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Using the electromagnetic induction survey method to examine the depth to clay soil layer (Bt horizon) in playa wetlands

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Abstract

Purpose Sediment accumulation has been and continues to be a significant threat to the integrity of the playa wetland ecosystem. The purpose of this study was to determine the vertical depth to the clay soil layer (Bt horizon) and thus to calculate the thickness of sediments accumulated in playa wetlands.

Materials and methods This study used the electromagnetic induction (EMI) survey method, specifically EM38-MK2 equipment, to measure the vertical depth to the clay soil layer at the publicly managed wetlands in the Rainwater Basin, Nebraska, USA.

Results and discussion The results indicated that the depth to the clay soil layer ranges from 21 to 78 cm ($n = 279$) with a mean sediment thickness of 39 cm. The annual sediment deposition rate since human settlement in the 1860s was calculated to be 0.26 cm year^{-1}. The results provided science-based data to support future wetland restoration planning and the development of decision support tools that prioritize conservation delivery efforts.

Conclusions Our research confirmed that the EMI technique is effective and efficient at determining the depth to the Bt horizon for playa wetlands. Additionally, these results supported previous studies and continue to indicate that a large amount of sediment has accrued in these playa wetlands within the Rainwater Basin area since settlement. Wetland restoration ecologists can use this information to prioritize future wetland restoration work that intends to remove culturally accumulated sediments above the clay soil layer. These findings provided a contemporary summary of wetland soil profile information that is typically used to develop restoration plans. This research also filled the critical knowledge gap about the thickness of the upper soils and the depth to Bt in publicly managed wetlands.

Keywords Bt horizon · Electromagnetic induction (EMI) survey · Playa wetlands · Sediment

1 Introduction

1.1 Importance of sediment issues for playa wetlands

Playa wetlands are nearly circular, shallow depressions located in semi-arid areas and are predominately formed by aeolian processes (Smith 2003). Playas are found in the lowest areas within closed watersheds and have a well-defined clay soil layer (Bt horizon, also called the clay pan) just beneath the wetland surface (LaGrange et al. 2011; NRCS 2015). This Bt layer impedes downward water movement and supports ponded conditions within these hydric soil footprints when adequate runoff occurs. Soils within the upper profile have more organic matter and larger particle sizes (NRCS 2015) that increase water storage capacity within these layers rather than facilitating surface water ponding. Therefore, the depth to Bt or the thickness of the soil layers above the Bt can significantly affect wetland function, integrity, and sustainability for playa wetlands (Tang et al. 2015a).

The Rainwater Basin region is located in the narrow corridor of the Central Flyway and plays a significant role as stopover habitat for waterfowl and shorebirds during spring migration (Gersib et al. 1992; Brennan et al. 2005). When ponding occurs, these wetlands provide abundant wetland-derived seed resources that support the North American Waterfowl Management Plan objectives (Drahota and Reichart 2015).

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Many of these playa wetlands have been protected as waterfowl production areas (WPAs) and wildlife management areas (WMAs) that provide habitats for millions of waterfowl and other waterbirds including the threatened piping plover (*Charadrius melodus*) and bald eagle (*Haliaeetus leucocephalus*), and the federally endangered whooping crane (*Grus americana*) (LaGrange 2005; USFWS 2007).

Sediment accumulation has been and continues to be a significant threat to the integrity of the playa wetland ecosystem (Haukos and Smith 2003; Tsai et al. 2010; Daniel et al. 2015; Tang et al. 2015a, b). LaGrange et al. (2011) highlighted the consequences of sediment accumulation on playa wetlands, specifically on hydrological function, water quality, plant community structure, and impacts to invertebrate and vertebrate populations. Sediment within any wetland footprint can increase the water storage capacity within the soil profile due to the interstitial pore space in the unconsolidated sediment, and therefore can reduce the frequency and duration of surface water expression (Tsai et al. 2010; Tang et al. 2015b). The surface runoff spates transport sediment from tilled uplands down slope to playa wetlands causing rapid accumulation compared with pre-settlement conditions when grassed uplands limited soil movement. Thus, the water storage capacity of playa wetlands has declined, and wetland function has been severely altered (Goldsborough and Crumpton 1998; O'Connell et al. 2013). As the accumulation of sediment increases in playa wetlands, the water storage capacity within the soils above the clay pan increases causing less frequent surface ponding, declines in water depth, and shorter hydroperiods (Tang et al. 2018). Although short-term gains in the ponded area may be noted after significant runoff events, the hydroperiod will be shorter and the overall ponded area will be smaller during average climate conditions given excessive sediment accumulation (Cariveau et al. 2011; Uden et al. 2015). Furthermore, the loss of water storage volume resulting from aggradation will not only provide favorable conditions for invasive plant species (Galatowitsch et al. 1999; Smith and Haukos 2002) but also make them more vulnerable to agricultural conversions (Tsai et al. 2010; Tang et al. 2015b). Since playa wetlands are extremely sensitive to sediment accumulation, the thickness of accumulated soils above the clay layer is an important measure used in wetland restoration planning. Therefore, this research focused on determining the thickness of soils accumulated above the Bt horizon on publicly owned playa wetlands. This research was designed to provide a scientific inventory of the current conditions that restoration ecologists can use to plan and prioritize wetland restoration decisions, particularly for sediment removal projects.

1.2 Culturally accelerated sediment on playa wetlands

Culturally accelerated sedimentation occurs when land management practices accelerate the deposition of soils into playa wetlands breaking the balance of input and output of sediment in the natural sedimentation process (LaGrange et al. 2011). The culturally accelerated sediment from the cropland portion of the watershed has become the primary input into playa wetlands (LaGrange et al. 2011). Figure 1 illustrates the juxtaposition of rapid soil accumulation above the clay soil layer (the Bt horizon) (modified from Schafersman 2011).

Before European settlement in the Great Plains, playa wetlands occurred in a landscape with minimal anthropogenic disturbances, and persisted through natural disturbances such as drought and deluge events, fire, and grazing. All of these disturbances facilitated some soil deposition events along with deflation events that frequently removed deposited soils. In the natural cycle of soil erosion, transport, and deposition into playas, the amount of sediment moving into wetlands would be offset by the amount of sediment moving out of playa wetlands via wind deflation. Therefore, the inputs and outputs of sediment, to some degree, were considered to be in balance (Gill 1996; Luo et al. 1997, 1999; Smith 2003; LaGrange et al. 2011).

However, from the mid to late 1800s, the European settlement of the central USA began to occur after the arrival of railroads to the Great Plains (LaGrange et al. 2011; Tang et al. 2015a). The Homestead Act passed by Congress in 1862 required claim holders to build a certain size dwelling, and improve the land by growing crops within 5 years of the homestead application (Potter and Schamel 1997). This facilitated rapid conversion of grassland into cropland in the late 1800s (Layton et al. 1927), and most of the landscape was converted to croplands in south-central Nebraska. These tilled lands were exposed to wind and water erosion for a good portion of the year. As a result, an increase in the rate and amount of eroded topsoil was transported and deposited into depressional playa wetlands.

The accumulated sediments are functioning as a subsurface storage area and can store a considerable amount of water. The capacity of playa wetlands has been reduced by excessive sediment inputs. In fact, many only pond water after a considerable amount of precipitation has occurred in a short amount of time because of excessive sediment accumulation. Consequently, playas with high amounts of sediment accumulation require larger runoff events to facilitate ponding (Cariveau et al. 2011). Therefore, contemporary hydroperiods are significantly different from the pre-settlement era. Culturally accelerated sedimentation expedites successional transitions from wetland habitat to mesic or even upland habitat, and without appropriate removal of excessive sediment inputs, many playas will continue to be dysfunctional (LaGrange et al. 2011; Tang et al. 2016, 2018). Mapping the shallow soil vertical profile is critical in understanding the pattern, distribution, and trend of sediment accumulation in playa wetlands. Thus, this study focused on determining the depth to Bt that is frequently used as a benchmark depth for designing and implementing wetland restoration projects.

1.3 Approaches for sediment estimation

Recent research has greatly improved our understanding of playa sediment issues in the Rainwater Basin (Renard et al. 1997; Beas et al. 2013a; Daniel et al. 2015, 2017; Tang et al. 2015b). Although the soil erosion models can be used to estimate soil erosion rates at the landscape level, the actual sediment deposition rate is subject to the transport capacity at the watershed scale (Renard et al. 1997). Cropland area at the watershed scale can have a significant influence on runoff rates, sedimentation rates, and declines in water storage capacity in playas (Daniel et al. 2015). The contemporary inundation patterns of playa wetlands in the Rainwater Basin have been affected by excessive amounts of sediment deposited (Tang et al. 2015b). Recent studies also examined the effectiveness of sediment removal on playa wetland performance. Sediment removal has been linked to improvements in plant community characteristics (Beas et al. 2013a), and improvements in carbon and nitrogen storage in playa wetlands (Daniel et al. 2017). Yet, we still lack the ability to efficiently evaluate the depth to Bt that will help develop prioritization models to support conservation delivery efforts, particularly for sediment removal projects.

Wetland researchers, managers, and restoration ecologists still rely heavily on traditional field drilling or sampling techniques that allow the examination of wetland sediment profiles (Gleason and Euliss 1998; Olson and Jones 2001; Tang et al. 2015a). Traditional point measurement sampling techniques, such as soil coring, mechanical probes, and pit excavation, can provide soil profile information, but they are destructive, labor-intensive, and time-consuming. Moreover, these traditional soil survey tools not only provide partial or limited information but also provide an incomplete characterization of the subsurface profile at a localized level that is helpful for developing wetland restoration plans. Soil scientists must apply both visual and tactile observations to make interpretations based on inferences made across typically large and extensive areas using wide-ranging core samples (Sudduth et al. 1999). By increasing our ability to provide detailed site-specific soil profiles rapidly, we can describe the variability and dynamics soil moisture and clay content. Therefore, determining the depth to Bt is essential to defining the potential hydrologic impacts caused by accumulated sediment that ultimately support comprehensive wetland restoration designs.

1.4 Electromagnetic induction technique in sediment research

The electromagnetic induction (EMI) technique has been used to characterize the spatial variability of soil properties, such as clay content and soil salinity since the 1970s (de Jong et al. 1979; Williams and Baker 1982). EMI can measure the apparent soil electrical conductivity by inducing an electrical current in the soil (Saey et al. 2009), but has not been adequately tested for use in playa wetlands. The EMI approach is non-invasive, and it can provide immediate results. Therefore, it can be used to develop the inventory of lateral changes in soil properties underlying soil surfaces (Saey et al. 2009). The measurement depth is primarily determined by the coil

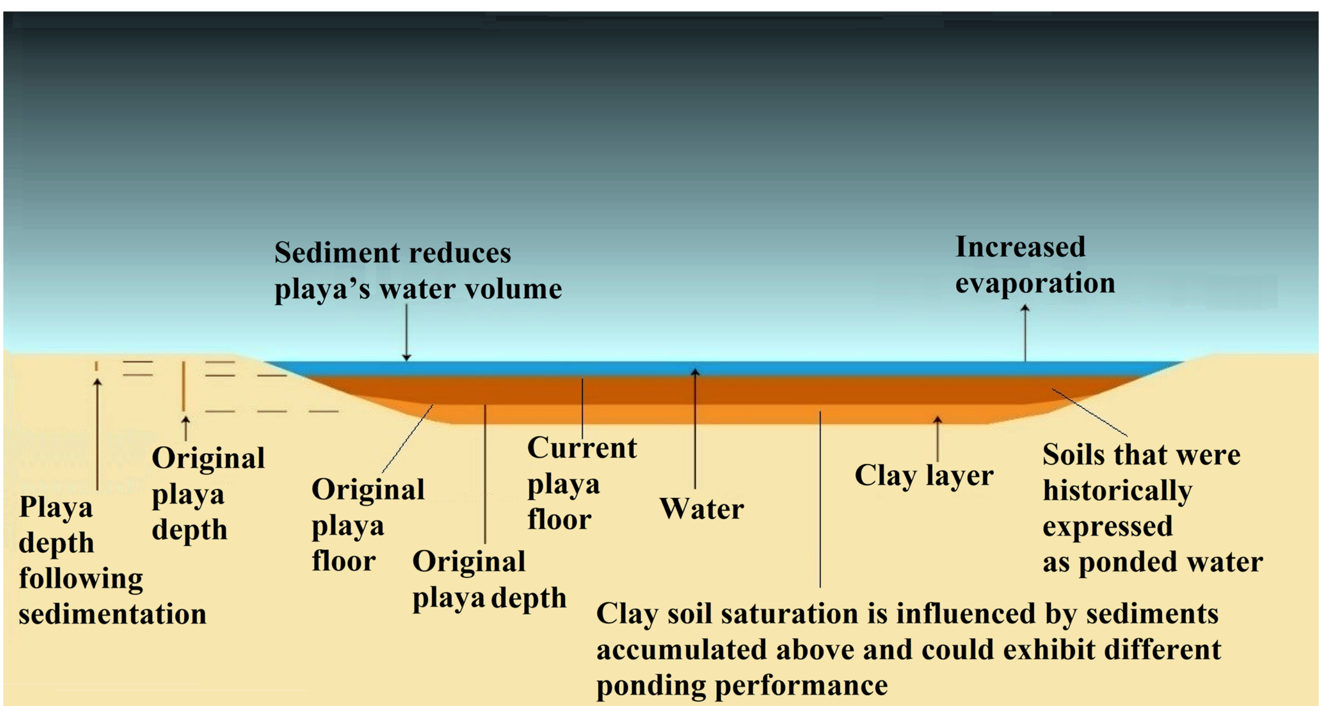


Fig. 1 Illustration of culturally accelerated sediment in playa wetlands (modified from Schafersman 2011). The darker orange area indicates new soils that may have been deposited since settlement and are filling the area with soil rather than ponded water

orientation, coil spacing, frequency of the induced current, and the height of the EMI instrument above the ground (Gebbers et al. 2007; Saey et al. 2009). Electrical conductivity in the soil is determined from the combination of physical and chemical soil properties (Corwin and Lesch 2005) such as soil porosity, soil salinity, clay content, water content, concentration of dissolved electrolytes in the soil solution, cation exchange capacity, composition of colloids, and temperature (Sundberg 1932; McNeill 1980; Rhoades et al. 1989; Sudduth et al. 1999; Saey et al. 2009; Sudduth et al. 2010; Herrero and Castaneda 2015). In principle, the apparent electrical conductivity at different depths can be obtained from the measures at different heights above the soil surface (Borchers et al. 1997). Soil electrical conductivity rises when soil conditions change such as increases in water content or percent clay (Freeland 1989).

The EMI techniques have been verified as useful tools to distinguish soil types and properties (Ammons et al. 1989; Heil and Schmidhalter 2012; White et al. 2012; Doolittle and Brevik 2014). Sudduth et al. (1995) used the EMI to examine the depths to clay pan, and Doolittle et al. (1995) used it to determine the thickness of floodplain splay deposits. In addition, EMI can effectively distinguish and map the soil conditions by detecting soil water content (Hezarjaribi and Sourell 2007; Huth and Poulton 2007) and soil salinity (Cook and Walker 1992; Yao et al. 2012; Herrero and Castaneda 2015). Doolittle et al. (1994) observed high correlations between the observed depth to claypan and the response of the EMI measurement in central Missouri. Sudduth et al. (1995) found high statistical agreement between the soil probe depth and the EMI estimated depth. Kitchen et al. (1996) used the EMI technique to estimate sand deposition by detecting the conductivity between sand and other soils. Boettinger et al. (1997) evaluated the soil depths for 216 soil sampling points at the 0–150-cm level and concluded that EMI is useful tool “for nondestructively assessing soil depth to a cemented subsoil horizon quickly and inexpensively.” Bork et al. (1998) evaluated the two EMI meters (EM38 and EM31 models) for non-destructively assessing soil depth to bedrock in grazing lands with different vegetation compositions. Freeland et al. (2001) pointed out the EMI technique is particularly suitable for the areas where subsurface properties are reasonably homogeneous.

The EMI response can clearly reflect the variations in properties (e.g., clay, water, or salt content) particularly when they dominate over other properties (Cook and Walker 1992; Freeland et al. 2001). The Natural Resource Conservation Service (NRCS 2002) found that the EMI method could accurately mirror soil survey maps and determined that the higher conductivity readings at greater soil depths was attributed to the increased clay and moisture contents of each soil horizon. This led them to recommend the EMI technique as a reliable method for soil survey verification and wetland delineation within the Platte River area of Nebraska (NRCS 2002). Saey et al. (2008) used the EMI technique (EM38DD model) to map the depth to the Tertiary clay substrate and reached a high correlation coefficient with independent observations. Saey et al. (2009) reconstructed the interface depth between the Quaternary loess and the Tertiary clay. Doolittle and Brevik (2014) summarized the EMI technique as a rapid, non-destructive, cost-effective approach to measure the spatial variability of soil properties. Based on the established literature, this study adopted the EMI technique and further expanded its application to identify the upper compacted layer of a playa wetland for precise wetland evaluations.

1.5 Research objectives

The overall goal of this research was to test the EMI survey technique to determine the depth to Bt in publicly managed wetlands in the Rainwater Basin, Nebraska, USA. We evaluated the EMI survey technique to determine if it can efficiently establish a depth to Bt and thus estimate the thickness of accumulated sediments in wetlands. This study was designed to fill a critical knowledge gap about the vertical profile of playa wetlands that can help to determine future wetland restoration priorities on public properties.

2 Methods

2.1 Study area

The location of the Rainwater Basin is shown in Fig. 2. The Rainwater Basin region encompasses 15,907 km^2 including all or parts of 21 counties on the Loess Plains of south-central Nebraska (LaGrange 2005). The Rainwater Basin consists of natural wind-formed wetland depressions that tend to have a northeast to southwest orientation (Kuzila and Lewis 1993). The landscape of the playa complex features flat to gently rolling plains which are formed by deep deposits of wind-blown loess, and the size of wetlands range from less than 4,000 m^2 (approximately 1 acre) to over 4.0 km^2 (approximately 1,000 acres). Each playa is identified by a characteristic hydric soil that typically has prolonged ponding during the growing season (Allen et al. 1972; Beas et al. 2013a). Our study sites in the Rainwater Basin comprise 93 publicly managed wetland watersheds including 35 Wildlife Management Areas (WMAs) managed by the Nebraska Game and Parks Commission and 58 Waterfowl Production Areas (WPAs) managed by US Fish and Wildlife Service.

2.2 Equipment and software

The EMI instrument used in this study is the EM38-MK2 manufactured by Geonics Limited, Mississauga, Ontario, Canada. The EM38-MK2 equipment includes a lightweight bar approximately 1 m in length that contains a transmitter and two receiver coils (Grisso et al. 2009). It is designed to be held by hand and take stationary electrical conductivity readings (Sudduth et al. 2010), and it includes a digital readout of electrical conductivity in millisiemens (mS) per meter with calibration controls. Collected data can be logged into a DAS 70-AR Data Acquisition System embedded in an Archer 2 Field Computer, connected by either Bluetooth wireless technology or an RS-232 serial cable.

We used the differential global positioning system receiver which has a 1–2-m accuracy to document each sampling point. At each sample location, EM38-MK2 was used to measure changes in the magnetic field between a transmitting and receiving coil. The device operated at a frequency of 14.5 kHz and provides measurements of ground apparent electrical conductivity (Quad-Phase). Two transmitter receiver coil separations occurred at 1.0 m and 0.5 m with 3 effective depth ranges at 0.75 m and 1.5 m in vertical dipole mode, and two ranged at 0.38 m and 0.75 m in horizontal dipole mode. In this study, the instrument was operated only in the vertical dipole mode with coil separation at 1.0 m to reach the effective signal detection depth of approximately 1.5 m. After the field data collection, the Interpex Limited 1X1Dv3 inversion software (Interpex.com at Golden, Colorado, USA) was used to generate one-dimensional conductivity versus depth profiles. The depth to Bt was derived from the inversion results.

2.3 Data source for soil series

The Soil Survey Geographic (SSURGO) database (NRCS 2015) was produced by the US Department of Agriculture's Natural Resources Conservation Service. We used the SSURGO data to determine the locations of hydric soil footprints and the soil classifications for each public wetland (Tang et al. 2015b). The primary wetland soil series found on public lands include Massie, Scott, Fillmore, Butler, and Rusco. According to the soil classifications, Massie series consists of very poorly and deep drained clay pan soil that formed in loess modified by water (NRCS 2015). Massie soils, when present, are typically in the lowest portions of hydric footprints and are often the wettest areas. Massie soils generally tend to function as semi-permanently ponded wetland habitat (Cowardin et al. 1979). The Scott soil series consists of very deep and poorly drained soils (NRCS 2015). Scott soils occur in the lower elevations of these depressional wetlands (NRCS 2015), but are above the Massie soil areas when present. Scott soils normally provide seasonal ponded habitat (Cowardin et al. 1979). Other soil types, such as Butlers, Fillmore, and Rusco soils, tend to be temporarily

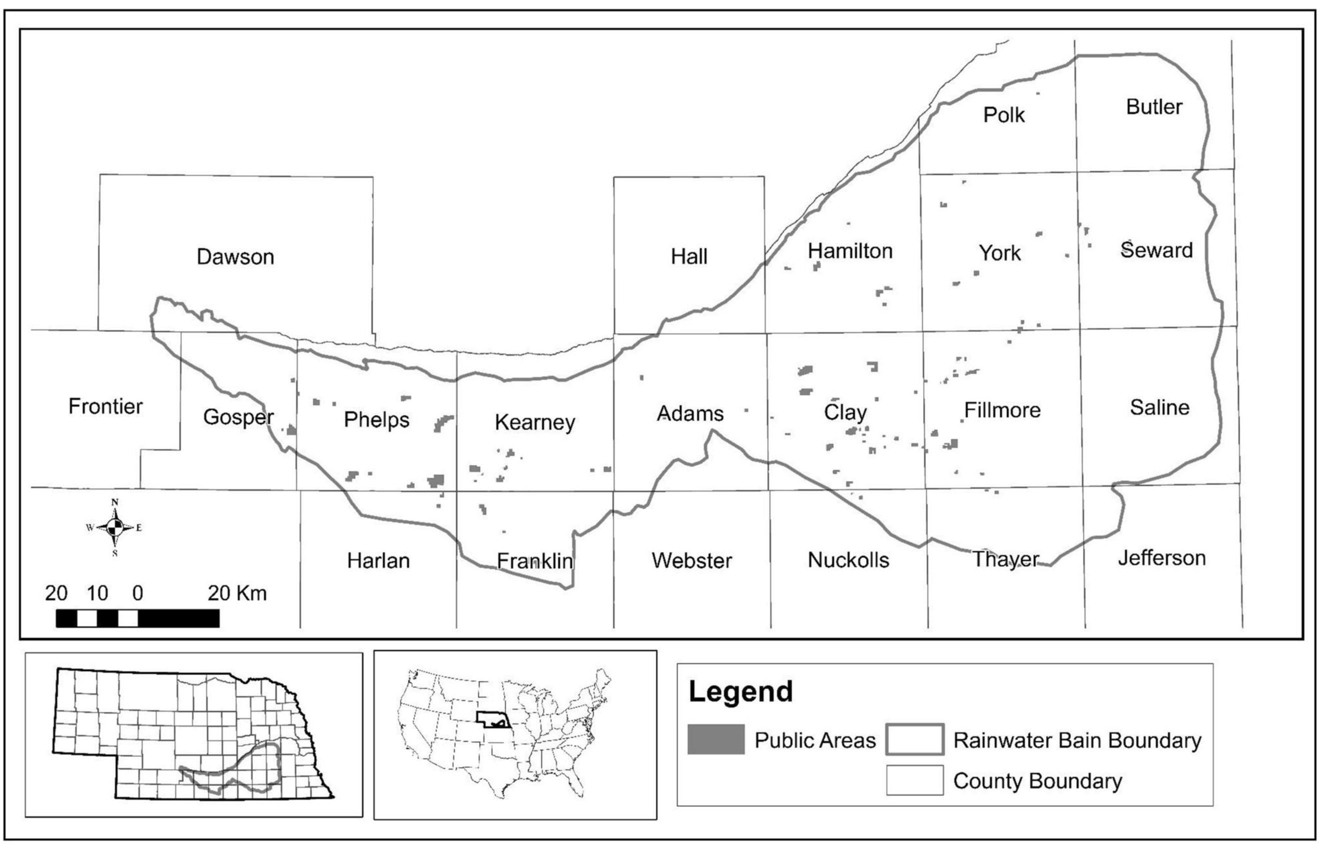


Fig. 2 Location of the publicly managed wetlands in the Rainwater Basin

ponded (Cowardin et al. 1979). Fillmore soils consist of somewhat poorly and very deep drained soils in shallow closed depressions or basins of stream terraces and uplands (NRCS 2015). The Fillmore series is normally positioned at lower elevations than the Butler series, but are higher than the Massie series when they occur within the same hydrologic soil footprint. Butler soils are somewhat poorly and very well drained soils that formed in loess or mixed loess (NRCS 2015). Butler soils are typically positions on flat or shallow depressions on loess uplands and high stream terraces (NRCS 2015), and are normally positioned higher than Fillmore and Scott soils in the landscape when they occur within the same hydric soil footprint. For example, at the Jones WPA watershed scale (as illustrated in the Fig. 3), the Holdrege soils dominated for 84.0% of the watershed areas. Holdrege soils are mainly distributed in uplands within the watershed. Each of the Fillmore silt loam soils and Scott silt loam soils occupy 4.9% of watershed while other soil types (Butler silt loam, Crete silt loam, Detroit silt loam, Uly and Coly silt loams) account for 6.2% of watershed areas.

2.4 Data collection and analysis methods

At each wetland footprint, we determined hydrologic routes, delivery points, and obvious silt plumes to minimize the possible influence from point-source deposition. Then, we identified areas that were likely to have uniform soil deposition within the lowest area of each hydric soil footprint, and then in an area that was easy to access. After these areas were located, we randomly positioned sampling areas. Then we cleared vegetation to create a bare soil area and collected three field measurements in the cleared area. The sampling strategy focused on the soils at the lowest elevation within each hydric soil footprint on public lands. Therefore, the data were collected with the following priority: (1) Massie series, (2) Scott series, (3) Fillmore series, and then the other soils. On the SSURGO soil map, these soil series are horizontally distributed on landscape, yet in the field these soil series represent different elevations at the watershed scale. We did remove dead plant material and live vegetation from the sample locations to ensure the ground conductivity meter was not influenced by electrical insulting material on the soil surface. Additionally, the soil surface was leveled off to create a relatively accurate reference plane when measuring the height above the referenced soil surface. Then, EM38-MK2 was deployed to collect conductivity values in both east-west and north-south directions in vertical mode with the transmitter receiver coil separation at 1.0 m. At the sample point, the instrument was held above the reference plane at a height of 0–150 cm, in 10-cm increments, and it stayed at each elevated increment for approximately 1 min to allow the unit to average the readings at each height increment.

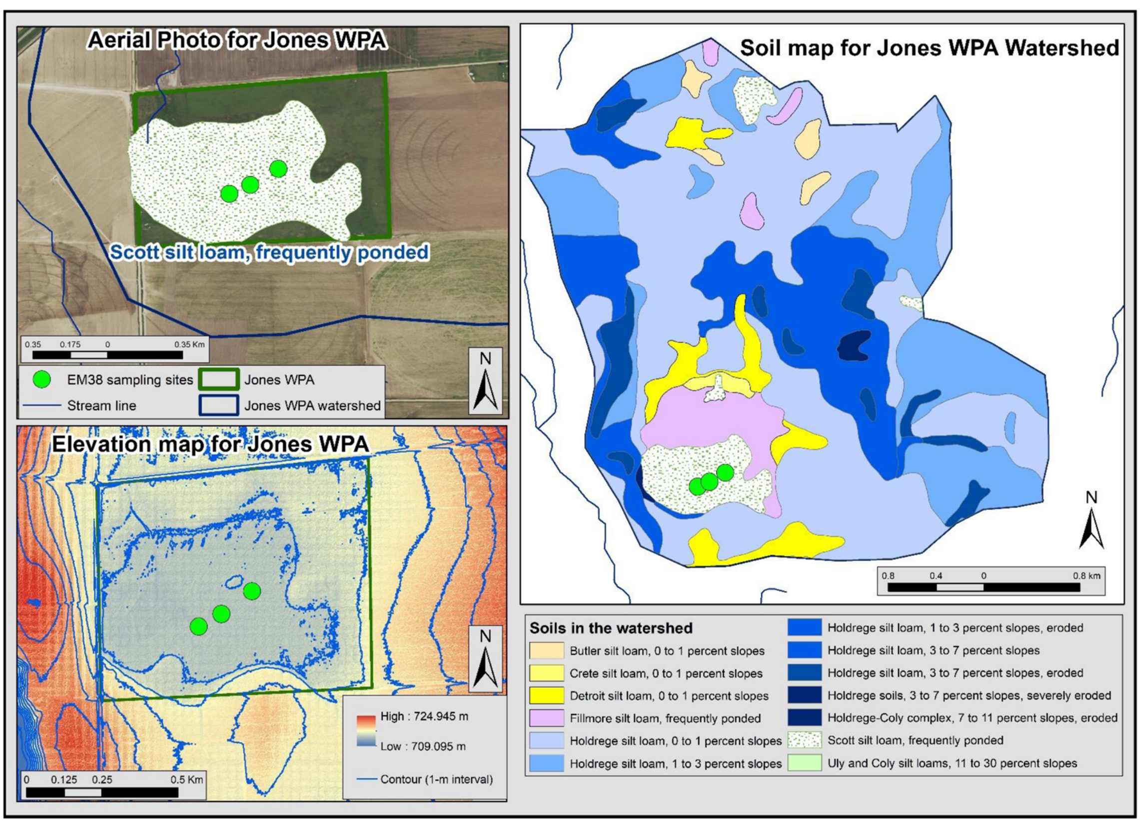


Fig. 3 Data collection and soil core sites in Jones WPA hydric soil footprint (left upper). The representative watershed is outlined in blue (right upper)

The EMI data were analyzed using the Interpex Limited 1X1Dv3 software (Golden, Colorado, USA). The Occam inversion smooth method was used to generate one-dimensional conductivity depth profiles for each replicate at each sample site. The data were visualized using 15–20 layers, depending on the best-fit pattern that minimizes the root mean square of the EMI conductivity value. After that, the multiple iteration inversion was used to analyze the equivalent layered pattern where only three split layers were displayed. Then, the equivalent layered pattern that best fits the conductivity value and the smoothing method were chosen to represent the vertical soil conductivity profile.

3 Results

3.1 Results validation in two wetlands

The EMI measurement results were validated with field survey data in two public wetlands. First, at Jones WPA in Phelps County, Nebraska, three soil cores were taken within the Scott soil series in 2012. The depth to Bt for those three cores were verified in the field by soil scientists (M. Kuzila and N. Dominy) and later quantified in the lab by visual observation by a University of Nebraska soil scientist (M. Kuzila). In this study, the data collection with EM38-MK2 was calibrated three times at each study wetland on August 16, 2016 (Fig. 3). The EMI sampling points were matched up with the soil cores points by using the same GPS coordinates. As illustrated in Fig. 4, these three split layers from top to bottom separately represent the sediment soil layer (includes the "A" and "E" horizons), the Bt horizon, and the subsoil layer underlying the clay pan. In these clay-pan depressional wetlands, the dominant factor causing the EMI measurement changes on the vertical sediment profile is the variations of soil types. Clay soil particles have a huge surface area to volume ratio; thus, it has a much higher exchange capacity than other soil types. Therefore, the reflection of the clay soil layer can appear much denser than the top layer of accumulated sediments. The first horizontal red line from the top surface is the marker to distinguish the accumulated soils above the Bt horizon. The inferred depth to Bt from the inversion results in Fig. 4 were 27 cm, 25 cm, and 28 cm at site 1, site 2, and site 3, respectively. The depths to the clay soil layer from the lab observations for the three sites were 25 cm, 20 cm, and 25 cm, respectively. The field observation photo is shown in Fig. 5. The soil scientists determined the depth to Bt was 25 cm, indicating that the thickness of accumulated soils was also 25 cm. This resulted in no significant difference between the two measurement strategies of soil core sampling and EMI measurements.

We further validated the utility of EMI method at the Kenesaw WPA in Adams County, Nebraska on September 6, 2016. Seven data collection sites were measured using the EM38-MK2 equipment. The first four sampling points located in higher elevations within the wetland footprint that were land leveled prior to acquisition. The EMI equipment indicated a mean depth to Bt of 46 cm (range 38 to 53 cm). Then, we sampled three points within an area near the lowest elevations of the wetland (the deepest areas were pits excavated in the 1970s so we did not sample within the lowest elevations). In this area, the mean depth to Bt was 29 cm (range was 24–35 cm). Interestingly, the soil core sampling in these same areas indicated that the depth to Bt was 45 cm and 30 cm, respectively. The EMI measurements were essentially consistent with the mean depth to Bt at within these two areas within the wetland footprint. Therefore, these results indicate that the EMI can serve as a reliable alternative tool to effectively detect the depth to Bt in the playa wetlands.

3.2 Depth to clay for all public wetlands

We evaluated 93 publically managed wetlands by conducting 3 field measurements within each wetland footprint for a total of 279 measurements. We found that the mean depth to Bt for all 279 sampling points was 39 cm with a standard deviation of 11 cm. There were large variations in depth to Bt across these wetlands that ranged from 21 to 78 cm (Fig. 6). Yet, we used all of the data to calculate a mean annual sediment deposition rate of 0.26 cm year^{-1} for all publicly managed wetlands with the assumption that rapid sedimentation has been occurring over the last 150 years given the increase in anthropogenic activities.

We found that a significant number of the wetlands had thick sediment accumulations above the Bt. Across all sample locations, the sediment thickness profiles at the 10th percentile, 25th percentile, 50th percentile, and 75th percentile reached 27 cm, 30 cm, 38 cm, and 45 cm, respectively.

We also calculated the mean values of the 3 sampling points in each wetland to ascertain the internal variations of the EMI measurements. Across all 93 wetlands, the mean was less than 6 cm. These results indicated a high level of internal consistency of the EMI measurements in each site. But, even in one wetland area, the EMI measurements may still be varied by many possible external factors (e.g., micro-topographic condition, vegetation condition, drainage pattern, agricultural disturbance). In this study, only five sites, accounting for 5% of the total 93 sites, had the internal variations larger than 11 cm, which was the overall standard deviation for all 279 samples.

We summarized the mean value of the depth to Bt in both ownership categories (Fig. 7)—WMAs and WPAs. For the 105 sampling points in the 35 state-managed WMA wetlands, the depth to Bt ranged from 24 to 78 cm. The mean thickness of accumulated soils above the Bt in these WMAs was 42 cm with a standard deviation of 12 cm, and the maximum depth to Bt was 78 cm at Southeast Sacramento WMA. For the 174

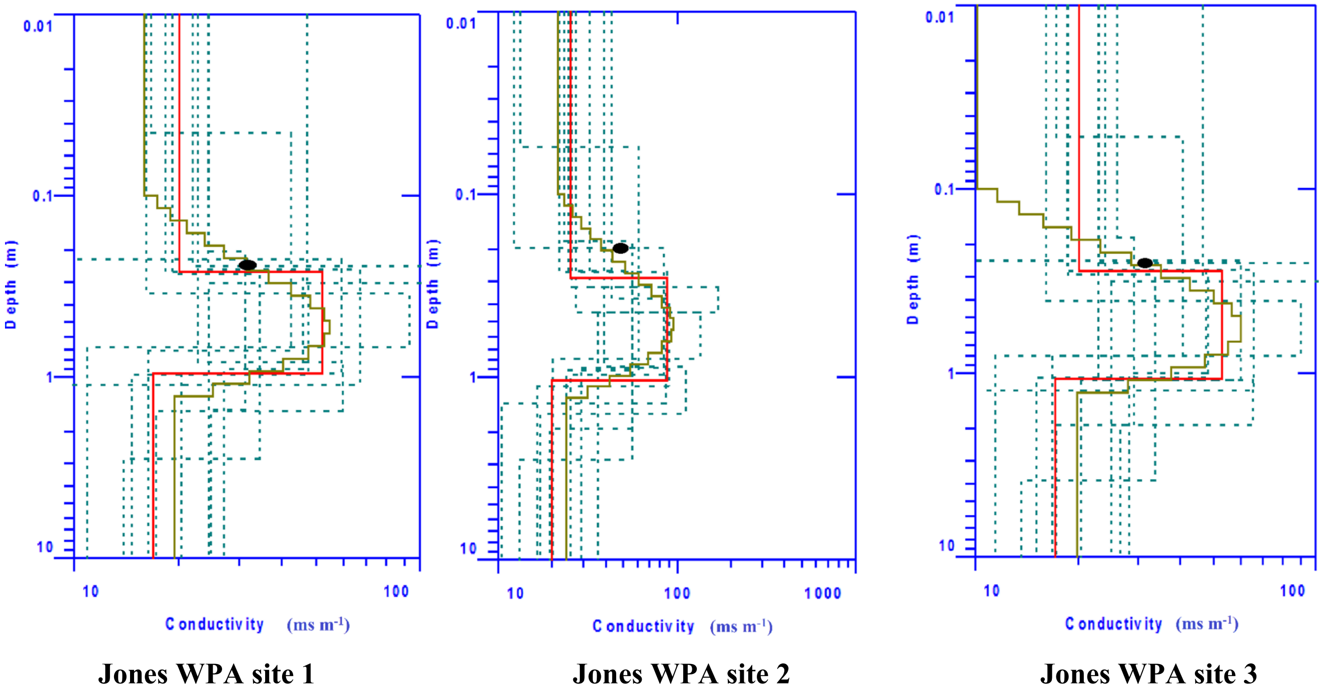


Fig. 4 Comparison of the EMI measurements and soil core samples at the three areas in Jones WPA (the top horizontal red line indicates the best-fit pattern from the Occam inversion smooth method based on the electromagnetic conductivity. The black dot indicates the field observation result from the soil cores)

sampling points within the 58 federally managed WPA wetlands, the depth to Bt ranged from 21 to 71 cm. The mean thickness of accumulated soils above the Bt in these wetlands was 37 cm with a standard deviation of 9 cm, and the maximum depth to Bt was 70 cm at Weseman WPA.

3.3 Depth to clay within specific wetland soil series

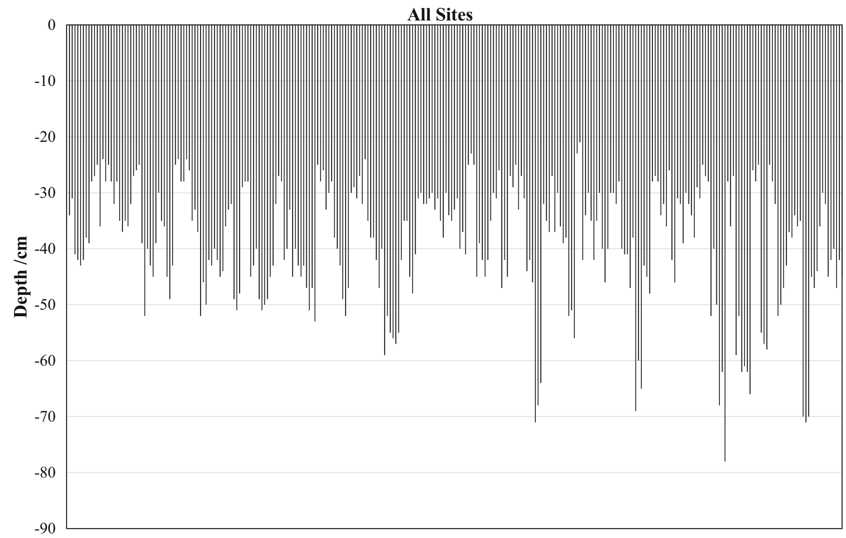
We determined the mean depth to Bt for each of the primary soil types within the lowest portion of the wetlands sampled (Fig. 8). For the 153 sampling points within the Massie soils, the Bt depth ranged from 23 to 78 cm. The mean thickness of accumulated soils above the Bt in the Massie soils was 40 cm with a standard deviation of 11 cm. In contrast, the 84 sampling points within the Scott soil areas had a range of depth from 24 to 62 cm. The mean thickness of accumulated soils in the Scott soils was 38 cm with a standard deviation of 10 cm. Similarly, for the 27 sampling points collected in the Fillmore soils, the depth to clay soil layer ranged from 24 to 66 cm. The mean thickness of accumulated soils in the Fillmore soils was 37 cm with a standard deviation of 12 cm. The one-way ANOVA statistical test resulted in no statistical difference ($p > 0.05$) for the mean depth within the different soil series.

Fig. 5 A soil core from Jones WPA indicating 25 cm of sediment accumulation above the Bt horizon (darker soils deeper in the profile)

3.4 Depth to clay under watershed context

We compared the depth to Bt and watershed size (Fig. 9) with the assumption that more sediment could accumulate within wetland footprints that have larger watersheds. Yet, the one-way ANOVA statistical test results did not indicate any significant differences ($p > 0.05$) in the thickness of accumulated soils above the Bt by watershed sizes. For the 63 sampling points in the smaller watersheds (area < 0.40 km^2), the mean thickness of accumulated soils above the Bt was found at 40 cm with a standard deviation of 11 cm. For the 156 sampling points in the medium-small watersheds (area between 0.41 and 2.02 km^2), the mean thickness of accumulated soils above the Bt was found at 39 cm with a standard deviation of 11 cm. For the 36 sampling points in the medium-large watersheds (area between 2.03 and 4.05 km^2), the mean

Fig. 6 The variability in depth to Bt across all publicly managed wetlands sampled by using the EMI method



thickness of accumulated soils above the Bt was found at 38 cm with a standard deviation of 10 cm. For the 24 sampling points in the large watersheds (area above 2.03 km^2), the mean thickness of accumulated soils above the Bt was found at 41 cm with a standard deviation of 10 cm.

We also evaluated watershed slope and the accumulation of soils above the Bt in these wetlands (Fig. 10). For the 66 sampling points in watersheds with a slope above 10°, the mean thickness of accumulated soils above the Bt reached 45 cm with a standard deviation of 12 cm. For the 99 sampling points in the watersheds with slopes of 5–10°, the mean thickness of accumulated soils above the Bt was 38 cm with a standard deviation of 11 cm. For the 114 sampling points in the watersheds with slopes less than 5°, the mean thickness of accumulated soils above the Bt was 37 cm with a standard deviation of 7 cm.

4 Discussion

4.1 Sediment accumulation rate

The thickness of soils, presumably the accumulations of organic matter and sediment deposited through fluvial and eolian processes, accumulated above the Bt horizon in each playa wetland represented the sediments in association with entire corresponding watershed. Soils originating from higher lands have been transported to the lowest areas in these hydrologically closed watersheds. We selected sample

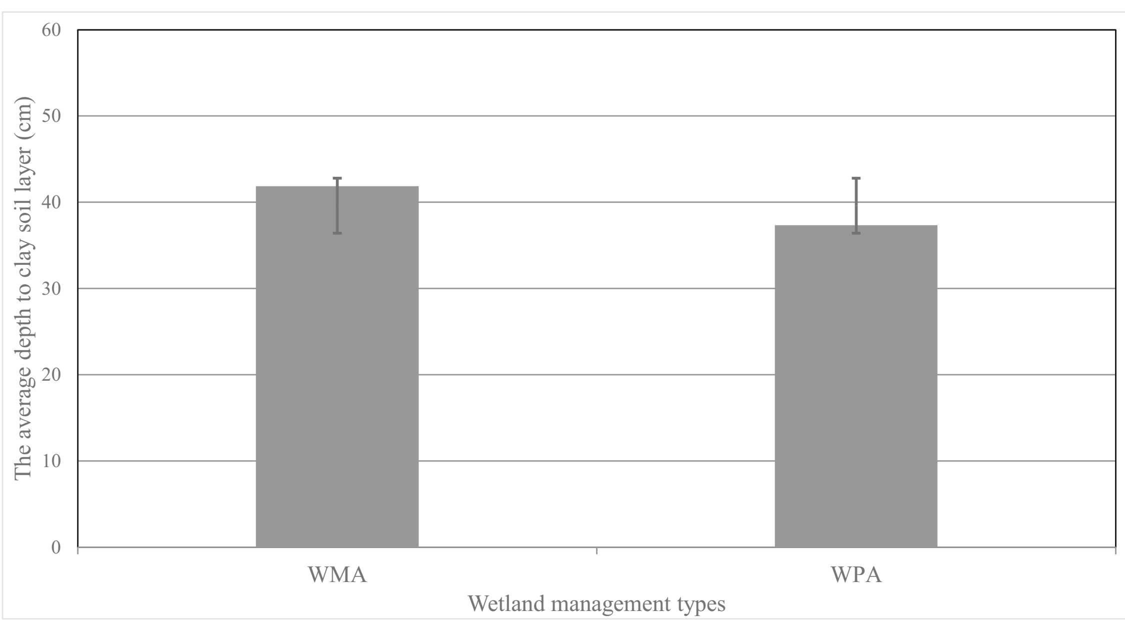


Fig. 7 The mean depth to the Bt horizon between State WMA and Federal WPA wetlands

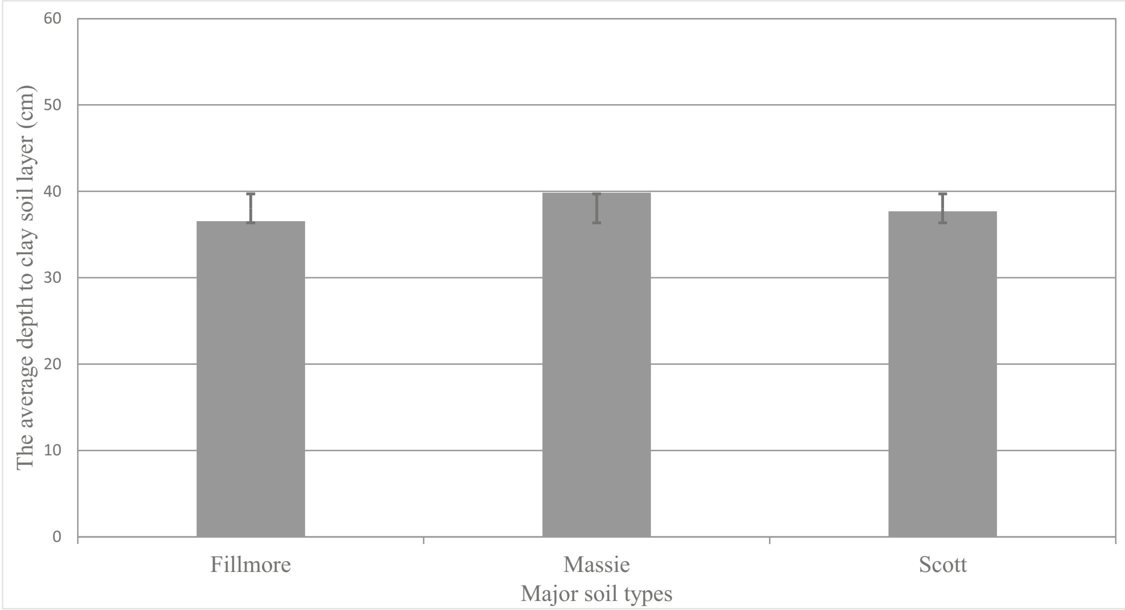


Fig. 8 The mean depth to Bt for the major hydric footprint soil types

locations at the lowest points within hydric soil footprints that were distal from watershed delivery points to avoid sampling in areas with excessive accumulations. We found that wetlands within the Rainwater Basin had similar sediment accumulation ranges in the High Plains playa wetlands. O'Connell et al. (2013) found that High Plain playa wetlands with either native grasslands or croplands had depth to the Bt horizon ranging from 1.0 to 78.0 cm. Based on our inversion analysis results and assuming no accumulation of soils above the Bt in some wetlands as indicated by Layton et al. (1927), the annual post-settlement sediment accumulation rates (0.14–0.52 cm year^{-1}) were similar to the 0.18–0.29 cm year^{-1} in five playa wetland sites found by Tang et al. (2015a) and were also similar to the mean of 0.34 cm year^{-1} found by Jones and Olson (1990).

The results of this study also showed that soils accumulated above the Bt horizon have been increased during the past century. Even though considerable variations existed, it is apparent that the depth to Bt is increasing based on the earliest documented soil conditions in contrast with contemporary soil conditions in Rainwater Basin wetlands. The available depth to Bt in the early Nebraska soil survey between 1919 and 1934 (Layton et al. 1927; and summarized in LaGrange et al. 2011), in the wetland soil surveys during 1997–2009 in the SSURGO

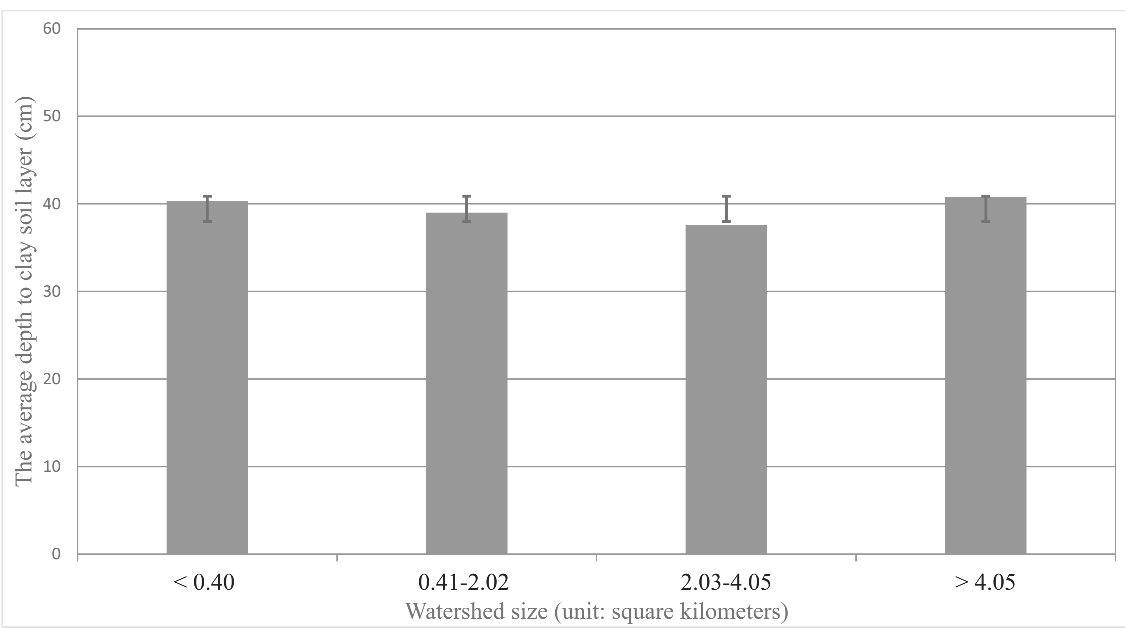


Fig. 9 The mean depth to the Bt horizon for wetlands within four watershed size categories

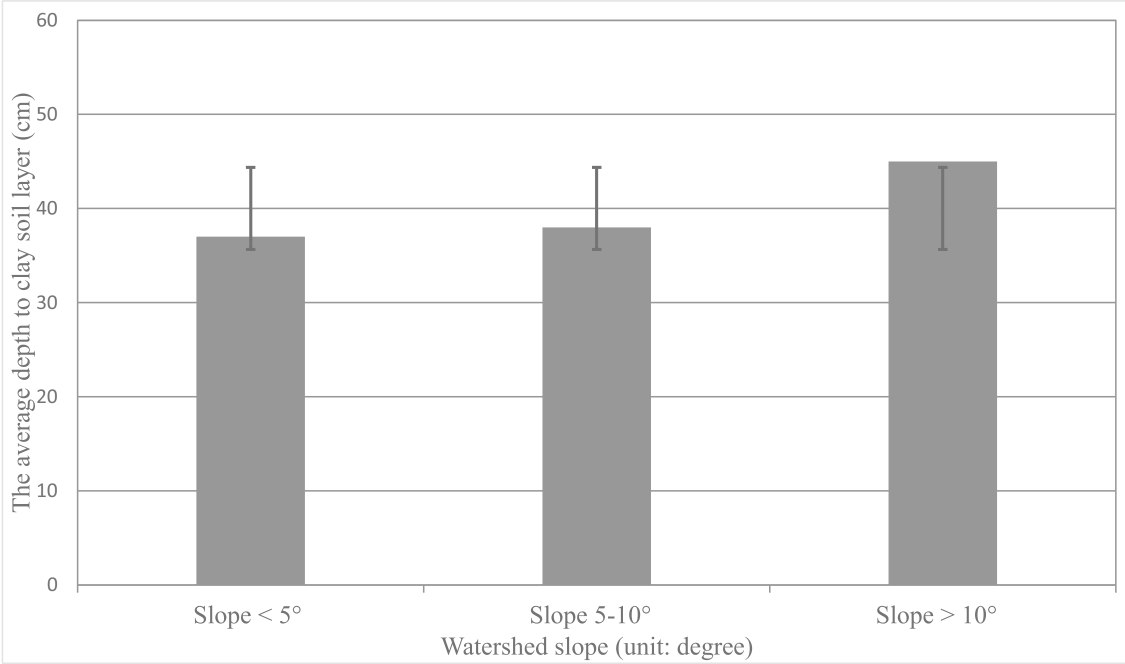


Fig. 10 The mean depth to the Bt horizon for three watershed slope categories that provide the primary source of runoff water to the wetland

data (NRCS 2015), and our EMI measurements in 2016 indicated a consistent progression of soil accumulation above the Bt. The mean depth to the contemporary Bt horizon has increased from the earliest documented soil conditions (1916–1934) because culturally accelerated sedimentation related to human activities that facilitated rapid soil movement into playa wetlands, yet these reference conditions are 50–70 years after many of these lands were first plowed. In early soil surveys published from 1916 to 1934, the depth to Bt for Fillmore soils had a range of 15–38 cm, and Scott soils had a range of 13–30 cm (LaGrange et al. 2011). The first soil survey did not describe a Massie soil series so these areas were classified into the Scott soils series. Although the sample size and locations were different between the SSURGO surveys and the EMI surveys, similar trends were observed in increased sediment accumulation among the playa wetlands. In the 1990s–2000s field soil survey results for playa wetlands in the Rainwater Basin, the depth to Bt in Fillmore soils had a sediment thickness range of 28–74 cm (our EMI result: 24–66 cm), Scott soils had a sediment thickness range of 8–23 cm (our EMI result: 24–66 cm), and the Massie soils had a sediment thickness range of 10–64 cm (our EMI result: 23–78 cm). Additionally, the sampling locations from the first soil surveys were likely different, but the increased soil thickness above the Bt in current playa soils is apparent. These early soil surveys within Rainwater Basin counties indicate the depth to Bt in Fillmore soils ranging from 15–38 cm and ranging from 2.5–64 cm in Scott soils (NRCS 2019).

The variations in the thickness of culturally accelerated sediment above the clay were a result of environmental influences associated with deposition and deflation events. The variability of culturally accelerated sedimentation actually reflects a series of complex hydrologic and climatic processes such as hydrological alteration, conservation practices within each watershed, eolian processes (both deposition and deflation), the grid road system, variations in topography that have occurred since settlement, and vegetative conditions that can add organic matter over time.

The rate of culturally accelerated sedimentation has not been physically measured but has been projected for Rainwater Basin wetlands. Yet, some studies have identified factors that directly influence accumulation rates such as climate conditions, soil types, wetland and watershed size, and land use types within the watershed (Tsai et al. 2007; Novotny 2008; O'Connell et al. 2013; Tang et al. 2015b). Among those factors, the variations in microtopography in combination with the human-made hydrologic alterations at watershed scale are all important roles in effecting sediment accumulation in wetlands. For example, roads cut off the pathway for natural runoff conveyance into wetlands, and concentration pits collect and store large volumes of water that would otherwise flow into wetlands. Both man-made features and agricultural infrastructure have wholly changed watersheds in the Rainwater Basin, and thus changed the patterns of sediment inputs into wetlands. Additionally, the upper soil horizon in many wetlands has been mixed by either hoof action as a result of wetland grazing or other management techniques such as discing or rototilling in wetlands (Tang et al. 2015a). In terms of micro-level topographic variations, wind erosion can naturally create different patterns of soil deposition including sediment dunes within wetlands that can also influence local sediment accumulation rates and can alter water movement patterns within the wetlands (LaGrange et al. 2011). During extended dry periods and drought years, Rainwater Basin wetlands can develop deep desecration cracks at the soil surface (Wilson 2010). The deep cracks in combination with the deep vegetative tap roots during droughts (Lindh et al. 2014) can contribute to creating conduits that facilitate downward water movement (Wilson 2010), and at times can even create visible vortices and therefore could initiate downward sediment movement. All of these factors can create high variability in sediment accumulation patterns within Rainwater Basin wetlands.

4.2 Policy implications for wetland conservation and restoration practices

To slow down the culturally accelerated sedimentation process, federal conservation programs have implemented practices such as the use of cover crops, no-till farming, and restoration of portions of watersheds from cultivated croplands to native grasslands in targeted areas (e.g., steeper slopes, water ways) across the Rainwater Basin area (Tsai et al. 2007; Cariveau et al. 2011). Establishing vegetative buffer strips along wetlands has been widely recognized as a simple, effective, and inexpensive conservation practice that can reduce sediment deposition in playa wetlands (Chaubey et al. 1994; Gleason 1996; Beas et al. 2013b; Smith et al. 2014; Tang et al. 2015b). If conservation programs focus on implementing practices that reduce the rate and amount of sediment delivered to wetlands, managers could then expect longer sustainability in a desirable condition after within-basin sediment removal restoration work has been completed. Therefore, these practices could also reduce the amount of water delivered to the wetland and therefore future research is needed to determine how wetland function is influenced by each practice considered.

Selecting cost-effective restoration practices can be challenging when faced with significantly altered habitats that provide limited ecosystem services, particularly when funds are limited (Adame et al. 2014). Sediment removal is recognized as a direct and effective way to restore wetlands impacted by the culturally accelerated sedimentation, and this treatment can increase ponded area, ponded volume, and average ponded depth in wetlands (Luo et al. 1997; Tsai et al. 2007; Cariveau et al. 2011). However, over-excavation can also remove portions of the clay soil layer and increase the potential of downward transmissivity, and thus pose unnecessary threats on the bio-geochemical and hydrological functions of wetlands. Therefore, it is important to precisely determine the depth to the clay soil layer before implementing any sediment removal restoration projects to reduce the risk of removing any of the Bt horizon.

5 Conclusions

This study demonstrated the utility of using the EMI technique to determine the depth to Bt for hydric soils and therefore the culturally accelerated sediment thickness in playa wetlands. Our research findings confirmed that the EMI technique is effective and efficient in determining the amount soil above the Bt horizon. The depth to Bt measurement provided important vertical profile information that can help characterize the spatial variability of soil deposition in playa wetlands. The detected sediment thickness values provided important scientific data that will help guide future wetland restoration projects that include sediment removal. The depth to Bt reflects the contemporary hydrologic capacity in playa wetlands. Our results provided an overall summary of contemporary sediment accumulations in all publicly managed wetlands. Essentially, compared with 1920s soil surveys, the increased soil thickness above the Bt in playa wetlands clearly demonstrated the impacts of human activities during the past century that have accelerated soil accumulation rates. The sediment thickness profiles highlighted the continuous long-term challenge of sediment management efforts for playa wetlands within agricultural landscapes.

Compared with the traditional methods used to determine the depth to Bt, such as soil cores, fly ash, and pollen analysis, this alternative method is rapid (with 10–15 min for each sampling point), simple (with standard operation manual), non-invasive (no need for machine or truck for soil core drilling), cost-effective (with the investment of the equipment and software cost for approximately US$20,000–30,000), and reusable (one machine for long-term use), and it requires only basic knowledge in geophysical and soil science. However, the geophysical interpretations can only be used as preliminary evaluations of site conditions. Remote and general site investigations (e.g., SSURGO soil review) are interpretive results that cannot be used as a replacement for direct ground-truth observations for each wetland. The use of the EMI technique could reduce the number of soil core observations needed, and therefore supplement the ground-truth soil profile observation and interpretation.

At the same time, we need to recognize the limitations of EMI methods. For example, the sediment thickness itself could be subject to many factors, including micro-elevation, on-site hydrological conditions, vegetation coverage, wetland size, watershed characteristics, climate conditions, and human activities. The EMI measurement could be effected by other operational or contextual variables, such as temperature, water table, salinity, and water quality conditions (e.g., total dissolved solids, salts, and minerals). There are also field limitations for using the EMI method in wetlands. It was extremely difficult to gain access to all parts of many wetlands because of ponding and/or extremely dense and tall vegetation. These conditions would create additional challenges if transects are needed for a full grid-type evaluation which might be needed when comprehensive restoration plans are desired. Thus, the practical application for continuous mobilized mapping using EM38-MK2 for the large wetland areas may be limited by site conditions. Therefore, data collection to represent the general sedimentation accumulation conditions of whole wetlands can only be made in the accessible hydric soil areas or when conditions are ideal such as after prescribed fire. Yet, this may prevent seasonal opportunities when environmental and habitat conditions facilitate efficient data collection. Furthermore, experimental random error is unavoidable in terms of the certain height of equipment above the ground because of the variations in micro-topography in wetlands in combination with the unsteadiness when elevating and holding the equipment by hand. In addition, the sampling locations also influence the representativeness of sediment thickness throughout certain wetlands. In particular, the road systems and culvert placement have significantly changed the natural hydrologic connectivity that is critical to playa wetland function in the Rainwater Basin.

The results of this study indicate that significant amounts of soil have accumulated above the Bt horizon in all 93 publicly managed playa wetlands in the Rainwater Basin area. Additionally, considerable variations in the depth to Bt, presumably recent sediment accumulation, were detected across all wetlands. This research indicates that it is possible to conduct a transect survey that would provide a detailed sediment profile to help visualize soil conditions at the wetland scale. Future wetland restoration designs should include careful evaluation of watershed features and within-basin factors that influence each wetland's hydrologic capacity such as the amount of organic matter within soil that has accumulated above the Bt layer. We recommend that the sediment profile of any wetland targeted for restoration be examined to determine the depth to Bt to help identify the amount of soil that could be removed to increase sustainability and wetland function and to maximize ponding frequency while also determining the maximum depth of removal to avoid removing the high clay content soils within the Bt horizon.

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Compliance with ethical standards

Conflict of interest The contents do not necessarily reflect the views and policies of the funding agencies, and do not mention the trade names or commercial products that constitute endorsement or recommendation for use. The findings and conclusions in this paper are those of the authors and do not necessarily represent the views of the U.S. Fish and Wildlife Service.

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