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Research article

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 filling of the wetland through 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.

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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

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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 et al., 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 (2013) 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 macroinvertebrates (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., 2012).

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 interspersion 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