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EFFECTIVENESS OF VEGETATION BUFFERS SURROUNDING PLAYA WETLANDS AT CONTAMINANT AND SEDIMENT AMELIORATION

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Abstract – Playa wetlands, the dominant hydrological feature of the semi-arid U.S. High Plains providing critical ecosystem services, are being lost and degraded due to anthropogenic alterations of the short-grass prairie landscape. The primary process contributing to the loss of playas is the filling of the wetland through the accumulation of soil eroded and transported by precipitation from surrounding cultivated watersheds. We evaluated effectiveness of vegetative buffers surrounding playas in removing metals, nutrients, and dissolved/suspended sediments from precipitation runoff. Storm water runoff was collected at 10-m intervals in three buffer types (native grass, fallow cropland, and Conservation Reserve Program). Buffer type differed in plant composition, but not in maximum percent removal of contaminants. Within the initial 60 m from a cultivated field, vegetation buffers of all types removed >50% of all measured contaminants, including 83% of total suspended solids (TSS) and 58% of total dissolved solids (TDS). Buffers removed an average of 70% of P and 78% of N to reduce nutrients entering the playa. Mean maximum percent removal for metals ranged from 56% of Na to 87% of Cr. Maximum removal was typically at 50 m of buffer width. Measures of TSS were correlated with all measures of metals and nutrients except for N, which was correlated with TDS. Any buffer type with >80% vegetation cover and 30 – 60 m in width would maximize contaminant removal from precipitation runoff while ensuring that playas would continue to function hydrologically to provide ecosystem services. Watershed management to minimize erosion and creations of vegetation buffers could be economical and effective conservation tools for playa wetlands.

Key Words: Buffer; Metals; Nutrients; Playas; Southern High Plains; Vegetation

1. Introduction

The principal surface hydrological features of the U.S. western High Plains are playa wetlands (Guthery and Bryant 1982). The distribution of the High Plains playas extends from

western Nebraska and eastern Wyoming southward into eastern New Mexico and northwest Texas (Osterkamp and Wood 1987). Playas are isolated, depressional, recharge wetlands representing the terminus point of a closed watershed (Smith 2003). The greatest density of playa wetlands occurs in the Southern High Plains in New Mexico and Texas (Smith et al. 2012). As the keystone habitat of the Southern High Plains, playas provide numerous ecosystem services individually, as an integrated wetland system, and by interacting with all other ecosystems in the region (Smith et al. 2011). Collectively, these wetlands are the primary sites for floodwater catchment, biodiversity, islands of refugia for native plant species, and focused recharge points to the underlying aquifer (Smith et al. 2012). They also are important sites for biomass production, water quality improvement, livestock water and forage, irrigation water, and recreation (Smith et al. 2011). In addition, playas are the primary wetland habitat for numerous wetland-dependent species that breed, migrate through, and winter in the Southern High Plains (Haukos and Smith 1994).

In a fully functional state, playas solely receive water from precipitation and associated runoff, with water loss occurring through evaporation, transpiration, and aquifer recharge (Smith 2003). When functional, these hydrologically-isolated wetlands go through unpredictable, natural wet-dry fluctuations, the duration (i.e., hydroperiod) and timing of which are the driving forces defining playa ecology (Haukos and Smith 1994). However, anthropogenic forces are directly and indirectly impacting the occurrence and ecology of playa wetlands (Johnson et al. 2012). Playas occur in one of the most agriculturally impacted regions in the Western Hemisphere as conversion to row-crop agriculture and livestock grazing have greatly altered the original short-grass prairie ecoregion (Bolen et al. 1989). Anthropogenic factors predominantly resulting from these activities have caused complete playa loss, with impairment of ecosystem

functions and services in many other playas. Approximately 70% of the Texas Southern High Plains has been converted to irrigated or nonirrigated cropland (McMahon et al. 2007) and Johnson et al. (2012) reported that only 0.2% of playas in the Southern High Plains had no wetland or watershed modification. Johnson et al. (2012) conservatively estimated at least 17% of historical playas had been subject to complete physical loss with no sign of presence; but if the playas where sediment has filled the hydric-clay soil volume were defined as physically lost, then potentially up to 60% of historical playas have disappeared from the landscape.

The magnitude of anthropogenic impacts are often higher in playas compared to other isolated wetlands because playas occur in watersheds with naturally highly erodible soils, causing them to be susceptible to deposition of sediment, chemical contaminants, and excess nutrients following runoff (Haukos and Smith 2003). Sediment deposition in playas is chiefly water-, not wind-deposited (Luo et al. 1999). Sediment accumulation in playas and its influence on identification of playa soils (Johnson et al. 2011), playa occurrence and volume (Luo et al. 1997, Tsai 2007, O'Connell et al. 2013, Daniel et al. 2014), hydroperiod (Tsai et al. 2007), plant community composition (Haukos and Smith 2004, O'Connell et al. 2012), and overall ecological function are well-documented (e.g., Smith and Haukos 2002, Smith et al. 2012). On average, playas with cultivated watersheds have had all of their original hydric-soil defined volume filled with sediment, and due to the effects of concentrated grazing (i.e., reduction in vegetation cover), playas in grazed grassland have lost approximately one-third to one-half of their hydric-soil defined volume (Luo et al. 1997, Daniel et al. 2014). Burris and Skagen (2012) predicted, based on soil loss and climate change models, that sediment accumulation rates will continue at contemporary levels until 2070, and without changes in watershed management nearly 90% of playas would fill with sediment by 2100 (irrespective of watershed condition).

In addition to sediment accumulation, transport of nutrients, contaminants, and pollutants have been detected in run-off from anthropogenically-altered playa watersheds. Thurman et al. (2000) reported that cotton and corn herbicides were detected in 97% of water samples from playas in the Texas Southern High Plains. Belden et al. (2012) found common individual herbicides (e.g., atrazine, metolachlor, pendimethalin, trifluralin) in sediments of ~10 – 45% of playas in the High Plains and Anderson et al. (2013) concluded that pesticide concentrations in playas may periodically be high enough to adversely affect aquatic organisms. Negative effects of heavy metals have been reviewed for amphibians (Freda 1991), birds (Scheuhammer 1987), and aquatic macro-invertebrates (Goodyear and McNeill 1999). However, levels and ecological roles of heavy metals in playa wetlands are unclear, but assumed to be accumulating in playa sediments and potentially affecting the ecology of individual playas (Venne et al. 2006). Transport of metals, nutrients, and other contaminants into playas is likely facilitated by sediment loads (Belden et al. 2013).

One method to reduce the impacts of altered watershed condition upon adjacent playas is to provide a surrounding buffer of vegetation (Haukos 1994, 1995). Wetland buffers reduce adverse impacts of adjacent land uses by preventing or reducing sediments, contaminants, and excess nutrients (e.g., fertilizer or manure) from reaching the wetland, while providing habitat for resident and migratory wildlife (Skagen et al. 2008). Factors hypothesized to affect playa buffer effectiveness include slope and size of the watershed, land use within the playa watershed, soil texture of the playa watershed, buffer vegetation species and structure, buffer width, buffer management and use, and frequency and intensity of rainfall events (Skagen et al. 2008).

Conservation groups have installed few playa buffers during the past 20 years, and U. S. Department of Agriculture Farm Bill Programs have served to incidentally (e.g., Conservation

Reserve Program; CRP) and intentionally (e.g., CRP Conservation Practice CP23a) buffer playas. Deliberately established buffers have primarily followed current recommendations for establishment of vegetative buffers around playas, including use of native plant species and minimum average width of 30-90 m (Smith 2003). Although buffers around playas may reduce excess nutrients and contaminants potentially reaching the wetland, they could negatively influence wetland flooding (Detenbeck et al. 2002). This is especially of concern with de-facto buffers that vary in species composition and width such as non-native plant species used in CRP that may intercept overland runoff water at a rate that prevents water from reaching the wetland (Cariveau et al. 2011, Bartuszevige et al. 2012, O'Connell et al. 2012). Lack of information on what criteria are necessary to balance contaminant removal and volume of water that flows into the wetland is hindering conservation efforts. Our objectives were to (1) determine the maximum potential removal of heavy metals, nutrients (e.g., N, P), and sediment in precipitation runoff by vegetation buffer strips surrounding playa wetlands, (2) estimate the most effective buffer width relative to removal of heavy metals, nutrients (e.g., N, P), and sediment in runoff, (3) evaluate the influence of factors other than width (e.g., slope, buffer type, percent cover in buffer, precipitation intensity) on the effectiveness of vegetation buffer strips, (4) measure the relationship between sediment load and concentrations of other contaminants in precipitation runoff, and (5) assess the relative impact of type of vegetation buffer on the volume of water entering playas.

2. Materials and Methods

2.1 Study area

Our study area was the Texas Southern High Plains. This is a semi-arid transitional region between the more mesic prairies to the northeast and the Chihuahuan Desert to the

southwest (Haukos and Smith 2004) and was historically dominated by short-grass prairie with interspersions of mixed-grass prairie in isolated relic areas of sandy soils (Wester 2007). Today, the landscape is dominated by agriculture supported by the pumping of the underlying Ogallala Aquifer for irrigation. Annual average rainfall varies from 33 cm in the western portion of the study area to 63 cm in the east. Precipitation events primarily occur from spring to early fall in the form of intensive and localized thunderstorms (Haukos and Smith 2004). Evaporation rates are high and can exceed 250 cm per year due to high summer temperatures and mild winters in conjunction with high winds (Bolen et al. 1989). The number of playas, based on historical identification of hydric soils, has been estimated to be approximately 19,400 within this region (Guthery et al. 1981) and the average playa size based on delineated historical hydric soil maps was 6.3 ha (Guthery and Bryant 1982).

To identify study sites, we initially focused on Playa Lakes Joint Venture and U.S. Fish and Wildlife Service Partners for Fish and Wildlife enhancement projects that included an established conservation native grass buffer component as part of a playa management plan. However, only three established buffer sites were present and accessible. Therefore, we randomly selected additional playas within or abutting to row-crop agriculture and that were surrounded by a vegetative buffer comprised of Conservation Reserve Program plantings (CRP; $n = 7$), native grassland (NG; $n = 11$, included the three conservation buffers), or unplanted fallow cover (i.e., cultivated at some point in the past, but no longer managed; FC; $n = 18$) (Johnson 2011). These buffer types comprise the majority of buffer vegetation composition and structure associated with playa wetlands.

2.2 Water collection

Precipitation runoff was collected using I-CHEM storm-water samplers and mounting kits (Forestry Suppliers Inc., Jackson, MS, USA). I-CHEM storm-water samplers collected up to 1 L of first-flush storm water. Once 1 L of sample was collected, a floating ball valve sealed off the sample collection port to prevent co-mingling of run-off water and loss of volatile analytes. Samplers were placed in each buffer flush with the soil surface beginning at the outer edge of the buffer (i.e., adjacent to cultivated field) and continued every 10 m up to 80 m with the final sampler installed adjacent to the playa floor for buffers >80 m. A combination of hand and power augers was used to create a hole slightly larger than the sampling device. To reduce contamination from excess soil from the excavated holes, sampler placement was offset by 2 m from the transect line and alternated sides of a direct line from the cultivated field to the playa floor. During placement, each sampler was covered to prevent soil contamination and lowered into the hole until it was flush with the soil surface. Once flush, displaced soil was used to back-fill the area between the sampler and the side of the hole, then excess soil was dispersed and the sampler cover was removed.

Precipitation event information was acquired through on-site rain gauges, West Texas Mesonet (<http://www.mesonet.ttu.edu/>), and information from landowners and local natural resource professionals. Once an event occurred that filled samplers, water was collected, associated vegetation measurements taken (see below), and, following cleaning, samplers were moved to the buffer(s) of a different playa(s). Within the same week that storm-water samplers filled, vegetation structure and composition of the buffer were measured. Vegetative basal cover and species composition were recorded between adjacent storm-water samplers using a point-frequency frame (Levy and Madden 1933) randomly placed approximately every 3.3 m along a transect from the cropland to playa edge (i.e., three estimates of cover between storm-water

samplers). Percent cover was calculated by dividing the number of pins that hit vegetation between adjacent samplers by the total number of pins. Nomenclature followed Barkley (1986); however, we recognized the change of *Buchloe dactyloides* to *Bouteloua dactyloides* (Columbus 1999). Watershed slope surrounding each sampled playa was indexed by the number of 1.5-m contours defining a specific playa basin as reported by Fish et al. (1998).

For first use, no decontamination of I-CHEM storm water samplers was necessary. Following the initial sample collection, mounting kits were washed with deionized water and exposed to an ultrasonic bath treatment for 3 min. Due to increased exposure to collected run-off, mounting kit screens and I-CHEM funnels were cleaned with an ultrasonic brush, soaked in a 10% nitric acid bath for 12 hrs, thoroughly rinsed with deionized water, and oven-dried. I-CHEM funnels were treated in the same manner. Nalgene wide-mouth 1-L bottles were used for sample collection. Each bottle was used only once and prepared through a process of rinsing three times with deionized water, then rinsing with 10% nitric acid for 20 seconds followed by seven rinses of deionized water. Decontamination protocols were verified effective by testing samples on an Inductively Coupled Plasma (ICP) instrument.

2.3 Water analyses

We used the protocols recommended by the U.S. Environmental Protection Agency for sample preservation and water analyses (Keith 1996). Immediately following collection, one 250 ml subsample per flooded I-CHEM sampler was lowered to a $\text{pH} < 2$ by the addition of nitric acid for subsequent metals analyses. Containers for water used in the metal analysis were acid-cleaned before use and remained capped until employed for sampling. Samples were stored on ice during transport. For long-term storage, samples were placed in a 4 C refrigerator.

To analyze for total metals, samples were digested (EPA method 3015), centrifuged, and decanted directly into sample tubes for analysis using Inductively Coupled Plasma-Atomic Emission Spectrophotometer (ICP). Element concentration measured (ppm) with ICP included total phosphorus and a suite of metals (i.e., Na, Mg, Al, K, Ca, V, Cr, Mn, Fe, Ni, Cu, Zn, As, Sr, Cd, and Ba) selected due to their level of toxicity, known occurrence within the study area (e.g., common in agrochemicals used in the Southern High Plains), or because of their inclusion in the standard material. The ICP calibration curve consisted of five standards ranging from 0.0 to 10 ppm (EPA method 6010; Venne et al. 2006). The Detection Limit (DL) was determined through the use of check standards and their reproducibility. If the percent variation was greater than 20% for any given standard concentration, then samples at or below that concentration were considered below the DL. To determine the quantitative lower limit, the DL was multiplied by 3. Samples outside of the calibration were diluted and retested. Final values for diluted samples were obtained through calculations that accounted for the dilution factor. The quantification limits after 3X increase and correction for dilution was <0.04 ppb for all heavy metals. To insure data quality, wash blanks, travel blanks, and refrigerator blanks were employed during water quality work.

Total suspended solids (TSS) (g/L) were measured by shaking each 1 L sample for 30 sec to ensure homogenization, passing an aliquot through a preweighed glass-fiber filter disc (Whatman 934-AH, 1.5 micron pore size), and dried in an oven at 103-105 C. The aliquot volume varied and was determined by the point at which a deposit remained on the filter or the 1-L sample was depleted. To evaluate the change in filter mass, each filter was weighed after drying at 103 – 105 C to a constant mass. The increase in the mass of the filter represented TSS (EPA method 160.2). In addition, nitrate (mg/L), as an index of N, was measured using a

spectrophotometer (EPA method 352.1) in 2008 or an equivalent method by ion chromatography (EPA method 300.0) in 2009. Total dissolved solids (TDS) (g/L) were measured in the laboratory using a LaMotte 2020-WE turbidity meter.

2.4 Hydrology

Runoff volume during inundation events was estimated using the curve number procedure (USDA-SCS 1972). The runoff curve number procedure was developed by USDA Natural Resources Conservation Service (NRCS) and is a widely used, empirically developed parameter for estimating runoff from rainfall. The effects of soil type, land use, and antecedent moisture conditions (AMC) are embodied in the procedure (Bedient et al. 2008). To calculate runoff volume with the runoff curve number method, the contributing watershed was delineated for each playa. The natural watershed boundary was delineated using 7.5-minute Digital Orthoquads with 1.5-m contour intervals (Ekanayake et al. 2009). Path lines were used to determine the natural watershed boundary for each playa by identifying high points surrounding each playa and drawing a single continuous path line creating a polygon connecting each of these locations along with additional path lines used to depict the directional flow of runoff.

To evaluate the effects of man-made structures on runoff volume, digital imagery was used to examine the structures found within each watershed. In this process, aerial and ground-level images were used to identify restrictive barriers and water-controlling structures, and determine whether or not the presence of these structures would cause a change in the contributing watershed area. Structures considered in this process included paved and unpaved roadways, open channels (barrow ditches), bridges, and culverts. Watershed boundaries were adjusted as necessary according to the effects of the structures present at each location. The

adjusted boundaries represented the current contributing watershed for each playa. Once delineated, polygons were created in ArcGIS and area (ha) estimates derived.

The USDA NRCS Web Soil Survey in combination with watershed delineations was used to estimate infiltration rates based on soils (<http://websoilsurvey.nrcs.usda.gov/app/HomePage.htm>). Selection of the runoff curve number (CN) was dependent on antecedent moisture conditions and cover type. Area associated with delineated cover types and land uses were determined through field observation, aerial photos, and area calculations using ArcGIS. Antecedent moisture conditions and storm-event precipitation totals were estimated through a combination of rain gauges and Next Generation Radar (NEXRAD) (http://www.srh.noaa.gov/rfcshare/precip_download.php). Following precipitation events, we imported precipitation information from NEXRAD into ArcGIS and overlaid precipitation estimates with the subset of playas fitted with storm water samplers. Average rainfall for a given playa watershed and storm event were estimated by averaging rainfall totals for four points in the surrounding watershed. Storm event intensity was estimated by dividing precipitation totals by the length of the storm as determined by NOAA radar data (www.ncdc.noaa.gov/oa/rada/radardata.html) and reported as centimeters per hour. Antecedent moisture condition was determined by downloading and evaluating precipitation amounts for the 5 days prior to the storm event of interest. The individual CN was extracted from the standard USDA-NRCS curve number table by using the AMC in combination with the types of cover. The composite CN for each watershed was calculated by weighting each curve number according to the proportion of the area within the watershed in each land use and soil type in combination with the AMC (proportion of area in land use in each soil class*proportion of the watershed in soil group=fraction of area) ($\sum \text{fraction of area} \times \text{individual CN} = \text{Composite CN}$).

Initially, estimates of runoff volume were conducted under the assumption of no buffer present ($Q = (P - I_a)^2 / (P - I_a + S)$ where Q =runoff, P =rainfall, I_a =initial abstraction ($0.2 \cdot S$), S =watershed storage in inches ($1000 - CN/10$)) (Bedient et al. 2008). To account for the impact of the buffer on runoff volume entering the playa, a second round of estimations was conducted where grassland buffers of each sampled playa were placed into the NRCS land-use description category based on buffer type. The percent change in Q between the estimations is reported as the percent change in runoff volume for each buffer type.

The actual volume of water in the playa was estimated and compared to the runoff volume estimate by the USDA-NRCS curve number method. Volume was estimated following Tsai et al. (2007) by using a cone to represent the playa and calculated as $V = (1/3\pi)(h_2r_2^2 - h_1r_1^2)$ where V = volume, h_2 = height at water surface to the base of the cone, r_2 = radius at the water surface, r_1 = radius of playa, and h_1 = height of cone from playa floor to the base of the cone. Playas were assumed to be circular and the radii of the playas were estimated using area estimates from the *Playa Lakes Digital Database for the Texas Portion of the Playa Lakes Joint Venture Region* (PLDD) (Fish et al. 1998) and the formula $r = \sqrt{\text{area} / \pi}$. Radius at the water surface (r_2) was calculated as radius + maximum estimated depth/slope. Slope (rise/run or elevation center-average elevation of 8 points/ Σ distance in meters from actual to visual and visual to playa floor) of the cone was estimated from field data collected on randomly selected playas ($n = 147$) in 2008 and 2009. The visual edge of the playa was determined based on change of slope, vegetation, and soil (Luo et al. 1997). Actual playa edge was defined as the hydric soil-defined boundary, where the soil changed from hydric soil to upland soil type. Actual edge was determined at two locations opposite each other on the playa by taking sediment cores along a transect perpendicular to the visual edge (i.e., change in slope and vegetation) (Luo et al. 1997).

Distances from the actual edge to the visual edge and visual edge to the playa floor were measured using an open-reel fiberglass tape. Measurements were summed and utilized as the run. The rise was obtained by using a level set located in the center of the playa to measure the change in elevation of the playa edge. The elevation change between the playa floor and each of the equidistant points ($n = 8$) along the visual edge of the playa was measured using the one-station method (Luo et al. 1997). The average of these points was used as the change in elevation or the rise. If playas lacked a discernable visual boundary, soil maps and aerial surveys were employed to estimate the historical boundary of the playa prior to soil sampling. The height bottom (h_1) was calculated as $\text{radius} \times \text{slope}$ and height at water surface (h_2) as $\text{height bottom} + \text{maximum estimated depth}$.

Maximum water depth was estimated by taking depth measurements in seven locations across the playa floor using a standard meter stick. Measurement locations were determined by placing one point in the center of the playa and one point on each of two edges. In addition, a depth measurement was taken 10 m from the water edge and between this measurement and the center point. Maximum estimated depth was a combination of the maximum water depth measured and evaporation ($\text{maximum estimated depth} = \text{observed depth} + \text{evaporation}$). Evaporation rates used were obtained from the National Weather Service and the Texas Water Development Board (<http://midgewater.twdb.state.tx.us/Evaporation/evap.html>). This site provided evaporation rates for each one-degree quadrangle in Texas. The quadrangle of interest was located and the monthly gross surface evaporation data was divided by the number of days in the given month to estimate a daily evaporation rate. Evaporation rates were multiplied by the number of days from the rain event to the day water measurements were taken.

2.5 Statistical Analyses

Total mean percent vegetation cover was derived from the between storm-water samplers pin-frame samples to assess differences in vegetation cover among buffer types (NG, CRP, FC). A completely random design analysis of variance (ANOVA) was used to compare vegetation cover among buffer types with $\alpha = 0.05$. Following a statistically significant *F*-test, a least significant difference test was used to compare mean vegetation cover between buffer types. The percent maximum removal of metals, nutrients, and sediment (hereafter contaminants) for each buffered playa was estimated by identifying the highest contaminant level in runoff samples regardless of location within the buffer, subtracting the lowest contaminant level found in subsequent samplers within the buffer for each playa, and then dividing by the highest value. An ANOVA was used to compare percent maximum removal among buffer types for each contaminant. Overall differences in average concentration among buffer distances were determined using a nonparametric Kruskal-Wallis (K-W) test for each contaminant with distance as the dependent variables. Following a statistically significant KW-test ($\alpha = 0.05$), a nonparametric pair-wise comparison test was used to compare concentration levels between sampler distances. Multiple linear regression was used to assess the relative influence of vegetation cover, slope, and precipitation intensity on maximum percent removal for each water quality variable. Marginal coefficients were tested for a difference from 0 using a *t* distribution. Pearson correlations between Total Suspended Solids and Total Dissolved Solids and other water variables (i.e., Na, Mg, Al, K, Ca, V, Cr, Mn, Fe, Ni, Cu, Zn, As, Sr, Cd, and Ba) were calculated to assess the influence of sediment on the level of contaminants and overall water quality in playas. STATISTIX ® 8 (Analytical Software, Tallahassee, Florida, USA) was used for all statistical analyses.

3. Results

3.1 Buffer characteristics

A total of 238 water samples were collected and analyzed from 36 playa buffers during 2008 and 2009. Average playa size ($\pm$ SE) was 17.22 ± 1.88 ha. Totals for rain events large enough to allow for filling of samplers ranged from 1.9 – 15.2 cm, with a mean ($\pm$ SE) of 6.4 ± 0.6 cm. Intensity of the rain events ranged from 0.3 – 20.5 cm per hour, with a mean ($\pm$ SE) of 2.5 ± 0.11 cm per hour. Buffer width varied from 10 to 250 m. However, 60% of the sampled playas had buffers ≤ 60 m and none of the remainder had greater than 3 common sampling distances >60 m; therefore, we restricted our analyses to a buffer distance to a maximum of 60 m from the edge of cultivation for all sampled playas, which was greater than the current recommended minimum buffer width of 30-40 m (Skagen et al. 2008). Residual vegetation (dormant or dead vegetation from the previous growing season) and bare ground were principal components in all buffer types (Table 1). Vegetation composition varied among buffer types (Table 1). Extant vegetation in Fallow Cropland (FC) was dominated by *Kochia scoparia*, *Agropyron smithii*, and *Sorghum halepense*. However, only 46% of vegetation species in FC were native species. The CRP buffer type was dominated by *Bouteloua gracilis*, *B. curtipendula*, *Bothriochloa laguroides*, and *Sorghum halepense*; 82% of the recorded species were native. However, unknown individuals comprised 20% of the vegetation in CRP buffer due to lack of distinguishing characteristics. Native Grass (NG) buffers were dominated by *Bouteloua dactyloides* and *B. gracilis*; 98% of recorded species were native (Table 1). Vegetation cover differed ($F_{2,35} = 3.53$, $P = 0.04$) among buffer types, with FC having 13% greater vegetation cover compared to NG buffer type (Figure 1).

3.2 Changes in water quality

There were no differences in maximum percent removal for any water quality variable among buffer types (Table 2). Therefore, we combined buffer types for subsequent analyses. Within the initial 60 m from a cultivated field, vegetation buffers of all types removed $>50\%$ of all measured

contaminants, including 83% of total suspended solids but only 58% of total dissolved solids (Figure 2). Buffers removed an average of 70% of P and 78% of N to reduce nutrients entering the playa (Figure 2). There was a wide range of mean maximum percent removal for metals ranging from 56% of Na to 87% of Cr. Maximum percent removal of contaminants was not influenced by precipitation intensity ($P = 0.29 - 0.93$; $\bar{X} = 1.06$, $SE = 0.32$, range = 0.11 – 8.08), percent cover ($P = 0.06 - 0.83$; $\bar{X} = 85.9$, $SE = 2.33$, range = 47.2 – 100), or an index to slope ($P = 0.12 - 0.98$; $\bar{X} = 1.33$, $SE = 0.13$, range = 0 – 3).

Average concentrations of contaminants as a function of buffer distance differed among sampler distances from a cultivated field for 17 of 18 contaminants and followed a similar pattern for nearly all water quality variables (Figure 3). In general, concentration levels were greatest at the buffer-cultivation edge (distance 0), then there was a relatively large decrease in concentrations between buffer distances 0 and 10 m, with continuing declines at distances 10 and 20 m from the cultivation edge to statistically consistent concentrations from distances 20 to 60 m, albeit steepness of the decline in concentration depended upon the water quality variable (Figure 3). Vanadium (V) was the lone exception to this pattern with no differences in concentrations among distances from a cultivated field. The lowest average concentration (i.e., distance of greatest percent removal) of contaminants was 50 m (66.7% of measured variables; Al, As, Ca, Cr, Fe, K, Mg, Mn, Ni, N, TDS, TSS), with 27.8% of variables having lowest concentrations at 60 m (Ba, Cu, Na, P, and Zn) and V with the lowest concentration at 30 m.

Measures of TSS were correlated with all measures of metals and nutrients except for N (Table 3). Metals such as Al, As, Ba, Fe, K, Mg, and Mn had correlation coefficients >0.80 . The magnitude of correlations between measures of metals and nutrients were much lower for TDS (Table 3). However, both N and P had increased correlation values with TDS rather than TSS, with

an increase from $r = 0.03$ to 0.54 for N; the correlation of P between TDS ($r = 0.46$) and TSS ($r = 0.49$) was only slightly greater.

3.3 Buffer influence on inundation

Eleven of 36 playas (CRP = 1, FA = 4, NG = 6) received sufficient precipitation runoff to measurably flood the wetland to water levels sufficient to assess the impact of different buffer types on hydrology. Playa area ($\pm$ SE) for the 11 playas was 18.38 ± 3.98 ha and ranged from 8.76 – 45.23 ha. Playa watershed area ($\pm$ SE) was 244.25 ± 36.77 ha and ranged in size from 117.36 to 402.67 ha, whereas the buffer area ($\pm$ SE) was 18.59 ± 5.29 ha and ranged from 4.90 – 52.16 ha with an average width of 51.2 m (SE = 7.05). Rain event totals ranged from 4.06 – 22.48 cm, with a mean ($\pm$ SE) of 11.61 ± 2.01 cm. Intensity of the rain events ranged from 0.31 – 1.58 cm per hour, with a mean of 0.51 ± 0.15 cm per hour. Three of the 11 playas were dropped from analysis due to the inability to accurately include the impacts of watershed, account for water being in the playa prior to the event of interest, or major playa manipulations that prevented accurate measurements. The one CRP playa that received a rainfall event large enough to flood the wetland experienced an estimated 5.71% reduction in runoff volume due to presence of a 50 m buffer. Grassland buffers reduced runoff on average by $0.76\% \pm 0.26$. At the scale of this study, fallow crop buffers did not affect estimated runoff volume (Table 4).

4. Discussion

Physical, structural, and functional losses of playas due to excess sediment and other contaminants are occurring at a rate greater than the meager conservation efforts can counteract them in the Southern High Plains (Johnson et al. 2012). Conservation groups have only been prescribing and implementing buffers occasionally in the Southern High Plains and their efforts have been inadequate. Indeed, only three were available for our study. However, when buffers

are implemented, the primary concern is the balance between maximizing contaminant removal and minimizing negative impacts to hydrology.

Buffer conditions assessed in this study (up to 60 m in width, >80% horizontal cover) successfully removed greater than 50 – 87% of sediment, nutrients, and metals. Maximum percent removal was not influenced by vegetation composition, amount of cover in the buffer, precipitation intensity, or an index of slope. The lack of variation within percent cover and index to slope likely contributed to this finding. A minimum level of precipitation intensity is necessary to create sufficient runoff to inundate a playa so the effect of this factor is limited to values greater than that necessary to produce sufficient runoff for water to be collected throughout the buffer. Shallow slopes or the index measure using elevation contours may not allow for us to discern the effect of this variable. Watershed slopes in this landscape are typically gentle with little change in elevation from the edge of the watershed to the playa. Only two playas had three 1.5-m interval depression contour lines and 55% of sampled playas had only one or zero 1.5-m interval depression contour lines. Bartuszevige et al. (2012), using mean slope of the watershed measured in degrees, reported that increasing watershed slope contributed, with other variables, to increasing probability of playa inundation. Vegetation cover was lowest for native grass buffer types, but maximum removal of contaminants did not differ among buffer types, indicating that any buffer with >80% cover should function similar to those in our study and vegetation composition was not influential in buffer effectiveness.

Inherently, many of the land use changes since the 1880s have increased overland flow into playa wetlands, such as the conversion of native vegetation to row crop agriculture, and inadequately managed grazing practices. Still others have decreased overland flow, such as the planting of non-native grass in the Conservation Reserve Program (Detenbeck et al. 2002, Cariveau et al. 2011, O'Connell et al. 2012). These watershed-level influences on hydrology

should be considered alongside buffer-level influences during the design of buffers especially because only one CRP-buffered playa became inundated.

Buffer effectiveness is related to multiple complex interacting factors. However, due to the lack of field studies, deficiencies in testing of hydrology and buffer efficiency model applicability to playa wetlands, and difficulty in the use of currently available models (e.g., Vegetative Filter Strip Model, Abu-Zreig 2001; Chemical Runoff and Erosion from Agricultural Management Systems, Knisel 1980), at present, buffer recommendations are currently based on watershed condition, political acceptability, and best professional judgment (Castelle et al. 1994, Skagen et al. 2008). Castelle et al. (1994) showed that appropriate buffer size for aquatic resources was in part determined by the contaminant(s) the buffer was used to remove. Average concentration of contaminants in precipitation runoff into playa wetlands was influenced by buffer width. For most metals, concentrations at 20 – 60 m were comparable; thus, any buffer width >20 m should maximize percent removal of metals in precipitation runoff. Removal of nutrients (N, P) had different patterns where N had continuously declining concentrations to 50 m; however, statistically, there were no differences in concentration level between 20 and 50 m due to relatively large variation among playas at shorter buffer distances. Concentrations of P remained relatively constant from 20 – 60 m. Similar patterns were found for TDS and TSS where concentrations were statistically similar from 20 – 60 m. These results indicate the importance of the first 20 m of buffers surrounding playa wetlands in removal of contaminants from precipitation runoff. Although the distance of lowest concentration varied among water quality variables, in general, a buffer distance of 40 – 50 m resulted in the lowest concentration of contaminants. Therefore, any buffer ≥ 40 m should effectively maximize percent removal of contaminants in precipitation runoff.

Vanadium was the only element that did not show decreasing concentrations based on buffer distance. One potential reason for this lack pattern is the instability of vanadium complexes in

environmental samples, which can be influenced in aqueous solutions by pH, salinity, and presence of other elements (Chen and Owens 2008). In addition, there remains inconsistent recommendations for pretreatment of water samples intended for analyses of vanadium (Chen and Owens 2008).

Balancing overland water flow with infiltration and contaminant removal rates is an important consideration in development of wetland conservation practices. Although limited to one playa, reduction in runoff volume was greatest for the vegetative buffer comprised of CRP cover (5.7% reduction). Vegetative buffers comprised predominantly of native species reduced overland flow volume by 0.76 ± 0.26 percent. The FC buffers in this study comprised <5% of the watershed, and differences in volume of precipitation runoff at this scale were not detectable using the NRCS curve number method. Thus, an influence of FC buffers on overland flow reduction was not observed in this study. These findings are based on buffer widths ≤ 60 m for playas in cultivated watersheds. Overland flow of precipitation runoff can be negatively affected when watersheds are comprised of cover types other than cropland. O'Connell et al. (2012) reported that playas with CRP watersheds were inundated 56% less than other watershed types. Bartuszevige et al. (2012) found that CRP at local (i.e., adjacent, within 100 m) scale reduced the odds of playa inundation by 18%. Whereas, CRP at the watershed scale reduced the probability of playa inundation in a declining linear relationship with increase proportion of CRP in the watershed. Cariveau et al. (2011) indicated that playas with native grassland watersheds had a probability of inundation three times greater than playas with CRP watersheds.

In addition to buffer distances of 40 to 60 m, native vegetation would provide additional ecological benefits in addition to removal contaminants from precipitation runoff, as these species have a high probability of germination, minimally impact hydrology and subsequent hydroperiod, and provide habitat for native grassland wildlife. Studies on the usefulness of buffers as wildlife habitat demonstrate reduced value to wildlife when non-native plantings are

used (Melcher and Skagen 2005). This reduced value is often a product of different composition and structure than what is on the landscape in a native state. The planting and establishment of native vegetation as a buffer zone for playa wetlands should create a level of cover comparable to that of unplowed native grassland, creating higher-quality habitat for grassland species.

The correlation between watershed land use and structure, function, and ecology of playas is well-established. Increasing the area of native grassland surrounding playas reduces susceptibility to negative outside influences (e.g., accelerated sedimentation). Although significant amounts of contaminants in precipitation runoff were removed by vegetative buffers, upland watershed management (e.g., no-till agriculture, terraces, grassed waterways, focused application of pesticides and nutrients) would likely further minimize the concentrations of contaminants entering the playa buffer. To monitor water quality of precipitation runoff entering playas, monitoring of total suspended solids would be indicative of metal and P loads given the high correlation with these elements. Additionally, monitoring total dissolved solids would also be suggestive of N because of the greater correlation with N than total suspended solids, which was also reported by Mihara and Ueno (2000). Increasing transfer of N and P via total suspended solids and total dissolved solids due to soil loss has been shown in several studies (e.g., Jordan et al. 1997, Ekholm et al. 2000, Mihara and Ueno 2000). Thus, sediment in precipitation runoff is the likely primary contributor of N and P into playas. Therefore, ideally landowners would manage the entire watershed and would include vegetative buffers as a management tool.

Wetland conservation could be enhanced by direct interactions between landowners and managers to develop a holistic management approach for playa wetlands at the watershed and hydric-soil scales. Given the alarming rate of playa loss and degradation (Johnson et al. 2012),

conservation organizations could reduce such rates by initiating and accelerating efforts in the High Plains to work with landowners to install buffers around playas. Additional methods for reducing the input of sediments, nutrients, and contaminants to playa wetlands include: conversion from irrigation to dryland farming, adoption of practices that require less disturbance of soil or impedes overland water flow (e.g., no tillage, contour tilling, or terracing), fine-tuning fertilizer and pesticide application quantities and application techniques (e.g., establishment of no-application zones around wetlands, avoiding application during high winds or directly before heavy precipitation events) (Skagen et al. 2008).

The impact of accelerated sediment accumulation and its associated disturbances on the structure and function of playas due to altered hydrology should be primary concerns of wetland managers in this semi-arid region. The correlation of sediment with contaminant transport reinforces the need for protection of playas in relation to excess sediment loading and this study has provided necessary field- and quantitative-based recommendations for buffers, the primary conservation measure used for the protection of playa wetlands and the biodiversity of the High Plains.

4.1 Conclusion

In summary, playa wetlands are being subjected to increased concentrations of nutrients, metals, and sediment via runoff during precipitation events. The use of 40-60 m wide vegetation buffer strips around playas significantly reduces nutrient, nearly all metal, and sediment loads in runoff that eventually inundates playas. Vegetation composition of the buffer strip has little effect on contaminant removal; however, undisturbed, dense vegetation such as found in unmanaged CRP buffers may intercept and reduce the volume of water reaching a playa wetland. Monitoring TDS and TSS will index the amount of metals and nutrients entering a playa due to

high correlations among these variables. Finally, watershed management would limit the amount of contaminants available to be transported by runoff because buffer strips do not remove 100% of contaminants present in precipitation runoff.

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Table 1. Species composition, number of pin hits, and percent of hits for a given plant species. Data were recorded between samplers and within each vegetative buffer type (Unplanted Fallow $n = 18$, Conservation Reserve Program [CRP] $n = 7$, Native Grassland $n = 11$), using the point-frame method during 2008 and 2009 in the Southern Great Plains of Texas.

Species	Landcover					
	Unplanted Fallow		CRP		Native Grassland	
	#	%	#	%	#	%
<i>Agropyron smithii</i>	103	6.44	16	0.91	75	1.77
<i>Amaranthus</i> sp.	41	2.56	12	0.69	14	0.33
<i>Ambrosia grayi</i>	49	3.06	14	0.80	4	0.09
<i>Ambrosia psilostachya</i>	0	0.00	0	0.00	1	0.02
<i>Aristida purpurea</i>	0	0.00	0	0.00	12	0.28
<i>Artemisia frigida</i>	0	0.00	0	0.00	6	0.14
<i>Aster subulatus</i>	1	0.06	0	0.00	0	0.00
<i>Bothriochloa laguroides</i>	0	0.00	64	3.66	0	0.00
<i>Bouteloua curtipendula</i>	0	0.00	77	4.40	0	0.00
<i>Bouteloua gracilis</i>	0	0.00	103	5.89	825	19.46
<i>Bromus unioloides</i>	20	1.25	0	0.00	6	0.14
<i>Buchloe dactyloides</i>	18	1.13	28	1.60	1086	25.61
<i>Chenopodium album</i>	6	0.38	0	0.00	5	0.12
<i>Chenopodium leptophyllum</i>	2	0.13	0	0.00	3	0.07
<i>Convolvulus equitans</i>	28	1.75	1	0.06		0.00
<i>Coreopsis tinctoria</i>	0	0.00	0	0.00	8	0.19
<i>Cynodon dactylon</i>	52	3.25	0	0.00	0	0.00
<i>Echinochloa crusgalli</i>	0	0.00	0	0.00	17	0.40
<i>Eleocharis</i> sp.	2	0.13	0	0.00	0	0.00

<i>Elymus canadensis</i>	0	0.00	0	0.00	3	0.07
<i>Erodium texanum</i>	0	0.00	0	0.00	4	0.09
<i>Oenothera canescens</i>	2	0.13	0	0.00	2	0.05
<i>Euphorbia marginata</i>	0	0.00	2	0.11	0	0.00
<i>Grindelia squarrosa</i>	21	1.31	0	0.00	4	0.09
<i>Helianthus annuus</i>	0	0.00	0	0.00	1	0.02
<i>Helianthus ciliaris</i>	46	2.88	21	1.20	48	1.13
<i>Hoffmannseggia glauca</i>	6	0.38	2	0.11	0	0.00
<i>Hordeum pusillum</i>	2	0.13	0	0.00	11	0.26
<i>Kochia scoparia</i>	257	16.06	5	0.29	8	0.19
<i>Lactuca serriola</i>	2	0.13	0	0.00	0	0.00
<i>Lepidium densiflorum</i>	0	0.00	0	0.00	2	0.05
<i>Leptochloa fascicularis</i>	0	0.00	27	1.54	0	0.00
<i>Lippia nodiflora</i>	17	1.06	2	0.11	18	0.42
<i>Malvella leprosa</i>	10	0.63	10	0.57	1	0.02
<i>Panicum capillare</i>	0	0.00	0	0.00	3	0.07
<i>Panicum obtusum</i>	0	0.00	0	0.00	31	0.73
<i>Paspalum paspaloides</i>	2	0.13	0	0.00	6	0.14
<i>Plantago patagonica</i>	0	0.00	0	0.00	1	0.02
<i>Polygonum pensylvanicum</i>	1	0.06	0	0.00	0	0.00
<i>Portulaca oleracea</i>	2	0.13	0	0.00	1	0.02
<i>Psoralidium tenuiflorum</i>	0	0.00	0	0.00	3	0.07
<i>Rorippa sinuate</i>	2	0.13	0	0.00	1	0.02
<i>Rumex altissimus</i>	3	0.19	0	0.00	0	0.00
<i>Rumex crispus</i>	1	0.06	0	0.00	0	0.00
<i>Salsola iberica</i>	0	0.00	3	0.17	0	0.00

<i>Schedonnadrus paniculatus</i>	0	0.00	0	0.00	3	0.07
<i>Setaria vulpiseta</i>	1	0.06	0	0.00	0	0.00
<i>Solanum elaeagnifolium</i>	0	0.00	2	0.11	3	0.07
<i>Sorghum halepense</i>	59	3.69	75	4.29	0	0.00
<i>Sphaeralcea coccinea</i>	0	0.00	0	0.00	8	0.19
<i>Verbena bracteata</i>	2	0.13	0	0.00	7	0.17
<i>Vernonia marginata</i>	0	0.00	0	0.00	1	0.02
<i>Xanthium strumarium</i>	0	0.00	3	0.17	0	0.00
Wheat	0	0.00	4	0.23	8	0.19
Bare soil	182	11.38	139	7.94	787	18.56
Residual vegetation	576	36.00	764	43.66	1086	25.61
Unknown individuals	61	3.81	354	20.23	116	2.74
Water	23	1.44	22	1.26	11	0.26
TOTAL	1600		1750		4240	

Table 2. Maximum average ($\pm$ SE) levels of percent contaminant removal or water quality change as a function of buffer type (Conservation Reserve Program [CRP] $n = 7$, Native Grassland $n = 18$, and Unplanted Fallow $n = 11$) in vegetative buffers surrounding 36 playa wetlands during 2008 and 2009 in the Southern Great Plains of Texas.

Water Measurements ^a	Buffer Type			F^b	P
	CRP	Fallow	Native Grassland		
TSS	85.43 (6.16)	79.76 (4.91)	83.44 (3.84)	0.30	0.75
TDS	57.53 (8.29)	57.62 (6.61)	58.85 (5.17)	0.02	0.98
Al	69.71 (8.14)	74.11 (6.65)	77.59 (5.54)	3.82	0.15
As	81.31 (8.81)	84.24 (7.20)	74.50 (5.99)	0.58	0.57
Ba	63.73 (8.47)	69.93 (6.92)	79.79 (5.75)	1.39	0.27
Ca	58.55 (9.86)	62.70 (8.05)	67.17 (6.70)	0.28	0.76
Cr	98.93 (11.21)	71.54 (9.15)	92.94 (7.62)	2.29	0.12
Cu	68.65 (8.51)	64.35 (6.95)	82.67 (5.78)	2.28	0.12
Fe	71.61 (7.62)	74.93 (6.22)	81.83 (5.18)	0.73	0.49
K	64.25 (7.81)	60.92 (6.38)	66.89 (5.31)	0.26	0.77
Mg	72.97 (8.00)	68.56 (6.53)	69.93 (5.44)	0.09	0.91
Mn	72.45 (7.54)	74.81 (6.12)	83.64 (5.12)	1.00	0.38
N	85.65 (10.45)	77.96 (8.34)	76.46 (6.52)	0.28	0.75
Na	58.63 (9.51)	57.38 (7.77)	54.66 (6.46)	0.07	0.93
P	72.04 (8.69)	59.43 (7.09)	76.13 (5.90)	2.13	0.34
Sr	50.01 (9.97)	65.78 (8.14)	67.21 (6.77)	1.10	0.35
V	89.95 (10.11)	77.81 (8.25)	82.30 (6.87)	0.43	0.62
Zn	60.60 (7.67)	65.64 (6.26)	76.69 (5.21)	1.81	0.18

^a TDS = Total Dissolved Solids, TSS = Total Suspended Solids

^b F -test degrees of freedom was 2, 35 for N, TDS, TSS; 2, 27 for Na, P, Sr, V, Zn,

Table 3. Correlations (r) between Total Suspended Solids (TSS) and Total Dissolved Solids (TDS) and other water characteristics (nutrients [N, P] and metal concentrations) in storm water samples ($n = 136$) in vegetated buffer areas surrounding ($n = 36$) playa wetlands in the Southern Great Plains of Texas during 2008 and 2009.

		Water Measurements																	
		Al	As	Ba	Ca	Cd	Cr	Cu	Fe	K	Mg	Mn	N	Na	Ni	P	Sr	V	Zn
TSS																			
	r^a	0.85	0.82	0.83	0.31	0.41	0.55	0.23	0.85	0.87	0.85	0.83	0.03	0.58	0.22	0.46	0.41	0.68	0.24
TDS																			
	r	0.19	0.23	0.20	0.23	0.08	0.08	0.00	0.18	0.29	0.15	0.13	0.54	0.01	0.00	0.49	0.32	0.04	0.03

^aSignificant correlation coefficients are indicated in bold ($\alpha = 0.05$).

Table 4. Estimated percent reduction in runoff reaching playa wetlands using the NRCS curve number (CN) method and as a function of vegetative buffer type surrounding 11 playas (Conservation Reserve Program [CRP] $n = 1$, Native Grassland $n = 6$, and Unplanted Fallow $n = 4$) during 2008 and 2009 in the Southern Great Plains of Texas.

Buffer Type	Watershed area (ha)	Buffer area (ha)	Precipitation (cm)	No Buffer		Buffer		Percent Reduced Runoff	Estimated Maximum Stored Volume (m ³)
				Composite CN	Estimated Runoff (m ³)	Composite CN	Estimated runoff (m ³)		
CRP	263.45	52.16	14.73	86	282508	84	266376	5.71	285979
Fallow	117.36	5.79	9.53	94	91870	94	91870	0.00	68156
Fallow	189.39	4.90	13.59	92	212063	92	212063	0.00	151587
Fallow	270.33	6.43	6.35	82	69718	82	69720	0.00	48456
Fallow	118.57	8.30	13.59	87	118267	87	118267	0.00	154018
Native Grass	137.59	23.97	15.57	88	166843	88	165488	0.81	153169
Native Grass	386.88	29.02	22.48	69	503388	69	501186	0.44	596826
Native Grass	402.67	26.87	4.06	83	44031	83	43686	0.78	55964
Native Grass	312.02	9.87	4.60	90	75601	90	75308	0.39	51082

Figure Captions

Figure 1. Average percent ($\pm$ SE) vegetation cover among three buffer types Native Grass (NG), Conservation Reserve Program (CRP), and Unplanted Fallow Cover (FC) for 36 playa wetlands during 2008 and 2009 in the Southern Great Plains of Texas. Means with the same upper case letter are not different.

Figure 2. Average maximum ($\pm$ SE) levels of percent contaminant (total suspended solids [TSS g/L], total dissolved solids [TDS g/L]), nutrients (N, P ppm) or metals (ppm) removal from precipitation runoff by vegetative buffers surrounding 36 playa wetlands during 2008 and 2009 in the Southern Great Plains of Texas.

Figure 3. Average ($\pm$ SE) concentration of contaminant (total suspended solids [TSS g/L], total dissolved solids [TDS g/L]), nutrients (N, P ppm) or metals (ppm) in precipitation runoff sampled in vegetation buffer surrounding 36 playa wetlands as a function of increasing distance from cultivation within the watershed (distance 0 – 60 m from buffer-cultivation edge) during 2008 and 2009 in the Southern Great Plains of Texas. A nonparametric Kruskal-Wallis (K-W) test was used to test for overall differences in mean contamination level among buffer distances at 10 m intervals from cultivation edge. Distances with the same letter do not differ in mean contaminant level.

Figure 1

Figure 2

Figure 3a

Figure 3b

Figure 3c