Title:

Short rotation willow on the prairie potholes' degraded marginal riparian lands: a potential land-use practice to manage soil salinity.

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Abstract

Land-use practice shift in the wetland riparian zone can influence groundwater table (GWT) fluctuations and salts dynamics, potentially leading to soil salinization. The risk of soil salinization linked with high water tables could better manage using high growing capacity and deep-rooted phreatophytic vegetation via 'biodrainage' approach. We evaluated the impacts of short rotation willow (SRW) plantation on soil and groundwater salinity linked to shallow GWT fluctuations and compared with adjacent annual crop (AC) and pasture (PA) in a field experiment. Groundwater salinity (EC_{gw}) along with depth to GWT and soil salinity (EC_{soil} at 0-60 cm depth) were measured along transects within each land-use practice in two prairie pothole region (PPR) wetland sites (A and B). The variations in EC_{gw} were significant ($p < 0.05$) across land-uses; however, inconsistent between sites. The positive correlation with EC_{gw}, $EC_{soil,}$ and total dissolved salts (TDS) indicated higher salinity and salt accumulation with increased depth to GWT in both sites. The EC_{soil} varied significantly ($p < 0.05$) among land-use practices; however, no consistent land-use patterns were observed between sites. Throughout the experimentation, site B consistently exhibited higher EC_{soil} (two-fold) than site A. Decreasing inclinations were observed in EC_{soil} with increasing SRW biomass at both depths (i.e., 0-30 and 30-60 cm) and vice versa. This study refines our knowledge of SRW linked potential hydrological alteration and its implication on salinity, which provides critical context for degraded marginal riparian wetland soil management in the PPR.

Keywords

Land-use practice; Short rotation willow (SRW); Soil salinity; Wetland riparian zone; Degraded marginal lands; Prairie pothole region (PPR).

1. Introduction

Land-use alterations are among the most critical anthropogenic causes of secondary salinization in the dry arid and semi-arid regions (Sakadevan & Nguyen, 2010). Salinization of groundwater and soil is an ancient and pressing environmental problem that threatens crop production and soil degradation within agroecosystems. A high water table with the contribution of additional dissolved salts to the shallow groundwater can further amplify soil salinity via the excess soil wetness caused by summer fallow and irrigation (Eilers et al., 1997). Phreatophytic vegetation, such as short rotation willow (SRW), has a substantial possibility of disrupting shallow groundwater table (GWT) and can trigger soil salinization (Jobbágy & Jackson, 2007). As the GWT becomes shallower, capillary rise and evapotranspiration (ET) bring groundwater and solutes upwards to the root zone and increase salinization risk (Nosetto et al., 2013).

The prairie pothole region (PPR) of the glaciated North American Great Plains consists of millions of small wetlands commonly known as "sloughs" or "potholes" and provide crucial habitat for migratory waterfowl and productive agricultural land (Dahl, 2010). Wetlands in the PPR are situated in a semi-arid region where potential evaporation is almost double the annual precipitation (Winter, 1989). Like many arid and semi-arid regions of the world, salinization is a major threat affecting agricultural productivity and land degradation in the PPR (Clearwater et al., 2016). In the PPR landscape, high GWT and soil salinization's primary reason are three groundwater conditions: artesian discharge, evaporative rings, and hillside seeps (Henry, 2003; Henry et al., 1987; LaBaugh et al., 2018). Wet-dry cycles drive the complex hydrological processes that control surface and sub-surface water dynamics (Valk, 2005) and salinity in the closed-basin PPR landscape (Heagle et al., 2013). Seasonally, mainly in the summer months, an evaporative ring of solute rich porewater can be created due to the out-seepage around the periphery of small ephemeral wetlands by a transient drawdown of the water table by transpiration of fringing phreatophytes (e.g., willow) (Meyboom, 1966b; Nachshon et al., 2013; Stolte et al., 1992; Winter & Rosenberry, 1995).

Processes that accumulate soluble salts can contribute to soil salinity in susceptible areas of the landscape and lead to soil degradation. Soil salinization is a natural process in the semi-arid prairies where a soil water deficit is likely, and the soil and groundwater commonly have a higher amount of mineral salts, including sodium, calcium, and magnesium sulfate (Nachshon et al., 2013). Within the PPR region, salinity begins in a landscape where the water requirements of the existing land-use practice are lower than the snowmelt and precipitation (Heagle et al., 2013). Notably, salinization can occur rapidly when water tables rise due to wetter years. Subsequently, rapid evaporation and transpiration can remove soil water, and soluble salts become concentrated near the soil surface (LaBaugh et al., 1998).

Soil salinization can make a productive agricultural crop non-productive and reduce crop yield, consequently reducing the farmers' economic returns. The estimated annual income loss was $257 million to Canadian farmers due to the soil salinity (Forge, 1998). However, the risk of salinization in the Canadian prairies decreased between 1981 and 2011 primarily due to the decrease in summer fallow (7 million ha, 78 % reduction), and it increased in the area of permanent cover (4.8 million ha, 14 % increase) with a most substantial portion of the change in Saskatchewan (over 3 million ha) (Bock, 2016). Nonetheless, there is 4 million ha of salt-affected abandoned marginal degraded land across the Canadian prairies, which is unsuitable for arable crop production, including 1.6 million ha in Saskatchewan, which has potential for salt-tolerant SRW plantation (Amichev et al., 2014).

Reducing the risk of salinization and improving the condition of saline soil demands proper soil-water management. The primary requirement for saline soil rehabilitation and management is the leaching of soluble salts beyond the active root zone, which can be achieved by chemical amendments and engineering approaches such as surface and sub-surface drainage to control salt and water balance (Henry et al., 1987). In addition to the salty drainage water disposal, this approach requires high maintenance and capital investment. However, an alternative solution to this can be 'biodrainage,' i.e., the use of plant species with high growth capacity and deep root systems (Heuperman et al., 2002; Minhas & Dagar, 2016; Singh & Lal, 2018; Stirzaker et al., 1999). According to Miller et al. (1981), the possible best solution is to better utilize the soil water through several successful management practices such as 1) growing deep-rooted plants (e.g., perennial crops), 2) flexible intensive cropping systems (e.g., reducing summer fallow), and 3) draining selected upland, freshwater wetlands. Hence, the process of soil salinization associated with high water tables might be better managed using deep-rooted phreatophytic SRW agroforestry land-use practice (Dagar & Minhas, 2016). Moreover, using SRW plantation can be a nature-based solution for soil salinity management and the recovery of degraded lands (Fernandes & Guiomar, 2018). However, it is imperative to assess how SRW water use varies under specific site conditions, considering evaporative demands, soil type, depth to GWT, and linked salinity (Dimitriou et al., 2009; Minhas & Dagar, 2016).

Reducing soil salinization risk requires a spatial and temporal assessment of risks and further development and implementation of beneficial management practices (BMPs) (Bock, 2016). The inclusion of a new land-use practice, e.g., SRW – with high biomass production and deep rooting phreatophytic nature – in the wetland riparian zones (i.e., in the fringes of the wetland) requires precise knowledge of the spatial and temporal variation of the shallow GWT and salinity.

Therefore, this multi-year field study's objective was to assess whether the SRW plantation would affect GWT depth and associated temporal and spatial distribution of soil salinity, compared to adjacent annual crop (AC) and pasture (PA) in the marginal riparian zones of PPR wetlands.

2. Materials and Methods

2.1. Study Site

The experimental field sites were established in two adjacent PPR wetlands (distance between the boundary of the two sites is approximately 200 m) in Indian Head, Saskatchewan, Canada (N 50° 30.605' and W 103° 43.011') (Fig. 1). The approximate extent of the area (estimated from FlySask2.ca) of site A was 1.8 ha, and site B was 1.3 ha. Saskatchewan Soil Survey Staff (1986) described both sites' soil as non-calcareous Black Chernozems of the Oxbow Association, poorly drained in depressions, with level to gently rolling topography formed on loamy glacial till. The background soil salinity (0-60 cm depth) at site A (mean = 0.92, maximum = 2.11, and minimum = 0.41 mS cm^{-1}) was approximately half of that at site B (mean = 2.07, maximum = 3.39, and minimum = 0.44 mS cm^{-1}); salinity at both sites was within the range of previously reported values suitable for the adaptation and growth of SRW variety in the Canadian prairie (Hangs et al., 2011). Site A can be categorized as non-saline and site B as weakly saline (Saskatchewan Soil Survey Staff, 1986). The 30-year climate norms (1981 to 2010) were 428.4 mm for annual total precipitation (321.7 mm rainfall and 110.5 mm snowfall), and +2.7°C for average annual temperature, with minimum and maximum values of −20.1 and +25.0 °C respectively at the Indian Head, Saskatchewan, Environment Canada Climate Station (Environment Canada, 2020).

The SRW variety *Salix dasyclados* Wimm. (cultivar 'India') was planted adjacent to PA and AC in the riparian zones (Hayashi & Rosenberry, 2002) of both wetland sites on 10th and 11th June of

2013. The estimated planting area of SRW was 0.46 ha in site A and 0.52 ha in site B (Fig. 1). The dormant hardwood cuttings of SRW were approximately 25 cm long and planted by inserting into the rotovated soil under black plastic mulch in a double row design (13,300 plants ha^{-1}). The distance between double rows was 2 m, and the rows were 30 cm apart. No fertilizer was applied to the sites after SRW planting; however, Glyphosate (StartUp®; 540 grams acid equivalent per liter, present as potassium salt) was used two to three times during the growing season to inhibit weed growth between the rows. In the past, at both SRW sites, only barley and oats had been grown, and the land may have been under fallow once or twice during the ten years prior to SRW planting. Typical crops in the study sites are barley, oats, or flax, as the land area of both sites are slightly saline (Mirck & Schroeder, 2013, 2018). The SRW aboveground biomass components were manually harvested after three consecutive growing seasons in October 2015 and considered a 3-year non-coppiced rotation cycle (including the plantation year, i.e., 2013). Harvesting of SRW biomass was completed using brush saws in a 2-m by 2-m grid in each site with three replications.

The unmanaged PA had been established 10-12 years before starting this experiment with an alfalfa (*Medicago sativa*) and bromegrass (*Bromus madritensis*) mixture and typically only subjected to light grazing in early May for three to four weeks. During the study period (2013-2015), the AC land was seeded to oats (*Avena sativa*); during the previous ten years, the AC land had been cultivated with barley, oats, and flax. The oats received 100 lbs of 50-20-0 fertilizer at seeding. Glyphosate (StartUp®; 540 grams acid equivalent per liter, present as potassium salt) was applied before sowing, and Prestige™ was used on the crop in June for broadleaf (and volunteer flax) weed control at recommended rates. The oats were seeded between May 7-15 and harvested during September 5-20 each year.

2.2. Groundwater Table and Electrical Conductivity Monitoring

A total of 28 shallow GWT monitoring wells (15 in site A and 13 in site B) were installed along transects across the sites (Fig. 1). Each transect either extended from the field boundary to the wetland edge and/or from one wetland edge to another to cover all landforms and was parallel to the groundwater flow direction. The wells were constructed with PVC tubing (6-cm diameter, 2-m length) screened with 0.5 cm holes equally distributed throughout and sealed with a bottom cap (Supplementary Fig. 1). The entire PVC well was wrapped with porous woven fabric (nylon) to prevent intrusion of soil sediments into the wells during groundwater monitoring. Wells were installed in July 2013 by drilling a 7-cm diameter borehole to 2-m depth with a Giddings soil corer (Giddings Machine Company, Windsor, Colorado, USA) mounted onto a tractor (Model # 3120; John Deere, Moline, Illinois, USA). The PVC monitoring wells were then inserted into the borehole and sealed with 20 cm of pelletized bentonite and soil mixture around each well's stock. The opening of the well was covered with a loosely fitted detachable PCV cap. For the construction and installation of shallow groundwater wells and water table monitoring, procedures were followed as described in USACE (2005) and Sprecher (2008).

Depth to GWT was monitored using a Mini-Diver data logger (Schlumberger Water Services, Kitchener, Ontario, Canada), installed into each groundwater monitoring well. Groundwater table data were collected continuously (at 30-min intervals) throughout two consecutive growing seasons (May to September 2014 and 2015). Groundwater electrical conductivity (EC_{gw}) was monitored after collecting groundwater samples (see *groundwater and soil sample collection* section) from each monitoring well and measured in the laboratory using a PC700 pH/mV/conductivity meter (Oakton, Vernon Hills, IL, USA).

An on-site weather station (Campbell Scientific Canada, Edmonton, Canada) was installed to measure climatic variables during the growing seasons of 2014 and 2015. Air temperature and relative humidity were recorded on an hourly basis. Monthly total precipitation and average air temperature were calculated from the obtained climatic data.

2.3. Groundwater and Soil Sample Collection

Groundwater samples were collected (Supplementary Fig. 1) monthly using a Masterflex E/S Portable Sampler – 115 VAC (Cole-Parmer® Instrument Company, Montreal, QC Canada) as procedures described in Vail et al. (2013). Instantly before sampling, each groundwater monitoring well was purged using the peristaltic pump, after which it was allowed to recharge to a representative part of the groundwater. The groundwater samples were collected in 250-mL polypropylene sample bottles and placed in coolers for transport to the laboratory, where they were refrigerated at 4°C until analyzed (see *laboratory analyses* section for details).

Soil samples were collected in May, July, and September of 2014 and 2015 within 1-m radius of each monitoring well (Supplementary Fig. 1); background samples were collected in 2013 (i.e., the SRW plantation year) from the exact locations (before monitoring well installation). At each point, an auger was used to collect soil samples from two depth increments: 0-30 cm, and 30-60 cm. Samples were transferred into a Ziploc® bag, labeled, and stored temporarily in a cooler for transport to the laboratory, where they were refrigerated at 4°C until analyzed.

2.4. Laboratory Analyses of Groundwater and Soil Samples

All collected soil samples were subdivided into two portions. The first portion was kept intact (i.e., field moist sample); gravimetric water content was calculated from the weight loss of approximately 20-g soil sample oven-dried for 24 hours at 105°C in an aluminum tin (Topp et al., 2008). Soil bulk density was calculated from the ratio of the mass of oven-dried soil (at 105°C) to the bulk volume of core soil collected from the field at desired soil layer (Hao et al., 2008). The volumetric soil water content (VSWC) was calculated from gravimetric water content and soil bulk density. The second portion was air-dried, ground, and passed through a 2-mm sieve for particle size distribution, cation exchange capacity (CEC), pH, electrical conductivity (EC), and ammonium acetate extractable Na^+, Ca^{2+}, Mg^{2+}, and SO_4^{2-}. The particle size distribution was determined by the modified pipette method (Kroetsch & Wang, 2008). CEC was measured by the ammonium acetate methods at pH 7 (Hendershot et al., 2008a). Soil pH was determined in 20 mL of deionized water with 10 g of air-dried soil samples (2 : 1 ratio) by digital pH meter (PC700 pH/mV/conductivity, Oakton, Vernon Hills, IL, USA) (Hendershot et al., 2008b). The EC_{soil} determined from the same soil extract used for pH measurement after 1 hour of shaking with an end-over-end shaker, then filtered through the highly retentive filter (No. 42, Whatman Inc., Piscataway, NJ), and measured using a digital EC meter (PC700 pH/mV/conductivity, Oakton, Vernon Hills, IL, USA) (Miller & Curtin, 2008). The Na^+, Ca^{2+}, Mg^{2+}, and SO_4^{2-} were measured using a 1M ammonium acetate (buffered at pH 7) extraction (Hendershot et al., 2008a) and analyzed by atomic emission (Na^+) and by atomic absorption (Ca^{2+} and Mg^{2+}) spectroscopy (Varian Spectra 220 Atomic Absorption Spectrometer; Varian Inc., Palo Alto, CA, USA). The SO_4^{2-} analysis was done via microwave plasma-atomic emission spectrometry (4100 MP-AES; Agilent Technologies, Melbourne, Australia). Total dissolved salts (TDS) were calculated from the measured EC with a conversion factor of 640 for EC values between 0.1 to 5 mS cm^{-1}, and 800 for EC > 5 mS cm^{-1} (Bresler et al., 2012). Soil exchangeable sodium percentage (ESP) and sodium adsorption ratio (SAR) were calculated by following equations 3.1 and 3.2 (Abrol et al., 1988), respectively, as follows:

$$ESP = \text{Exchangeable } \frac{Na^{+}}{CEC} \times 100 \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots (1)$$

$$SAR = Na^{+} / \sqrt{Ca^{2+} + \frac{Mg^{2+}}{2}} \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots (2)$$

2.5. EM38 Survey for Soil Electrical Conductivity and Salinity Mapping

In September of 2014 and 2015, soil apparent electrical conductivity (EC_a) across both sites was measured with an electromagnetic induction meter (EM38-MK2; Geonics Limited, Mississauga, ON, Canada) attached to an All-Terrain Vehicle (ATV) (Supplementary Fig. 1). The EM38 is specially developed to measure EC_a with two modes: vertical (response comes from 1 m soil depth) and horizontal (response comes from 0.5 m soil depth) (McNeill, 1980). A vertical (up to 1 m) survey was conducted to capture EC_a of the root zone under all land-uses except PA in 2014 at both sites. Between-row EM38 measurements were taken under SRW and AC; however, for PA, 1-m spacing was maintained. All measurements were taken following standard calibration (McNeill, 1984). Data were calibrated through the correlation developed between temperature adjusted EC_a and salinity (EC) measured from the collected soil samples in the laboratory (soil to water ratio of 1:2) to improve the reliability. Obtained EM38 data were used to create soil salinity maps using ArcGIS v10.5 (ESRI, Redlands, California, USA) ordinary kriging (Cassel, 2007).

2.6. Elevation Survey and Mapping

Universal Transverse Mercator (UTM) coordinates and corresponding elevations were taken from the field experimental sites in a 1-m by 1-m grid using a Leica GS15 (Leica Geosystems, Heerbrugg, Switzerland) real−time kinetic Global Positioning System (rtkGPS). The elevation data were collected from three different land-use practices from both sites during the Fall of 2015. Based on the rtkGPS surveys, digital elevation models (DEMs) were created in ArcGIS v10.5 (ESRI, Redlands, California, USA) using ordinary kriging.

2.7. Statistical Analysis

Statistical analyses and data visualization were completed using R version 3.4.4 for Windows (R Core Team, 2018). Data visualization was performed through bar plots, box plots, and line graphs using the "ggplot2" package. Data were tested for normality by the Shapiro-Wilk test and histogram. Homogeneity of variances or homoscedasticity was tested by Levene's test using the "car" package. Pearson correlation among the measured groundwater and soil variables were performed using the "psych" package. When required, the square root transformation was performed to improve the assumption of normality and homoscedasticity of the data. Analysis of variance (ANOVA) with linear mixed-effects models was used to test for significant differences across land-use practices, years, and depths (for soil only) in measured variables using the "lmerTest". The mixed approach was selected due to its suitability for unequal variances, nestedness, and unbalanced design. Mean comparisons of quantified variables were accomplished using Tukey Honest Significant Differences (TukeyHSD) test via "TukeyC". A principal component analysis was performed using "FactoMineR" and "factoextra" to investigate the relationship among relevant groundwater and soil characteristics measured under different land-use practices, years, and depths (for soil only).

3. Results

3.1. Groundwater Table and Salinity

In terms of GWT, no consistent land-use patterns were observed between sites. The depth to GWT significantly varied among the land-use practices within site B ($p < 0.001$) but was not significant within site A ($p = 0.325$) (Table 1), indicating other factors were controlling the observed variability. The depth to GWT was significantly higher ($p < 0.001$) in 2014 than in 2015 under all land-use practices in both sites (Table 1 and Supplementary Table 1), suggesting that the rise in the GWT was possibly triggered due to the higher precipitation events throughout the wet year (i.e., during 2014) (Fig. 2). The spatial variation of depth to GWT was affected by wet versus dry years (i.e., wet = 2014, and dry = 2015), with the corresponding groundwater and EC_{soil} in both sites shown in Fig. 2. In both years, the most significant fluctuations in GWT depth were observed between June and August, when the precipitation was highest (Fig. 2), revealing that groundwater responded to precipitation patterns.

Like depth to GWT, the variations in EC_{gw} among land-use practices were significant ($p < 0.001$) (Table 1), but the effect was not consistent between sites (Fig. 2 and Supplementary Table 1), suggesting that other underlying soil factors perhaps controlled the observed variability. Site B showed higher EC_{gw} (two-fold) than site A. The EC_{gw} varied significantly ($p < 0.05$) between years and among months in site B, but not significantly ($p > 0.05$) in site A (Table 1).

Significant positive correlation (r = 0.40, $p < 0.001$) was observed between depth to GWT and EC_{gw} in site A, whereas the correlation was positive but non-significant (r = 0.14, $p > 0.05$) in site B, suggesting that the EC_{gw} and TDS increased with the increase in depth to GWT (Supplementary Fig. 2). The correlations between the depth to GWT and groundwater Na^+, Ca^{2+}, Mg^{2+}, and SO_4^{2-} were positive and significant ($p < 0.05$) in both sites (Supplementary Fig. 2). Significant positive correlations (r = 0.63, $p < 0.001$ in site A, and r = 0.18, $p < 0.05$ in site B) between the depth to GWT and elevation indicated that higher elevation resulted in the lowered GWT depth (i.e., higher depth to GWT) (Supplementary Fig. 2).

3.2. Soil Salinity

The EC_{soil} significantly differed ($p < 0.001$) among land-use practices (Table 2), but no consistent land-use patterns were observed between sites (Supplementary Table 2). Site B consistently showed higher EC_{soil} (two-fold) than site A. Total dissolved salts (TDS) followed identical patterns as measured EC_{soil} (Supplementary Table 2). The correlations among EC_{soil} and Na^+, Ca^{2+}, Mg^{2+}, and SO_4^{2-} were positive at both sites (Supplementary Fig. 3).

The spatial distribution of apparent soil electrical conductivity (EC_a) from EM38 survey showed higher soil EC_a in site B, which confirms higher overall soil salinity (i.e., EC_{soil}) compared to site A (Figure 3). However, correlations (r = 0.03 in site A, and r = 0.01 in site B) between site elevations and EC_{soil} were not significant ($p > 0.05$), suggesting that the soil salinity was not spatially dependant on the elevation at either of the experimental sites (Supplementary Fig. 4).

Positive correlations were observed between soil clay contents and EC_{soil}, however, significant in site A (r = 0.25, $p < 0.01$) and not significant in site B (r = 0.04, $p > 0.05$) (Supplementary Fig. 3). Significant ($p < 0.001$) negative correlations (r = - 0.60 in site A, and r = - 0.47 in site B) were observed between depth to GWT and VSWC in both sites (Supplementary Fig. 4), suggesting shallower GWT depth (i.e., high GWT) significantly increased the VSWC. The correlations between depth to GWT with EC_{soil} and TDS were positive and not significant (r = 0.10, $p > 0.05$) in site A, but positive and significant (r = 0.34, $p < 0.01$) in site B. Whereas the observed positive and significant (r = 0.57, $p < 0.001$ in site A, and r = 0.32, $p < 0.01$ in site B) correlations between EC_{gw} and EC_{soil} indicate that the raised EC_{gw} may cause an increased EC_{soil} (Supplementary Fig. 4).

A declining trend was observed in EC_{soil} with higher SRW biomass at the depth of 0-30 cm (R^2 = 0.56, p = 0.053 from site A, and R^2 = 0.81, p = 0.002 from site B), and 30-60 cm (R^2 = 0.02, p = 0.635 from site A, and R^2 = 0.64, p = 0.017 from site B) from both sites (Supplementary Fig. 5), suggesting that the SRW biomass could potentially reduce soil salinity. However, the inverse situation might be possible, i.e., the SRW biomass production was lower under high soil salinity.

4. Discussion

Within the PPR, the development of soil salinity is primarily due to the transport of naturally occurring salts in the near-surface by the capillary movement of soil moisture (Richardson et al., 1994; van der Kamp & Hayashi, 2009). In this study, the EC_{gw} at the two experimental sites ranged from 0.47 to 16.66 mS cm^{-1}, suggesting a naturally high salt load. However, the exact location and extent of soil salinity are highly dependent on the aquifer characteristics, including thickness and permeability, and the distribution of ET potential along with the slope position (Stolte et al., 1992).

The PPR wetland system's geochemistry is linked to the ET rate, recharge hydrology, ionic mobility, and exchange (Arndt & Richardson, 1988). Hence, groundwater discharge management in the low-lying areas in the PPR is crucial for limiting soil salinity (Henry et al., 1987). The relationship between EC_{gw} and EC_{soil} with TDS was 1:1 in this study, indicating elevated SO_4^{2-} concentrations in throughflow and discharge wetlands as suggested by (Arndt & Richardson, 1989). In contrast, recharge wetlands are free of calcite ($CaCO_3$) and gypsum ($CaSO_4.2H_2O$) and do not show similar geochemical properties. Accordingly, this study's soils contained a high amount of SO_4^{2-} in both sites (although site B contained 3 to 4 times more than site A); this indicated a discharge or throughflow wetland system.

The soil salinity (i.e., EC_{soil}) may increase in the near-surface (at the uppermost soil layer) if deep-rooted vegetation (e.g., natural willow ring in the prairie) is replaced by shallow-rooted crops, especially in the areas that contain natural deposits of salts (Henry et al., 1987). However, in this study, no consistent land-use pattern was observed in both experimental sites. Salinization is most likely to occur during the time of the year when evaporation exceeds the infiltration and percolation (Henry et al., 1987). Hence, the same situation may have prevailed in 2015 (relatively dryer year with higher EC_{gw}) in both of our experimental field sites.

Studies around the world have shown a substantial lowering of the GWT with high rates of ET under agroforestry plantation compared to adjacent grassland (Jobbágy & Jackson, 2004), grassland and cropland (Nosetto et al., 2012), and the presence of riparian willows (Doody & Benyon, 2011; Doody et al., 2007). In the PPR, phreatophytic native willows are commonly found in the discharge areas, commonly referred to as 'willow rings' (Meyboom, 1966b). The average water table fluctuation caused by native willow rings in Saskatchewan has been reported as 3 cm day^{-1} in the south-central region (Meyboom, 1966a), and 5 – 10 cm day^{-1} in the south (Mirck & Schroeder, 2018). On the other hand, the root system of SRW is relatively shallow (around 50 cm from the surface) and more concentrated near the soil surface, and the average transpiration rate of SRW stays between 30 to 45 cm per year (Dimitriou et al., 2009). In this study, the SRW stands had not yet achieved their full potential biomass development during the first rotation cycle; therefore, the SRW would not have had a significant impact on the depth of GWT as it would not yet have achieved its full ET potential.

In long-term monitoring of water table and soil salinity build-up from southern Saskatchewan, Canada, conversion from AC to deep-rooted perennials *Bromus inermis* Leyss. (bromegrass) and *Medicago sativa* L. (alfalfa) caused the PPR wetlands to dry up within four to six growing seasons (van der Kamp et al., 1999). However, the neighboring wetlands with natural willow rings in the riparian zone and the cultivated area did not alter the water table levels. Furthermore, the deep-rooted perennial grasses in the permanent cover transpired most of the water, mainly trapped from snow, snowmelt infiltration, and rainwater (during summer months) into the soil (van der Kamp et al., 2003). On the other hand, land-use practices such as continuous cropping and permanent cover (e.g., growing perennial forage) in the upland area can prevent soil salinization in adjacent low-lying areas by limiting the amount of water leaching through the soil (Eilers et al., 1997).

Land-use-driven vegetation changes especially shifts between agroforestry plantation and grassland can alter water balances and soluble salt fluxes. In a study with phreatophytic plantations and adjacent native grasslands in Pampas, Argentina, Jobbágy & Jackson (2007) found that the phreatophytic discharge could control solute transfers from groundwater via 1) affecting solute transport to the rooting zone by altered groundwater flow within the aquifer, and 2) concentrating solutes in the rooting zone by water uptake plus solute exclusion. Nosetto et al. (2013) similarly observed solute exclusion as a dominant salinization mechanism under tree plantations. However, soil salinization processes linked with enhanced groundwater consumption from agroforestry plantations discussed here in the literature are mainly associated with tree age that was > 12 years old. In contrast, in our study, the SRW plantation age was within the first rotation (i.e., 3-year rotation cycle) and may not have fully developed robust root system, hence less impact on soil salinization.

In the long run, the surplus water from adjacent cropland could maintain sustained groundwater supply where the agroforestry plantation trees are spread to maximize the capture of excess water from the surrounding landscape (Heuperman et al., 2002). Establishing deep-rooted plants under dry climatic conditions can significantly lower the GWT and, as a result, can reverse the natural process of soil salinization (Nosetto et al., 2008; Schofield, 1992). Consequently, the fundamental intention for saline soil management is to move salt downward from the root zone by lowering the water table (removing excess water) via the establishment of suitable land-use practice (i.e., use of deep-rooted salt-tolerant perennial vegetation), which has a higher transpiration rate, e.g., SRW (Mirck & Schroeder, 2018; Mirck & Zalesny, 2015). In this way, EC_{soil} and the salts (TDS) can be reduced in the 0-30 cm rooting zone and can improve the soil conditions that allow better crop production. Hence, the most cost-effective and environmentally sustainable, and nature-based option to manage salts and excess water in the discharge area in a wetland system could be the 'biodrainage' approach (Heuperman et al., 2002; Minhas & Dagar, 2016).

Soil salinity management in the wetland discharge area is challenging, particularly in the semi-arid climate and with glacial geology that results in unique hydrological conditions and distributions of naturally accumulated salts in the subsurface (Nachshon et al., 2013). The SRW land-use practice has the phreatophytic nature and high biomass production capacity that can better utilize the excess water in marginal riparian areas to reduce salinity in the semi-arid PPR (Eilers et al., 1997; Hangs et al., 2011; Mirck & Schroeder, 2018; Stolte et al., 1992). Therefore, as a supplement to the growing deep-rooted other perennial plant species, planting SRW either as agroforestry or as a riparian buffer in the degraded marginal land could be part of the beneficial management practices and nature-based solution, e.g., as a biodrainage option. However, this study's findings suggest that this management strategy's full potential would only be realized over the long term as SRW stands mature.

5. Conclusions

The land-use practices significantly impacted the GWT depth in site B, but not in site A, indicating inconsistent land-use effect patterns between sites. The higher precipitation events throughout the wet year (i.e., during 2014) resulted in a shallower depth to GWT across land-use practices. The EC_{gw} among land-use practices varied significantly. The EC_{soil} varied among land-use practices but was inconsistent between sites, whereas there were no observed variations among depths, years, and months in both sites. Field observation exhibited a declining trend in EC_{soil} with increasing SRW biomass at both depths (i.e., 0-30 and 30-60 cm) from both sites; however, given the lack of consistent patterns, it is more likely that the SRW biomass production was lower in locations with inherently higher soil salinity.

Despite the anticipated high-water consumption, the plantation of SRW in the riparian zones had a minimal drawdown impact on the GWT during this study period. Our experimental situation is attributable to plantation stands that had not yet achieved their maximum potential biomass production during the first rotation cycle. Therefore, salinity management through 'biodrainage' model might not be efficient during the first rotation cycle because the SRW plantation has not yet achieved its full ET potential and inevitably an ineffectual impact on the GWT fluctuation. Yet, over the longer term, SRW plantations have vast potentials to reduce and manage salinity at the surface soil (0-60 cm) in the areas of low to moderate salinity (EC_{soil} 1-6 mS cm^{-1} range) as a part of the best management practices and nature-based solution, particularly on the degraded marginal riparian lands within the PPR.

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List of Tables

Table 1 Analysis of variance (ANOVA) for measured depth to GWT and EC_{gw} from three different land-use practices at two sites during the growing season of 2014 and 2015.

Response variable	Sources of variation		Site A		Site B	
		df	*F* - value	*p* - value	*F* - value	*p* - value
Depth to GWT	Land-use	2	1.232	0.325 ^{ns}	10.281	<0.001 ***
	Year	1	44.091	<0.001 ***	43.737	<0.001 ***
	Month	4	2.305	0.061 ^{ns}	1.133	0.345 ^{ns}
EC_{gw}	Land-use	2	16.859	<0.001 ***	24.353	<0.001 ***
	Year	1	0.607	0.437 ^{ns}	5.167	0.025 *
	Month	4	2.062	0.089 ^{ns}	3.637	0.008 **
TDS	Land-use	2	16.912	<0.001 ***	10.281	<0.001 ***
	Year	1	0.540	0.464 ^{ns}	43.737	<0.001 ***
	Month	4	2.195	0.073 ^{ns}	1.133	0.345 ^{ns}

[a] *, **, *** Indicate there is a statistically significant difference at $p \leq 0.05$, $p \leq 0.01$ and $p \leq 0.001$ level of significance, respectively; ^{ns}, is not significantly different ($p > 0.05$).

[b] EC_{gw} = groundwater electrical conductivity, GWT = depth to groundwater table, TDS = total dissolved salts

Table 2 Analysis of variance (ANOVA) for measured VSWC, EC_{soil}, TDS, ESP, and SAR in soils from different land-use practices from two sites during the growing season of 2014 and 2015.

Response variable	Sources of variation	Site A			Site B	
		df	*F* - value	*p* - value	*F* - value	*p* - value
VSWC	Land-use	2	2.419	0.092 ^{ns}	3.901	0.022 *
	Depth	1	19.040	<0.001 ***	26.384	<0.001 ***
	Year	1	4.023	0.046 *	3.489	0.064 ^{ns}
	Month	1	5.411	0.005 **	1.059	0.349 ^{ns}
EC_{soil}	Land-use	2	8.961	<0.001 ***	63.357	<0.001 ***
	Depth	1	0.997	0.320 ^{ns}	2.519	0.115 ^{ns}
	Year	1	0.776	0.380 ^{ns}	0.001	0.978 ^{ns}
	Month	2	1.218	0.298 ^{ns}	1.718	0.183 ^{ns}
TDS	Land-use	2	8.961	<0.001 ***	63.357	<0.001 ***
	Depth	1	0.997	0.320 ^{ns}	2.519	0.115 ^{ns}
	Year	1	0.776	0.380 ^{ns}	0.001	0.978 ^{ns}
	Month	2	1.218	0.298 ^{ns}	1.718	0.183 ^{ns}

[a] *, **, *** Indicate there is a statistically significant difference at $p \leq 0.05$, $p \leq 0.01$ and $p \leq 0.001$ level of significance, respectively; ^{ns}, is not significantly different ($p > 0.05$).

[b] VSWC = volumetric soil water content, EC_{soil} = soil electrical conductivity, TDS = total dissolved salts, ESP = exchangeable sodium percentage, SAR = sodium adsorption ratio.