) *US Geological Survey artificial recharge workshop proceedings, Sacramento, CA, April 2–4, 2002*. USGS Open-File Report, 02-89, 88pp.

Allison GB, Cook PG, Barnett SR *et al*. (1990) Land clearance and river salinisation in the western Murray Basin, Australia. *Journal of Hydrology*, **119**, 1–20.

Allison GB, Hughes MW (1983) The use of natural tracers as indicators of soil–water movement in a temperate semi-arid region. *Journal of Hydrology*, **60**, 157–173.

Andraski BJ, Scanlon BR (2002) Thermocouple psychrometry. In: *Methods of Soil Analysis, Part 4, Physical Methods* (eds Dane JH, Topp GC), pp. 609–642. Soil Science Society of America, Madison, WI.

Bonan GB (1997) Effects of land use on the climate of the United States. *Climatic Change*, **37**, 449–486.

Bruce BW, Oelsner GP (2001) Contrasting water quality from paired domestic/public supply wells, central High Plains. *Journal of America Water Resources Association*, **37**, 1389–1403.

Cook PG, Walker GR, Jolly ID (1989) Spatial variability of groundwater recharge in a semiarid region. *Journal of Hydrology*, **111**, 195–212.

Dawes WR, Gilfedder M, Stauffacher M *et al*. (2002) Assessing the viability of recharge reduction for dryland salinity control: Wanilla, Eyre Peninsula. *Australian Journal of Soil Research*, **40**, 1407–1424.

Dennehy KF (2000) *High Plains regional ground-water study*. USGS Fact Sheet, FS-091–00, 6pp.

Fahlquist L (2003) *Ground-water quality of the Southern High Plains aquifer, Texas and New Mexico, 2001*. USGS Open-File Report, 03-45, 59pp.

Flint AL, Campbell GS, Ellet KM *et al*. (2002) Calibration and temperature correction of heat dissipation matric potential sensors. *Soil Science Society of America Journal*, **66**, 1439–1445.

Gee GW, Campbell MD, Campbell GS *et al*. (1992) Rapid measurement of low soil water potentials using a water activity meter. *Soil Science Society of America Journal*, **56**, 1068–1070.

Gee GW, Wierenga PJ, Andraski BJ *et al*. (1994) Variations in water balance and recharge potential at three western desert sites. *Soil Science Society of America Journal*, **58**, 63–71.

Halvorsen AD, Reule CA (1980) Alfalfa for hydrologic control of saline seeps. *Soil Science Society of America Journal*, **44**, 370–374.

Healy RW, Cook PG (2002) Using ground-water levels to estimate recharge. *Hydrogeology Journal*, **10**, 91–109.

Hirsch RM, Slack JR (1984) A nonparametric trend test for seasonal data with serial dependence. *Water Resources Research*, **20**, 727–732.

Izbicki JA (2002) Geologic and hydrologic controls on the movement of water through a thick, heterogeneous unsaturated zone underlying an intermittent stream in the western Mojave Desert, southern California. *Water Resources Research*, **38**, 1–14.

Jobbágy EG, Jackson RB (2004) Groundwater use and salinization with grassland afforestation. *Global Change Biology*, **10**, 1299–1312.

Jury WA, Gardner WR, Gardner WH (1991) *Soil Physics*. John Wiley & Sons, New York.

Knowles TR, Nordstrom P, Klemt WB (1984) *Evaluating the ground-water resources of the High Plains of Texas*. Texas Department of Water Resources, Report 288, 119pp.

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

Luckey RL, Becker MF (1999) *Hydrogeology, water use, and simulation of flow in the High Plains aquifer in northwestern Oklahoma, southeastern Colorado, southwestern Kansas, northeastern New Mexico, and northwestern Texas*. USGS Water Resources Investigations Report, 99-4104, 68pp.

McGuire VL (2001) *Water-level changes in the High Plains aquifer*. USGS Fact Sheet, FS-029-01, 2pp.

McMahon PB, Bohlke JK, Lehman TM (2004) *Vertical gradients in water chemistry and age in the Southern High Plains aquifer, Texas, 2002*. US Geol. Surv. Scientific Inv. Rept., 5053, 53pp.

McMahon PB, Dennehy KF, Bruce BW *et al*. (in press) Storage, flux, and transit time of chemicals in thick unsaturated zones under rangeland and irrigated cropland, High Plains, USA. *Water Resources Research*.

McMahon PB, Dennehy KF, Ellett KM *et al*. (2003) *Water movement through thick unsaturated zones overlying the central High Plains aquifer, southwestern Kansas, 2000–2001*. USGS Water Resources Investigation Report, 03-4171, 32pp.

Miller MR, Brown PL, Donovan JJ *et al*. (1981) Saline-seep development and control in the North American Great Plains: hydrogeological aspects. *Agricultural Water Management*, **4**, 115–141.

Moore N, Rojstaczer S (2002) Irrigation's influence on precipitation: Texas High Plains, U.S.A. *Geophysical Research Letters*, **29**, 2-1–2-4.

Newman BD, Sala OE, Wilcox BP (2003) Conference promotes study of ecohydrology of semi-arid landscapes. *EOS, Transactions, American Geophysical Union*, **84**, 13, 17.

O'Connell MG, O' Learey GJ, Connor DJ (2003) Drainage and change in soil water storage below the root-zone under long fallow and continuous cropping sequences in the Victorian Mallee. *Hydrological Processes*, **54**, 663–675.

Odening WR, Strain BR, Oechel WC (1974) The effect of decreasing water potential on net CO_2 exchange of intact desert shrubs. *Ecology*, **55**, 1986–1095.

Olson CG, Porter DA (2002) Isotopic and geomorphic evidence for Holocene climate, Southwestern Kansas. *Quaternary International*, **87**, 29–44.

Petheram C, Zhang L, Walker GR *et al*. (2000) Towards a framework for predicting impacts of land-use on recharge: 1. A review of recharge studies in Australia. *Australian Journal of Soil Research*, **40**, 397–417.

Phillips FM (1994) Environmental tracers for water movement in desert soils of the American Southwest. *Soil Science Society of America Journal*, **58**, 14–24.

Pielke RAS, Avissar R, Raupach M *et al*. (1998) Interactions between the atmosphere and terrestrial ecosystems:

influence on weather and climate. *Global Change Biology*, **4**, 461–475.

Pitman AJ, Narisma GT, Pielke RAS *et al*. (2004) Impact of land cover change on the climate of southwest Western Australia. *Journal of Geophysical Research*, **109**, doi: 18110.11029/12003JD00437.

Prudic DE (1994) *Estimates of percolation rates and ages of water in unsaturated sediments at two Mojave Desert sites, California-Nevada*. USGS Water Resources Investigation Report, 94-4160, 19pp.

Qi SL, Konduris A, Litke DW (2002) *Classification of irrigated land using satellite imagery, the High Plains aquifer, nominal 1992*. USGS Water Resources Investigation Report, 02-4236, 31pp.

Richards WF (1990) Land transformation. In: *The Earth as Transformed by Human Action* (eds Turner BL, Clark WC, Kates RW *et al*.), pp. 561–575. Cambridge University Press, Cambridge.

Roark DM, Healy DF (1998) *Quantification of deep percolation from two flood-irrigated alfalfa fields, Roswell Basin, New Mexico*. USGS Water Resources Investigation Report, 98-4096, 32pp.

Rodriguez-Iturbe I (2000) Ecohydrology: a hydrologic perspective of climate-soil-vegetation dynamics. *Water Resources Research*, **36**, 3–9.

Salama R, Hatton T, Dawes WR (1999) Predicting land use impacts on regional scale groundwater recharge and discharge. *Journal of Environmental Quality*, **28**, 446–460.

Savage MJ, Ritchie JT, Bland WL *et al*. (1996) Lower limit of soil water availability. *Agronomy Journal*, **88**, 644–651.

Scanlon BR (1991) Evaluation of moisture flux from chloride data in desert soils. *Journal of Hydrology*, **128**, 137–156.

Scanlon BR, Andraski BJ (2002) Miscellaneous methods for measuring matric or water potential. In: *Methods of Soil Analysis, Part 4, Physical Methods* (eds Dane JH, Topp GC), pp. 643–670. Soil Science Society of America, Madison, WI.

Scanlon BR, Goldsmith RS (1997) Field study of spatial variability in unsaturated flow beneath and adjacent to playas. *Water Resources Research*, **33**, 2239–2252.

Scanlon BR, Keese K, Reedy RC *et al*. (2003) Variations in flow and transport in thick desert vadose zones in response to paleoclimatic forcing (0–90 kyr): field measurements, modeling, and uncertainties. *Water Resources Research*, **39**, doi: 1110.1029/2002WR001604.

Scanlon BR, Levitt DG, Keese KE *et al*. (2005) Ecological controls on water-cycle response to climate variability in deserts. *Proceedings of the National Academy of Sciences USA*, **102**, 6033–6038.

Service RF (2004) As the west goes dry. *Science*, **303**, 1124–1127.

Smith SD, Monson RK, Anderson JE (1997) *Physiological Ecology of North American Desert Plants*. Springer-Verlag, Berlin.

Stonestrom DA, Prudic DE, Laczniak RJ *et al*. (2003) *Estimates of deep percolation beneath native vegetation, irrigated fields, and the Amargosa-River channel, Amargosa Desert, Nye County, Nevada*. USGS Open File Report, 03-104, 83pp.

Texas Water Development Board (2003) *Groundwater Database Reports*. Accessed June 2003 at URL http://www.twdb.state.tx.us/publications/reports/GroundWaterReports/GWDatabaseReports/GWdatabaserpt.htm

Tilman D, Fargione J, Wolff B *et al*. (2001) Forecasting agriculturally driven global environmental change. *Science*, **292**, 281–284.

Tyler SW, Chapman JB, Conrad SH *et al*. (1996) Soil-water flux in the southern Great Basin, United States: temporal and spatial variations over the last 120,000 years. *Water Resources Research*, **32**, 1481–1499.

US Department of Agriculture (2003) *National agricultural statistics service data base*. Accessed July 2003 at URL http://www.nass.usda.gov/

Vogelmann JE, Howard SM, Yang L *et al*. (2001) Completion of the 1990s National Land Cover Data set for the conterminous United States from Landsat Thematic Mapper data and ancillary data sources. *Photogrammetric Engineering and Remote Sensing*, **67**, 650–662.

Vorosmarty C, Lettenmaier D, Leveque C *et al*. (2004) Humans transforming the global water system. *Eos*, **85**, 509–520.

Walker GR, Jolly ID, Cook PG (1991) A new chloride leaching approach to the estimation of diffuse recharge following a change in land use. *Journal of Hydrology*, **128**, 49–67.

Walvoord MA, Phillips FM, Stonestrom DA *et al*. (2003) A reservoir of nitrate beneath desert soils. *Science*, **302**, 1021–1024.

Walvoord MA, Stonestrom DA, Andraski BJ (2004) Constraining the inferred paleohydrologic evolution of a deep unsaturated zone in the Amargosa Desert. *Vadose Zone Journal*, **3**, 502–512.