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Modeling sediment accumulation in North American playa wetlands in response to climate change, 1940–2100

Lucy Burris · Susan K. Skagen

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Abstract Playa wetlands on the west-central Great Plains of North America are vulnerable to sediment infilling from upland agriculture, putting at risk several important ecosystem services as well as essential habitats and food resources of diverse wetland-dependent biota. Climate predictions for this semi-arid area indicate reduced precipitation which may alter rates of erosion, runoff, and sedimentation of playas. We forecasted erosion rates, sediment depths, and resultant playa wetland depths across the west-central Great Plains and examined the relative roles of land use context and projected changes in precipitation in the sedimentation process. We estimated erosion with the Revised Universal Soil Loss Equation (RUSLE) using historic values and downscaled precipitation predictions from three general circulation models and three emissions scenarios. We calibrated RUSLE results using field sediment measurements. RUSLE is appealing for regional scale modeling because it uses climate forecasts with monthly resolution and other widely available values including soil texture, slope and land use. Sediment accumulation rates will continue near historic levels through 2070 and will be sufficient to cause most playas (if not already filled) to fill with sediment within the next 100 years in the absence of mitigation. Land use surrounding the playa, whether grassland or tilled cropland, is more influential in sediment accumulation than climate-driven precipitation change.

1 Introduction

Shallow depressional wetlands across the North American Great Plains provide important ecosystem services, including ground-water recharge, storm-water retention, carbon storage, and provision of resources and habitats for the maintenance of biodiversity (Gurdak and Roe 2010; Haukos and Smith 1994; Smith et al. 2011). These geographically dispersed and

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hydrologically isolated wetlands provide critical habitats for plants, invertebrates, and vertebrates, including many taxa of amphibians, mammals, and migrant wetland-dependent birds such as sandhill cranes (*Grus canadensis*), waterfowl, and shorebirds (Ghioca-Robrecht et al. 2010; Haukos and Smith 1994; Skagen et al. 1999; Webb et al. 2010). Collectively, prairie wetlands are highly vulnerable to land conversion, agricultural policies and practices, hydrology alteration, and water pollution. Today, ground-water withdrawal, increasing rates of sedimentation, and enhanced likelihood of drought due to climate change pose far-reaching conservation challenges for wetland-dependent biota.

Precipitation-filled playa wetlands, or shallow ephemeral wetlands lined with an impermeable subsurface clay layer, dispersed across the west-central Great Plains are especially affected by climate dynamics. Playas may be inundated annually or only once per decade, depending upon recent rainfall, geographic location, surrounding land use and size (e.g., Johnson et al. 2011b), and these hydrologic cycles have important effects on biotic response (e.g., phenology and composition of plant and animal communities). Wet-dry fluctuations, created by the interplay of inundation, percolation, evaporation, and transpiration, provide ephemeral habitats ranging from shallow water to dry ground with associated variation in flora and fauna (Cariveau et al. 2011; Euliss et al. 2004).

Climate predictions from atmospheric-oceanic general circulation models (AOGCMs) suggest higher summer temperatures across the western Great Plains and reduced summer precipitation in the south (Christensen et al. 2007: Fig. 11.12). Whereas precipitation-based runoff is projected to decrease due to the combination of increased temperature and reduced precipitation (Intergovernmental Panel on Climate Change [IPCC] 2007: Fig. 3.5), recent trends have shown an increase in storm intensity (fewer storms delivering more precipitation [Karl et al. 1995]). The combined effects of temperature increases, precipitation changes, and surrounding landscape may significantly impact runoff-driven sedimentation rates and playa inundation period.

Among many land use impacts, sedimentation is a primary threat to the continued existence of playas at the regional scale (Smith 2003). Although sedimentation is a natural process, sediments accumulate at a faster rate in playas in agricultural settings than in grasslands (Johnson 2011; Luo 1994; Luo et al. 1997; Tsai et al. 2007, 2010) as a result of runoff-borne materials. Of playas that existed in the Southern High Plains (eastern New Mexico, Oklahoma panhandle, and western Texas) before 1970, 60 % may no longer fulfill vital ecological functions (e.g., ground-water recharge, provision of habitat and resources for plant and animal communities) due to anthropogenic sedimentation and modification and 20 % may be completely gone (Johnson 2011). Native biota, and especially migratory birds, positively respond to the presence of individual playas, to their timing and length of inundation (hydroperiod), their area, and the area and density of playas in the surrounding landscape (Cariveau and Pavlacky 2009; Smith 2003).

Sediment delivery represents a critical nexus between land-use patterns and weather (climate) because heavy precipitation on exposed soil causes erosion, overland transport and deposition of sediments. Therefore, the need for understanding sedimentation patterns and processes to develop soil conservation practices that consume little water (allowing it to pass to wetlands) is critical as climate change increases vulnerability of shallow depressional wetlands.

To address this need, we examined the relative roles of projected changes in precipitation and land use context in the playa sedimentation process across the west-central Great Plains. We modeled rates of sedimentation, sediment depths, and resultant playa wetland depths from 1940 through 2100. We estimated erosion using the Revised Universal Soil Loss Equation (RUSLE; Renard et al. 1997), an analysis approach at the regional scale which retained evaluation of individual playa wetlands and their surrounding landscapes, and included upland land use, topography, erosivity (estimated from precipitation), soil texture, and conservation practices. RUSLE was an appealing approach for both landscape scale analysis and future scenario evaluation due to its small number input parameters and is currently used when erosion analyses are conducted with limited historical information or beyond field-scales. We used regional monthly climate predictions (Coulson et al. 2010) that included influences such as latitude, longitude, and elevation. The resulting spatially explicit predictions of playa longevity based on land use patterns and climate will inform conservation planning for these critical wetlands and the diverse biota dependent upon them.

2 Methods

2.1 Study area

The playa region of the west-central Great Plains (Fig. 1) extends from Nebraska through the panhandle of Texas, including parts of Colorado, New Mexico, Kansas, and Oklahoma (ca 780,000 km^2, 30.8–43.6°N latitude, 96.4–105.3°W longitude). Because the area is generally encompassed by the Great Plains Landscape Conversation Cooperative (GPLCC), we used their boundary for our study extent (http://www.fws.gov/gis/data/national/index.html). The vast Ogallala Aquifer underlies much of this area. Vegetation is dominated by warm and cool season short grass prairie and agricultural crops (Smith 2003).

Playa wetlands are distinguished by lack of outlet drainage, circular to ellipsoidal shapes, hydric soils and vegetation, and impermeable subsurface clay horizons (Smith 2003). In general, they range in size from less than 1 ha to several hundred hectares (Smith 2003). We limited our analyses to playas and excluded reservoirs, lakes, rivers, stock ponds, and rainwater basins. Due to modifications by plowing, road bisection, and dredging, our estimate of roughly 80,000 (see below) water bodies on the west-central Great Plains may underestimate the number of playa wetlands present before 1860 and overestimate the

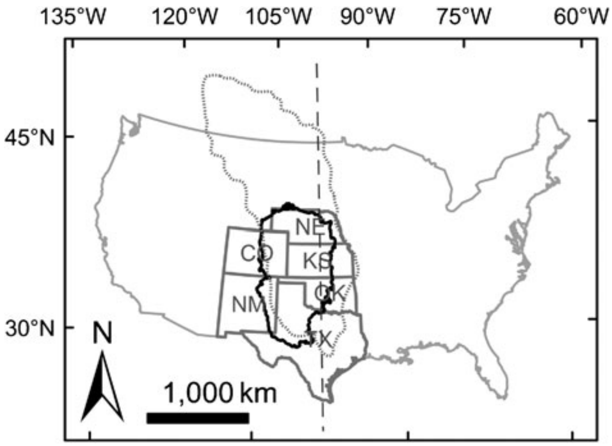

Fig. 1 Boundary of the west-central Great Plains study area, coincident with the Great Plains Landscape Conservation Cooperative (*solid line*), and full extent of Great Plains (*hatched line*). Study area was truncated to the east of 98.4°W (*dashed vertical line*), removing 1 % of all playas. State names: CO, Colorado; KS, Kansas; NE, Nebraska; NM, New Mexico; OK, Oklahoma; SD, South Dakota; TX, Texas; WY, Wyoming

number of functional playas remaining (Johnson 2011). Our study period began in 1940 when irrigated agriculture intensified (Allen et al. 2005; Brown et al. 2005; Luo 1994; Rosenberg et al. 1999).

2.2 Model and data preparation

We used the RUSLE (Renard et al. 1997) to model sediment loss from uplands surrounding playas:

$$T_{up} = RKLSCP \tag{1}$$

where T_{up} was annual soil mass loss per unit area, t ha^{-1} yr^{-1} with t in metric tons. R was soil rainfall erosivity, MJ mm ha^{-1} hr^{-1} yr^{-1}. The time interval of R determined the time interval for the units of T_{up}. K was the soil erodibility factor, t ha hr ha^{-1} MJ^{-1} mm^{-1}. LS was the dimensionless topographic factor. T_{up} then represented the mass of sediment lost from a single upland unit area per unit time. C was the cover management factor, dimensionless, determined from land cover. P was the conservation practice factor, dimensionless, and customarily set to one for large areas unless comparing specific field practices.

The volume of sediment displaced from the upland was equal to the sediment deposited in the playa, $V_{playa}=V_{upland}$, where $V_{upland}=T_{up}/\rho_{up}$ summed over the upland area, where ρ_{up} (g cm^{-3}, bulk density). Sediment depth was the volume of sediment divided by the area of the playa, SDepth=$V_{playa}/A_{playa.}$ In these calculations, we made the following simplifying assumptions: (1) all eroded sediment arrived at the playa, and (2) playas were flat-bottomed and evenly filled. We recognized that sediment texture, slope steepness, aeolian redistribution, presence of prior sediments, and runoff intensity all contribute to the actual location of sediment deposition within the playa. However, we lacked sufficient information for each playa to make more than a simplified assumption of uniform filling. To account for redeposition of sediments along the slope and inherent RUSLE assumptions, and to avoid using an arbitrary sediment delivery ratio, we calibrated our estimated sediment depths to field measurements reported by Luo (1994) (see below).

We used a geographical information system (GIS, ESRI ArcGIS 9.3 1999–2008) to create spatial data sets and performed raster calculations to evaluate SDepth at each playa in the study area (process model available in Online Resource, Fig. 1). Data sets included playas, land use, elevation, soil properties, roads, and precipitation clipped and projected to a common footprint and datum (details available in Online Resource, Details of RUSLE Computations). We developed a playa dataset of ca 78,000 potential playas; discarding any playa that had low circularity, non-eroding uplands, or undefined soil erodibility.

Gridded monthly total precipitation estimates were obtained from Coulson et al. (2010) for historic years, 1940–2000, based on PRISM (Parameter-elevation Regressions on Independent Slopes Model, http://www.prism.oregonstate.edu/) normals from 1961–1990. Coulson et al. (2010) also provided monthly future precipitation for 2010–2100 using the PRISM normals and three AOGCMs: Australian Commonwealth Scientific and Industrial Research Organization CSIRO-MK3.5, Canadian Centre for Climate Modeling and Analysis model CGCM2.1MR, and Japanese Centre for Climate System Research Model for Interdisciplinary Research on Climate MIROC Version 3.2; and three IPCC 4th Assessment Report (IPCC 2007) emission scenarios: A1B, A2, and B1 (broadly, representing continuing current greenhouse gas emission rates, significant emissions increases, and emissions reduction, respectively). All precipitation values were upscaled from a 2.5 arc minute (PRISM data) or downscaled (AOGCM models) to a 5 arc minute grid. Coulson et al. (2010) downscaled the three AOGCM data sets using first a change factor approach (the difference in monthly temperature at each AOGCM location from the 30-year PRISM 1961–1990 mean or the ratio of monthly precipitation to 30-year PRISM mean) and then using ANUSPLIN software (a thin-plate spline approach using multivariate linear regression with a smooth nonparametric function where the resulting climate variable is dependent upon latitude, longitude, and elevation [Hutchinson 1995; McKenney et al. 2005; Price et al. 2004]) to interpolate the change factor values for the prediction period. We summed monthly precipitation values to obtain annual precipitation at each grid point. For visualization purposes we averaged annual precipitation for two historic and three future periods: 1941–1970, 1971–2000, 2011–2040, 2041–2070, and 2071–2100. We selected a 30-year reference interval of 1971–2000 to include the IPCC baseline period of 1980–1999 (Christensen et al. 2007). To assess the range of variability of the predictions, we examined the annual precipitation at each grid point and selected the "driest" and "wettest" model/scenario during each future period. The driest (wettest) model/scenario had the most grid points with the lowest (highest) precipitation.

Using average annual precipitation (P) and our driest and wettest model/scenarios, we calculated annual R-factor values after Renard and Freimund (1994:Eqs. 11, 12).

$$R = 0.04830P^{1.61}, \text{when } P <= 850 mm/yr \text{ or} \tag{2}$$

$$R = 587.8 - 1.219\ P + 0.00415\ P^2, \text{when } P > 850\ mm/yr \tag{3}$$

The R-factor at each grid point for each year was summed for four periods: 1941–1992, 2011–2040, 2041–2070, and 2071–2100. The historic period 1941–1992 allowed comparison to field results reported in Luo (1994). Future periods were selected to provide consistent 30-year intervals. Using ArcGIS, these point values were used to generate an R-factor surface using inverse distance weighting with a power of 2 and 12 nearest points. We extracted the R-factor value at the centroid of each watershed and assumed it to be valid across the watershed (average watershed size~1 km^2).

Meso-scale (~30-m) upland land use was extracted from the National Land Cover 2001 dataset (http://datagateway.nrcs.usda.gov). Conservation Reserve Program (CRP) land use was not explicitly indicated in this dataset, but we confirmed grassland presence for a small sample of known CRP locations. We were unable to distinguish between native and introduced grasslands or to identify buffer strips below the 30-m resolution of the dataset. We assumed land use (grassland vs. cropland) was unchanged from 1940 through 2000 (44.5 % and 44.9 % agricultural lands in non-metropolitan areas in 1949 and 2000, respectively; Brown et al. 2005: Appendix; Parton et al. 2005) continuing to 2100. We obtained large-scale (sub-county) C-factor values from the 1997 National Resources Inventory (Natural Resources Conservation Service 2000). Land cover and C-factors were compared by state (Online Resource Table 1) to link C-factor values to generalized land use categories of grassland (C-factor=0.01), pasture (0.02), cropland (0.22), and other (0). Within the cropland category, C-factor values ranged from hay (0.02) to wheat (0.12–0.21) to irrigated corn (0.35) and cotton (0.35–0.7). Use of a generalized crop C-factor potentially underestimated or overestimated erosion in particular locations but avoided the need to consider within and between year crop shifts.

We averaged sediment estimates across the nine climate model/emission scenarios to obtain a single estimate of sediment depth for each playa. We calibrated our estimated sediment depths to field measurements reported by Luo (1994) who collected sediment samples from 40 random playas situated in grassland (20 playas) or cropland with watershed areas of at least 64 ha, playa areas of 4–10 ha, upland slopes of less than 3 %, playa depths of less than 2 m, circular or ellipsoidal shapes, “fine” or “medium” texture, and located in eleven Texas counties.. Since we did not know specifically which playas were measured, we extracted all playas in our dataset which matched these criteria. We defined “grass” and “crop” uplands as those composed of least 95 % grassland or cropland, respectively. We compared our mean sediment depths to the mean depths reported by Luo (1994) by land use. Using a mixture model, we developed a linear relationship setting Luo’s reported average sediment depth equal to our RUSLE predicted sediment depth (SDepth) multiplied by a local sediment delivery index based on the relative contributions of watershed grass and crop use.

$$\begin{aligned} &LuoAvgSedimentDepth \approx CalSDepth \\ &= SDepth\left(a_1 \frac{\%grass}{\%grass + \%crop} + a_2\left(1 - \frac{\%grass}{\%grass + \%crop}\right)\right) \end{aligned} \tag{4}$$

where a_1 and a_2 were determined by the model fit of our data to Luo’s data. We calibrated our results using this relationship.

We used the average annual deposition rate from 1941–1992 to estimate sedimentation through 2010. Using projected R-factors, we estimated future accumulation for the periods 2011–2040, 2041–2070, and 2071–2100 averaged across the nine climate model/scenario combinations. Within each period, we averaged the annual accumulation rates to obtain a single average rate for the 30-year period. Playa fill times were determined by the number of years needed to fill the remaining playa depth (minimum playa perimeter elevation – minimum playa interior elevation). We assumed surface elevations corresponded to the vintage of the underlying DEM (which ranged from 1920–1959, which we averaged to 1940, 1960–1979 averaged to 1970, and 1980–1999 averaged to 1990 [USGS National Elevation Dataset Data Source Index, http://ned.usgs.gov/usgs_gn_ned_dsi/viewer.htm]). Most DEM values were from 1970 DEMs, so generally our estimated playa depth loss due to sediment accumulation started in 1970. We counted playas filled by 1992, during 1993–2010, during 2011–2040, during 2041–2070, and during 2071–2100, and those that remained unfilled.

2.3 Analysis

To examine area-wide precipitation patterns, we extracted average annual precipitation at 70 random locations across the study area for each of our comparison periods 1941–1970, 2011–2040, 2041–2070, 2071–2100 and calculated the difference of the average annual precipitation at those points from the reference period of 1971–2000. To minimize possible spatial correlation at small scales, our points were separated by a minimum of 70 km (Augustine 2010). We used an analysis of variance (ANOVA, SAS Institute SAS 9.3 2002–2010) to test for the average difference in precipitation for the four periods and constructed 95 % confidence intervals on the mean difference by period to test for non-zero mean differences.

We compared calibrated RUSLE predictions area-wide to field data by Luo (1994). Because comparable field measurements have not been found for other areas, we used quantile regression (Koenker 2010; R Development Core team 2010) to compare our RUSLE predicted sediment accumulation at playas matching criteria from Luo (1994) for the period from 1941–1992. Because of the small numbers of matching playas found in Kansas, Nebraska, New Mexico, and Oklahoma, we assigned playas to three groups: playas in Luo’s study area, southern playas (all other Texas playas, New Mexico, and Oklahoma), and northern playas (Colorado, Kansas, Nebraska). We computed 95 % confidence intervals for the mean accumulated sediment depth by surrounding land use (crop or grass) at 0.1, 0.25, 0.5, 0.75, and 0.9 quantile intervals (which contain cumulatively 10 %, 25 %, etc., of the response variable values). We used ANOVA (R Development Core Team 2010) on land use and period factors for mean sediment accumulation at playas which matched Luo's criteria that were unfilled by 2070 for two 50-year periods: 1940–1992 (historic precipitation) and 2010–2062 (average, driest scenarios, and wettest scenarios).

3 Results

Nearly 90 % of playas of the west-central Great Plains were vulnerable to sediment infill by 2100 (Fig. 2, Table 1). Based on average future precipitation, land use had a stronger influence on sedimentation rates than precipitation; sediment accumulation decreased by ca 4 cm across 50 years (t=22.8, df=3968, P<0.0001; Table 2) compared to an increase in sedimentation due to land use change from grassland to cropland of ca 24 cm across 50 years (t= −3.82, df=3968, P<0.0001). Sediment accumulation in cropland settings also was significantly higher than in grasslands under both the driest and wettest scenarios (Table 2). Under the driest scenarios with reduced sediment accumulation, 563 of 3968 (14 %) playas filled one 30-yr period later than predicted under average predicted precipitation; under the wettest scenarios 384 (10 %) playas filled one period sooner.

During the reference period 1971–2000, precipitation within the study region ranged from ca 285 to 720 mm/yr, increasing from west to east (Fig. 3, left). The 30-year average annual precipitation during this reference period was greater than precipitation during the historic period of 1941–1970 and greater than projected average precipitation in future periods (2011–2040, 2041–2070, 2071–2100) (Fig. 3, right; ANOVA $F_{3,276}$=5.64, P=0.0009; mean difference (95 % confidence interval of the difference)= −35.1 (−41.9, −28.2), −31.4 (−38.6, −24.1), −13.0 (−23.1, −2.9), −25.5 (−33.4, −17.7), respectively). Our driest and wettest model/scenario combinations varied by period: during 2010–2040 and 2041–2070 MIROC A1B was the driest at 5762 of 9172 grid points and 5403 points, respectively; during 2071–2100 MIROC A2 was the driest at all 9172 points. The wettest combinations during these periods were CSIRO A2 (at 4136 points), CSIRO B1 (9172) and CSIRO A1B (8826), respectively. The coefficient of variation of the precipitation at each grid point across all nine model/scenario combinations and periods (n=27517) averaged 12.8 %, ranging from 4.6 % to 22.5 %. The coefficient of variation for the R-factors based on these precipitation values averaged 21.5 % (7.7 %–37.8 %).

The final proposed playa layer for our entire study area contained ca 78,000 playas. Playa attributes varied from north to south and east to west across the region, with more numerous playas in Nebraska, Texas, and Kansas; larger playas and watersheds in the south; and more grassland matrix in the west (Table 1, Fig. 4). In comparison with Ekanayake et al. (2009), our average watershed area of 107.6 ha compared favorably with their estimates of 110.32 for survey (considered to be "truth") and 118.76 for minimum distance methods. The vast majority (93 %) of our playas were less than 0.3 m deep. Due to shallow depth and upland cultivation, playas in the northern portion of our study area were at risk of earlier sediment infill (loss of depth between perimeter low point and playa bottom) than those in the south.

We calculated calibration factors to relate erosion to sedimentation rates based on work by Luo (1994) who identified 11,939 playas in his study area; in the same area we found 12,232 playas. Of these, 1975 fit Luo's criteria exclusive of land use, and 230 of these also matched his more stringent land use criteria (176 exclusively in ≥95 % cropland and 54 in

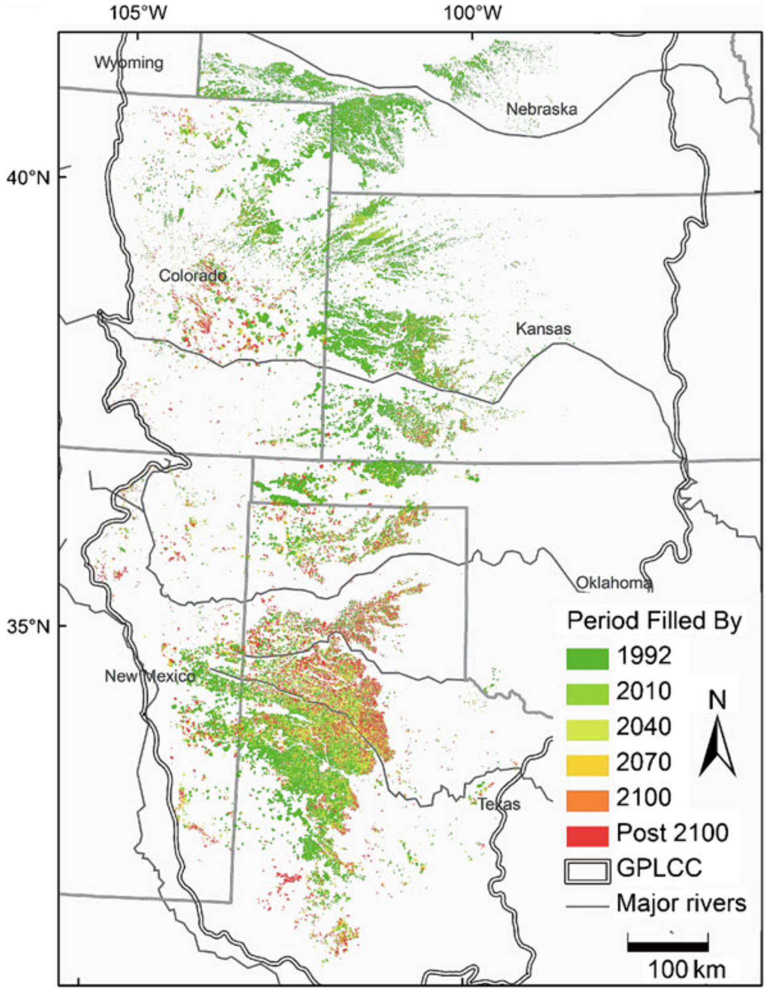

Fig. 2 Period (ending 1992, 2010, 2040, 2070, or 2100) by which each playa within the study area will be filled with sediment based on calibrated RUSLE (Revised Universal Soil Loss Equation) estimation and historic and downscaled annual precipitation predictions though 2100. To improve visibility, playa watersheds are shown; each watershed contains a single playa

≥95 % grassland). In this 11-county area, our estimated mean sediment depth at all playas matching Luo's criteria was 3.14 cm/50 years (2.56, 3.72; 54 playas) in grasslands and 64.55 cm/50 years (59.19, 69.91; 176 playas) in croplands compared to Luo's sample means of 4.62 (2.83, 6.42; 20 playas in grassland) and 42.12 (31.4, 52.8; 20 playas in cropland), respectively. Our resulting calibration factors were a_1=1.473 and a_2=0.653 (Eq. 4, above). Our average calibrated sediment depths were 4.61 (3.76, 5.46) in grassland and 42.36 (38.85, 45.87) in cropland. Proportionally fewer playas matched Luo's criteria in the remainder of our study area due to mixed land use, playa size, and soil texture differences. The southern region (the remaining Texas counties, New Mexico, and Oklahoma) contained ca 14,000 playas of which 115 met Luo's criteria, and the northern part of our study area (Colorado, Nebraska, and Kansas) contained ca 52,000 playas of which 206 met Luo's criteria.

Table 1 Playa counts (Total) by state and estimates of number (#) and percentage of playas predicted to fill during five time periods based on RUSLE estimated sediment accumulation pooling over three future climate scenarios (A1B, A1, B2) and three climate models

		Year filled by										Total filled by 2100		Not filled by 2100	
		1992		2010		2040		2070		2100					
State	Total[a,b]	#	%[c]	#	%	#	%	#	%	#	%	#	%	#	%
CO	8060	3544	44 %	1114	14 %	734	9 %	391	5 %	286	4 %	6069	75 %	1991	25 %
KS	21700	12904	59 %	4245	20 %	2136	10 %	973	4 %	443	2 %	20701	95 %	999	5 %
NE	22158	19046	86 %	1659	7 %	728	3 %	269	1 %	126	1 %	21828	99 %	330	1 %
NM	2265	722	32 %	358	16 %	277	12 %	197	9 %	105	5 %	1659	73 %	606	27 %
OK	2277	1441	63 %	297	13 %	179	8 %	105	5 %	49	2 %	2071	91 %	206	9 %
TX	21711	5383	25 %	4313	20 %	3466	16 %	1929	9 %	1142	5 %	16233	75 %	5478	25 %
Total	78171	43040	55 %	11986	15 %	7520	10 %	3864	5 %	2151	3 %	68561	88 %	9610	12 %

[a] Assumed minimum playa depth of 0.3 m

[b] Total playas proposed to exist using 2011 data. Many playas may already have filled and no longer be recognized on the landscape

[c] Percentage of total playas within state

Table 2 Least squares means of sediment accumulation depth (95 % CI, cm) by land use and climate for a subset of playas (those predicted to fill after 2070, n=3,968) during historic (1940–1992) and future (2010–2062) time periods with historic and predicted precipitation conditions (average, driest, and wettest). Playas were situated in either grassland or cropland settings

Time period	Surrounding upland		Difference between land uses
	Grassland (n=1,148 playas)	Cropland (n=2,820 playas)	
1940–1992	27.1 (24.6, 29.6)	51.8 (50.2, 53.4)	24.7 (21.7, 27.6)***
Average precipitation, 2010–2062	24.6 (22.1, 27.1)	47.7 (46.1, 49.3)	23.1 (20.2, 26.1)***
Difference from 1940–1992	−2.5(−6.1, 1.0)^{ns}	−4.1 (−6.3, −1.8)**	
Dry scenarios, 2010–2062	18.9 (16.4, 21.4)	35.6 (34.0, 37.2)	16.7 (13.7, 19.6)***
Difference from 1940–1992	−8.2 (−11.7, −4.7)***	−16.2 (−18.4, −13.9)	
Wet scenarios, 2010–2062	32.9 (30.4, 35.4)	63.3 (61.7, 64.9)	30.4 (24.5, 33.4)***
Difference from 1940–1992	5.77 (2.2, 9.3)**	11.5 (9.3, 13.8)***	

** P<0.001, *** P<0.0001, ns P>0.05

All |t|>3.2, df≥2296

In general, we obtained good agreement between calibrated sediment depth at these playas and Luo’s sample of playas based on overlapping 95 % confidence intervals at all quantile values (Fig. 5) in both grassland and cropland settings, indicating RUSLE results were congruent regardless of geographic location. In essence, at geographic locations outside of the calibration counties, we found a similar distribution of sediment depths as Luo as assessed by quantiles (e.g., in cropland settings 25 % of the playas had <40 cm of sediment regardless of location, 50 %<60 cm, and so forth).

4 Discussion

Land use surrounding playas is more influential than climate-driven precipitation change in determining rates of sediment accumulation within the playas. Within a 30-year time period, playas embedded in cropland matrices may accumulate nearly twice the depth of sediments as playas surrounded by grassland. In contrast, our results suggest that contemporary and future predicted rates of sediment accumulation within a given land use context do not differ. Smith et al. (2011: Fig. 3) also observed a stronger influence of land use than climate on playa hydroperiod based on temperature simulations and proposed sediment removal as a practice that potentially can restore playa hydrology.

Accelerated sediment accumulation in cropland playas is detrimental to native biota. As little as 0.5 cm of sediment is sufficient to suppress seedling and invertebrate emergence in the prairie pothole wetlands of North America (Gleason et al. 2003), and if this pattern holds in playa systems, rapid accretion of sediment will depress aquatic invertebrates, the key protein for resident and migrating wetland-dependent birds. Wetland-dependent plant richness is reduced by loss of wetland volume and presence of upland agriculture (Tsai et al. 2012). Further, sediments can also mask underlying hydric soils, compromising wetland delineation efforts and thus environmental protection, leaving more playas vulnerable to development (Johnson et al. 2011a).

The spatial patterns of predicted playa longevity presented here may help identify landscapes for playa conservation or more intensive research.. Many playas within the west-

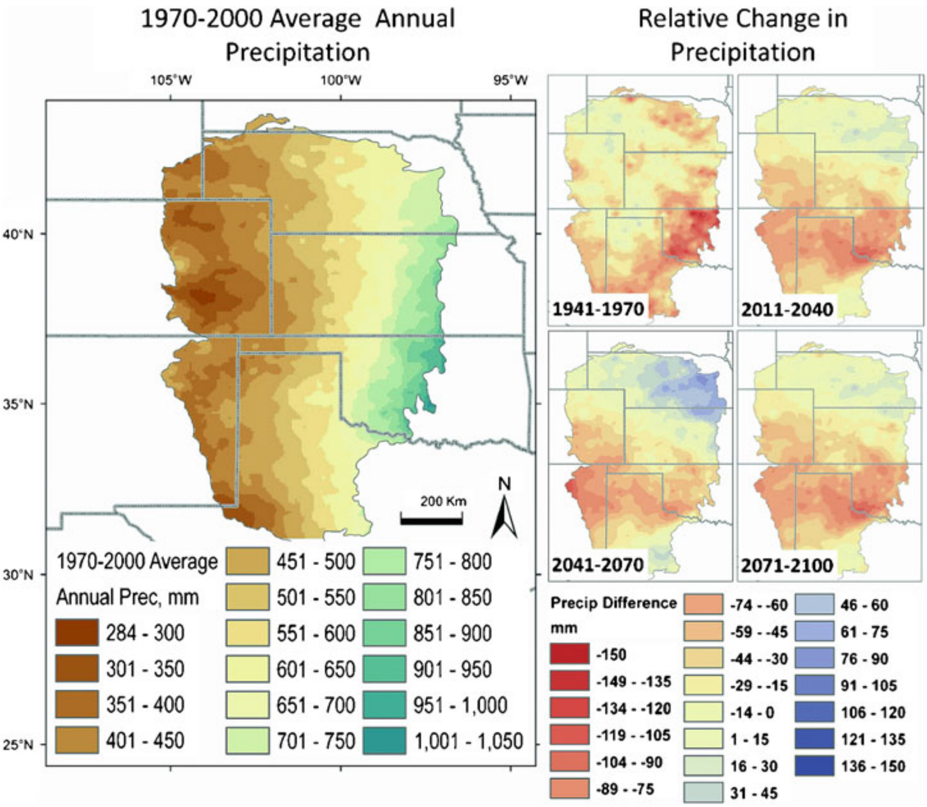

Fig. 3 *Left*: Average annual precipitation, mm, across the study area for the reference period of 1971–2000. *Right*: Difference in average annual precipitation during four 30-year periods and the reference period (1971–2000). *Red areas* have less precipitation in an average year than the reference period; *blue areas* have more. *Light yellow areas* reflect little change. Data from 1940–2000 and 1971–2000 are based on PRISM (Parameter-elevation Regression on Independent Slope Model, from Coulson et al. 2010). Prediction periods (2011–2040, 2041–2070, 2071–2100) used the Canadian Centre for Climate Modeling and Analysis model CGCM2.1MR, scenario A1B (continuing current greenhouse gas emission rates). At this scale, the other eight model/emission scenario combinations showed only minor differences from these maps

central Great Plains have the potential to fill with sediment within 100 years, although the longevity of playas varies across our study area. Playa lifespan is shorter in the northern half of our study area where watersheds are smaller and contain greater relative coverage of cropland. On the other hand, the southern portion of our study area is predicted to experience reduced precipitation which may reduce the likelihood of playa inundation. Given the importance of functional playas, especially in the role of aquifer recharge (Smith 2003), playas with longer expected lifespan may merit additional evaluation for protection. Our models suggest locations for application of more refined tools such as dynamically downscaled climate models that can address storm intensity; LIDAR-based [light detection and ranging] elevation data; local land use identification; and recently developed erosion models for field scale assessments, e.g., Water Erosion Prediction Project (WEPP) and Modified Universal Soil Loss Equation (MUSLE; Soil and Water Conservation Society 2003).

Although climate change likely will not accelerate sediment accumulation rates, it will exacerbate other stressors on wetland condition and quality of wildlife habitat. Higher

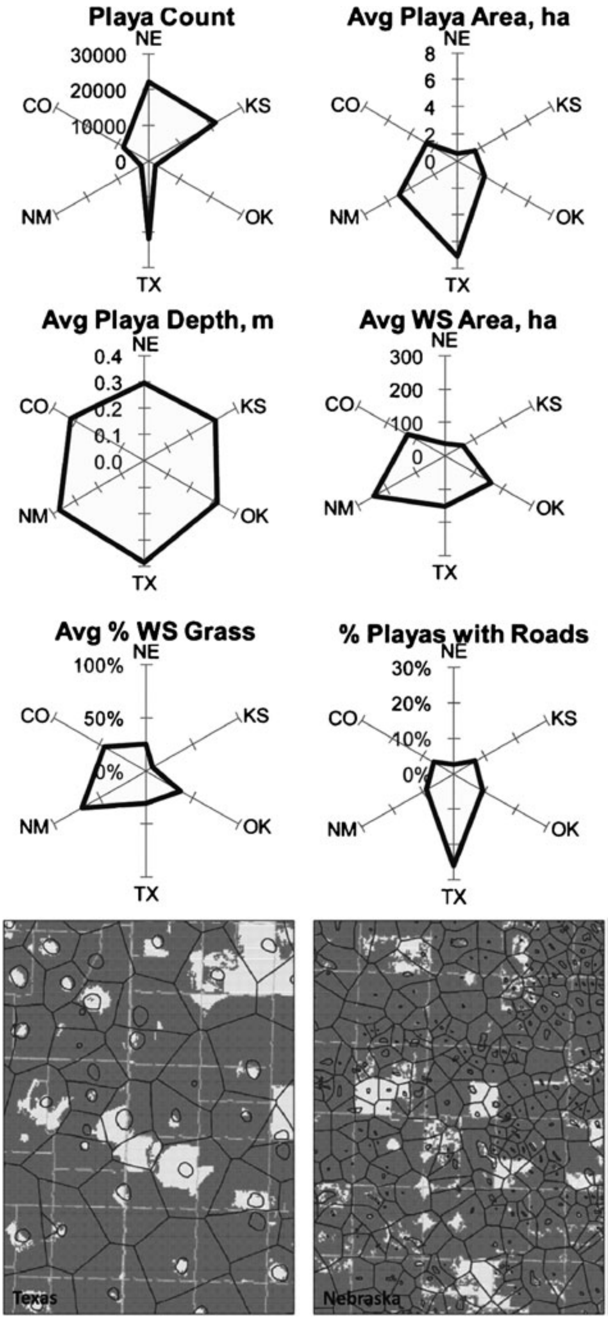

Fig. 4 *Top*: Variation in playa wetland and watershed characteristics by state. WS: watershed; Avg% WS Grass: average percentage of watershed in native grassland. State names: CO, Colorado; KS, Kansas; NE, Nebraska; NM, New Mexico; OK, Oklahoma; TX, Texas. *Bottom*: Variation in watershed extent in Texas and Nebraska. *Circular lines* delineate playa boundaries, *angular lines* watersheds, *light background* is grassland/grass, *dark background* is cropland, *light dashed lines* are roads (~1.6 km intervals)

temperatures in concert with reduced precipitation will shorten hydroperiods due to increased evaporation and transpiration rates and less moisture (Christensen et al. 2007). Reduced playa depths due to sedimentation will compound this problem as available water

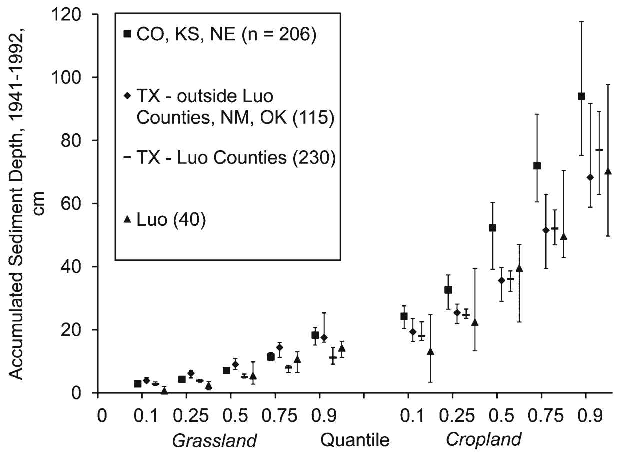

Fig. 5 Expected quantile values and 95 % confidence intervals by geographic area (Luo [1994] source counties; all other counties in Texas, all counties in Oklahoma and New Mexico; and all counties in Colorado, Kansas, and Nebraska) and land use (all grassland or all cropland) for playas meeting Luo (1994) criteria (see Methods)

is spread over a larger surface area (Tsai et al. 2007). Attempts to restore playa functionality through erosion control practices, such as vegetative buffers, may not have the desired effect and may be further confounded by climate change. For example, although Conservation Reserve Program (CRP) plantings (generally nonnative) around wetlands do reduce sedimentation rates and overland contaminants reaching the wetlands (Brinson and Eckles 2011; Johnson 2011; Skagen et al. 2008), playas surrounded by CRP are less likely to be inundated than those in either crop or grassland (Cariveau et al. 2011; O'Connell et al. 2012). This effect is likely to be exacerbated by increasing rates of evapotranspiration.

Although there have been no major changes in the overall amount of agricultural land in this region since the 1940s, future shifts in land use and agricultural practices may be forced by the current unsustainable nature of water use practices in concert with climate change (Brown et al. 2005; Parton et al. 2007). The future of irrigated agriculture in the Great Plains region is intricately linked with the status of the Ogallala aquifer. Since the beginning of major irrigation development in the 1940s, irrigation water has been drawn from the aquifer at unsustainable rates, rates that far exceed recharge, and climatic changes toward warmer and drier conditions are predicted to exacerbate this shortfall (Allen et al. 2005; Rosenberg et al. 1999). In the future, agricultural practices may shift to more sustainable approaches, including added emphasis on dryland rather than irrigated crops or the integration of crop and livestock systems (Allen et al. 2005; Parton et al. 2007). These future agricultural practices potentially may reduce soil erosion and yield lower rates of sediment accumulation in wetlands.

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Modeling sediment accumulation in North American playa wetlands in response to climate change, 1940-2100

Climatic Change

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Details of RUSLE Computations

A. Playa Identification

We combined several existing spatial layers of potential playa wetland locations and geometries (Kansas Data Access and Support Center [http://www.kansas.org]; M. McLachlan, D. Pavlacky unpublished data) to create a single dataset containing roughly 84,000 playa wetlands on the west-central Great Plains. We assumed the playa data to be the current best estimate of playa location and extent. Due to file size processing limitations and because few playas occurred east of 98.4°W, we truncated the playa dataset here. Not including this playa-sparse portion of the study area in our analyses resulted in a loss of about 0.33% (788 ha) of total playa wetland area and ca 1% (1049) of all playas.

We applied a circularity threshold of ≥ 0.75 to identify circular and ellipsoidal playas. We estimated playa circularity (approximation to a circle) by

$$c = \frac{\sqrt{A_{playa} / \pi}}{P_{playa} / 2\pi} \qquad \text{(ESM-1)}$$

where A_{playa} was playa area and P_{playa} was playa perimeter. We removed non-circular playas and interior stock ponds and pits. To define drainage loci or "watersheds", we combined the playa layer with other surface water features (flow paths, rivers, canals, reservoirs, and lakes) obtained from the high-resolution National Hydrography dataset (http://nhd.usgs.gov/index.html). In the absence of high resolution DEMs, we created watersheds using nearest neighbor to assign landscape pixels to the nearest water and retained only those watersheds containing playas. We compared our watershed areas to those determined by Ekanayake et al. (2009) for the minimum distance method.

B. Soil K-Factor

Soil K-factor (erodibility, multiplied by 0.1317 for SI units [Foster et al. 1981]), texture (% sand, % silt, and % clay), and bulk density were obtained from SSURGO2.0 (USDA Natural Resources Conversation Service Soil Survey Spatial and Tabular Data, http://datagateway.nrcs.usda.gov). Data were limited to the uppermost-recorded major soil horizon in each soil mapping unit.

C. Slope and Playa Depth

We extracted playa elevations and upland slopes using a 10-m digital elevation model (DEM, National Elevation Data set, http://datagateway.nrcs.usda.gov), the highest resolution available with coverage of the complete study area. For this DEM, vertical resolutions of 0.01 m, 0.1 m, and 1 m were reported for 50, 25, and 25% of the area, respectively (http://ned.usgs.gov/downloads.asp). We used XTools Pro (version 6.2, Data East 2003-2010) to create points at 10-m intervals around each playa perimeter and determined the elevation at each point. The difference between the lowest point on the playa perimeter and the lowest point within the playa established the playa depth. Although we were able to discern playa watersheds using the 10-DEM, vertical resolution was not sufficient to consistently differentiate within-playa elevations. Based on average playa depth from field measurements (Luo 1994; Tsai et al 2010), we assigned a depth of 0.3 m to playas with calculated depths of < 0.3 m. Because playas are extremely shallow, we likely overestimated the depth and increased the potential sediment capacity of some playas. We computed the topographic factor (LS) after Renard et al. (1997:Eqns. 4-1, 4-4, adjusted for SI units per Foster et al. [1981])

$$LS = \left(\frac{l}{22.13}\right)^{\mu}\left(10.8\sin slope + 0.03\right) \qquad \text{(ESM-2)}$$

where 22.13 was the length of a RUSLE reference plot (m), l was the length (m) of the field under test, and slope was the slope angle (degrees) of the field under test. The length exponent, μ, (McCool et al. 1989:Table 2) and slope term (McCool et al. 1987:Eqn. 10) were both based on shallow slopes and low to moderate rill/interrill ratios. Using a GIS approach, the field length under test was equal to the cell length (grid dimension, m).

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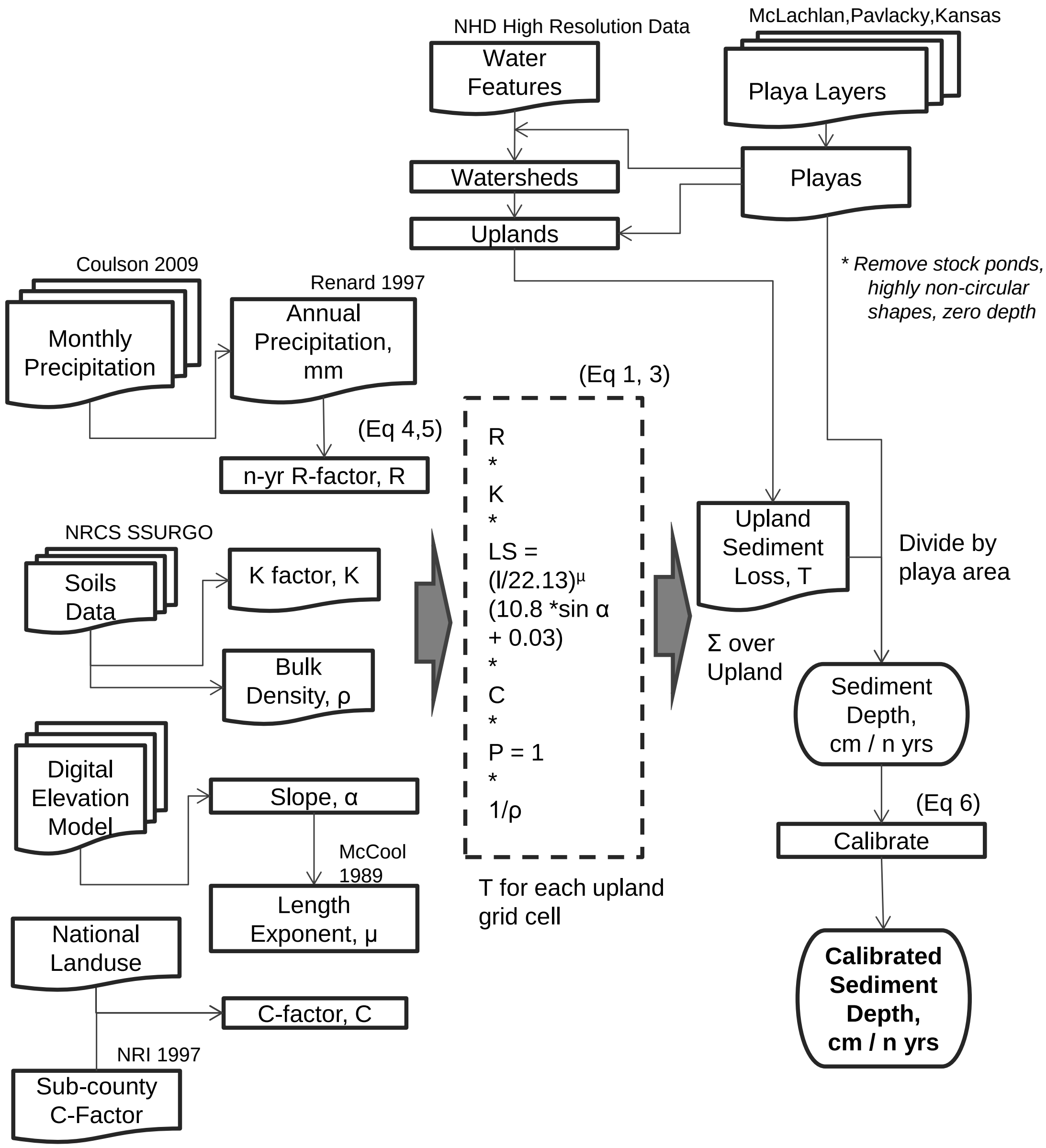
Captions

Fig 1 Data processing flow chart for implementation of RUSLE (revised Universal Soil Loss Equation) using GIS (geographical information system)

Table 1 Correspondence of National Resources Inventory (NRI) Land Use categories, crop types, and C-factor values by state to National Land Cover (NLC) categories and determination of average C-factor by generalized land use (cropland, pasture, or grassland) for the west-central Great Plains



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NRI Land Use										
		C-Factor by State						NLC Category	Average	% Land
Code	Use[a]	CO	KS	NE	OK	NM	TX		C-factor	Area
11	Corn		0.16-0.24	0.15-0.16			0.35	82	0.22	26
12	Sorghum		0.18-0.28				0.35	82		
13	Soybeans			0.13-0.18				82		
14	Cotton						0.35-0.7	82		
111	Wheat	0.32	0.23	0.1-0.16	0.12-0.21	0.13-0.18	0.15-0.25	82		
142	Hay/legume	0.02	0.02	0.02		0.02		81	0.02	9
170	Other/summer fallow	0.17-0.37	0.17-0.21	0.11-0.13				81		
211	Pasture grass			0.01	0.01-0.04		0.01	81		
410	Other farm land / CRP	0-0.01	0.01	0-0.01	0-0.04	0.03	0.01	81		
250	Grassland	0	0	0	0	0	0	71	0.01	55
	Other							Other	0	8

[a] C-factors for major land uses only. Average C-factor based on weighted average of land use by acreage in the category. Although grassland had an average C-factor of 0, it was assigned a small value (0.01) to distinguish it from non-eroding surfaces that also have a C-factor of 0 and to reflect the field evidence of some erosion.