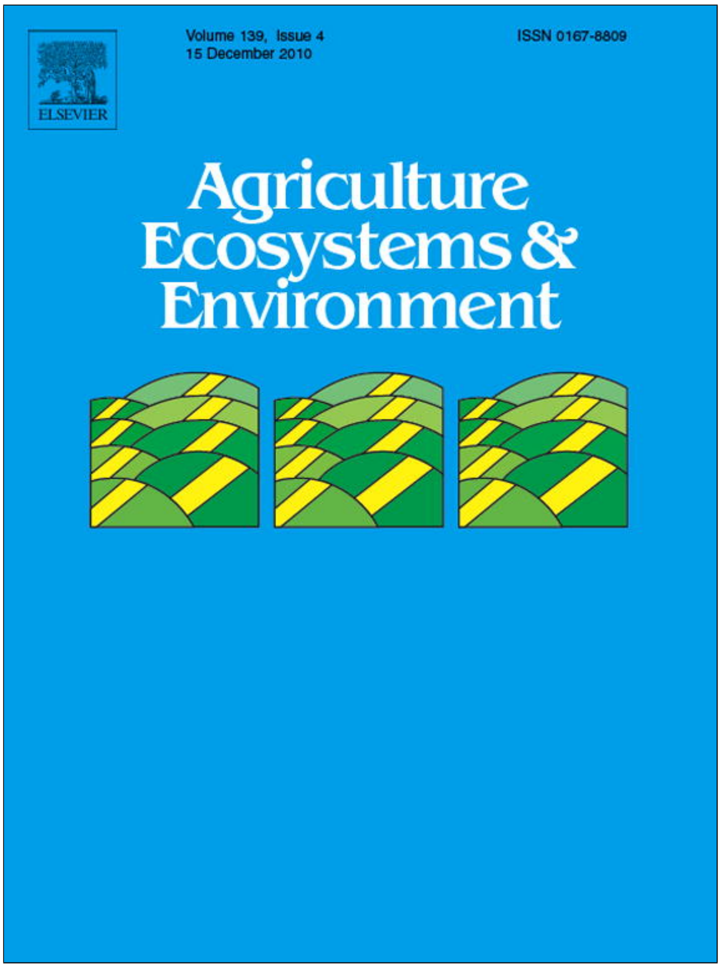
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Impact of agroecosystems on groundwater resources in the Central High Plains, USA

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ABSTRACT

Agroecosystems impact water resources by consuming most fresh water through irrigation and by changing water partitioning at the land surface. The study assesses impacts of agroecosystems on groundwater resources in the Texas Central High Plains (37,000 km^2 area) by evaluating temporal variations in groundwater storage and quality. Percolation/recharge rates were estimated using groundwater Cl data and using unsaturated zone matric potential and water-extractable chloride and nitrate from 33 boreholes beneath different agroecosystems. Total groundwater storage decreased by 57 km^3 since the 1950s when irrigation began and individual well hydrographs had declines $\leq$1.3 m/yr. The renewable portion of groundwater is controlled by percolation/recharge, which is related to soil texture and land use. In fine–medium (f–m) grained soils, there is no recharge beneath natural ecosystems or rain-fed agroecosystems; however, recharge is focused beneath playas and drainages. In medium–coarse (m–c) grained soils, percolation/recharge is low (median 4.8 mm/yr) beneath natural ecosystems and is moderate (median 27 mm/yr) beneath rain-fed agroecosystems. Although irrigation increased percolation under all soil types (median 37 mm/yr), irrigation return flow has not recharged the aquifer in most areas because of deep water tables. Groundwater depletion (21 km^3 over 52 yr) is 10 times greater than recharge (11 mm/yr; 2.1 km^3) where water table declines are greatest ($\geq$30 m). Therefore, current irrigation practices are not sustainable and constitute mining of the aquifer, which is being managed as a nonrenewable resource.

1. Introduction

1.1. How do agroecosystems impact groundwater quantity?

Irrigated agroecosystems affect water demand by consuming ~90% of global fresh-water resources during the past century (Shiklomanov, 2000). Overabstraction of groundwater for irrigation has resulted in large groundwater level declines, particularly in the SW US, NW and W India, and the North China Plain (Siebert et al., 2005). By changing partitioning of water at the land surface among evapotranspiration (ET), runoff, and recharge, agroecosystems also alter the distribution of green water (soil moisture from precipitation) and blue water (surface water and groundwater). In contrast to irrigated agroecosystems, which deplete groundwater resources, conversion of natural ecosystems to rain-fed agroecosystems increases groundwater resources by enhancing percolation/recharge by up to two orders of magnitude in semiarid regions, such as southeast Australia (Allison et al., 1990) and SW Niger (West Africa) (Favreau et al., 2009). Percolation refers to deep drainage below the root zone that has not reached the water table to recharge the aquifer.

1.2. How do agroecosystems impact groundwater quality?

Vegetation plays a large role in controlling water quality because the process of ET is similar to desalinization in that it excludes salts, resulting in a buildup of salts in soils from bulk precipitation (precipitation + dry fallout), unless the salts are flushed through the soil profile by percolation/recharge. In many semiarid regions, large reservoirs of salts, including Cl, ClO_4, SO_4, F, and sometimes NO_3, have accumulated from bulk precipitation as a result of long-term drying under natural vegetation over millennia (Scanlon et al., 2009). Agroecosystems increase vulnerability of groundwater to contamination by increasing percolation/recharge rates mobilizing these salts and reducing time lags to reach the aquifer (McMahon et al., 2006). This mobilization of natural salt

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and nutrient inventories has been documented in the Murray Darling Basin in Australia and the Amargosa Desert and High Plains in the US (Stonestrom et al., 2003; McMahon et al., 2006; Scanlon et al., 2009). In addition to mobilizing these salts and nutrients that had accumulated over millennia in the soil profile through increased percolation/recharge, irrigation also adds salts to the system because irrigation water has much higher salt concentrations than precipitation. The impact of irrigation on water quality through salt loading depends strongly on the quality of the irrigation water. Irrigation can result in soil salinization and/or aquifer salinization over time.

1.3. What effect does soil texture have on percolation/recharge rates?

Field studies and a simple water balance model showed that recharge decreases by about an order of magnitude (30–3 mm/yr) with increasing clay content from 0 to 20% under cropland in the Murray Basin (Australia) for mean annual precipitation ranging from 310 to 380 mm (Kennett-Smith et al., 1994). A review of recharge studies in Australia showed that recharge rates within land use categories (annual vegetation and trees) varied across soil types (Petheram et al., 2000). Large variations in recharge in clayey soils were attributed to preferential flow. Modeling analyses of recharge in Texas, US, showed maximum recharge rates in sandy soils and large reductions in recharge with soil textural variability (Keese et al., 2005). More recent studies in the Nebraska Sand Hills and adjacent silt loam soils noted reductions in recharge and corresponding large increases in ET from sand to silt loam (Wang et al., 2009). These studies indicate that soil texture can play an important role in controlling recharge.

1.4. What techniques can be used to assess impacts of agroecosystems on groundwater resources?

Groundwater level hydrographs can be used to quantify impacts of agroecosystems on groundwater depletion through irrigation pumpage and groundwater increase through recharge. Water table fluctuations have been used to quantify changes in recharge in response to land use changes in many regions (Sophocleous, 1991; Favreau et al., 2009). Soil profiles in the unsaturated zone provide records of long-term impacts of land use on subsurface water quantity and quality and link surface processes with aquifers. The Cl mass balance (CMB) approach or the Cl front displacement (CFD) approach has been used to quantify changes in percolation/recharge in different land use settings (Walker et al., 1991; Stonestrom et al., 2003; Scanlon et al., 2007). The CMB approach balances Cl input from bulk precipitation with Cl output in percolation/recharge and is used to estimate percolation/recharge. The CFD approach uses the Cl bulges that accumulated under natural ecosystems over millennia as a marker to track percolation. The Cl front marks the upper part of the Cl bulge. The presence of bomb pulse tritium has been used to distinguish prebomb (before 1950s) and postbomb tritium water and track unsaturated zone pore water movement related to irrigation return flow (McMahon et al., 2006). Modeling is also a useful tool to evaluate recharge related to agroecosystems and assess different controls on percolation/recharge, such as climate, soils, and crop rooting depths (Kennett-Smith et al., 1994; Keese et al., 2005).

The US High Plains (450,000 km^2 area), one of the largest agricultural areas in the US, provides an excellent study area to examine impacts of agroecosystems on water resources. This region constitutes one of the most intensively irrigated areas in the US, representing 30% of the nation's groundwater used for irrigation (Maupin and Barber, 2005). Groundwater depletion for irrigation from the High Plains or Ogallala aquifer is greatest in the Central High Plains (CHP) and in the north part of the Southern High Plains (SHP) with maximum groundwater level declines of ≥45 m since irrigation began in the 1950s (McGuire, 2009). Because of the importance of the High Plains for agricultural production and overabstraction of water from the High Plains aquifer, numerous studies have been conducted in this region. Regional recharge was estimated to be 11 mm/yr on the basis of the CMB approach applied to groundwater Cl data mostly in the north part of the SHP (Wood and Sanford, 1995). Recharge is attributed primarily to focused flow beneath ephemeral lakes or playas with rates of 60–120 mm/yr on the basis of unsaturated zone Cl and tritium data (Wood and Sanford, 1995; Scanlon and Goldsmith, 1997). Although there are ~53,000 playas in the High Plains, they only occupy ~0.4% of the land surface. Lack of percolation/recharge adjacent to playas under natural ecosystems in f–m g soils in the CHP is evidenced by bulge-shaped Cl profiles that have been accumulating Cl since Pleistocene times ~10,000 yr ago; however, these studies are restricted to <1% of the area of the Texas CHP (Scanlon and Goldsmith, 1997). Conversion of natural ecosystems to rain-fed agroecosystems increased percolation/recharge rates in sandy soils in the SHP to a median value of 24 mm/yr (4.8–92 mm/yr) based on 19 unsaturated zone Cl profiles and groundwater level rises in the southeast part of the SHP (Scanlon et al., 2007). Irrigated agroecosystems generally resulted in moderate percolation/recharge rates in the SHP (18–97 mm/yr) similar to the range (4.8–92 mm/yr) under rain-fed agroecosystems (McMahon et al., 2006; Scanlon et al., in press). Limited profiling (two profiles) in irrigated areas in the CHP in Kansas resulted in recharge rates of 39 and 54 mm/yr (McMahon et al., 2006). At many of these sites, irrigation return flow has not reached the water table. However, previous groundwater models of the region simulated increased recharge under irrigated areas ranging from 24% of irrigation application in the 1940s and 1950s to 2% of irrigation application in the 1990s, assuming irrigation efficiency increased with transition from predominantly flood irrigation to sprinkler systems over time (Luckey and Becker, 1999; Dutton et al., 2001).

Although there have been numerous studies conducted in the High Plains, there is very little information on groundwater recharge throughout much of the CHP in Texas (Fig. 1). There is considerable interest in the renewability of groundwater resources in this region to support widespread irrigation practices. In addition, Mesa Water has purchased water rights for ~600 km^2, and the Canadian River Municipal Water Authority (CRMWA) has rights for ~1000 km^2 to supply the City of Amarillo. Mesa Water had plans to transport water from this region to large municipalities in Texas. The Canadian River is also a gaining river, and there is concern that overabstraction of groundwater would reduce baseflow discharge to the river and change it from a gaining to a losing river. There is also interest in enhancing recharge in the region to increase groundwater supply.

The objective of this study was to evaluate impacts of agroecosystems on water resources in the CHP, including effects on water quantity and quality and assessing water sustainability issues. Impacts of irrigation on groundwater quantity were evaluated using water level data from the 1950s to present and examining individual well hydrographs. Effects of agroecosystems on groundwater quality were assessed using groundwater total dissolved solids (TDS), Cl, and NO_3 data. Estimating recharge is also critical to determin the renewability of groundwater resources. Previous studies were limited to recharge estimation under natural ecosystems, including 9 playa and 13 adjacent interplaya sites, mostly in fine-grained (clay loam) soils (Scanlon and Goldsmith, 1997) and under rain-fed and irrigated agroecosystems at one site in clay loam soils (Fig. 1, Scanlon et al., 2008a,b, in press.). However, these studies covered <1% of the current study area. The current study expands on the previous work by including unsaturated zone matric potential and water extractable Cl and NO_3 under natural (14 profiles)

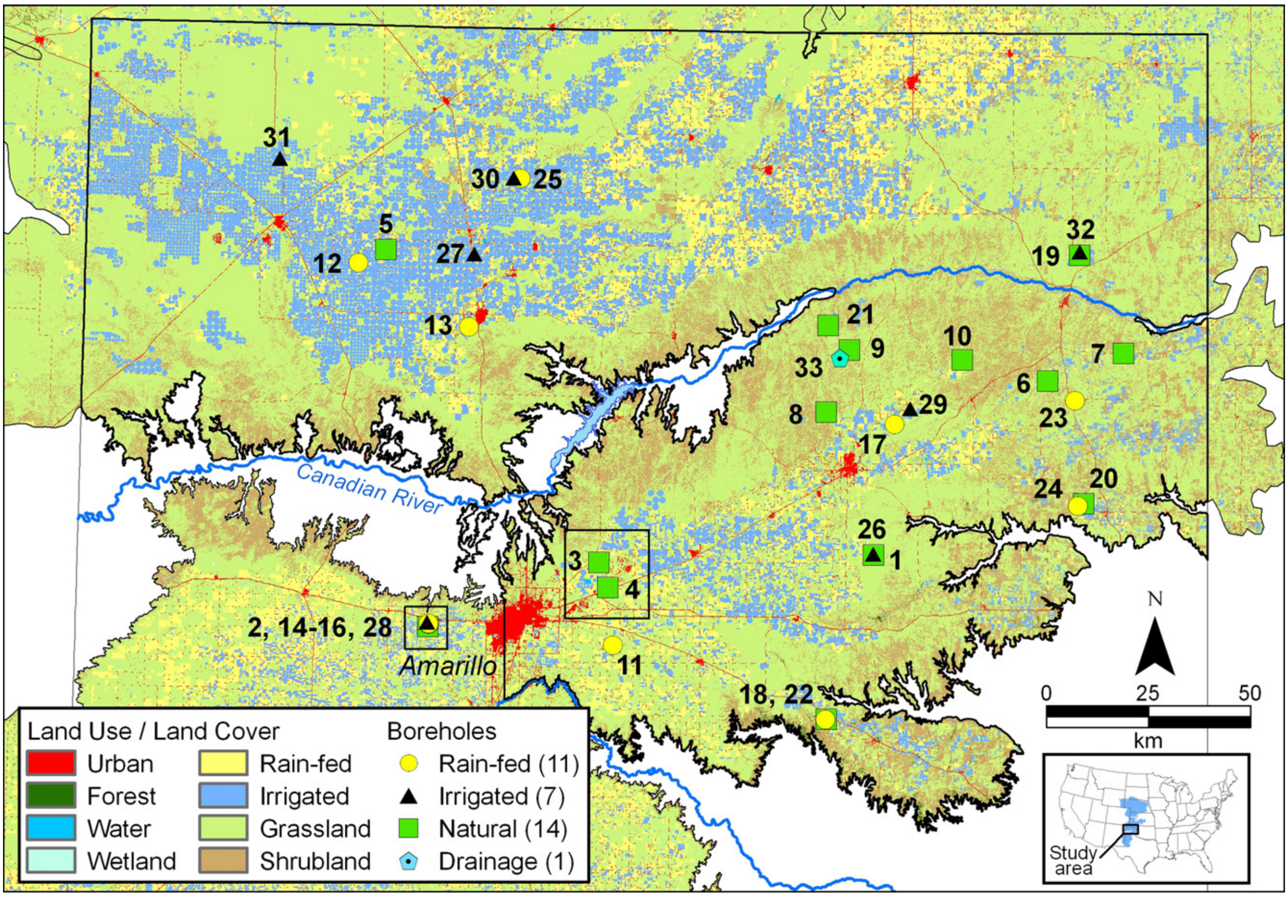

Fig. 1. Land use/land cover and borehole locations in the study area. Land use represents generalized National Land Cover Data (2001; Homer et al., 2007) land use categories excluding irrigated areas. Irrigated areas are based on satellite coverage from ∼1992 (Qi et al., 2002) and delineation of center pivot circles by National Agricultural Imagery Program (NAIP) coverage from 2006.

ecosystems and rain-fed (11 profiles) and irrigated (7 profiles) agroecosystems throughout the entire study area (Fig. 1). Profiles in a range of soil textures from clay loam to sand were used to examine effects of textural variability on impacts of agroecosystems on percolation/recharge. Implications of groundwater depletion and recharge for water resources management in the Texas CHP are also discussed.

2. Materials and methods[1]

2.1. Site characteristics and history

The Texas part of the CHP occupies a 37,000 km^2 area. Long-term mean annual precipitation ranges from 380 mm in the west to 620 mm in the east, with a median value of 525 mm/yr (1971–2000, PRISM). Precipitation occurs mostly in the summer, with 65% during May through September, generally in the form of convective thunderstorms. There is only one major river in the region, the Canadian River, which was dammed upstream to create Lake Meredith in 1965.

Soil textures are variable and include clay loam soils (29% of area), with shrink–swell characteristics throughout much of the central region (Table 1, Figs. 2 and S1). Soils are coarser in the west (17% of area), and sand dunes have also been mapped in this region (Muhs and Holliday, 1995). Loamy textured soils on slopes of >8% are found adjacent to the Canadian River and around the margins of clay loam areas (40% of area). The coarsest soils are found in the east, mostly sands (15% of area). Dunes have been mapped north of the Canadian River in this region (Muhs and Holliday, 1995). Variations in soil texture are related to spatial variability in soil permeability, with permeabilities <0.3 m/d associated with clay loam soils and up to 7.7 m/d in sandy soils in the southeast (Fig. S2). Topography and soil texture are related, with lower slopes (median 1.0%) associated with clay loam and loamy soils in the northwest (median 2.4%) and much steeper slopes associated with loamy soils generally in the eastern half of the study area marginal to clay loam areas (median 6–14%) (Fig. S3). Because of the generally flat topography throughout much of the CHP, most surface water drains internally to ∼4,100 playas (Fig. 2). Playas account for ∼1% of the area regionally, but 89% of playas are concentrated in clay loam areas representing 3.4% of the area in these soils. Drainages are concentrated along the slopes adjacent to the Canadian River (Fig. S3).

Land use in the CHP of Texas consists mostly of grassland (53%), shrubland (16%), and cropland (28%) (Fig. 1). Most of the cropland is found in clay loam soils (62%), followed by loamy soils with low slopes (24%) and decreasing cropland in loams with steeper slopes (Table 1). There is very little (3%) cropland in sands. Approximately 17% of the land surface is irrigated (Fig. 1). The area equipped for irrigation represents 62% of cultivated land; however, these areas currently may not be fully irrigated because of water shortages in some regions. All irrigation water is derived from groundwater, mostly the High Plains/Ogallala aquifer. Irrigation accounts for ∼95% of groundwater abstraction (Dutton et al., 2001).

[1] Supporting information is provided with this paper (see online version) in the form of figures and tables labeled with a prefix S (i.e., Figure S1).

Table 1
General soil category characteristics.

Category	% of total area	% of cultiv. area	% of playa area	K_{sat} (m/d)	Slope (%)	Clay (%)	Rain-fed (%)	Irrigated (%)	Cultiv. (%)	CMB rech. (mm/yr)
Clay loams, >35% clay	29	62	89	0.2	1.0	43	29	33	62	12
Loams, <5% slope	17	24	7.6	1.1	2.4	26	7.9	32	40	8
Loams, 5–8% slope	13	9.4	2.2	1.4	6.0	25	7.4	14	21	10
Loams, >8% slope	27	2.0	0.7	1.5	13.9	20	0.8	1.3	2.1	12
Sands, <14% clay	15	2.8	0.0	4.3	8.7	10	1.5	4.0	5.5	16

Category: generalized soil map category (Fig. 2), *% of total area*: percentage of study area represented by the category, *% of cultiv. area*: percentage of total cultivated area found in the category, *% of playa area*: percentage of total playa floor area found in the category, K_{sat}: median saturated hydraulic conductivity based on STATSGO (USDA, 1994), *Slope*: median ground surface slope based on 30-m digital elevation model, *clay*: median soil clay content based on SSURGO (USDA, 1995), *rain-fed*: percentage of the category area that is rain fed, *irrigated*: percentage of the category area that is irrigated, *cultiv.*: percentage of the category area that is cultivated, *CMB rech.*: median recharge in the category based on groundwater chloride mass balance (excluding areas with groundwater chloride to sulfate mass ratios greater than 1, Fig. 7).

Dominant crops include wheat (63% of cropland), corn (27%), and sorghum (10%), according to 2007 data from the National Agricultural Statistics Service (NASS, www.nass.usda.gov). The total cultivated area decreased beginning in the early 1980s from a mean of 11,000 km^2 (1972–1982) to 8500 km^2 (1987–2007) (Fig. S4a). This analysis is based on only the area planted to the major crops listed, which represents 23% of the CHP area. The area cultivated with corn increased from ~7% to ~24% whereas wheat area remained relatively constant (~65 to ~75%) and sorghum area declined from ~25–30% to ~15% (Fig. S4b). All corn is irrigated in this region whereas only ~30% of wheat and 35% of sorghum is irrigated. The area planted with corn began to increase in the mid-1980s and has ranged from 20% to 28% since ~1993. The corn crop value peaked at $380 to $460 million in 2007–08, although the area planted did not change significantly.

The Ogallala Formation (Tertiary age, Eocene) ranges from 0 to 240 m in thickness in the CHP, consisting of fluvial gravel, sand, and silt and eolian sand and silt (Gustavson and Holliday, 1999). Variations in aquifer thickness reflect paleotopography at the base of the High Plains aquifer, with large aquifer thickness in paleovalleys (Canadian River and Panhandle paleovalleys) dominated by fluvial deposits and much thinner eolian deposits in paleouplands (Amarillo Uplift) and blanketing the entire area. The top of the Ogallala Formation consists of a 2 m thick calcrete layer called the Caprock

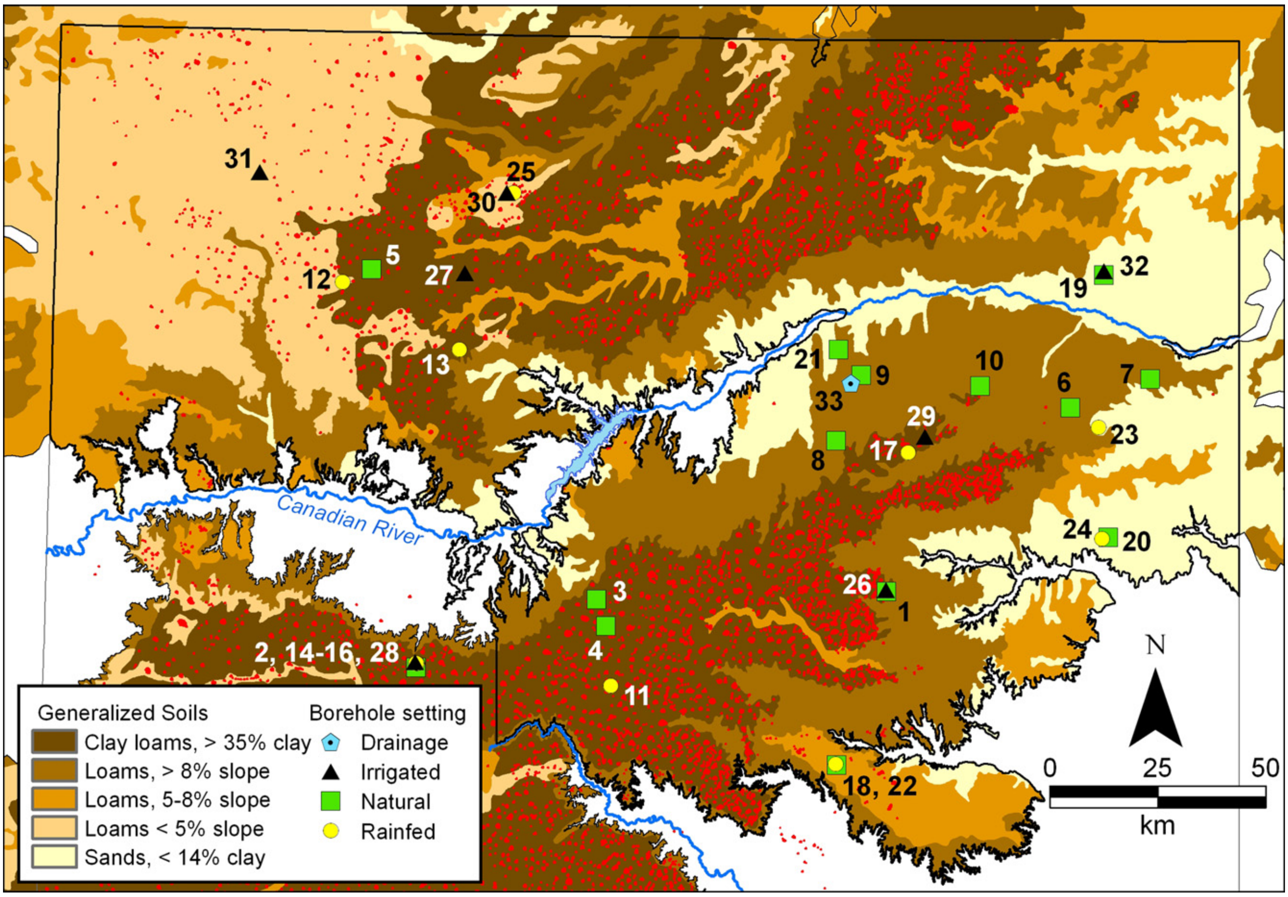

Fig. 2. Generalized soil map based on STATSGO (USDA, 1994) with playas shown in red. Clay loam soils occupy 29% of Texas CHP and include Pullman series south of the Canadian River and Sherm, Gruver, and Darrouzett series north of the Canadian River. Loams with <5% slopes are found mostly in the west (17% area) and include Dallam, Rickmore, and Vingo series. Loams with slopes >5% near the Canadian River and around margins of clay loam areas (40% of area) include Mobeetie, Burda, and Veal series. Sands are mostly found in the east (15% of area) and include Circleback, Likes, Springer, and Delwin series. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

Table 2
Parameters and estimated inputs for profiles located in natural rangeland ecosystems and rain-fed agroecosystems with fine–medium grained soils. Rain-fed profiles show little or no impact from cultivation and represent essentially natural conditions.

Profile	Interval (m)	Clay (%)	θ (m^3/m^3)	MP (m)	Cl_{Max} (mg/L)	N_{Max} (mg/L)	Cl (mg/L)	N (mg/L)	P (mm/yr)	Cl_P (mg/L)	Age_P (yr)
Natural (fine–medium grained soils)											
1	0.5–11.7	42	0.21	−210	3,200	17	850	13	575	0.26	13,600
2[a]	1.0–14.6	42	0.16	−210	3,000	17	670	12	485	0.24	11,600
3[b]	1.0–14.1	41	0.21	−170	4,200	–	590	–	515	0.24	13,700
4[b]	1.0–17.6	41	0.16	−270	2,500	–	600	–	520	0.24	13,100
5	1.2–5.9	26	0.16	−230	1,500	7.3	1,100	4.2	445	0.22	8,000
6	1.8–5.8	29	0.10	−120	2,200	4.7	1,600	2.4	585	0.26	3,500
7	1.5–10.6	29	0.25	−69	880	1.1	520	0.6	600	0.26	6,900
8	0.9–11.7	27	0.20	−83	560	3.3	440	1.3	565	0.24	7,100
9	1.8–22.6	26	0.09	−96	1,400	4.0	960	2.2	555	0.24	12,700
10	0.9–12.2	33	0.23	−87	2,600	5.6	1,100	2.3	585	0.24	20,400
Median		31	0.18	−150	2,400	5.2	760	2.4	560	0.24	12,200
Rain-fed (fine–medium grained soils)											
11[a]	1.0–8.0	44	0.21	−100	1,200	140	870	22	520	0.24	10,100
12	1.2–8.9	26	0.20	−69	900	19	540	13	445	0.22	8,300
13	0.6–15.7	45	0.18	−79	1,000	280	330	20	445	0.22	8,900
14[a]	1.1–13.3	42	0.18	−79	600	62	460	11	485	0.24	8,600
15	0.9–18.3	42	0.19	−120	1,100	99	400	17	485	0.24	11,900
16	0.9–13.7	42	0.21	−110	1,400	74	570	20	485	0.24	13,400
17	0.9–18.3	43	0.21	−100	1,300	12	410	5.7	580	0.24	11,200
Median		42	0.20	−100	1,100	74	460	17	485	0.24	10,100

Profile: profile number (Fig. 1), *interval*: borehole depth interval below the root zone, also indicates total borehole depth, *clay*: depth-weighted mean soil clay content to a depth of ~1.8 to 2.0 based on SSURGO, θ: depth-weighted mean water content, MP: depth-weighted mean matric potential, Cl_{Max} and N_{Max}: peak chloride and NO_3-N concentrations, Cl and N: depth-weighted mean Cl and NO_3-N concentrations, *P*: mean annual precipitation (1971–2000, Prism Climate Group, www.prism.oregonstate.edu), Cl_P: mean Cl concentration in bulk annual precipitation from NADP (~1980–2007), Age_P: age of water at maximum profile depth assuming chloride in precipitation is the only input.

[a] Scanlon et al. (2008a).

[b] Scanlon and Goldsmith (1997).

Calcrete, which is overlain by the eolian Pleistoceneage Blackwater Draw Formation.

2.2. Physical and chemical measurements

In this study, 28 boreholes were drilled, and 5 additional boreholes reported in this paper were drilled in previous studies (Fig. 1 and Tables 2–4). The boreholes represent different land use settings in the CHP, 14 under natural rangeland ecosystems, 11 under rain-fed and 7 under irrigated agroecosystems, and 1 in a drainage. The methods used in this study are similar to those described in Scanlon et al. (2007). Most boreholes were drilled using a direct push drill rig (Model 6620 DT, Geoprobe, Salina, KS), and continuous soil cores were extracted. Soil cores were analyzed for water content, matric potential, and water extractable anion concentrations (Cl, NO_3). Borehole depths range from 2.8 to 23 m (Tables 2–4).

Table 3
Parameters, estimated inputs, and calculated percolation rates and lag times to the water table for profiles located in natural rangeland ecosystems and rain-fed agroecosystems with medium–coarse grained soils.

Profile	Interval (TD) (m)	Clay (%)	θ (m^3/m^3)	MP (m)	Cl_{Max} (mg/L)	N_{Max} (mg/L)	Cl (mg/L)	N (mg/L)	P (mm/yr)	Cl_P (mg/L)	Pe_V (mm/yr)	Pe_{CMB} (mm/yr)
Natural (medium–coarse grained soils)												
18	1.0–8.3	13	0.09	−98	80	44	27	11	585	0.26	–	8.7
19	2.4–12.2	18	0.06	−24	250	1.3	83	<0.5	575	0.26	–	2.8
20	1.0–10.7	7	0.08	−130	60	120	33	35	580	0.26	–	5.9
21	1.0–11.7	25	0.05	−69	110	32	51	8.2	545	0.24	–	3.7
Median		16	0.07	−84	95	38	42	9.6	578	0.26	–	4.8
Rain-fed (medium–coarse grained soils)												
22	1.0–8.4 (15.2)	25	0.19	−1.8	33	2.7	8.1	1.1	585	0.26	48	30
23	1.0–7.5 (9.4)	29	0.10	−61	29	9.3	20	6.9	595	0.26	6.7	9.8
24	1.0–9.6	19	0.12	−3.1	24	36	6.9	23	580	0.26	10	36
25	0.9–7.3	18	0.17	−35	24	17	13	2.6	455	0.22	17	10
Median		22	0.15	−19	27	13	11	4.8	583	0.26		27[a]
Drainage (coarse-grained soil)												
33	0.6–8.4	8	0.17	−2.3	74	2.5	18	0.7	555	0.24	–	17[b]

Profile: profile number (Fig. 1), *interval* (TD): borehole depth interval below the root zone impacted by natural conditions or rain-fed agriculture and total depth (TD) if greater than the profile base, *clay*: depth-weighted mean soil clay content to a depth of ~1.8 to 2.0 based on SSURGO, θ: depth-weighted mean water content, MP: depth-weighted mean matric potential, *inventory*: mass of *Cl* and NO_3-N (*N*) in the profile interval normalized by interval thickness, Cl_{Max} and N_{Max}: peak Cl and NO_3-N concentrations, Cl and N: depth-weighted mean Cl and NO_3-N concentrations *P*: mean annual precipitation (1971–2000, Prism Climate Group, www.prism.oregonstate.edu), Cl_P: mean chloride concentration in bulk annual precipitation from NADP (~1980–2007), Pe_V: percolation based on velocity method, Pe_{CMB}: percolation based on CMB method.

[a] Median of maximum values.

[b] Minimum value that does not account for runon.

Table 4
Profile parameters, estimated inputs, and calculated percolation rates and lag times to the water table for profiles located in irrigated agroecosystems with fine- to coarse-grained soils.

Profile	Interval (TD) (m)	Clay (%)	θ (m^3/m^3)	MP (m)	Cl_{Max} (mg/L)	N_{Max} (mg/L)	Cl (mg/L)	N (mg/L)	P (mm/yr)	Cl_P (mg/L)	I (mm/yr)	Cl_I (mg/L)	t_I (yr)	Pe_V mm/yr	Pe_{CMB} (mm/yr)
Irrigated (fine–medium grained)															
26	1.0–4.6 (9.7)	42	0.26	−56	46	73	40	47	575	0.26	420	9.0	25	35	100
27	0.9–6.1 (9.8)	45	0.23	−26	82	34	29	18	445	0.22	635	8.8	20	52	150
28	1.0–3.7 (18)	42	0.25	−16	170	80	110	48	485	0.24	440	8.6	19	36	33
29	0.3–10.4 (15.2)	43	0.31	−10	270	170	210	60	585	0.24	300	35	55	49	54
30	0.9–5.0	36	0.26	−20	400	56	180	42	455	0.22	635	36	36	29	170
31	1.2–2.8	25	0.19	−59	650	430	470	290	440	0.20	635	6.8	37	–	–
Median		42	0.26	−23	220	77	150	48	470	0.23	540	8.9	31	36	100
Irrigated (coarse-grained)															
32	0.6–10.4	13	0.14	−13	1,000	30	200	8.0	575	0.26	200	4.0	35	38	110
Median[a]		42	0.25	−20	270	73	180	47	485	0.24	440	8.8	35	37	110

Profile: profile number (Fig. 1), *interval* (TD): borehole depth interval below the root zone impacted by irrigated agriculture and total depth (TD) if greater than the profile base, *clay*: depth-weighted mean soil clay content to a depth of ~1.8 to 2.0 based on SSURGO, θ: depth-weighted mean water content, MP: depth-weighted mean matric potential, Cl_{Max} and N_{Max}: peak Cl and NO_3-N concentrations, Cl and N: depth-weighted mean chloride and NO_3-N concentrations, *P*: mean annual precipitation (1971–2000, Prism Climate Group, www.prism.oregonstate.edu), Cl_P: mean chloride concentration in bulk annual precipitation from NADP (~1980–2007), *I*: mean annual irrigation application from land owner records, Cl_I: chloride concentration in irrigation water, t_I: time of irrigation, Pe_V: percolation based on velocity method, Pe_{CMB}: percolation based on CMB method.
[a] Median values include all profiles.

Core subsamples (25 g) were leached using 40 mL of double deionized water to analyze chemical parameters. The mixture was shaken for 4 h in a reciprocal shaker, centrifuged at 7000 rpm for 20 min, and the supernatant was filtered (0.2 μm filter). Core subsamples were then oven dried at 105 °C for 48 h to determine gravimetric water content. Water-extractable concentrations of Cl and NO_3 were measured using ion chromatography (Dionex ICS 2000; EPA Method 300.0). Water-extractable ion concentrations are expressed on a mass basis as mg ion per kg of dry soil and were calculated by multiplying ion concentrations in the supernatant by the extraction ratio (g water/g soil). Ion concentrations are also expressed as mg ion per L of soil pore water and were calculated by dividing concentrations in mg/kg by gravimetric water content and multiplying by water density.

Total potential data are used to determine the direction of water movement because water moves from regions of high to low total potential (sum of matric, gravitational, and osmotic potentials). Osmotic potential is generally ~10% of total potential and is usually ignored in low-salinity environments (Scanlon et al., 2003). Matric potentials were measured in the laboratory on soil samples collected in the field. Tensiometers (Model T5, UMS, Munich, Germany) were used to measure matric potentials in the wet range (≥−8 to 0 m) and a chilled-mirror psychrometer (Model WP4T, Decagon Devices, Pullman, WA) was used to measure water potentials (matric + osmotic potential) in the dry range (≤−8 m).

2.3. Data analysis

2.3.1. Groundwater data

The impact of agroecosystems on regional groundwater quantity was evaluated by examining groundwater level changes between predevelopment (~1955) and 2007 (McGuire, 2009). Time series of individual well hydrographs were also evaluated to determine rates of groundwater level declines through time in different areas.

Regional recharge rates were estimated using the CMB approach applied to groundwater Cl concentrations (Cl_{GW}) from the Texas Water Development Board (TWDB) database (www.twdb.state.tx.us). According to the CMB approach, Cl input (Cl_P) from bulk precipitation (*P*) balances Cl output from recharge (R_{CMB}):

$$P \times Cl_P = R_{CMB} \times Cl_{GW}; \quad R_{CMB} = \frac{P \times Cl_P}{Cl_{GW}} \qquad (1)$$

where Cl_{GW} is Cl concentration in groundwater. Concentrations of Cl in precipitation were obtained from the National Atmospheric Deposition Program (http://nadp.sws.uiuc.edu/). The Cl concentrations in precipitation were doubled to account for dry fallout, which is consistent with total Cl fallout based on prebomb $^{36}Cl/Cl$ ratios at Amarillo, Texas (Scanlon and Goldsmith, 1997). The above analysis assumes that there is no input of Cl from irrigation water based on limited unsaturated profile data from McMahon et al. (2006) and Scanlon et al. (2008b). However, groundwater Cl can also be derived from underlying more saline aquifers. Mass concentration (mg/L) ratios of groundwater Cl/Br and Cl/SO_4 were used to distinguish Cl of meteoric origin from precipitation from Cl derived from upward flow of saline groundwater from deeper aquifers. Water quality data for 2285 wells in the Texas portion of the CHP were obtained from the TWDB database (www.twdb.state.tx.us) and for an additional 260 wells in neighboring states (Oklahoma and New Mexico) from the National Water Information System (NWIS) database (waterdata.usgs.gov/nwis). The Cl and SO_4 concentration data represent samples analyzed between 1938 and 2009 (median 1976). The Br concentration data for the region are much more limited, with data for 455 wells representing samples analyzed between 1991 and 2009 (median 1999).

Recharge rates were also estimated using the water table fluctuation method (Healy and Cook, 2002) applied to rising groundwater levels based on data from the TWDB database. The impact of agroecosystems on groundwater quality was also evaluated by comparing median concentration values of total dissolved solids (TDS) and NO_3-N for TWDB database samples analyzed before and after 1981. Both of these constituents can be impacted by land use change from natural ecosystems to agroecosystems through flushing of accumulated salts in the unsaturated zone (Scanlon et al., 2007) and through mobilization of mineralized soil organic nitrogen and/or nitrogen fertilizers (Scanlon et al., 2008b).

2.3.2. Unsaturated zone data

Concentrations of Cl in soil water can be used as a qualitative indicator of percolation below the root zone or recharge at the water table. Chloride from bulk precipitation moves into the subsurface with infiltrating water. Vegetation takes up water through evapotranspiration and excludes Cl. Therefore, areas of low percolation/recharge have large Cl buildups or bulges because most of the infiltrated water is taken up by vegetation and little, if any, water percolates below the root zone. In contrast, areas of high percolation/recharge have low Cl concentrations because Cl is flushed through the soil profile, such as beneath playas (Scanlon and Goldsmith, 1997).

Natural rangeland ecosystems are often characterized by large bulges of Cl that accumulated over millennia, indicating no percolation/recharge. These Cl bulges serve as a marker to delineate increases in percolation below the root zone or recharge at the water table caused by land use change. Downward displacement of these bulges can be used to estimate percolation (Pe_V) and/or recharge (R_V) rate from the velocity of the Cl front (v_{cf}, upper part of the bulge where Cl concentration increases sharply with depth):

$$Pe_V = R_V = v_{cf} \times \bar{\theta} = \frac{\Delta z}{\Delta t}\bar{\theta} \tag{2}$$

where $\bar{\theta}$ is the mean volumetric water content over this depth interval. Percolation/recharge rates can be calculated if the depth interval over which the increased water flux (Δz) and corresponding time (Δt) can be identified.

The CMB approach can be applied to Cl concentrations in unsaturated zone pore water (Cl_{UZ}) to estimate percolation (Pe_{CMB}) and/or recharge (R_{CMB}) (Allison and Hughes, 1983). In irrigated areas, Cl input from irrigation water needs to be considered also:

$$P \times Cl_P + I \times Cl_I = Pe_{CMB} \times Cl_{UZ} = R_{CMB} \times Cl_{UZ};$$

$$Pe_{CMB} = R_{CMB} = \frac{P \times Cl_P + I \times Cl_I}{Cl_{UZ}} \tag{3}$$

where Cl_P, Cl_I, and Cl_{UZ} are Cl concentrations in precipitation, irrigation water, and unsaturated zone pore water, respectively. Fertilizers can also contribute Cl to soil pore water; however, landowners indicated that they did not apply fertilizers containing Cl where the boreholes were drilled in this study. The irrigation term is excluded in non irrigated sites. Concentrations of Cl in irrigation water were measured in this study using ion chromatography or were estimated from nearby wells from the TWDB groundwater database (www.twdb.state.tx.us). The velocity based-approach for estimating percolation/recharge (Eq. 2) has much fewer inputs, and each input can be estimated from the measured Cl profile; therefore, percolation/recharge rates from this approach are considered to be more reliable than those based on the CMB approach where practicable. The accumulation time represented by Cl in the unsaturated zone can be determined by dividing the cumulative total mass of Cl for the depth interval related to a particular land use by the Cl input:

$$t = \frac{\int_0^z \theta \times Cl_{uz}\, dz}{P \times Cl_P + I \times Cl_I} \tag{4}$$

There is a time lag between increases in percolation below the root zone and when the water reaches the water table to become recharge. In this process, a wetting front migrates ahead of the Cl front by an amount equal to the displaced water in the depth zone impacted by increased percolation. The velocity of the wetting front and Cl front would be the same if there was no initial water in the profile, and differences between the two velocities increase as initial water content in the profile increases (Scanlon et al., 2007). Recharge occurs when the wetting front reaches the water table. Wetting front velocity (v_{wf}) can be determined from the percolation rate based on the Cl front velocity (Pe_V) or the CMB (Pe_{CMB}) approaches and the difference in mean volumetric water content between post cultivation (i.e., new water, θ_n), and pre cultivation (i.e., old water, θ_o) and the time lag (t_L) for recharge can be calculated as follows:

$$v_{wf} = \frac{Pe_V \text{ or } Pe_{CMB}}{\bar{\theta}_n - \bar{\theta}_o} \quad t_L = \frac{WT_d - z}{v_{wf}} \tag{5}$$

where z is the root zone depth (~1 m) or some other pre specified depth, and WT_d is water table depth.

3. Results and discussion

Groundwater-fed irrigation has large-scale impacts on water resources in the study area, causing steep declines in groundwater levels. The renewability of these resources is evaluated by quantifying percolation/recharge rates using unsaturated zone data (Sections 3.2–3.4) and groundwater data (Section 3.5). Impacts of agroecosystems on groundwater quality are also evaluated (Section 3.6), and recharge rates from this study are compared with those from other studies (Section 3.7). The implications of the findings from this study for water resources management are described in Section 3.8.

3.1. Impacts of irrigated agroecosystems on groundwater quantity

Irrigation in the CHP accounts for ~95% of groundwater abstractions in the region. Regional groundwater level declines >7.5 m represent 41% of the area and 95% of the total water storage change, whereas declines >30 m represent only 10% of the area but account for 36% of the storage change (Table 5 and Fig. 3) (McGuire, 2009). Groundwater level changes of ±3 m cover 48% of the area, indicating no systematic variation over time in this area. Representative hydrographs show declines of ≤1.3 m/yr over long times (Fig. 4). There is no systematic variation in the time series of water level declines, with most showing fairly uniform declines over time, and the data generally do not suggest greater declines during predominantly flood irrigation and lesser declines during predominantly center pivot sprinkler irrigation. Some hydrographs show leveling off of declines at later times that may be related to proximity of water levels to the base of the aquifer (Fig. 4c), and others show more rapid declines later in time (Fig. 4a). Many individual hydrographs show declines ≥30 m in the northwest, where ~80% of cultivated land is equipped for irrigation (Fig. 1). The largest declines south of the Canadian River correspond to intense abstraction by the CRMWA for the city of Amarillo (population 175,000) based on a well field located in a thick part of the aquifer related to a paleovalley. Some regions in the extreme southeast show rising water tables (~1% of CHP) in very sandy soils as described previously.

3.2. General relationship between soil texture/land use and percolation/recharge

The impacts of soil texture and land use on percolation/recharge rates are best described using data from the unsaturated zone profiles (Fig. 5). In f–m grained soils, there is no percolation/recharge beneath natural ecosystems or rain-fed agroecosystems (Fig. 5a and Table 2). In contrast, in m–c grained soils, percolation/recharge rates are low beneath natural ecosystems (median 4.8 mm/yr) and moderate beneath rain-fed agroecosystems (median 27 mm/yr, Table 3). Inventories of Cl are inversely related to percolation/recharge rates: large inventories under f–m grained soils correspond to absence of percolation/recharge, and much lower inventories under m–c grained soils correspond to higher percolation/recharge rates (Fig. 5b and Table S1). Irrigation increases percolation/recharge beneath all agroecosystems (median 36 mm/yr, f–m grained soils, 38 mm/yr, m–c grained soils) (Table 4). Inventories of Cl are variable under irrigated agroecosystems, depending on the quality of the irrigation water (r = 0.91) and the percolation/recharge rate.

Table 5
Water level changes in the study area and sub-regions between predevelopment (~1955) and 2007, according to McGuire (2009).

Region	Area (km^2)	% of total area	ΔS (km^3)	% of ΔS	CMB flux (mm/yr)	CMB recharge (km^3/yr)	Total recharge (km^3)	Deficit
CHP Texas	37,000	100	−57	100	11.0	0.41	21	−2.7
>7.5 m WL decline	15,300	41	−54	95	10.3	0.16	8.2	−6.6
>15 m WL decline	11,400	31	−48	83	10.8	0.12	6.4	−7.4
>30 m WL decline	3600	10	−21	36	11.2	0.04	2.1	−9.9
Cultivated	10,500	28	−32	55	11.4	0.12	6.2	−5.1
Irrigated	6500	17	−22	38	11.0	0.07	3.7	−5.8

Region: analysis region defined by different land use or degree of water level change (Figs. 1 and 3), *area*: analysis region area, *% of total area*: percentage of study area, ΔS: change in storage between ~1955 and 2007, *% of* ΔS: percent of total storage change in the analysis region, *CMB flux*: median annual recharge rate estimated from groundwater chloride mass balance, *CMB recharge*: estimated annual recharge volume, *total recharge*: total estimated recharge volume between ~1955 and 2007, *deficit*: ratio between recharge and change in storage between ~1955 and 2007.

The impact of different soil textures and land uses on subsurface water movement can also be distinguished in a 3-D plot of median Cl and NO_3-N concentrations and matric potential (Fig. 6 and Tables 2–4). In f–m grained soils, profiles under different land use settings fall into distinct groups in this plot. Because unsaturated zone Cl concentrations are inversely proportional to percolation/recharge Eq. (3), large Cl bulges under natural ecosystems and rain-fed agroecosystems indicate that there is little or no percolation/recharge in these settings, whereas lower Cl concentrations under irrigated agroecosystems indicate that percolation/recharge (36 mm/yr) is occurring. Matric potential data are consistent with Cl data, with low matric potentials under natural ecosystems and rain-fed agroecosystems indicating dry soils and much higher matric potentials under irrigated agroecosystems indicating much wetter soils. Low NO_3-N concentrations under natural ecosystems are attributed to accumulation of NO_3-N as soil organic N (SON). Higher NO_3-N concentrations under rain-fed and irrigated agroecosystems are attributed to addition of fertilizers and mineralization and nitrification of SON associated with the beginning of cultivation, similar to the SHP (Scanlon et al., 2008b).

In m–c grained soils, all profiles show evidence of percolation/recharge. Under natural ecosystems (four profiles), low Cl and moderate matric potentials indicate low percolation/recharge rates (median 4.8 mm/yr) (Fig. 6 and Table 3). Under rain-fed agroecosystems (four profiles), lower Cl and higher matric potentials indicate higher percolation/recharge rates (median 27 mm/yr) relative to those under natural ecosystems. Only one profile was drilled under irrigated agroecosystems in sandy loam soils, which has moderate Cl and high matric potentials, with a percolation/recharge rate of 38 mm/yr. There is no systematic variation in NO_3-N among land use settings in m–c grained soils (Tables 3 and 4).

The following sections emphasize the impacts of land use on subsurface flow by comparing vertical profiles of matric potential, Cl, and NO_3-N within the same soil texture f–m grained soils (Fig. 7) and m–c grained soils (Fig. 8).

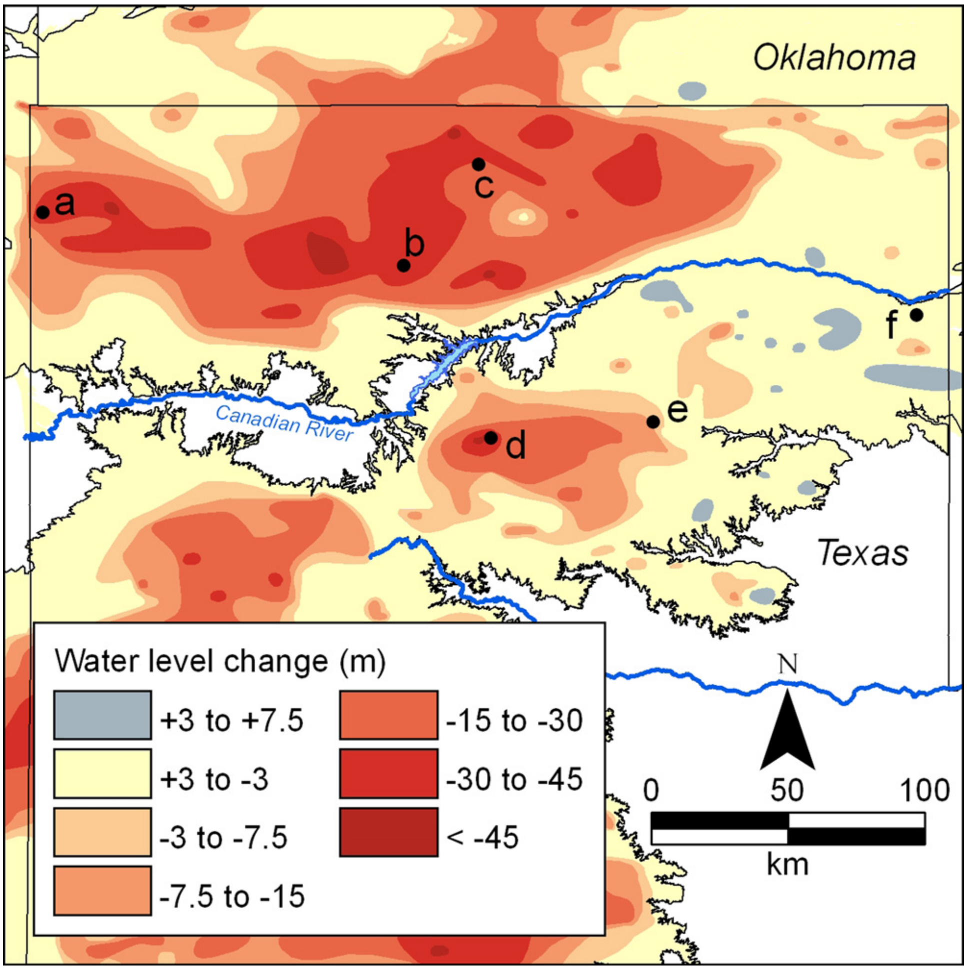

Fig. 3. Spatial distribution of water level elevation changes between predevelopment (~1955) and 2007 in the Texas Central High Plains on the basis of data compiled by McGuire (2009). These changes in water levels were calculated by subtracting predevelopment water level data from 2007 data. In contrast, water level changes and water storage changes in McGuire (2009) were calculated by subtracting predevelopment data from 2000 data and annual changes through 2007. Points in this map represent locations of groundwater hydrographs shown in Fig. 4.

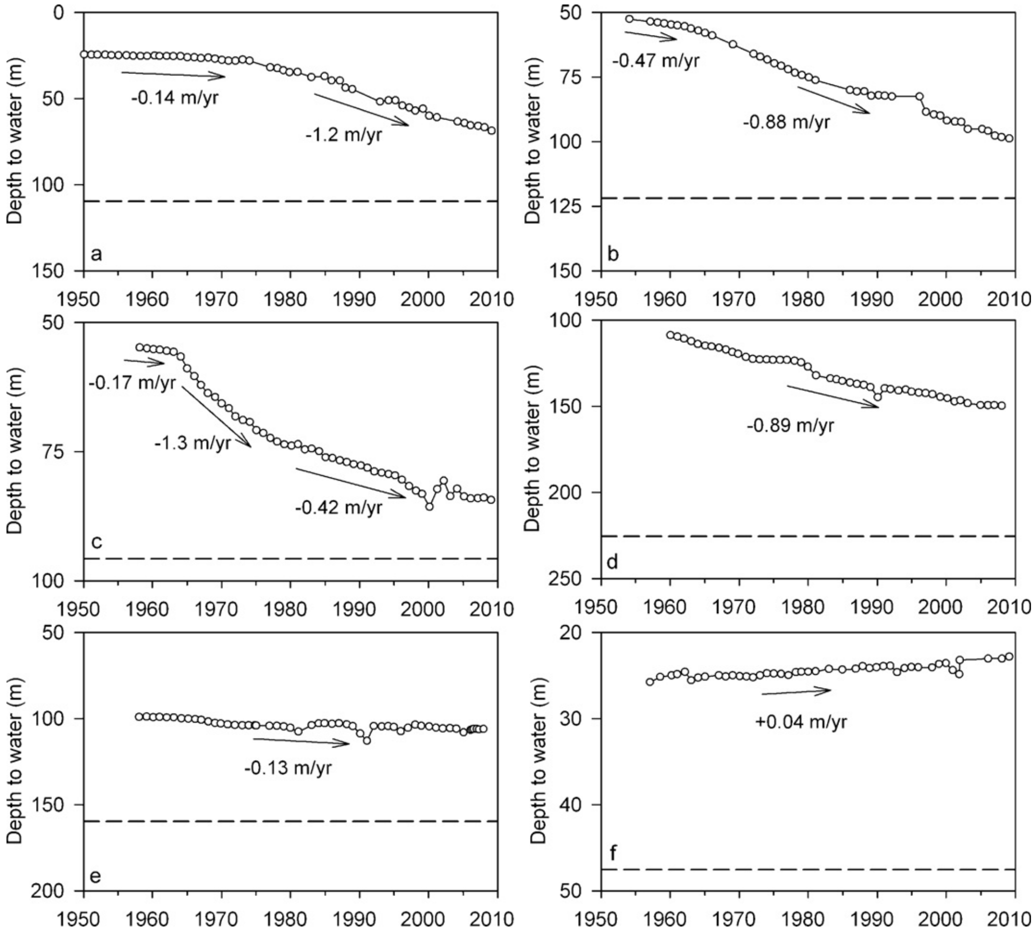

Fig. 4. Representative groundwater well hydrographs for locations shown in Fig. 3. Dashed lines represent approximate depth of aquifer base. Depth to water rate changes are based on the slopes of linear regressions fit to measurements during periods of generally consistent changes.

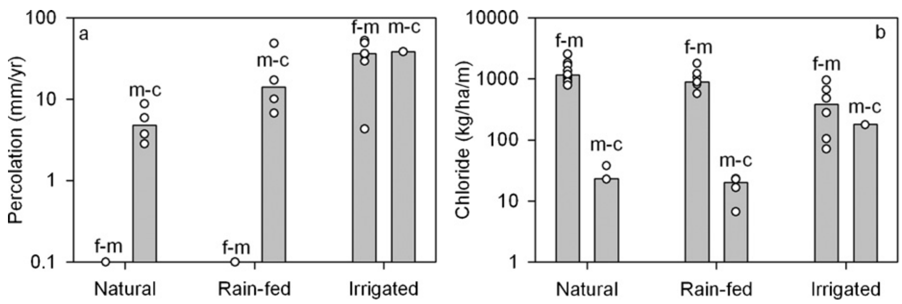

Fig. 5. Distribution of (a) borehole profile percolation rates and (b) borehole Cl inventory normalized by borehole depth for different land use settings (natural, rain-fed, and irrigated) and soil textures (f–m: fine–medium grained, m–c: medium–coarse grained). Bar heights represent median values; points represent data values.

3.3. Percolation/recharge rates in fine–medium grained soils

Natural ecosystems (10 profiles) in f–m grained soils have no percolation/recharge in interplaya settings, as shown by large Cl bulges with high peak concentrations (median 2400 mg/L; Table 2, Figs. 7b and S5). Large Cl inventories are found in these Cl bulges (median 1200 kg/ha/m), which represent accumulations since Pleistocene times (median 12,200 yr) (Tables 2 and S1). Some (seven) of these profiles are in clay loam soils; the remaining profiles (three) are in loam and fine-sandy loam soils. Water potential profiles are consistent with Cl data, with upward total potential head gradients (water potential + gravitational potential) indicating upward water movement, discharging through evapotranspiration (ET) (Fig. 7a). Previous modeling analyses reproduced the Cl and matric potential data using long-term drying since Pleistocene times (Scanlon et al., 2003). The lack of recharge beneath these interplaya natural ecosystem settings is consistent with results from previous studies in the Texas High Plains (Wood and Sanford, 1995; Scanlon and Goldsmith, 1997). Recharge in this setting is focused beneath ephemeral lakes or playas (Fig. 2). Most (89%) of the playas in the Texas CHP are in clay loam soils. Areas with low playa densities generally have surface drainages (Fig. S3) that can also focus recharge, as shown by data from profile 33 in a drainage with an estimated recharge rate of 17 mm/yr (Table 3 and Fig. S6). This recharge rate is considered a minimum because data on the amount of runon into the playas and Cl concentration in runon were not available to include in the chloride mass balance recharge estimate. Therefore, drainages, along with playas, focus recharge in areas of natural ecosystems. Concentrations of NO_3-N are generally low in these profiles (median 2.4 mg/L) and are attributed to NO_3-N accumulation mostly as soil organic nitrogen (SON) rather than as NO_3-N, similar to natural ecosystems in the SHP (Scanlon et al., 2008b). Slightly higher NO_3-N values are found in profiles 1 and 2 (12, 13 mg/L), which may reflect some accumulation as NO_3.

Beneath rain-fed agroecosystems (seven profiles) there has been little or no percolation in f–m grained soils during the past 10,100 yr (median accumulation time), as shown by large Cl bulges, similar to profiles under natural ecosystems in the same soil types (Figs. 7d and S7, Table 2). Median peak Cl concentration is half of that found under natural ecosystems and peak Cl concentrations are slightly deeper (median 3.4 m) than those beneath natural ecosystems (median 2.4 m). Matric potentials are generally higher than those beneath natural ecosystems and suggest that the profiles may be affected slightly by downward movement of a pressure front that precedes

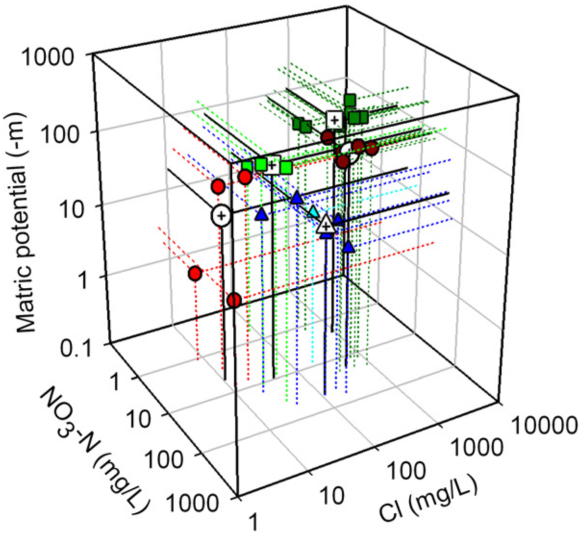

Fig. 6. Relationship between depth-weighted mean values of matric potential and Cl and NO_3-N concentration for different ecosystem setting borehole profiles. *Fine–medium grained soils*: median of profile mean values: natural ecosystems: Cl: 760 mg/L; matric potential: −150 m; NO_3-N, 2.4 mg/L; rain-fed agroecosystems: Cl: 460 mg/L; matric potential, −100 m; NO_3-N 17 mg/L; irrigated agroecosystems: Cl 150 mg/L; matric potential −23 m; NO_3-N 48 mg/L. *Medium–coarse grained soils*: median of profile mean values: natural ecosystems: Cl: 42 mg/L; matric potential: −84 m; NO_3-N, 9.6 mg/L; rain-fed agroecosystems: Cl: 11 mg/L; matric potential, −19 m; NO_3-N 4.8 mg/L; irrigated agroecosystems: profile 32: Cl 200 mg/L; matric potential −13 m; NO_3-N 8.0 mg/L. Symbol shapes and primary colors represent ecosystem setting (natural: green squares, rain-fed: red circles, irrigated: blue triangles). Symbol color tones represent general soil texture (finer-grained soils: darker tones, coarser-grained soils: lighter tones). Median values shown with white-fill symbols. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

solute fronts; however, the Cl bulge has generally not been displaced below the root zone in these profiles. Concentrations of NO_3-N are much higher under rain-fed agroecosystems near the root zone and may be attributed partly to fertilizer application within the root zone. In addition, mineralization and nitrification of SON may have occurred below the root zone associated with the beginning of cultivation because soil water in this zone predates fertilizer application given Cl accumulation times Eq. (4). This process was found to occur in many regions of the SHP that are related to initiation of cultivation in the SHP (Scanlon et al., 2008b).

Beneath irrigated agroecosystems (six profiles) in f–m grained soils, percolation rates range from 29 to 52 mm/yr using the CFD method (Eq. 2) (Figs. 7f and S8, Table 4). This excludes one profile (31), which is too shallow to provide a reliable estimate. Low median Cl and high median matric potential in the irrigation-impacted zones in all profiles are consistent with percolation under irrigated agroecosystems. The Cl fronts found at the bases of three of the six profiles (26, 28, and 29) correspond to the upper part of large Cl bulges that represent accumulations under natural ecosystems that were displaced downward by increased percolation under irrigated agroecosystems (Fig. S8). Downward displacement of Cl bulges is evident when comparing nearby profiles in natural and irrigated agroecosystems in clay loam soils (Fig. 7b and f). The Cl bulges serve as markers for the depth intervals impacted by irrigation return flow in these profiles. Because profiles under rain-fed agroecosystems in these soils show no displacement of the Cl bulge (Fig. 7d), all displacement is attributed to percolation under irrigation and none to rain-fed management that preceded irrigation. The presence of Cl bulges indicates that increased percolation in these profiles has not recharged the aquifer. The remaining three profiles do not have a Cl bulge at the base that may result from the profiles being too shallow (e.g., profile 33, 2.8 m deep) or may reflect higher percolation. One of the profiles (profile 27) has low Cl concentrations at the base relative to current measurements in irrigation water at this site (Fig. S8), suggesting that either 40 to 80% of irrigation return flow percolated through the profile or a period of rain-fed agriculture is being reflected. Assuming the latter results in an estimated percolation rate for the irrigated section of the profile of 52 mm/yr, similar to those estimated for other profiles. Calculating percolation using the entire profile depth may underestimate actual percolation for profiles that do not include a Cl bulge (e.g., profile 30). The resultant median percolation rate for irrigated agroecosystems, excluding the shallow profile (31) is 36 mm/yr. Water table depths under these profiles range from 76 to 112 m (Table S1); therefore, it is highly unlikely that irrigation is recharging the aquifer at any of these sites.

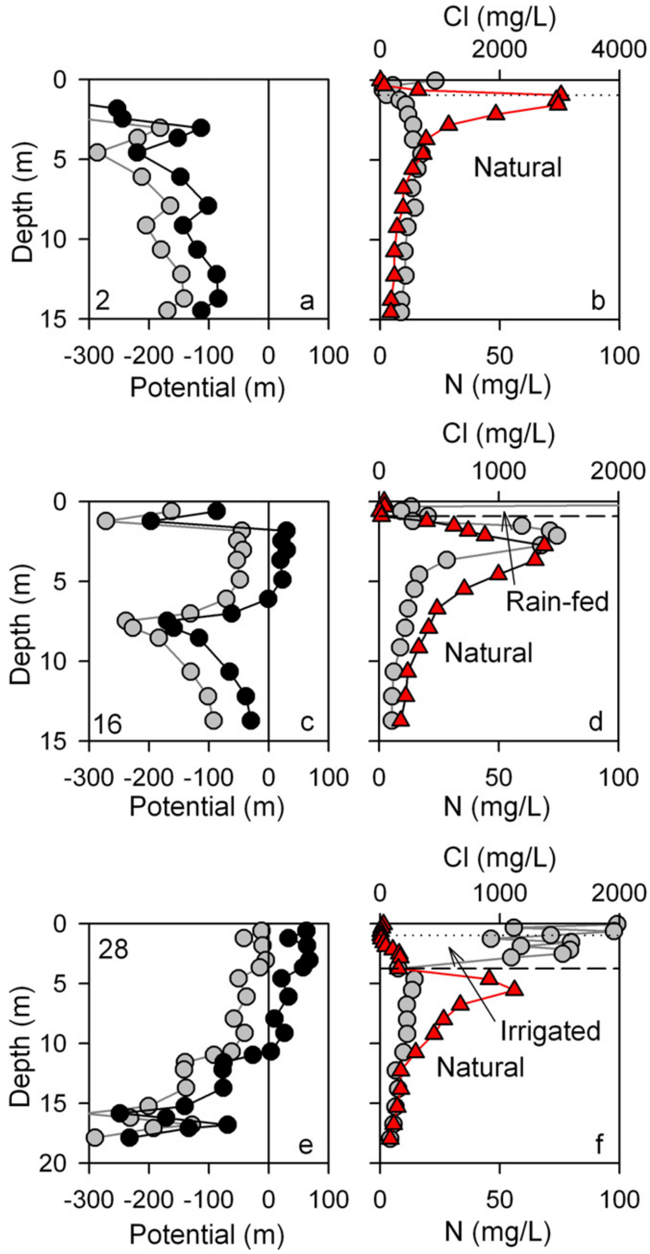

Fig. 7. Profiles in f–m grained soils including natural (2), rain-fed (16), and irrigated (28) profiles of potential (water potential: gray circles, total potential: black circles), Cl (red triangles), and NO_3-N (gray circles) for profiles located in f–m grained soils. Dotted horizontal lines represent the base of the root zone. Dashed horizontal lines represent the base of the rain-fed or irrigation impacted zone with respect to Cl concentrations. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

3.4. Percolation/recharge rates in medium- to coarse-grained soils

Beneath natural ecosystems in m–c grained soils (four profiles) in the southeast part of the CHP and along the Canadian River, areally, distributed percolation/recharge occurs (median 4.8 mm/yr, 1% of precipitation), as shown by relatively low Cl (Table 3, Figs. 8b and S9). Moderately high NO_3-N levels in these sandy soils (median 9.6 mg/L) are attributed to accumulation of some N as NO_3-N rather than as SON.

Beneath rain-fed agroecosystems (four profiles), low Cl concentrations (median 11 mg/L) indicate moderate percolation/recharge rates in the northwest and southeast parts of the CHP (Figs. 8d and S10, Table 3). Two of the profiles did not extend deeply enough for the entire section impacted by rain-fed agriculture to be sampled; therefore, percolation/recharge estimates based on profile depth (10 and 17 mm/yr) represent a lower bound on actual rates. The higher percolation/recharge rate based on the CMB approach for one of the profiles (36 mm/yr) may be more reliable; however, the lower CMB recharge estimate for the other profile (10 mm/yr) may

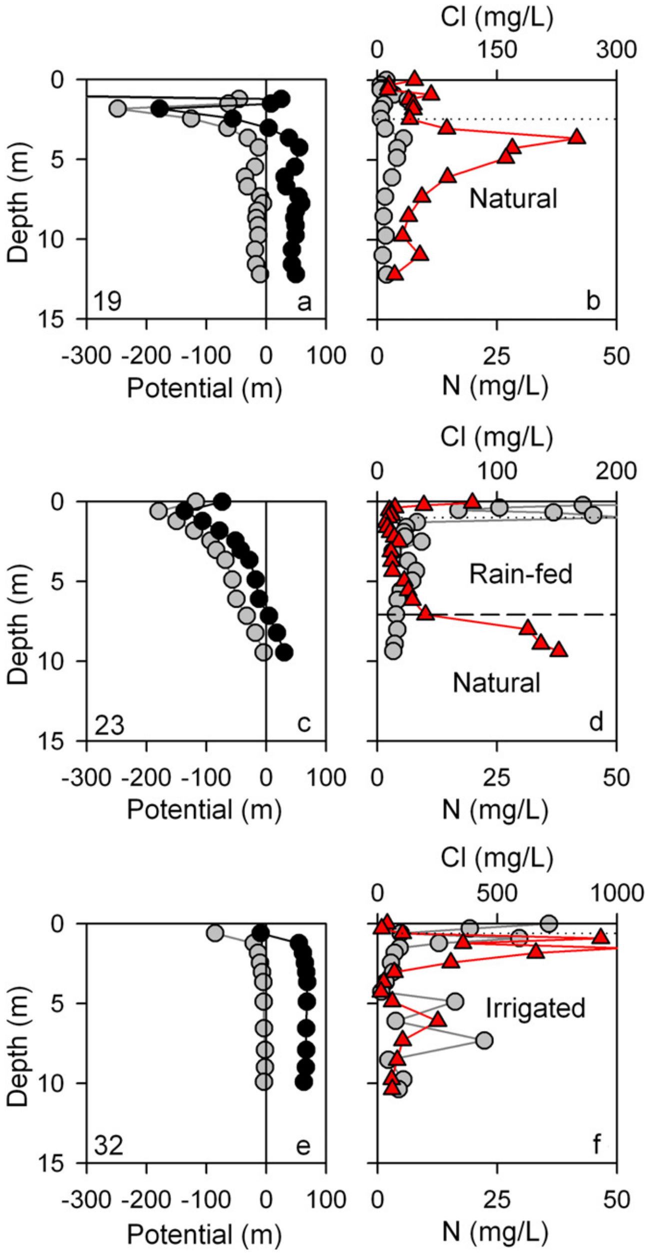

Fig. 8. Profiles in m–c grained soils including natural (19), rain-fed (23), and irrigated (32) profiles of potential (water potential: gray circles, total potential: black circles), Cl (red triangles) and NO_3-N (gray circles) for profiles located in m–c grained soils. Dotted horizontal lines represent the base of the root zone. Dashed horizontal lines represent the base of the rain-fed impacted zone with respect to Cl concentrations. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

reflect residual Cl in more clayey layers. The maximum percolation/recharge rates are considered more reliable because various processes can reduce these rates, such as residual Cl in the profile from incomplete flushing. The resultant median percolation/recharge rate based on maximum values of Pe_v or Pe_{CMB} is 27 mm/yr (5% of precipitation). Percolation/recharge rates beneath these rain-fed agroecosystems are generally much higher than those beneath natural ecosystems (median 4.8 mm/yr). Profile mean matric potentials are also much higher, indicating wetter soils.

There is only one profile (32) under irrigated agroecosystems in m–c grained soils, which has moderate Cl and high matric potential, with a resultant percolation/recharge rate of 38 mm/yr, similar to rates under f–m grained soils (Fig. S11 and Table 4). The percolation/recharge rate represents a lower bound because the entire irrigated section is not sampled in the profile; there is no Cl front at the base of the profile.

3.5. Regional recharge estimates based on groundwater chloride data and groundwater hydrographs

The unsaturated zone data described in the previous sections indicate that recharge to groundwater is focused beneath playas and surface water drainages and is areally distributed in coarser-grained soils to the southeast and west. The regional recharge estimates based on groundwater Cl data integrate recharge from playas and drainages (Figs. 9 and S12). Ratios of Cl/Br and/or Cl/SO_4 were used to evaluate Cl contribution from upward movement of more saline water from underlying geologic units (Figs. S13 and S14). Typical Cl:Br mass ratios of precipitation range from 50 to 150, and ratios in fresh groundwater generally range from 100 to 200, whereas ratios in groundwater affected by salt dissolution range from 1000 to 10,000 (Davis et al., 1998).

Ratios of Cl/Br throughout much of the CHP are within the range of those typical of precipitation and fresh groundwater; however, high Cl:Br ratios in the east (300–700) include a previously mapped saline plume (~250 km^2 in area) attributed to topographically driven, upward cross-formational discharge of paleowater from the underlying Dockum Formation that had mixed with salt dissolution zone water from the underlying evaporite confining unit of Permian age (Mehta et al., 2000). In addition to discharging along the north part of the Amarillo Uplift, this cross-formational discharge also occurs along the Canadian River and to the northeast. The high Cl/Br ratios are attributed to low Br concentrations typical of recrystalized halite. Ratios of $Cl:SO_4$ greater than 1 are also characteristic of this saline plume and are generally consistent with the high Cl:Br ratio (>300) area, suggesting that groundwater throughout this region may be impacted by upward cross-formational flow. Therefore, groundwater Cl data should provide a lower bound on actual recharge rates in this region.

Estimated recharge rates range from 4 to 23 mm/yr in the west half of the CHP, where groundwater Cl data are not affected by deep brines (Fig. 9). Recharge rates ≤5 mm/yr in the eastern region may not be reliable because of additional Cl supplied by upward cross-formational flow. High recharge rates (14–41 mm/yr) along the eastern escarpment are attributed to very sandy soils (clay content 5–12%, Fig. S1). Ratios of Cl/Br and Cl/SO_4 in this region suggest that it should not be impacted by upward movement of more saline water. The recharge rates are generally consistent with estimated higher recharge from unsaturated profile data in coarser-grained soils (2.8–48 mm/yr, Tables 3 and 4) (Table S2). The area where Cl/SO_4 ratios are >1.0 (12,000 km^2) closely matches the area where Cl/Br ratios are >300 and recharge estimates in this area were excluded from regional calculations. Ratios of Cl/SO_4 more accurately delineate the zone impacted by cross-formational saline flow than ratios of Cl/Br because of much higher data density. The resulting median recharge rate is 11 mm/yr, both regionally and under cultivated areas.

Groundwater level hydrographs can also be used to estimate recharge. Hydrographs throughout much of the CHP show large declines related to irrigation pumpage (Section 3.1). The only region with rising groundwater levels, indicative of recharge, is in sandy zones in the southeast part of the CHP. Recharge rates estimated from the hydrographs range from 4.3 to 68 mm/yr (median 14 mm/yr) (Table S2). Total groundwater level rises range from 1.0 to 5.7 m over times ranging from 9 to 53 yr (median 30 yr). These recharge rates are within the range of those estimated from unsaturated zone Cl data in m–c grained soils (2.8–48 mm/yr, Table 3).

3.6. Impacts of irrigated agroecosystems on groundwater quality

Groundwater quality is relatively stable over time, as shown by the similarity between regional median total dissolved solids (TDS) and NO_3-N concentrations in groundwater sampled before and after 1981 (TDS: 314 and 311 mg/L; NO_3-N: 1.6 and 1.7 mg/L, respectively) (Table S3). Median TDS increased and decreased slightly in different counties, and median NO_3-N increased by ≤0.7 mg/L and decreased by ≤1.7 mg/L in different counties. The stability of groundwater quality is consistent with the dominant recharge source being playas and drainages, which have high-quality water.

Nitrate concentrations in groundwater are generally low, mostly ≤4 mg/L NO_3-N, which is considered background (Fig. S15). Wells with NO_3-N exceeding the EPA maximum contaminant level (MCL) are generally concentrated along the southeast near the escarpment. This region is extremely sandy with median clay content of ≤13% (SSURGO, USDA, 1995) and generally has the shallowest water table depths in the Texas CHP region (~15 to 20 m deep). Precipitation is also high in this region (600 mm/yr). A total of 32 sampled groundwater wells exceed the EPA MCL in this region, only 3 of which are located adjacent to concentrated animal feeding operations. Two unsaturated zone profiles in this region, beneath natural ecosystems and rain-fed agroecosystems, have high NO_3-N (profile 24, 23 mg/L, and profile 20, 35 mg/L; Table 3), indicating that N from precipitation may have been accumulating as NO_3-N, rather than as SON, beneath natural ecosystems during pre cultivation times. This natural NO_3-N source may be contributing to high groundwater NO_3-N in this region.

3.7. Comparison with recharge rates from other studies

Percolation/recharge rates from this study compare favorably with those from McMahon et al. (2003, 2006) for the CHP in Kansas, which range from 5 mm/yr in the Cimarron National Grassland (CNG) to 39 and 54 mm/yr under two irrigated

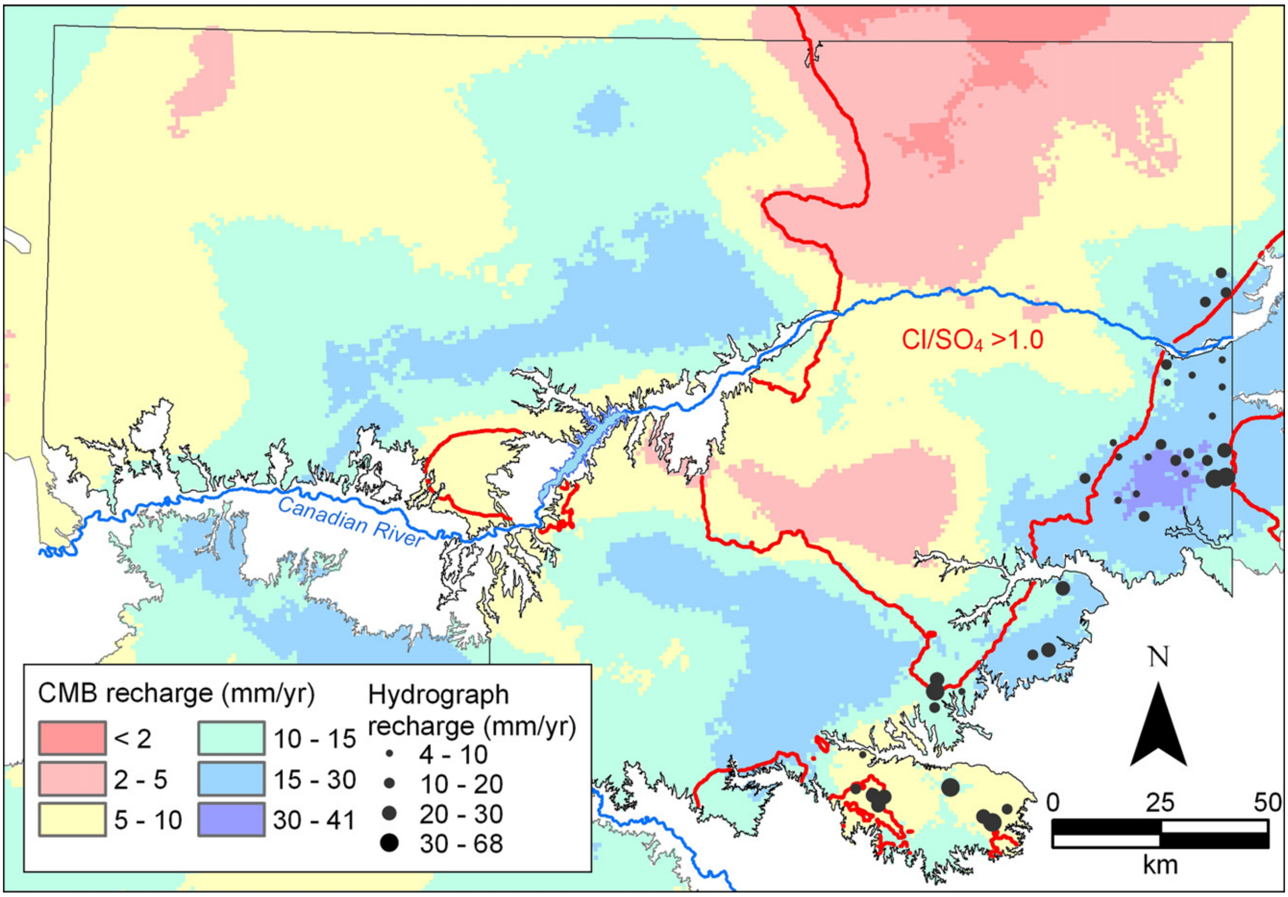

Fig. 9. Spatial distribution of groundwater recharge in the Texas Central High Plains based on the chloride mass balance (CMB) method using groundwater Cl concentrations from 2545 wells from the TWDB database and recharge rates calculated from water level increases for hydrographs of wells located in coarse-grained soils in the southeast region of the study area. Red line delineates region where groundwater $Cl/SO_4 > 1.0$ (Supporting information, Fig. S14). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

sites in central Kansas. Soil texture at the natural ecosystem site (CNG) is moderately coarse grained with ≤10% clay near the surface (STATSGO), and sand content ranges from 35 to 94% at depth. The percolation/recharge rate for this natural ecosystem site was based on the CMB approach and generally corresponds with those under natural ecosystems in m–c grained soils in the Texas CHP (median 4.8 mm/yr, Table 3). Percolation/recharge rates under irrigated agroecosystems in the Kansas CHP (39 and 54 mm/yr) are similar to those estimated in the Texas CHP in this study (29–52 mm/yr) (Table 4). Recharge rates for the irrigated sites in Kansas were based on penetration depth of bomb tritium in the profiles. The regional recharge rate in the Texas CHP (11 mm/yr) is similar to that estimated from previous studies in western Kansas (mean 7 mm/yr) (Sophocleous, 2004). Higher recharge rates were estimated in younger aquifers to the east in the Kansas CHP: Great Bend Prairie aquifer in central Kansas, 37 mm/yr (Sophocleous, 1992), and Equus Beds, 42 mm/yr in nonsand dune areas, 165 mm/yr in sand dune areas (Sophocleous, 2004). The higher recharge rates in these regions are attributed to coarser soils and higher precipitation (mean annual precipitation ∼640 and 760 mm). Regional recharge rates have not been estimated using groundwater Cl data in the CHP in Kansas. Groundwater NO_3-N concentrations in the Kansas part of the CHP can also be used as an indicator of recharge. A probability map of detecting groundwater NO_3-N concentrations exceeding background (4 mg/L) based on logistic regression shows low probability (0–20%) throughout much of the CHP region (Gurdak and Qi, 2006). Groundwater NO_3-N concentrations ranged from 0.25 to 61 mg/L in 30 samples, with screened intervals within 3 to 6 m of the water table at depths of 30 to 58 m beneath irrigated sites in the Kansas part of the CHP (Bruce et al., 2003). A total of 60% of the wells exceeded background, indicating recharge under many of the irrigated sites.

Differences in percolation/recharge under agroecosystems in the Texas CHP and the SHP are attributed to differences in soil texture, which is generally coarser grained throughout much of the SHP. Whereas there is no percolation/recharge under rain-fed agriculture in the f–m grained soils that dominate the CHP, recharge under rain-fed agroecosystems in the SHP ranges from 5 to 92 mm/yr (median 24 mm/yr) (Scanlon et al., 2007). High groundwater NO_3-N in much of the SHP also provides evidence of more widespread recharge under agroecosystems in this region relative to the CHP (Scanlon et al., 2008b, in press).

3.8. *Implications for water resources management*

Results from this study indicate that water resources are demand driven, with groundwater abstractions greatly exceeding recharge. During predevelopment, groundwater recharge equals groundwater discharge. Water for irrigation can be derived from a decrease in groundwater storage, a decrease in groundwater discharge to rivers and springs, and/or an increase in groundwater recharge. Although irrigated agriculture only represents 17% of the land surface, it has markedly reduced groundwater storage. Groundwater depletion between predevelopment (∼1955) and 2007 totals ∼57 km^3 (McGuire, 2009). Low predevelopment recharge in the CHP (∼11 mm/yr) results in low baseflow discharges to rivers and springs, providing very little flow for capture by irrigation and supported by groundwater modeling analyses (Dutton et al., 2001). Although recharge is higher in sandy soils, only 3% of the cultivated area is in these soils because of steeper slopes (median 8.7%) (Table 1). Whereas many groundwater modeling studies simulate large increases in recharge from irrigation return flow (Luckey and Becker, 1999; Dutton et al., 2001), detailed unsaturated zone profiling in this study indicates that percolation beneath irrigated agroecosystems is restricted to the shallow subsurface in much of the study area and water table depths in other regions are so great that it is unlikely that irrigation return flow actually recharges the aquifer, with the exception of a limited area of sandy soils in the southeast. Because groundwater is the only source of recharge in the CHP, the rate of water table decline should exceed the velocity of irrigation return flow, reducing the potential for recharge from irrigation return flow. Therefore, nearly all irrigation water is derived from groundwater storage, and water levels should continue to decline as long as irrigation is practiced.

Groundwater in this region cannot be managed sustainably because recharge is too low. Therefore, local groundwater conservation districts have allowed average depletion of 50% of the aquifer thickness within 50 yr, beginning in 1998. In the western region of the Texas CHP, a 60% depletion has been proposed. Groundwater levels are monitored each year to track declines. Irrigation pumpage since pre development (approximately 50 yr) has already resulted in ≥50% reduction in aquifer saturated thickness in many regions; therefore, irrigation pumpage will have to be reduced to meet the regulated 50% reduction in remaining aquifer thickness. Groundwater dating in the Kansas part of the CHP indicates that ages range from

<2600 yr near the water table to 12,800 ± 900 yr toward the base of the aquifer (McMahon et al., 2004), essentially spanning the Holocene period and indicating that much of the water that is being pumped is fossil water.

One option to improve water resources in this region is through managed aquifer recharge, as is practiced in California (Izbicki et al., 2008). Previous studies examined use of deep plowing to 0.7 m and artificial recharge ponds installed adjacent to playas to increase recharge (Scanlon et al., 2008a). Deep plowing with a recharge rate of 70 mm/yr could increase volumetric recharge by 0.12 km^3/yr if conducted over a 1700 km^2 area. However, deep plowing would require a lot of energy to conduct and runs counter to the no till practices currently being promoted to reduce erosion. Installing ponds (0.4 ha in area) adjacent to 100 playas with a recharge rate of 0.4 m/day would increase volumetric recharge by 0.016 km^3/yr. These increases in volumetric recharge represent ≤10% of the groundwater storage declines caused by irrigation (∼1 km^3/yr). Because the system is demand driven, reductions in groundwater demand for irrigation may have greater impacts. Colaizzi et al. (2008) suggested improved irrigation scheduling using an ET network, increased irrigation efficiency by switching from gravity systems to center pivot systems, and changing from high water to low water demand crops, such as switching corn to cotton.

4. Conclusions

Irrigation has abstracted large amounts of groundwater in the Texas CHP, resulting in water table declines of ≥15 m over 11,400 km^2. Individual hydrographs show rates of water table declines of ≤1.3 m/yr. The total amount of groundwater depleted between predevelopment (∼1955) and 2007 is ∼57 km^3. The renewability of groundwater resources depends on the recharge rate.

Percolation/recharge rates are controlled by soil textures and the distribution of agroecosystems. In f–m grained soils there is no percolation/recharge beneath natural ecosystems or rain-fed agroecosystems, as evidenced by large Cl bulges and low matric potentials, consistent with previous studies that were limited to <1% of the study area. In m–c grained soils, there are low percolation/recharge rates beneath natural ecosystems (median 4.8 mm/yr) and higher rates beneath rain-fed agroecosystems (median 27 mm/yr), as evidenced by low to moderate Cl and high matric potentials. Percolation beneath irrigated agroecosystems is independent of soil texture (median 37 mm/yr), as shown by low to moderate Cl and high matric potentials. However, these percolation rates are unlikely to recharge the aquifer because of deep water tables in most regions.

Regional recharge in the Texas CHP is estimated to be 11 mm/yr, on the basis of groundwater Cl data, which provide an integrated estimate of recharge derived from focused flow beneath playas and drainages. Irrigation return flow is restricted to the shallow subsurface throughout most of the study area and generally does not recharge the aquifer. Low groundwater NO_3-N concentrations (≤4 mg/L) throughout most of the CHP are consistent with regional recharge data because water from playas and drainages has low NO_3-N levels. Groundwater storage depletion (21 km^3) since predevelopment (1950s) in areas of largest groundwater level declines (>30 m) is 10 times greater than regional recharge rates in these areas (2.1 km^3). Much of the groundwater that is being pumped is fossil water that was recharged during the past 12,000 yr, according to ^{14}C dating conducted in the CHP in Kansas. Because water demand for irrigation greatly exceeds supply from recharge, future water management should focus on reducing water demand for irrigation.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.agee.2010.10.017.

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