

Soil organic carbon in playas and adjacent prairies, cropland, and Conservation Reserve Program land of the High Plains, USA



Jessica L. O'Connell^a, Dale W. Daniel^{b,*}, Scott T. McMurry^b, Loren M. Smith^b

^a Department of Marine Sciences, University of Georgia, Athens, GA 30602-3636, USA

^b Department of Integrative Biology, Oklahoma State University, Stillwater, OK 74078, USA

ARTICLE INFO

Article history:

Received 14 January 2015

Received in revised form 8 September 2015

Accepted 18 September 2015

Available online 13 October 2015

Keywords:

Agriculture

Conservation program

Depressional wetlands

Global climate change

Grasslands

Land use

Playa watersheds

Soil carbon sequestration

ABSTRACT

Soil organic carbon (SOC) is an important reservoir for atmospheric CO₂ associated with climate warming. The High Plains, USA, lacks region-wide SOC estimates within playa wetlands and their adjacent watershed. Croplands often have less SOC than grasslands, and the Conservation Reserve Program (CRP; former croplands planted to grass) may return SOC. Our goals were to estimate SOC within playa wetlands and investigate small scale differences within wetland catchments across a broad agriculturally modified landscape. We estimated SOC (kg m⁻²) to 50 cm depth from 4 soil cores/catchment (in playas and 10, 40 and 100 m into uplands) at 56, 52, and 54 sites in native grassland, CRP and cropland, respectively. At a subset of sample locations within each land use type, we estimated SOC to 1 m depth to characterize SOC missed by shallow sampling. In playa wetlands, CRP SOC from 0 to 50 cm was 18% greater than croplands, but native grassland playa SOC did not differ from other land-uses. From 0 to 1 m, SOC in native grassland wetlands and uplands was 20% greater than the same habitats within croplands, while CRP lands were intermediate. Native grassland playa SOC also was 16% greater than in surrounding native short grass prairie. Playas therefore represent an important SOC repository in the High Plains ecoregion. CRP playas and uplands may require an additional 10–30 years to resemble native grassland SOC. SOC increased with playa area throughout CRP and native grassland catchments, suggesting playa hydrogeomorphology influences adjacent upland SOC. High Plains playas store 20.8 Tg C and cropland conversion caused a cumulative loss of 2.0 Tg C from 82,000 ha of playas. Currently, CRP enrollment on over 25,000 ha of playas has returned 0.2 Tg C (95% CI: 0.1–0.3), only half the historic SOC lost by cropland conversion within CRP playas. To promote SOC storage, native grasslands and large playas should be preserved and CRP enrollments should be maintained over long timescales.

© 2015 Elsevier B.V. All rights reserved.

1. Introduction

Terrestrial soils are significant reservoirs for carbon, and play an important role in the climate system (Post et al., 1982; Swift, 2001). Conversion of natural ecosystems to other land-uses generally results in losses of soil organic carbon (SOC) from soils by reducing organic residue inputs, increasing SOC loss by soil erosion, and exposing stored SOC to microbial activity (Daniel et al., 2015; Lal, 2003; Schlesinger and Andrews, 2000; Syswerda et al., 2011). Land management practices and land restoration may restock some of the SOC pool lost following land-use change (Guo and Gifford, 2002; Post and Kwon, 2000). However, the efficacy of many management and restoration strategies for increasing SOC remains

uncertain (Powlson et al., 2011), due in part to spatial variability in factors regulating C storage.

The High Plains region of the United States was historically upland prairie habitat with depressional recharge wetlands (playas) scattered throughout. Though only occupying 2% of the High Plains landscape, playas may play an important role in C sequestration for the region (Smith et al., 2011). As a result of varying amounts of seasonal precipitation, playas experience plant community shifts during wet and dry phases (Smith and Haukos, 2002). Environmental differences among wetlands and uplands may result in different local SOC storage estimates (Raich and Schlesinger, 1992; Trumbore, 1997). Climate also differs across the High Plains, such that the southern portions are drier and hotter than the northern portions (High Plains Regional Climate Center, 2011). Until now, catchment SOC within High Plains playas and adjacent uplands has not been investigated across broad landscapes and climate gradients.

* Corresponding author. Fax: +1 405 744 7824.

E-mail address: dale.daniel@okstate.edu (D.W. Daniel).

Wetland loss due to land conversion has resulted in net fluxes of C into the atmosphere (Armentano and Menges, 1986; Maltby and Immirzi, 1993) and in the High Plains, cropland is the largest land-use alteration affecting playas (Luo et al., 1999; O'Connell et al., 2012). Cultivation disrupts soil structure, increases bulk density and decomposition of SOC (Burke et al., 1995; Post and Kwon, 2000). Cropland playas also are often cultivated when dry, concomitantly with surrounding uplands. Further, cropland playas are subject to continuous disturbance from deposition of eroded upland sediments (Luo et al., 1997). Excessive upland sediment accumulation within playas reduces playa volume and hydroperiod (Tsai et al., 2007), potentially altering SOC formation processes. Upland sediment accumulation in wetlands also buries wetland hydric soils, possibly altering SOC estimates (Maynard et al., 2011).

Cropland soils removed from cultivation and converted to grassland can restore SOC to historic levels given adequate time (Leifeld et al., 2011; Post and Kwon, 2000). In the United States, the largest coordinated conversion of cropland to perennial grassland has been through the United States Department of Agriculture (USDA) Conservation Reserve Program (CRP), initiated in 1985 (Young and Osborn, 1990). Gebhart et al. (1994) showed that retired cropland soils placed into the CRP gained an average of 1.1 tons C ha⁻¹ year⁻¹ with the potential to sequester 45% of the C released annually from U.S. agriculture. In the High Plains, CRP covers approximately 15% of the landscape (O'Connell et al., 2012) and is established on more than 2.8 million ha of playa watersheds, making it the dominant conservation program influencing playas (Smith et al., 2011).

Though CRP establishment generally increases SOC, its influence on C storage potential of associated playas is unknown. Grasses used in most CRP plantings in the High Plains were introduced tall-grasses planted in both the uplands and wetland basins (O'Connell et al., 2012). These grasses provide barriers to overland sheet flow in catchments (the largest input to playa hydroperiod), reducing soil erosion, but also reducing inundation frequency in wetlands (Cariveau et al., 2011; Detenbeck et al., 2002; O'Connell et al., 2012; Van der Kamp et al., 2003). Less inundation will alter soil moisture and influence SOC in playas and their upland catchments.

Our goals were to estimate SOC within playa wetlands and investigate small scale differences within wetland catchments across a broad agriculturally modified landscape. Specifically, we quantify SOC in playas and surrounding uplands among dominant land-uses (native grassland, cropland, and CRP lands) to identify land management influences on SOC across an entire region (the High Plains, USA). Spatial variability in factors regulating C storage can increase the uncertainty of the impacts many land management strategies may have on SOC (Powlson et al., 2011) thus, land use decisions that have the potential to increase SOC at regional scales should either account for these variables or be robust across them.

2. Materials and methods

2.1. Study area

We sampled playas and their immediate watersheds from the High Plains region, covering six states from western Nebraska and eastern Colorado, south to eastern New Mexico and western Texas (Fig. 1). Playas here are in high density, with up to 75,000 individual wetlands occurring in the region (Playa Lakes Joint Venture (PLJV), www.pljv.org). The High Plains is largely a flat expanse of short-to mixed grass prairie, little topographical variation and a semiarid climate (Smith, 2003). Total average annual precipitation ranged from 21–63 cm along a west–east gradient over 1971–2000 (www.ncdc.noaa.gov). Average annual temperature ranged from 7.2 to

18.3 °C along a north–south gradient over 1971–2000 (www.ncdc.noaa.gov) (Fig. 1). Agricultural cultivation has increased since the 1940s and wheat, cotton, corn, and sorghum are common crops (Bolen et al., 1989; USDA, 2009). Approximately 30% of cultivated lands in the High Plains are irrigated (Scanlon et al., 2012). Remaining native prairie is largely grazed by domestic livestock (Samson et al., 2004).

Geologically, the High Plains is a depositional environment, and much of the land surface consists of eolian and alluvial material deposited over older soils (Kelly et al., 2008). Depositional and erosional processes were spatially variable and occurred numerous times during the Holocene, resulting in paleosols more than 2 m deep in some areas (Kelly et al., 2008). Soil age ranges from older than 600,000 years to that of recent age (Kelly et al., 2008). Soils throughout the Great Plains are most often Mollisols, Alfisols, Aridisols and Entisols (USDA–NRCS, 2006). Playas have hydric (reduced soils with gleyed soil matrices), clay Vertisol soils, of Randall, Lipan, Ness, Lofton, Stegall, Pleasant, and Scott clays (Smith, 2003; Soil Survey Staff, 2011). Playas average 7 ha (Daniel et al., 2014) and are generally circular in shape, especially in the south (Bolen et al., 1989).

2.2. Data collection and experimental design

We sampled 162 wetlands and adjacent watersheds (56 in native grassland, 52 in CRP, and 54 in cropland) to 50 cm depth, in a random design stratified by playa density/region and county. Of these, we sampled to 100 cm depth in 17, 15, and 17 sites each in native grassland, CRP, and cropland, respectively, to account for patterns of SOC storage in deeper soils (Fig. 1). To select sample sites, we initially chose locations in native grasslands because these are most limited, and then paired these with nearby sites in the other land-uses.

We sampled SOC during the growing season in 2009. We first surveyed playa area with a GPS (Trimble GeoXT) by walking playa visual edges (Luo et al., 1997). We delineated playa visual edges by noting changes in vegetation from hydrophytic to upland plants and changes in topography from sloped basin edge to flat upland (Luo et al., 1997). We refined playa boundary estimates by coring to locate heavy clay, gleyed hydric soil edges (Luo et al., 1997). We cored along transects perpendicular to the visual edge boundary and used two such transects on opposite sides of playa basins (Tsai et al., 2007). The hydric soil edge was where soil color and texture changed from wetland soils of heavy clay Vertisols with matrix chroma <3 to coarser, browner upland soils (Luo et al., 1997; Tsai et al., 2007). We used Muncell soil color charts to confirm hydric and upland soil classifications (Schoeneberger et al., 2002). See Luo et al. (1997) for chromas distinguishing Randall clay playa wetland soil series from upland soils. A similar process was followed for other wetland soil series.

To estimate patterns in SOC with soil depth, we took intact soil cores from playas at multiple depth intervals within the soil profile: 0–5 cm, 5–25 cm and 25–50 cm. Sample playas did not contain surface water. We additionally collected 50–75 cm and 75–100 cm depth intervals from playa subsets. We used soil cores, slide hammers and augers to sample soils (AMS Inc., American Falls, ID, USA). We used a 5.08-cm soil core and a slide hammer to collect the 0–5 cm depth interval. For other depths, we used a 3.81-cm soil core and slide hammer. To minimize compaction, we collected each depth interval separately, extracted it, and then used a 7.62 cm soil auger to excavate a wider pit down to the next depth interval surface. This minimized friction and suction on the soil core, as well as compaction of collected soil. Some compaction did occur, but to account for this we used the internal volume of the core, rather than of the retained soil as our soil volume estimate (Bronson et al., 2004).

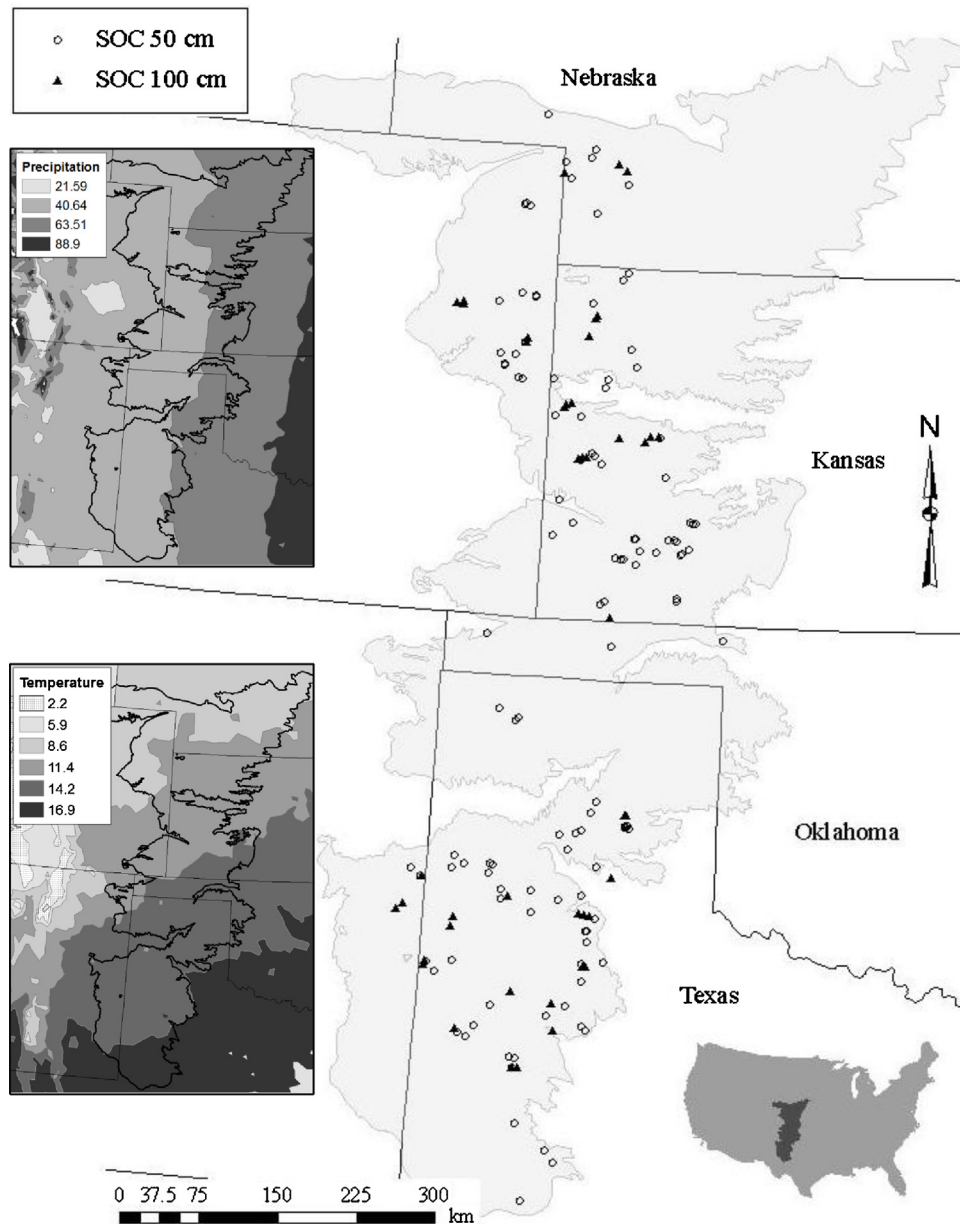


Fig. 1. Sites where SOC was sampled to either 50 cm or 100 cm in playas and uplands in the High Plains, USA. The border of the High Plains is shaded. Average annual temperature (°C) and precipitation (cm) from 1971 to 2000 (www.ncdc.noaa.gov) are inlayed.

In a similar manner, we collected soil samples along a transect extending from the playa visual edge southwest into the immediate upland watershed at 10 m, 40 m and 100 m from the playa. Some playas have lunettes (small ridges or dunes) on their southeast side (generally windward of playas) (Bowen and Johnson, 2012; Smith, 2003) and therefore, we chose to extend transects southwest to avoid lunettes and ensure sampling consistency. At playa subsets in the upland, 50–75 cm and 75–100 cm depth intervals were collected, but only at the 40 m point. At the time of soil collection, we recorded all plant species growing within a 0.5 m radius of sampling points. We converted to species m^{-2} by multiplying species richness by 1.27.

Soil samples were placed in sealed plastic freezer bags, stored on ice in the field and chilled to 4°C until laboratory analysis (USDA-NRCS, 2004). In the laboratory, soils were weighed then oven-dried to constant weight at 105°C. Soils were then ground to

pass through a 2-mm sieve. Rock and other fragments greater than 2 mm were rare, but were weighed when present. We used fluid displacement within a 50-ml graduated cylinder to estimate the volume of >2 mm fragments. Soil bulk density ($g\ dry\ soil/cm^3$) was calculated as: soil bulk density (d_b) = (weight of oven dry soil <2 mm)/(total soil volume), where volume was the internal volume of the soil core for that depth interval minus the volume of fragments >2 mm (Lal et al., 2001).

To determine soil moisture, we calculated percent soil moisture ($g\ H_2O/100\ g\ dry\ soil$) as percent water on a dry-mass basis, using the formula: percent soil moisture = $100 \times (\text{soil wet-mass} - \text{soil dry-mass}) / \text{soil dry-mass}$ (Sumner, 2000). We weighed 30 soil samples, at random, soon after collection in the field to determine wet mass. We compared this mass over time to verify that soils stored in sealed plastic freezer bags retained field moisture until sampled. Therefore, we measured wet weight of soil samples in the laboratory before drying.

To calculate root biomass, we removed, weighed and recorded coarse root biomass greater than 2 mm from dried soil samples prior to soil C analyses. We estimated percent dry weight of coarse belowground biomass as: percent belowground biomass = $100 \times$ dry belowground biomass/soil dry-mass (Sumner, 2000).

Soil texture data were ascertained using SSURGO soil maps. Sand, silt, and clay fractions were determined from the soil type in the upland watershed at each site. We used a weighted average of sand, silt, and clay fractions for the depth range of 0–50 cm. Based upon patterns of playa formation and development, playa soils are high clay, hydric Vertisol soils with gleyed reduced soil matrices. High clay content of playa soils is consistent with our field observations and with other studies that have shown increasing clay concentrations at lower slope positions (Zhao et al., 2009). Moreover, this has been validated by past studies describing playa pedology (Allen, 1972; Luo et al., 1999; Nelson and Western Energy and Land Use Team, United States, 1984; Soil Survey Staff, 2011).

We measured percent total soil carbon by weight with LECO TruSpec carbon and nitrogen analyzers (LECO Corporation, St. Joseph, MI, USA). Soils in the semiarid High Plains often contain inorganic carbonates (CaCO_3) (Reeves, 1970). We tested for inorganic carbonates by reacting soil subsamples with 2 M HCl (Schumacher, 2002). When present, we measured percent inorganic carbon with a modified pressure calciminer method (Sherrod et al., 2002). We calculated percent soil organic carbon (pSOC) as percent total carbon minus percent inorganic carbon in soils (Abreu et al., 2011). We converted pSOC to soil organic carbon (SOC) (kg m^{-2}) with the formula:

$$\text{SOC} = \frac{(\text{pSOC} \times d_b \times l_1) + (\text{pSOC} \times d_b \times l_2) + \dots (\text{pSOC} \times d_b \times l_n)}{10}$$

where l is the thickness of each sampled layer in cm. Dividing by 10 is needed to convert from g cm^{-3} to kg m^{-2} (Lal et al., 2001). Similarly, we estimated soil moisture (kg m^{-2}) and belowground biomass (g m^{-2}) by substituting percent soil moisture [moisture (g)/dry soil (g)] and percent dry root biomass [roots (g)/dry soil (g)] for pSOC in the above equation. All subsequent analyses use these areal measurements for SOC, soil moisture, and root biomass.

2.3. Statistical approach

Our goals were to compare SOC among sample points (playa and 10, 40 and 100 m into the uplands) and land-uses (native grassland, cropland, CRP). We used ordinary least squares (OLS) regression (i.e., linear models) in program R (version 3.0.2, the R Foundation for Statistical Computing) to compare SOC among land-use and sampling points over 5 cm, 50 cm and 1 m depths. To account for non-normal distribution, we log transformed SOC, resulting in reduced heterogeneity of variances and increased normality of residuals, though we present untransformed means in our figures for ease of interpretation.

We used the *spdep* package in R to investigate spatial dependency of our sample playas. We first calculated Moran's I for Ordinary Least Squares (OLS) linear model residuals using 1000 Monte Carlo permutations of the data. If spatial autocorrelation was present ($P < 0.1$), we investigated the nature of the spatial dependency using Lagrange Multiplier diagnostics. We observed spatial dependence in error terms ($P < 0.1$) and proceeded with spatial simultaneous autoregressive (SAR) modeling with spatially correlated errors (ESAR). We investigated spatial dependence in error over several scales including distance based and nearest neighbor-based approaches. Both P values and model Akaike's Information Criterion (AIC) suggested that a nearest neighbor approach that included all points within a catchment plus up to two points in the nearest catchments provided the best data fit. We compared Moran's I again for ESAR model residuals to insure

spatial dependency had been accounted for. To estimate final goodness of fit, we compared AIC, R^2 (OLS regression) and explained deviance (ESAR models) (Kühn, 2007). Explained deviance can be calculated as $(1 - [\text{residual sums of squares best fit model}] / [\text{residual sums of squares from an intercept only model}])$. We used a similar approach to compare soil moisture, soil texture, species richness and root biomass with distance from playas, along with land-use \times distance interaction over the 0–50 cm depth.

To elucidate which factors were important to SOC stocks among sample sites, we also investigated landscape level and geomorphology related variables. We modeled each land-use separately. Potential predictors were sediment depth, upland vs. wetland habitat, latitude, longitude and wetland area, as well as habitat interactions with each parameter. In croplands, we also added whether playa basins were tilled, or only surrounding uplands, and habitat \times tilling interactions. We downloaded average annual precipitation and temperature from 1971 to 2000 from the National Climate Data Center (www.ncdc.noaa.gov) to use as potential explanatory variables. However, temperature was highly correlated with latitude and precipitation was highly correlated with longitude ($r > 0.9$). Latitude and longitude provided finer grained predictors with better fit to SOC, as indicated by AIC. To refrain from using multicollinear variables as predictors, we elected to use latitude and longitude as proxies for climate gradients. ESAR models again were needed to account for spatial dependency in error terms. Variables to include in final models were selected using forward selection. We started with an intercept only model and added parameters that dropped AIC by > 4 .

Predicted SOC totals were also calculated within playas by land-use. For this analysis, we used the PLJV land-use and playa location data sets (www.pljv.org, accessed November 2012). We used these data because they were the most current (as of November 2012) for playa area and land cover estimates. The PLJV playa data used multiple sources to map wetland areas including the National Wetlands Inventory, the Soil Survey Geographic database, along with satellite imagery (www.pljv.org). The PLJV land cover data also used multiple sources including USGS Gap data, NatureServe Ecological Systems data, satellite imagery, SSURGO soils maps, Farm Service Agency Common Land Unit data, and other local sources as available (www.pljv.org). Because some of these data sources cannot be publicly distributed to protect land owner privacy rights (e.g., Farm Service Agency Common Land Unit data), GIS experts at PLJV used their data to summarize playa hectares by land-use indicating a total of 238,000 ha of playa wetlands (not including watersheds) within the High Plains (108,000 ha in native grassland, 25,000 ha in CRP, 82,000 ha in cropland and 23,500 ha in other land-use types) (A. Daniels, PLJV, unpublished data). Similar to the approach taken by Euliss et al. (2006), we applied our 1 m depth estimates of SOC in cropland, native grassland and CRP playas to the PLJV land-use and area estimates of wetlands throughout the High Plains (i.e., $\text{playa SOC}_{\text{land-use}} = e^{[\ln(\beta_{\text{land-use}} \pm \text{SE})]} \times \text{area}_{\text{land-use}}$), where $\beta_{\text{land-use}}$ is the playa parameter estimate (Table 2). To calculate historic playa wetland stores, we assumed that SOC within native grassland playas (these playas have never been plowed) represented historic SOC and applied our grassland playas estimate to all wetlands in the High Plains (i.e., historic SOC is $\beta_{\text{grassland}} \times 238,000$ ha). We calculated SOC loss within cropland playas by subtracting cropland playa SOC from historic playa SOC. We did not measure all possible correlates of SOC. Unmeasured correlates contribute to residual error in our final models. However, explained variance allows us to assign a 95% confidence interval to SOC differences by land-use. We calculated these intervals using SE by land-use (Table 2).

Table 1
OLS and ESAR model results of SOC (kg m^{-2}) among land uses and sample points, separately for each depth interval (soil surface to 5 cm, 50 cm, and 100 cm) in the High Plains, USA. Model results of soil moisture, species richness, and root biomass among land use and sample point are presented. Within land use (0–50 cm depth) models of SOC with potential predictors (sediment depth, habitat, latitude, longitude and wetland area) are also presented. Moran's I for model residuals are provided. *P* values for Lambda, the test statistic for spatially correlated errors also are provided for ESAR models.

Model	OLS: AIC	R^2	<i>F</i> , <i>P</i>	<i>P</i> for Moran's-I	ESAR: AIC	Exp. Dev.	Lambda <i>P</i>	<i>P</i> for Moran's-I
ln(SOC 0–5 cm) ~ land-use	1473.6	0.01	$F_{2,683} = 2.9, P < 0.054$	<0.001	1421.5	0.37	<0.001	0.444
ln(SOC 0–50 cm) ~ land-use × sampling point	802.9	0.06	$F_{11,653} = 2.7, P = 0.002$	<0.001	733.20	0.61	<0.001	0.269
ln(SOC 0–100 cm) ~ land-use + sampling point	146.6	0.05	$F_{1,221} = 5.9, P = 0.016$	0.023	141.80	0.56	0.008	0.539
ln(cropland SOC 0–50 cm) ~ latitude	249.6	0.03	$F_{3,89} = 2.2, P = 0.09$	<0.001	231.6	0.64	<0.001	0.441
ln(CRP SOC 0–50 cm) ~ playa area (ha)	208.8	0.04	$F_{1,197} = 7.4, P = 0.007$	<0.001	187.4	0.67	<0.001	0.456
ln(grassland SOC 0–50 cm) ~ habitat + playa area (ha) + latitude	346.3	0.06	$F_{3,213} = 4.5, P = 0.005$	<0.001	330.7	0.55	<0.001	0.362
Soil moisture ~ land-use + sampling point	1036.6	0.11	$F_{5,650} = 15.8, P < 0.001$	<0.001	944.8	0.58	<0.001	0.358
Species richness ~ land-use × sampling point	3074.2	0.27	$F_{11,653} = 22.4, P < 0.001$	<0.001	2949.4	0.37	<0.001	0.386
Root biomass ~ land-use + sampling point	2276.4	0.12	$F_{11,645} = 7.7, P < 0.001$	<0.001	2254.7	0.17	<0.001	0.578

3. Results

3.1. SOC differences among land-use types

In playa wetlands, SOC from 0 to 5 cm was 15% greater within native grasslands and CRP than croplands, whereas CRP and grassland were not different from each other (Tables 1 and 2 and Fig. 2a). SOC was 18% less in cropland playas than CRP from 0 to 50 cm (Tables 1 and 2 and Fig. 2b). From 0 to 100 cm, SOC was 20% greater in native grassland playas than in cropland, while CRP was intermediate to other land-uses (Tables 1 and 2 and Fig. 2c).

In uplands immediately adjacent to the wetlands, SOC at 0–5 cm did not differ among sampling points. For 0–50 cm, SOC was not different among land-use at 10 m from the wetland edge (Tables 1 and 2 and Fig. 2a and b), but was 28% and 21% greater in

grassland than cropland at 40 m and 100 m from the wetland edge, respectively (Tables 1 and 2 and Fig. 2b). At the 40 m point, grassland 0–50 cm SOC also was greater than in CRP (Tables 1 and 2 and Fig. 2b). Elsewhere, grassland and CRP uplands were equivalent to 50 cm depth. For 0–100 cm, SOC was 20% greater in native grasslands than in cropland, while CRP was intermediate to other land-uses (Tables 1 and 2 and Fig. 2c).

3.2. Within land-use comparisons: playa vs. upland SOC and comparing SOC with increasing distance from playas within a land-use

Watershed areas adjacent to playas did not store greater SOC than watershed areas further from wetlands. For 0–5 and 0–50 cm in uplands, SOC was not related to distance from playas in any land-use (Tables 1 and 2 and Fig. 2a and b). For surface SOC estimates (5 and 50 cm), SOC also was similar among upland and playa points within a land-use (Fig. 2a and b). Over 0–100 cm, playa SOC was higher than their immediate upland counterparts, averaging 1.3 kg m^{-2} higher within croplands and 1.5 kg m^{-2} higher within CRP and native grasslands (Table 2, $P = 0.07$; Fig. 2c).

Within a land-use, all playas and watersheds were not equivalent to each other. In native grassland and CRP, wetland area had a positive relationship with SOC (kg m^{-2}) from 0 to 50 cm in both playa and uplands, suggesting sites with larger wetlands had greater SOC per unit area. Playa area interactions with sampling point were not important (Tables 1 and 3). Latitude also influenced SOC without sampling point interactions, such that SOC increased along a northerly gradient in cropland and grassland (Tables 1 and 3). Northerly gradients also were highly correlated with increasing average annual precipitation (Fig. 1). Therefore, latitude and wetland area were influential in determining SOC in uplands and wetlands. Depth of accumulated upland sediments over the playa hydric soil surface was not related to SOC in either CRP or cropland ($P > 0.2$).

3.3. Patterns that provide context for SOC development: soil texture, soil moisture, species richness and belowground biomass

Soil texture did not differ among land-uses over 0–50 cm sampling depths. The sand fraction averaged $30.1 (\pm 0.17)$ and did not differ among land-uses ($F_{2, 166} = 1.89, P = 0.15$). Average silt content was $41.3 (\pm 0.15)$ and also did not differ among land-uses ($F_{2, 166} = 1.58, P = 0.21$). Similarly, clay content averaged $28.6 (\pm 0.11)$ and did not differ among land-uses ($F_{2, 166} = 1.59, P = 0.21$). Soil moisture ($\text{kg H}_2\text{O m}^{-2}$) totaled over the 0–50 cm depth, differed among land-use and across sampling locations within a site. As expected, within a land-use, wetland points were consistently

Table 2
Asymptotic coefficients from ESAR modeling of SOC (kg m^{-2}) within the High Plains, USA, from the soil surface to 5 cm, 50 cm, and 100 cm among land-use and sample point.

Parameter	Coef	SE	Z	<i>P</i>
Model: ln(SOC 5 cm) ~ land-use				
Land-use				
Crop playa (intercept)	0.41	0.06	-6.87	<0.001
CRP playa	0.14	0.09	1.62	0.106
Grass playa	0.16	0.09	1.89	0.058
Model: ln(SOC 50 cm) ~ land-use × sample point				
Land-use				
Crop playa (intercept)	1.70	0.06	30.15	<0.001
CRP playa	0.23	0.08	2.89	0.004
Grass playa	0.09	0.08	1.20	0.229
Sample point				
10 m	0.09	0.07	1.36	0.175
40 m	-0.05	0.07	-0.73	0.464
100 m	-0.01	0.07	-0.09	0.927
Land-use × sample point				
CRP × 10 m	0.10	0.10	1.0	0.332
CRP × 40 m	-0.08	0.10	-0.8	0.422
CRP × 100 m	-0.11	0.10	-1.1	0.270
Grass × 10 m	-0.21	0.10	-2.12	0.034
Grass × 40 m	0.23	0.10	2.41	0.016
Grass × 100 m	0.20	0.10	2.07	0.038
Model: ln(SOC 100 cm) ~ land-use + sample point				
Land-use				
Crop playa (intercept)	2.15	0.11	19.7	<0.001
CRP playa	0.11	0.12	0.8	0.404
Grass playa	0.20	0.13	1.6	0.109
Sample point				
40 m	-0.16	0.09	-1.8	0.071

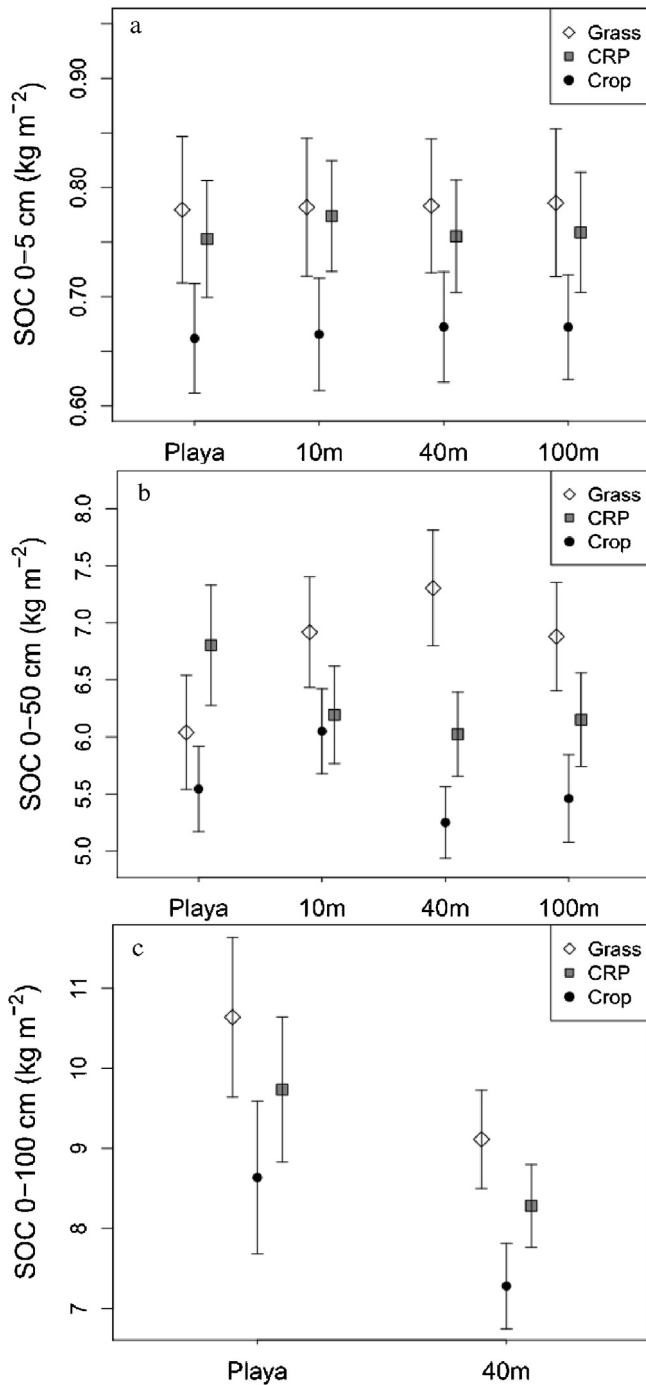


Fig. 2. SOC (kg m^{-2}) (mean \pm SE) within the High Plains, USA, from the soil surface to (a) 5 cm, (b) 50 cm, and (c) 100 cm. Playa = playa center, and 10 m, 40 m and 100 m are samples taken at these distances from the playa visual edge along a transect heading southwest.

moister than other points; 10 m points were intermediate, and at 40 m from the playa edge, soil moisture decreased to a value that remained constant at the more distal 100 m sample point (Tables 1 and 4). Among land-uses, croplands had the highest soil moistures, whereas CRP and grassland soil moistures were similar to each other and drier than croplands.

Species richness also varied among land-uses and among upland vs. wetland habitats within native grasslands. Cropland species richness was less than other land-uses at all sample points and ranged from 33 to 50% of that in CRP or native grasslands

Table 3

Best ESAR models of geomorphological or landscape level variables relating to ln (SOC) from 0 to 50 cm in playas and uplands by land-uses in the High Plains, USA.

Parameter	Coef	SE	Z	P
Cropland				
Intercept	0.67	0.57	1.18	0.238
Latitude	0.03	0.02	1.84	0.066
CRP				
Intercept	1.47	0.07	26.5	<0.001
Playa area (ha)	0.02	0.01	1.84	0.065
Grassland				
Playa (intercept)	0.15	0.91	0.16	0.871
Upland	0.15	0.07	2.26	0.018
Playa area (ha)	0.01	0.005	2.19	0.029
Latitude	0.04	0.02	1.74	0.081

(Tables 1 and 4). CRP and grassland playas had similar richness (~ 4 species m^{-2}). In CRP, richness remained similar among playa and upland points. Upland species richness was highest in native grasslands (~ 5 – 6 species m^{-2}); upland points within grassland were similar to each other (Tables 1 and 4).

Belowground biomass also differed among land-use and was less in in croplands than in CRP or native grasslands (Tables 1 and 4). CRP playas had 22% greater root mass than native grassland playas, but within uplands, native grasslands had $\sim 30\%$ greater root mass than CRP. Within a land-use, cropland 10 m sampling points had $\sim 40\%$ higher belowground biomass than other cropland sampling points. Within CRP, a similar pattern was observed with a 23% increase in root mass at the 10 m sampling point, whereas other points were similar to each other. Within native grasslands, belowground biomass was $\sim 50\%$ higher in uplands than in wetlands. Ten and 40 m points were similar to each other, but the 100 m point had 15% greater root mass than the other upland sampling points.

3.4. Estimating landscape level SOC within playas to 1 m

Currently, grassland, cropland and CRP playas store 20.8 Tg C (95% CI: 16.4–26.3 Tg C) in the top 1 m of soil in the High Plains. Grassland playa wetlands (108,000 ha) contain 11.4 Tg (95% CI: 8.9–14.5), CRP (25,000 ha) contains 2.4 Tg (95% CI: 1.9–3.1), while croplands (82,000 ha) contain 7.0 Tg C (95% CI: 5.7–8.7). Conversion of native grassland playas to cropland caused an estimated 2.0 Tg C loss (95% CI: 1.4–3.0 Tg C) over 107,000 ha of wetlands. Conversion of the remaining 108,000 ha of native grass playas to cropland could result in an additional 2.3 Tg C loss (95% CI: 1.5–

Table 4

Soil moisture, plant richness, belowground biomass in playas and uplands by land-uses in the High Plains, USA.

Land-use	Playa	10 m	40 m	100 m
Soil moisture (kg m^{-2})				
Cropland	1.2 \pm 0.1	1.0 \pm 0.1	0.8 \pm 0.1	0.8 \pm 0.1
CRP	1.0 \pm 0.1	0.8 \pm 0.1	0.6 \pm 0.1	0.6 \pm 0.1
Grassland	1.0 \pm 0.05	0.7 \pm 0.04	0.6 \pm 0.04	0.6 \pm 0.04
Species richness				
Cropland	2.2 \pm 0.3	2.1 \pm 0.3	1.7 \pm 0.3	1.8 \pm 0.3
CRP	4.2 \pm 0.3	4.1 \pm 0.3	4.2 \pm 0.3	3.7 \pm 0.3
Grassland	4.1 \pm 0.3	5.8 \pm 0.3	5.9 \pm 0.3	5.4 \pm 0.3
Root biomass (g m^{-2})				
Cropland	216 \pm 18	378 \pm 23	180 \pm 23	194 \pm 24
CRP	449 \pm 24	584 \pm 33	431 \pm 33	479 \pm 33
Grassland	353 \pm 24	662 \pm 33	692 \pm 33	790 \pm 33

3.5 Tg C). Losses of C in adjacent uplands by converting native prairie to cropland would further contribute to total C losses across the landscape. Currently, CRP enrollments over 25,000 ha of wetlands (A. Daniels, PLJV, unpublished data) have returned 0.2 Tg C (95% CI: 0.1–0.3), only half of the historic SOC levels in these playas.

4. Discussion

Land-use alteration had a significant impact on SOC in playas and upland watersheds across the High Plains. As a result, wetlands embedded in cropland settings contained less SOC than those in native grassland, and CRP perennial grass establishment of former croplands returned half of the historic SOC lost through cultivation. Soil depth played an important role in SOC estimates. To a depth of 50 cm, playa SOC was equivalent with upland SOC, however, over 1 m soil depths, sampling locations as well as land-use differences were evident, such that playa SOC generally was higher than in surrounding uplands. Shallow soil sampling can mask important SOC patterns, and therefore our study benefited from our 1 m soil sampling protocol. From the soil surface to 1 m soil depths, grassland playas had 20% greater SOC than cropland playas and 16% greater SOC than native grassland uplands. Our results are similar to those of Wiesmeier et al. (2013) who also found that differences in SOC between cropland and grasslands may be more apparent in subsurface soils. Indeed, native grassland playas are important sites of SOC within the High Plains and should be preserved from crop production where SOC promotion is a goal.

4.1. SOC estimates within croplands

In our study, croplands averaged roughly 20% less SOC than native grasslands in both playa and upland points from 0 to 1 m depths. There are multiple possible explanations for this land-use difference. First, when dry, cropland playas often were tilled along with uplands (O'Connell et al., 2013b, 2012). Soil tilling within croplands causes compaction and disrupts soil structure, accelerating decomposition and reducing SOC (Burke et al., 1995; Post and Kwon, 2000). Second, upland watershed cultivation causes eroded topsoil to accumulate in wetlands downslope (Daniel et al., 2014; Luo et al., 1997). These sediments are depleted in SOC, resulting in reduced playa SOC estimates at the soil surface. Maynard et al. (2011) suggested that burial of seasonal wetland hydric soils by eroded agricultural sediments may protect soil organic carbon within hydric soils. They argued that sediment deposition reduces decomposition of the original hydric soil layer. However, depth of eroded soils in playa basins was not related to SOC storage in cropland and CRP playas in this study.

Sedimentation also reduces playa basin volume and duration of water ponding (Tsai et al., 2007, 2010), potentially altering C accumulation. Moreover, sedimentation and tilling alter plant communities, such that sediment accumulation reduces seed germination rates, while tilling removes aboveground plants (O'Connell et al., 2013b). Cropland playas exhibited declines in belowground biomass and richness compared with other land-uses, further evidence of altered plant communities. Tillage was likely the dominant process driving SOC estimates; upland and playa SOC were similar in croplands and tilling was the only process common to both.

4.2. SOC estimates within CRP lands

Estimated over 1 m soil depth, wetlands embedded in CRP had intermediate SOC compared with native grasslands and croplands. Thus, CRP establishment replenishes some SOC lost through cultivation. Multiple lines of evidence suggest that soil water

availability likely was a limiting factor influencing CRP estimates. First, when CRP grasses were established, playa restoration was not a goal. Rather, the program aimed to conserve topsoil within highly erodible croplands by establishing perennial grasses, the vast majority of which were exotic. Accumulated upland sediments within playas were not removed and upland grasses were planted in some playas as well as their adjacent uplands. As a result, plant communities differed little among CRP playas and uplands, in terms of species richness, and belowground biomass (this study) and wetland vegetation was under represented (O'Connell et al., 2012, 2013a). Thus, while CRP playas had greater SOC than CRP uplands, plant communities largely did not differ and therefore could not have driven upland-wetland differences. However, CRP playas were inundated less frequently than grassland and cropland playas, likely because CRP grasses have higher biomass than short grass prairie and provide a barrier to overland run-off (Cariveau et al., 2011; O'Connell et al., 2012). Second, the variable that differed most among upland and playa points was greater soil moisture in playas. Finally, playa area was positively related to SOC within CRP catchments. Larger playas have larger watersheds to collect precipitation, and subsequently have longer hydroperiods (Tsai et al., 2007). Thus, CRP soils with greater water availability had greater SOC.

Post and Kwon (2000) reviewed studies of SOC in CRP lands and found that arid climates and low productivity generally retard SOC accumulation to rates of 0.1–0.03 kg C m⁻² year⁻¹ in re-established grasslands. Post and Kwon's (2000) rates are within the range of possible rates of accumulation for our CRP playas. For example, our CRP playas and uplands were each roughly 1 kg C m⁻² greater than in the same habitats within croplands. CRP enrollments began in the High Plains in 1985, 24 years before our sampling. Because of landowner privacy rights, we could not obtain length of time within CRP enrollment for our wetlands. However, if we assume an average length of enrollment of 6, 12, or 18 years, our accumulation rates probably were within the range of their estimates (e.g., 0.16, 0.08 and 0.05 kg C m⁻² year⁻¹, respectively).

While CRP playas were inundated less often than cropland and native grassland playas (O'Connell et al., 2012), they certainly had higher soil moisture than CRP uplands. Although conditions created by CRP establishment on playas are sufficient for restocking lost soil C, proper wetland restoration techniques would likely increase the rate of C storage. Euliss et al. (2006) demonstrated that the rate of C storage in restored prairie wetlands was 5 times higher than that of restored grasslands. Thus, if SOC restoration is a land management goal, we recommend removing accumulated upland sediment from CRP wetlands and planting only historically common native species within CRP wetlands and uplands.

4.3. Within land-use comparisons: wetland vs. upland and catchment differences

Soil moisture was not related to distance from playas further than 10 m from wetlands. Others also have observed that soil moisture gradients do not exist down slopes in semiarid ecosystems (Singh et al., 1998; Yonker et al., 1988), attributing this to high evapotranspiration, and few precipitation and runoff events. Similarly, species richness and belowground biomass exhibited only small differences across upland points. Therefore, the main contribution of playas to landscape level SOC is within the playa hydric soil footprint.

However, all playas and adjacent watersheds were not equivalent within a land-use. Playa area and latitude, a proxy for temperature gradients, were related to SOC. In croplands, playa area likely was not important both because tilling is a strong

disturbance that can mask other drivers, and because cropland playas were filled in with upland sediments and thus were generally small (O'Connell et al., 2012). Interestingly, in native grassland and CRP, where playa area was important, sampling point interactions with playa area were not. Thus, uplands surrounding large playas have greater SOC per unit area than sites with small playas. Large playas have larger watersheds that collect more precipitation than smaller playas. Soil moisture increased with latitude and longitude across all land-uses (data not presented). Thus again, cooler and moister soils contained greater SOC, all else being equal.

4.4. Influences on landscape level SOC estimates

In general, playa SOC was less than in other wetlands, probably because constantly fluctuating hydrology within wetlands promotes rapid decomposition (Anderson and Smith, 2002; Bernal and Mitsch, 2008; Maynard et al., 2011). In contrast, longer inundation periods within other wetland types, promotes anaerobic soils and SOC accumulation. Prairie potholes, depression wetlands common to northern Great Plains prairies where climate is cooler and wetter, stored 10 kg C m⁻² to a depth of 30 cm in reference conditions (Euliss et al., 2006). Playas stored approximately 6.0 kg C m⁻² to a depth of 50 cm in native grasslands.

However, this does not suggest that playas are unimportant for SOC. Playas in native grasslands had 20% greater SOC than those in cropland to a 100 cm depth and 16% greater SOC than in native grassland uplands. Cropland conversion of playas conservatively caused 1.8 Tg C loss and conversion of upland watersheds will certainly increase C loss. Therefore native grassland playas and watersheds are important to High Plains SOC because they have higher SOC than surrounding habitats. Land-use conversion of playas and uplands promotes CO₂ export to the atmosphere.

When considering SOC storage, length of time since land-use conversion is an important co-variable that we did not account for. Cultivation history began pre-1910 within the High Plains and was patchy across the landscape (USDA census of agriculture historical archive, agcensus.mannlib.cornell.edu). Balesdent et al. (1988) identified that labile SOC is lost during the first 30–40 years of cultivation. Recalcitrant SOC remained over a 100 years period in their long term study of croplands (Balesdent et al., 1988). Cihacek and Ulmer (1995) also identified variable rates of C loss from cultivated soils, ranging from 10% for croplands <10 years old to 50% for croplands greater than 40 years old. Length of time within a land-use was not available for individual locations because of land owner privacy rights. There also were no suitable public datasets for estimating land-use change. For example, the National Land Cover Database for the United States (landcover.usgs.gov) does not contain CRP as a cover class and was only 75% accurate for classifying croplands within the year we surveyed (data not presented).

Therefore, we relied on literature values to approximate the influence of time on SOC estimates. Our croplands lost ~20% of SOC compared with grasslands. Additional SOC losses might be possible with longer cultivation times, given that older croplands lose up to 50% of historic SOC (Cihacek and Ulmer, 1995). For CRP we are also interested in rates of SOC gain. We applied Post and Kwan's (2000) accumulation rates for semi-arid CRP lands (0.1–0.03 kg C m⁻² year⁻¹). In our study, native grasslands contained 1 kg additional C m⁻² than CRP, and CRP contained 1 kg additional SOC than croplands, within a habitat. Thus, we estimate that average length of time for CRP enrollments we sampled was between 10 and 24 years. Therefore, an additional 10–30 years will be necessary for CRP SOC to resemble native grasslands. Croplands planted to grass sometimes have greater SOC than nearby cultivated croplands

(Potter et al., 1999), but often do not approach SOC in native prairies (Huggins et al., 1997; Purakayastha et al., 2008), even after 60 years since planting (Potter et al., 1999); but see Reeder et al. (1998).

4.5. Conservation recommendations

Information presented in this paper provides insight for regional and nationwide C modeling efforts that can guide future climate mitigation. Playas are important for SOC in the High Plains, storing approximately 20.8 Tg C in the top 1 m of soil. Additionally, under proper management practices, playas may replenish lost C stores faster than adjacent uplands. Certain practices such as cultivating playas results in substantial C losses and should be avoided. Untilled playas also have concomitant benefits for plant communities (O'Connell et al., 2012). Removing land parcels containing playas from active cultivation and placing them in CRP can replenish some C lost from agricultural activities, however, rates of replenishment in playas embedded in CRP would likely increase if natural hydroperiods were restored by removing upland sediments from playa basins. Additionally, planting native short grass species rather than introduced tall-grasses in CRP playa watersheds would increase frequency of inundation and likely rates of C accumulation.

Sites with larger playas had greater SOC per unit area than those with small playas and should be preferentially preserved to promote SOC in the High Plains. Further, northern playas had greater SOC than southern ones. Thus, conservation could be targeted in these areas if resources were limited and SOC management is a primary management goal. Playas and uplands still remaining in native grassland should be preserved because these store the greatest amount of C, losses of which will take decades to regain.

Acknowledgements

We thank Amber Staley Williams, K. Marie Russo, Guido Ufer, Jason Warren, Dave Haukos, Sam Fuhlendorf, Lynn Nymeyer, Joe Hartman, William Burnidge, Nathan Andrews, Alex Daniels, Megan McLachlan, Buffalo Lake National Wildlife Refuge, Cimarron and Comanche National Grasslands, and The Nature Conservancy for their kind assistance. Laboratory funding was provided by Oklahoma State University and field costs were supported by the CEAP Wetlands project (NRCS-USDA).

References

- Abreu, S.L., Godsey, C.B., Edwards, J.T., Warren, J.G., 2011. Assessing carbon and nitrogen stocks of no-till systems in Oklahoma. *Soil Tillage Res.* 117, 28–33. doi: <http://dx.doi.org/10.1016/j.still.2011.08.004>.
- Allen, B.L., 1972. The Mineralogy and Chemistry of High Plains Playa Lake Soils and Sediments: Project Completion Report. Agronomy Department, Texas Tech University.
- Anderson, J.T., Smith, L.M., 2002. The effect of flooding regimes on decomposition of *Polygonum pensylvanicum* in playa wetlands (Southern Great Plains, USA). *Aquat. Bot.* 74, 97–108. doi: [http://dx.doi.org/10.1016/S0304-3770\(02\)00049-9](http://dx.doi.org/10.1016/S0304-3770(02)00049-9).
- Armentano, T.V., Menges, E.S., 1986. Patterns of change in the carbon balance of organic soil-wetlands of the temperate zone. *J. Ecol.* 74, 755. doi: <http://dx.doi.org/10.2307/2260396>.
- Balesdent, J., Wagner, G.H., Mariotti, A., 1988. Soil organic matter turnover in long-term field experiments as revealed by carbon-13 natural abundance. *Soil Sci. Soc. Am. J.* 52, 118. doi: <http://dx.doi.org/10.2136/sssaj1988.03615995005200010021x>.
- Bernal, B., Mitsch, W.J., 2008. A comparison of soil carbon pools and profiles in wetlands in Costa Rica and Ohio. *Ecol. Eng.* 34, 311–323. doi: <http://dx.doi.org/10.1016/j.ecoleng.2008.09.005>.
- Bolen, E.G., Smith, L.M., Schramm, H.L., 1989. Playa lakes: prairie wetlands of the Southern High Plains. *Bioscience* 39, 615–623.
- Bowen, M.W., Johnson, W.C., 2012. Late quaternary environmental reconstructions of playa-lunette system evolution on the central High Plains of Kansas, United States. *Geol. Soc. Am. Bull.* 124, 146–161. doi: <http://dx.doi.org/10.1130/b30382.1>.

- Bronson, K.F., Zobeck, T.M., Chua, T.T., Acosta-Martinez, V., Pelt, R.S., van Booker, J.D., 2004. Carbon and nitrogen pools of Southern High Plains cropland and grassland soils. *Soil Sci. Soc. Am. J.* 68, 1695–1704.
- Burke, I.C., Lauenroth, W.K., Coffin, D.P., 1995. Soil organic matter recovery in semiarid grasslands: implications for the Conservation Reserve Program. *Ecol. Appl.* 5, 793–801.
- Cariveau, A., Pavlacky, D., Bishop, A., LaGrange, T., 2011. Effects of surrounding land use on playa inundation following intense rainfall. *Wetlands* 31, 65–73. doi: <http://dx.doi.org/10.1007/s13157-010-0129-4>.
- Cihacek, L., Ulmer, M., 1995. Estimated soil organic carbon losses from long-term crop-fallow in the Northern Great Plains of the USA. In: Lal, R., John Kimble, M., Elissa Levine, R., Bobby Stewart, A. (Eds.), *Soil Management and the Greenhouse Effect*. CRC Press, Boca Raton FL, USA, pp. 85–92.
- Daniel, D.W., Smith, L.M., Haukos, D.A., Johnson, L.A., McMurry, S.T., 2014. Land use and conservation reserve program effects on the persistence of playa wetlands in the High Plains. *Environ. Sci. Technol.* 48 (8), 4282–4288.
- Daniel, D.W., Smith, L.M., Belden, J.B., McMurry, S.T., Swain, S., 2015. Effects of land-use change and fungicide formulations on non-target soil fungi in playa wetlands of the U.S. High Plains. *Sci. Total Environ.* doi: <http://dx.doi.org/10.1016/j.scitotenv.2015.01.066>.
- Detenbeck, N.E., Elonen, C.M., Taylor, D.L., Cotter, A.M., Puglisi, F.A., Sanville, W.D., 2002. Effects of agricultural activities and best management practices on water quality of seasonal prairie pothole wetlands. *Wetl. Ecol. Manag.* 10, 335–354. doi: <http://dx.doi.org/10.1023/a:1020397103165>.
- Euliss, N.H., Gleason, R.A., Olness, A., McDougal, R.L., Murkin, H.R., Robarts, R.D., Bourbonniere, R.A., Warner, B.G., 2006. North American prairie wetlands are important nonforested land-based carbon storage sites. *Sci. Total Environ.* 361, 179–188. doi: <http://dx.doi.org/10.1016/j.scitotenv.2005.06.007>.
- Gebhart, D.L., Johnson, H.B., Mayeux, H.S., Polley, H.W., 1994. The CRP increases soil organic carbon. *J. Soil Water Conserv.* 49, 488–492.
- Guo, L.B., Gifford, R.M., 2002. Soil carbon stocks and land use change: a meta analysis. *Glob. Change Biol.* 8, 345–360. doi: <http://dx.doi.org/10.1046/j.1354-1013.2002.00486.x>.
- High Plains Regional Climate Center, 2011. *Historical Climate Data Summaries [WWW Document]*. URL <http://www.hprcc.unl.edu/data/historical/index.php> (accessed 01.02.20).
- Huggins, D.R., Allen, D.L., Gardner, J.C., Karlen, D.L., Bezdicke, D.F., Rosek, M.J., Alms, M.J., Flock, M., Miller, B.S., Staben, M.L., 1997. Enhancing carbon sequestration in CRP-managed land. In: Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A. (Eds.), *Management of Carbon Sequestration in Soil*. CRC Press, Boca Raton, FL, USA, pp. 323–334.
- Kelly, E.F., Yonker, C.M., Blecker, S.W., Olson, C.G., 2008. Soil development and distribution in the shortgrass steppe ecosystem. In: Lauenroth, W.K., Burke, I.C. (Eds.), *Ecology of the Shortgrass Steppe: A Long-term Perspective*. Oxford University Press, Cary, NC, USA, pp. 30–54 (Chapter 3).
- Kühn, I., 2007. Incorporating spatial autocorrelation may invert observed patterns. *Divers. Distrib.* 13, 66–69. doi: <http://dx.doi.org/10.1111/j.1472-4642.2006.00293.x>.
- Lal, R., 2003. Soil erosion and the global carbon budget. *Environ. Int.* 29 (4), 437–450.
- Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A. (Eds.), 2001. *Assessment Methods for Soil Carbon*. CRC Press, Boca Raton, FL, USA.
- Leifeld, J., Ammann, C., Nefel, A., Fuhrer, J., 2011. A comparison of repeated soil inventory and carbon flux budget to detect soil carbon stock changes after conversion from cropland to grasslands. *Glob. Change Biol.* 17, 3366–3375. doi: <http://dx.doi.org/10.1111/j.1365-2486.2011.02471.x>.
- Luo, H.R., Smith, L.M., Allen, B.L., Haukos, D.A., 1997. Effects of sedimentation on playa wetland volume. *Ecol. Appl.* 7, 247–252.
- Luo, H.R., Smith, L.M., Haukos, D.A., Allen, B.L., 1999. Sources of recently deposited sediments in playa wetlands. *Wetlands* 19, 176–181.
- Maltby, E., Immirzi, P., 1993. Carbon dynamics in peatlands and other wetland soils: regional and global perspectives. *Chemosphere* 27 (6), 999–1023.
- Maynard, J.J., Dahlgren, R.A., O'Geen, A.T., 2011. Soil carbon cycling and sequestration in a seasonally saturated wetland receiving agricultural runoff. *Biogeosciences* 8, 3391–3406. doi: <http://dx.doi.org/10.5194/bg-8-3391-2011>.
- Nelson, R.W., Western Energy and Land Use Team, United States, 1984. *Playa Wetlands and Wildlife on the Southern Great Plains: A Characterization of Habitat*. The Team, Washington, D.C.
- O'Connell, J.L., Johnson, L.A., Smith, L.M., McMurry, S.T., Haukos, D.A., 2012. Influence of land-use and conservation programs on wetland plant communities of the semiarid United States Great Plains. *Biol. Conserv.* 146, 108–115. doi: <http://dx.doi.org/10.1016/j.biocon.2011.11.030>.
- O'Connell, J.L., Johnson, L.A., Beas, B.J., Smith, L.M., McMurry, S.T., Haukos, D.A., 2013a. Predicting dispersal-limitation in plants: optimizing planting decisions for isolated wetland restoration in agricultural landscapes. *Biol. Conserv.* 159, 343–354. doi: <http://dx.doi.org/10.1016/j.biocon.2012.10.019>.
- O'Connell, J.L., Johnson, L.A., Daniel, D.W., McMurry, S.T., Smith, L.M., Haukos, D.A., 2013b. Effects of agricultural tillage and sediment accumulation on emergent plant communities in playa wetlands of the U.S. High Plains. *J. Environ. Manag.* 120, 10–17. doi: <http://dx.doi.org/10.1016/j.jenvman.2013.01.035>.
- Post, W.M., Kwon, K.C., 2000. Soil carbon sequestration and land-use change: processes and potential. *Glob. Change Biol.* 6, 317–327. doi: <http://dx.doi.org/10.1046/j.1365-2486.2000.00308.x>.
- Post, W.M., Emanuel, W.R., Zinke, P.J., Stangenberger, A.G., 1982. Soil carbon pools and world life zones. *Nature* 298, 156–159.
- Potter, K.N., Torbert, H.A., Johnson, H.B., Tischler, C.R., 1999. Carbon storage after long-term grass establishment on degraded soils. *Soil Sci. Soc. Am. J.* 64, 718–725.
- Powlson, D.S., Whitmore, A.P., Goulding, K.W.T., 2011. Soil carbon sequestration to mitigate climate change: a critical re-examination to identify the true and the false. *Eur. J. Soil Sci.* 62, 42–55. doi: <http://dx.doi.org/10.1111/j.1365-2389.2010.01342.x>.
- Purakayastha, T., Huggins, D., Smith, J., 2008. Carbon sequestration in native prairie, perennial grass, no-till, and cultivated Palouse silt loam. *Soil Sci. Soc. Am. J.* 72, 534–540. doi: <http://dx.doi.org/10.2136/sssaj2005.0369>.
- Raich, J.W., Schlesinger, W.H., 1992. The global carbon dioxide flux in soil respiration and its relationship to vegetation and climate. *Tellus B* 44, 81–99. doi: <http://dx.doi.org/10.1034/j.1600-0889.1992.t01-1-00001.x>.
- Reeder, J.D., Schuman, G.E., Bowman, R.A., 1998. Soil C and N changes on conservation reserve program lands in the Central Great Plains. *Soil Tillage Res.* 47, 339–349. doi: [http://dx.doi.org/10.1016/s0167-1987\(98\)00122-6](http://dx.doi.org/10.1016/s0167-1987(98)00122-6).
- Reeves Jr., C.J., 1970. Origin, classification, and geologic history of caliche on the southern High Plains, Texas and eastern New Mexico. *J. Geol.* 352–362.
- Samson, F.B., Knopf, F.L., Ostlie, W.R., 2004. Great Plains ecosystems: past, present, and future. *Wildl. Soc. Bull.* 32, 6–15.
- Scanlon, B.R., Faunt, C.C., Longuevergne, L., Reedy, R.C., Alley, W.M., McGuire, V.L., McMahon, P.B., 2012. Groundwater depletion and sustainability of irrigation in the US High Plains and Central Valley. *PNAS* 109, 9320–9325. doi: <http://dx.doi.org/10.1073/pnas.1200311109>.
- Schlesinger, W.H., Andrews, J.A., 2000. Soil respiration and the global carbon cycle. *Biogeochemistry* 48, 7–20. doi: <http://dx.doi.org/10.1023/A:1006247623877>.
- Schoenberger, P., Wysocki, D., Benham, E., Broderson, W., 2002. *Field Book for Describing and Sampling Soils, Version 2*. NRCS, USDA, Lincoln, NE, USA.
- Schumacher, B.A., 2002. *Methods for the Determination of Total Organic Carbon (TOC) in Soils and Sediments*. U.S. EPA, Las Vegas, NV, USA.
- Sherrod, L.A., Dunn, G., Peterson, G.A., Kolberg, R.L., 2002. Inorganic carbon analysis by modified pressure-calimeter method. *Soil Sci. Soc. Am. J.* 66, 299–305.
- Singh, J.S., Milchunas, D.G., Lauenroth, W.K., 1998. Soil water dynamics and vegetation patterns in a semiarid grassland. *Plant Ecol.* 134, 77–89. doi: <http://dx.doi.org/10.1023/A:1009769620488>.
- Smith, L.M., 2003. *Playas of the Great Plains*. University of Texas Press, Austin, TX, USA.
- Smith, L.M., Haukos, D.A., 2002. Floral diversity in relation to playa wetland area and watershed disturbance. *Conserv. Biol.* 16, 964–974.
- Smith, L.M., Haukos, D.A., McMurry, S.T., LaGrange, T., Willis, D., 2011. Ecosystem services provided by playa wetlands in the High Plains: potential influences of USDA conservation programs and practices. *Ecol. Appl.* 21, S82–S92. doi: <http://dx.doi.org/10.1890/09-1133.1>.
- Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture. *Official Soil Series Descriptions*. Available online (accessed 05.10.11).
- Sumner, M.E., 2000. *Handbook of Soil Science*. CRC Press, Boca Raton, FL, USA.
- Swift, R.S., 2001. Sequestration of carbon by soil. *Soil Sci.* 166, 858–871. doi: <http://dx.doi.org/10.1097/00010694-200111000-00010>.
- Syswerda, S.P., Corbin, A.T., Mokma, D.L., Kravchenko, A.N., Robertson, G.P., 2011. Agricultural management and soil carbon storage in surface vs. deep layers. *Soil Sci. Soc. Am. J.* 75, 92. doi: <http://dx.doi.org/10.2136/sssaj2009.0414>.
- Trumbore, S.E., 1997. Potential responses of soil organic carbon to global environmental change. *Proc. Natl. Acad. Sci.* 94, 8284–8291.
- Tsai, J.S., Venne, L.S., McMurry, S.T., Smith, L.M., 2007. Influences of land use and wetland characteristics on water loss rates and hydroperiods of playas in the Southern High Plains, USA. *Wetlands* 27, 683–692.
- Tsai, J.-S., Venne, L.S., McMurry, S.T., Smith, L.M., 2010. Vegetation and land use impact on water loss rate in playas of the southern High Plains, USA. *Wetlands* 30, 1107–1116. doi: <http://dx.doi.org/10.1007/s13157-010-0117-8>.
- USDA, 2009. *2007 Census of Agriculture*.
- USDA-NRCS, 2004. *Soil Survey Laboratory Methods Manual (Soil Survey Investigations Report No. 42)*, Soil Survey Investigations. U. S. Department of Agriculture, Natural Resources Conservation Service, Lincoln, NE, USA.
- USDA-NRCS, 2006. *Land Resource Regions and Major Land Resource Areas of the United States, the Caribbean, and the Pacific Basin*. U.S. Department of Agriculture Handbook 296, Washington, D.C., USA.
- Van der Kamp, G., Hayashi, M., Gallén, D., 2003. Comparing the hydrology of grassed and cultivated catchments in the semi-arid Canadian prairies. *Hydrol. Process.* 17, 559–575. doi: <http://dx.doi.org/10.1002/hyp.1157>.
- Wiesmeier, M., Hübner, R., Barthold, F., Spörlin, P., Geuß, U., Hangen, E., Kögel-Knabner, I., 2013. Amount, distribution and driving factors of soil organic carbon and nitrogen in cropland and grassland soils of southeast Germany (Bavaria). *Agric. Ecosyst. Environ.* 176, 39–52.
- Yonker, C., Schimel, D., Paroussis, E., Heil, R., 1988. Patterns of organic-carbon accumulation in a semiarid shortgrass steppe, Colorado. *Soil Sci. Soc. Am. J.* 52, 478–483.
- Young, C.E., Osborn, C.T., 1990. Costs and benefits of the conservation reserve program. *J. Soil Water Conserv.* 45, 370–373.
- Zhao, Z., Chow, T.L., Rees, H.W., Yang, Q., Xing, Z., Meng, F.-R., 2009. Predict soil texture distributions using an artificial neural network model. *Comput. Electron. Agric.* 65, 36–48. doi: <http://dx.doi.org/10.1016/j.compag.2008.07.008>.