MONITORING AND MODELING SOIL SALINITY DYNAMICS AT GREENHOUSE AND FIELD SCALES

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GREENHOUSE AND FIELD SCALES

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An integrated monitoring and modeling approach was undertaken in greenhouse and field scale in terms of three main studies. In the first study, the changes in soil salinity and several other chemical properties were investigated in an irrigation district during a period that experienced severe drought followed by above-normal precipitation. Soil salinity, represented by the electrical conductivity (EC) of the saturated paste, decreased for the top layers and increased for the bottom layers during the study period, suggesting that some level of leaching occurred. However, the change in EC was not statistically significant when averaged over the top 1.5 m of the soil profile.

In the second study, a greenhouse experiment was conducted followed by numerical simulation to study performance of HYDRUS model in simulating the measured data. Six scenarios were defined based on irrigation water salinities and leaching application in soil columns under Bemudagrass. Soil profile salinity increased by increase in irrigation water salinity. Fresh water leaching was able to reduce soil salinities, which is a valid technique to offset the hazardous effects of saline irrigation and enable utilization of more saline irrigation water that are generally considered unsuitable for irrigation. HYDRUS simulation model performance was acceptable in predicting observed data.

In the third study, the potential land application of saline water was studied using the validated model from the second study. Using HYDRUS model, a combination of 36 scenarios were simulated using various irrigation water quality and quantity in three different locations of Panhandle, Central, and Southwest Oklahoma. Based on the results of this experiment, similar to the second study, soil solution salinity increased by increase in irrigation water salinity. Hydrologic characteristics of the land applied location had a significant effect on the amount of salts accumulated in the root zone. Elevated precipitation in Central Oklahoma made it more suitable location for discharge of saline water. However, hyper-saline water added extended amounts of salts to the soil and downstream water resources. Further studies are required to investigate the environmental sustainability of the saline irrigation considering the impacts of salts released to the environment which can contaminate soil and downstream water resources.

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INTRODUCTION

Background

Ever increasing world population has intensified the competition over natural resources between agriculture, industry, municipalities, and the environment (Ayars, et al., 2015). The world population is anticipated to reach 9.3 billion by 2050 (Bruinsma, et al., 2003). In order to feed this ever-increasing population more crop production is required. On the other hand, land is being degraded at a rate of 6-7 million hectares per year (Bullock, et al., 2005). Currently about 29% of the global land, which is home to 3.2 billion people, is degraded (Nkonya, et al., 2016). The exact number of people affected by land degradation is larger as many people directly or indirectly depend on goods and services provided by the degraded land. The gap between increasing need for agricultural products to supply the food and feed demands on one side and the loss of land to degradation can only be filled with higher yields and intensified, yet environmentally sustainable, agricultural production practices.

Irrigation has enabled increased yields and multiple cropping in traditionally single-cropped regions (Ghassemi, et al., 1995). This in return has provided a sense of stability and security for utilization of expensive fertilizers and pesticides, ensuring the crop growth even in areas with unstable and uneven precipitation patterns (Hillel, 2000). Despite the above-mentioned benefits, irrigation has negative aspects to it. The competition over fresh water resources due to the need to feed a growing population and the lack of good quality water, has made it inevitable to seek for alternative irrigation water resources which are of lower quality and have higher salt content (Berezniak, et al., 2017). Use of marginal quality irrigation water, however, adversely affects the crop productivity due to its high salt content. In most arid and semi-arid areas in United States the salts present in irrigation water include chlorides, carbonates, sulfates, and bicarbonates of calcium, sodium, and potassium. While an optimal level of soil salinity and sodicity can improve soil structural properties through aggregation and stabilization of soil structure, high salinity and sodicity are harmful and even lethal to plants and makes soil water unavailable for root uptake (Olson, 1997; Pearson et al., 2006). Sodicity which refers to the amount of sodium present in the soil solution, has a significant impact on soil structural stability. Even the best quality irrigation water contains some salt which introduces soluble salts to the soil. Salts tend to accumulate in the root zone, when the water is evapotranspired, which leads to soil salinity and sodicity (Crowin, et al., 2007).

Human induced salinization, also called secondary salinization, ages back to the start of irrigation application (Ghassemi, et al., 1995). The secondary salinization is mainly driven by development of irrigation facilities. It is also affected by activities like land clearing and replacement of native vegetation with short rooted agricultural crops, which led to dryland salinization. Human induced salinization is fast to develop and slow and expensive to reclaim. In the early 1990s around 45 million ha of irrigated land was salt spoiled. This area has increased to more than 62 million ha (20%) of the world’s 310 million ha irrigated land area (Ghassemi, et al. 1995). During the last 20 years, each day more than 2000 ha of irrigated land in arid and semi-arid areas, stretched over 75 countries, has been lost to salinization (Qadir, et al., 2014). Approximately 1.5% of the irrigated land permanently loses its productivity due to salinization, annually (Zeng, et al., 2014). Salt degradation is specific but not limited to arid and semi-arid areas, where rainfall is not enough to percolate through soil profile and leach salts below the root zone.

Globally more than $441 per hectare was lost to salt induced degradation in 2013, which yields around $27.3 billion in annual global loss (Qadir, et al., 2014). For example, in the Colorado River basin, salt induced degradation in irrigated land led to $750 million economic loss annually. These costs would be even higher, if the extensive costs associated with losses in property and business value in areas close to salt degraded farms, infrastructure deterioration such as roads and buildings, and the social costs are taken into account. Furthermore, there is an environmental cost associated with the extended water and wind erosion from salt induced farms. Although there are no metrics to take into account the sustainability and social aspects of the salt induced soil degradation, it is important to recognize the extent to which society, the environment, and economy is affected by it.

All irrigation water resources contain soluble salts that can accumulate in the root zone. The salt accumulation depends on many factors like the quality of the applied water, precipitation, evapotranspiration, and leaching. The extent of soil salinization and thus its negative effect on crop productivity can be controlled by irrigation management and remediation techniques (Corwin, et al., 2007; Yurtseven, et al., 2013). Mathematical models have been developed to predict soil water and salute dynamics. These models consider variables such as soil type, climate, and crop factors and are considered useful tools in evaluating different management strategies to control salinization (Yurtseven, et al., 2013).

Statement of the Problem

Salinity affects physical soil properties by binding fine particles together and forming aggregates. This process is called flocculation and improves soil properties through increased soil aeration, root penetration, and growth (Warrence, et al., 2002). Although soil salinity improves soil structural properties through aggregation and stabilization, high salinity is harmful and even lethal for plant growth, therefore, soil salt levels should be optimized (Olson, 1997). Soil salinity, particularly in arid and semi-arid areas like western Oklahoma, restricts crop growth by increasing the osmotic pressure of the soil water and thus making it more difficult for the roots to uptake water and nutrients. Furthermore, some salts cause ion toxicity in the plants and disturb the nutritional balance that naturally exists.

Concerning the crucial role of irrigation in intensified agriculture on one hand and its contribution to salinization and thus declines in crop productivity on the other hand, there is a gap in knowledge on how to both control salinization and maintain irrigated agricultural productivity. Different aspects of this problem have been investigated in the past. However, there are lack of solutions that can be applied in different regions with variable agro-climatological conditions. The proposed research is focused on this knowledge gap and attempts to investigate different strategies to cope with soil salinization in irrigated agriculture. This research is innovative in terms of its inclusion of soil salinity dynamics in the root zone in periods of extreme weather events. As a changing climate is expected to bring a higher likelihood and frequency of extreme events, it is of great importance to study the impacts of intense dry and wet cycles on salinity variations in the crop root zone. This study will also provide farmers with practical information to prepare for and cope with salinity without halting the agricultural production or jeopardizing the sustainability of irrigated agriculture.

Many methods have been developed to determine the impact of irrigation water quality and irrigation management on salt dynamics in the root zone. The most precise method is taking soil samples from different locations and running soil tests in the laboratory. This technique, however, is time consuming, expensive, and labor-intensive. Moreover, it is difficult to take samples at the quantity and quality that would represent the soil conditions of the entire study area (Rasouli, et al., 2013). Therefore, the most economically viable method is to use a combination of monitoring and modelling techniques to assess soil salinity dynamics in response to irrigation regimes for better decision-making (Yurtseven, et al., 2013). Numerous mathematical models have been developed in the past to predict water and solute transfer in the soil. The HYDRUS model developed by Šimůnek, et al. (2008) is one of the most commonly used analytical tools for simulating the water movement and solute transport in the soil.

This study focuses on the effect of various irrigation water qualities and quantities on soil salinity. The potential land application of various qualities of saline water is studied using the HYDRUS model. Potential surface discharge of produced water (PW), taking into account the salinity of PW in Oklahoma, is an emerging concern and therefore addressed in this study. Optimization of model parameters for conditions in Oklahoma such as soil type, climate, irrigation water quality, and main crop types have not been conducted to the best of our knowledge.

Goal and Objectives

The goal of this research is to study soil salinity dynamics under variable climatological, agricultural and management conditions. In this regard, specific objectives are defined as follows:

1. To study variations in root zone salinity as impacted by periods of extreme dry and wet conditions using monitoring methods at field and irrigation district scales.
2. To investigate the effects of irrigation water quality and leaching on soil salinity using monitoring and modelling techniques at the greenhouse scale.
3. To compare different scenarios of land application of saline water under variable climatic conditions, saline water quality and quantity using the validated model.

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CHAPTER II

SOIL SALINITY VARIATIONS IN AN IRRIGATION DISTRICT DURING A PERIOD OF EXTREME DRY AND WET CYCLES

Abstract

Salinization of irrigated lands is a major challenge in supplying required food and feed sources to meet the increasing global population. In this study, the changes in soil salinity and other chemical properties were investigated in an irrigation district during a period that experienced severe drought followed by above-normal precipitation. Soil salinity, represented by the electrical conductivity (EC) of the saturated paste, decreased for the top layers and increased for the bottom layers during the study period, suggesting some level of leaching occurred. However, the change in EC was not statistically significant when averaged over the top 1.5 m of the soil profile. The change in exchangeable sodium percentage (ESP) was not significant over the study period either. In contrast, average pH and calcium concentrations increased and decreased significantly during the study period, respectively. EC and ESP data were used in soil classification. Sixty percent of all sampled sites were classified as saline at the beginning of the dry-wet period and dropped to 50% at the end of this period. All tested parameters were temporally stable, preserving their spatial rank during the study period. Overall, four years of extreme drought with no irrigation application succeeded by a period of above-normal rainfall reduced soil salinity in the top 90 cm of the soil and moved salts downward to the deeper soil layers. This reduction in surface soil salinity, although statistically insignificance, is beneficial for seedling establishment. However, levels of pH, EC, and ESP appear to be high and may cause yield loss at some of the sampling locations.

Keywords: Electrical conductivity, Sodicity, Cotton, Drought, Oklahoma

Introduction

Ever increasing world population in recent decades has led to intensification of competition over natural resources between agriculture, industries, and municipalities. In addition, crop production must increase to provide food for the growing population, mainly through increasing crop intensity. Irrigation has enabled increased yields and multiple cropping in traditionally single-cropped regions (Ghassemi et al. 1995). This in return has provided a sense of stability and security for utilization of fertilizers and pesticides, ensuring a viable crop yield even in areas with unstable and non-uniform precipitation patterns (Hillel 2000). Despite the above-mentioned benefits, irrigation can have negative effects. One potential adverse impact is related to salt concentrations. Salts and nutrients in the water are added to the soil during each irrigation event. In addition, the salts already present in the soil are mobilized in the irrigation process. Irrigation can also raise the water table and bring the salts in the groundwater to the root zone, affecting crop growth (Hillel 2000; Tanji and Kielen 2002; Kijne et al. 1988).

In the early 1990s around 45 million ha of irrigated land was salt spoiled (Oldeman et al. 1990). This area has increased to more than 62 million ha, or about 20% of the world's total irrigated area (Ghassemi et al. 1995; Qadir et al. 2014). According to Qadir et al. (2014), more than 2000 ha of irrigated land has been lost to salinization every day during the last 20 years. The estimated economic loss was more than $441 per hectare in 2013, which totals around $27.3 billion in annual global loss. These loss estimates would have been even higher if the extensive costs associated with losses in property and business values, infrastructure deterioration, and the negative impacts on social structure and stability of the communities were taken into consideration. Furthermore, there is an environmental cost associated with the increased water and wind erosion from salt induced farms.

Soil remediation for removal of salts from the root zone is both costly and time consuming and, in many cases, the damage is irreversible. Therefore, the economic and sustainable solution to the salinity issue is to prevent rather than cure (Ghassemi et al. 1995). Effective prevention (or minimization) requires a comprehensive understanding of salt dynamics and responses to spatially variable agricultural, climatological, and hydrological characteristics. Chang (2007) categorized the scale of spatial and temporal variations in water dependent properties of soil in four groups of micro (laboratory), macro (greenhouse), mega (field), and system (watershed/district). Previous studies have investigated the complex solute dynamics in the soil profile, with a major focus on short term responses and micro to macro scales of controlled environments (Feikema and Baker 2011; Razzouk and Whittington 1991; Tanton et al. 1995). Such studies have provided in-depth information about salt movement in the soil. However, this knowledge may have limited practical applications as some of the major factors that complicate salt dynamics under natural, uncontrolled conditions of large-scale ecosystems were ignored (Armstrong et al. 1996).

Some researchers have conducted long-term salinity studies at larger geographical scales (e.g. irrigation district). Ballantyne (1978) analyzed EC of soil profiles at 64 sites for a period of 11 years and found salt movement occurred well below the apparent root zone. They concluded that the annual net salinity change was site-specific and further studies were required to understand the trend of salt change and reasons behind it. They also pointed out that monitoring salinity at specific sites over the long term could help better understand salt movement patterns in soil profiles. Herrero and Pérez-Coveta (2005) assessed variations in soil salinity and sodicity of an irrigation district in northeast Spain during a 24-year period from 1975 to 1999. Considering the sampling period and large number of soil samples, they were able to determine a general desalinization trend which increased by soil sampling depth. However, the lack of data on crop management practices prevented them from determining what triggered the observed pattern. In Hetato Irrigation District in China, Wu et al. (2008) studied long-term changes in salinity and reported that installation and improvement of drainage systems at district and field scales along with maintaining a large portion of fallow fields led to successful salinity control in the cropped areas.

Although these few studies have provided valuable information about long-term soil salinity dynamics at large scales, none of them (to the best of our knowledge) has included periods of extreme weather events such as droughts and floods. As a changing climate is expected to bring a higher likelihood and frequency of extreme events, it is of great importance to study the impacts of intense dry and wet cycles on salinity variations in the crop root zone. The present study explored the dynamics of soil salinity and associated chemical properties across an irrigation district in southwest Oklahoma over eight growing seasons. About half of this period was characterized by an extreme drought, in which irrigation application was halted due to unavailability of water supply. This dry period was followed by historic precipitation events that replenished local water resources in a short period. The research hypothesis was that this no-irrigation period followed by intensive precipitation should have a significant influence on leaching salts and creating a more favorable condition for crop production.

Material and Methods

Study Area

The area studied was the Lugert-Altus Irrigation District (LAID), which occupies over 190 km^2 in southwestern Oklahoma (Autbee 1994). An irrigation water right was obtained by the LAID from the State of Oklahoma in 1939, allowing the district to use up to 105 million cubic meter per year from the North Fork Red River for irrigation purposes (Autbee 1994). This makes LAID the largest surface water irrigation district in Oklahoma. The water is stored in Lugert-Altus Reservoir (Lake Altus) and released during the growing season for agricultural irrigation through a network of main canals (83 km) and laterals (351 km), while open drains (42 km) provide the required water removal from the crop root zone (USBR 2005). Figure 2.1 demonstrates the location of LAID and its water reservoir in southwest Oklahoma, along with the irrigation canal network and the sampling locations used in the present study. The average size of fields within LAID is 53 hectare (ha) and most of them are flood irrigated using furrow system. Upland cotton is the most dominant crop in the study area, with a growing season that spans from May to September and an irrigation season from early July to late September in most years.

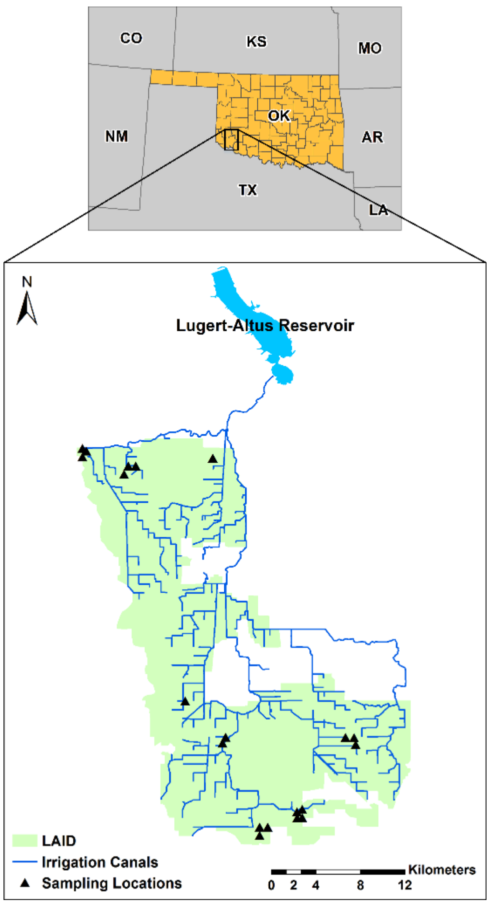

Figure 2.1. The Lugert-Altus Irrigation District (LAID) in southwest Oklahoma. Sampling locations are also identified within LAID.

The two major soils in the study area are Hollister silty clay loam (fine, smectitic, thermic Typic Haplusterts) and Roark loam (fine, mixed, superactive, thermic Pachic Argiustolls), which account for 41% and 15% of the total LAID area, respectively. Hollister soils have the parent material of Calcareous clayey alluvium. A typical profile for this soil is silty clay loam for the top 0.2 m of the soil, followed by silty clay to 0.4 m depth and then clay for deeper layers. The soils are naturally non-saline to slightly saline with moderate amounts of total available water. At an elevation varying from 350 to 500 meters (Evers *et al*. 1998), LAID has a sub-humid climate with hot and dry summers. Table 2.1 summarizes average growing season (May to September) and annual weather parameters for a 20-year period.

In 2011, southwest Oklahoma experienced record low precipitation (262 mm) and entered a period of severe drought. This drought led to drastic reductions in Lake Altus water levels, which serves as the sole source of water for LAID. As a result, the release of water to LAID irrigators was terminated for the first time in its history. This in return led to the decline of irrigated area to near zero (Taghvaeian et al. 2015) and had a devastating impact on the regional cotton industry. Heavy spring rains in 2015 ended the drought and refilled the reservoirs (Krueger et al. 2017). In May 2015 alone, 281 mm of rainfall was recorded at an Oklahoma Mesonet weather station (Altus) located within LAID. This was 3.5 times greater than the 20-year average participation of 80 mm for the same month and weather station.

Table 2.1.

Average growing season (May to September) and annual meteorological parameters for the period of 1997-2016.

Parameter	**Growing season**	**Year**
Total Prec.[a] (mm)	322	610
Mean R_s[b] (MJ m^{-2})	21.8	17.0
Min T_{air}[c] (°C)	18.6	9.4
Max T_{air} (°C)	32.2	23.3
Mean T_{air} (°C)	25.2	16.1
Min RH[d] (%)	34.4	37.2
Mean VPD[e] (kpa)	1.6	1.0
Mean U_2[f] (m s^{-1})	3.0	3.2

[a] Precipitation

[b] Total daily accumulation of solar radiation

[c] Air temperature

[d] Relative humidity

[e] Vapor Pressure Deficit

[f] Wind speed at 2.0 m above the ground

Soil Sampling Procedure

Soil sampling was conducted at two times, covering a period of over eight years. The first sampling took place in October 2007, when soil cores were extracted at twenty locations across the LAID using a deep soil core sampler (Giddings Machine Company, Inc., Col., USA). Each core was 1.5 m deep and was divided into five sub-cores of equal lengths upon extraction of the core. The sub-cores represented soil layers 0.0-0.3, 0.3-0.6, 0.6-0.9, 0.9-1.2, and 1.2-1.5 m. The sampling depth of 1.5 m was selected because it represents the active crop root zone, where soil salinity has the largest impact on crop yield. In addition, pushing the sampler to deeper layers could have caused soil compaction and consequently errors in salinity estimates of each sub-core.

The Global Positioning System (GPS) coordinates of all sampling locations were recorded and used in February 2016, when the same locations were visited, and the same procedure was followed to take soil cores. The spatial distribution of sampling points covered the entire irrigation district, with seven, six, and seven samples taken from the north, central, and south regions, respectively (Figure 2.1). Out of the twenty sampling locations, twelve were classified as Hollister soil and four as Roark soil.

Soil Testing Methods

Soil samples were oven dried overnight at 65 ºC and ground to pass through a 2-mm sieve. Soil pH and salinity parameters were determined using the 1:1 water to soil extraction and converted to the saturated paste equivalent based on the conversion factors described by Richards (1954). Briefly, 100 g of oven-dried soil was mixed with 100 mL of deionized water (USDA 1954). After reaching equilibrium (about 4 hours), the suspension was extracted using a low-pressure filter press apparatus. The electrical conductivity (EC) and pH were measured by a conductance cell and pH electrode. An inductively coupled plasma (ICP) spectrometer (SPECTRO Analytical Instruments

GmbH, Germany) was used to quantify the amounts of boron (B), potassium (K), calcium (Ca), magnesium (Mg), and sodium (Na) in the extract (Soltanpour *et al*. 1996). Exchangeable Sodium Percentage (ESP) was calculated using the equations provided in USDA (1954).

Statistical Analysis

All statistical analysis for significant differences were conducted using the General Linear Model procedure in Minitab V.13 (Minitab Inc., Pen., USA). Analysis were based on One-way ANOVA along with Tukey's pairwise comparison test at a family error of 0.05 (95% confidence interval). The General Linear Model is a flexible and useful statistical model as it assumes an exponential family model for the response. Once a significant difference between groups was found, the Tukey's test was used to determine where the significant difference lied.

In addition, two approaches were implemented to assess the temporal stability of measured soil parameters during the course of this study. The first approach was the nonparametric Spearman's rank correlation outlined in Vachaud *et al*. (1985). In this test the degree of change in the ranking of each sampling site compared to a previous sampling date is determined by estimating the Spearman's rank correlation coefficient (r_s) as

$$r_s = 1 - \frac{6\sum(R_{i,j} - R_{i,j'})^2}{n(n^2-1)} \qquad (2.1)$$

where $R_{i,j}$ is the rank of the measured parameter at the site i and time j (2007 in the case of this study), $R_{i,j'}$ is the rank of the same site at sampling time j' (2016), and n is the number of samples (20). A r_s value of unity indicates no change in the ranking, or otherwise a perfect temporal stability of the parameter of interest. The estimated r_s values are compared against the critical r_s to identify their statistical significance. In this study, the critical r_s was determined as 0.447 and 0.570 at the significance levels of 0.05 and 0.01, respectively (Ramsey 1989).

The second approach was based on the linear regression between measured soil properties at two sampling periods as explained in Kachanoski and Jong (1988) and Douaik *et al*. (2007):

$$Z_{i,j'} = I_{i,j} + S_{i,j} Z_{i,j} \tag{2.2}$$

in which, I is the intercept, S is the slope and *Z* is the soil property of interest measured at location *i* and two times of *j* and *j'*. Douaik *et al*. (2007) defined four different scenarios based on the possible values of I and S:

i. I=0 and S=1: There is no change in the measured soil property by time (perfect stability),

ii. I≠0 and S=1: The mean soil property changed by time, and the change was spatially uniform (static),

iii. I=0 and S≠1: The mean soil property did not change by time, and changes at different locations were non-uniform (dynamic),

iv. I≠0 and S≠1: The mean soil property changed by time and the change was non-uniform (dynamic).

Results and Discussion

Variations in Soil Chemical Properties

Variations in pH, concentration of several major ions, exchangeable sodium percentage, and electrical conductivity of the soil extracts were investigated at different soil layers during the study period to identify potential impacts of wet and dry cycles on profiles of these parameters in irrigated soils of the study area.

pH

Soil pH is an important parameter as it impacts the availability of nutrients (Thomas 1996). High pH levels can lead to decreased availability of positively charged ions, while negatively charged ions become more soluble (Wallender and Tanji 2011; Bohn *et al*. 2001; Kent and Lauchli 1985). In this study, pH values had a range of 6.7-9.3 in 2007 and 7.3-9.4 in 2016. The average pH significantly increased from 7.7 to 8.1 during the study period ($p = 0.01$). For cotton production, the desired range of root zone pH is between 5.6 and 8.0, with an optimum range of 6.0 to 6.5. The pH had small variations among the soil layers and these variations were not statistically significant on either sampling dates (Figure 2.2). The change in pH over time was depth dependent, ranging from a 0.5 unit increase for the shallowest layer to a 0.2 unit increase at the 0.9-1.2 m layer. This change was statistically significant for the top three layers (0.0-0.9 m).

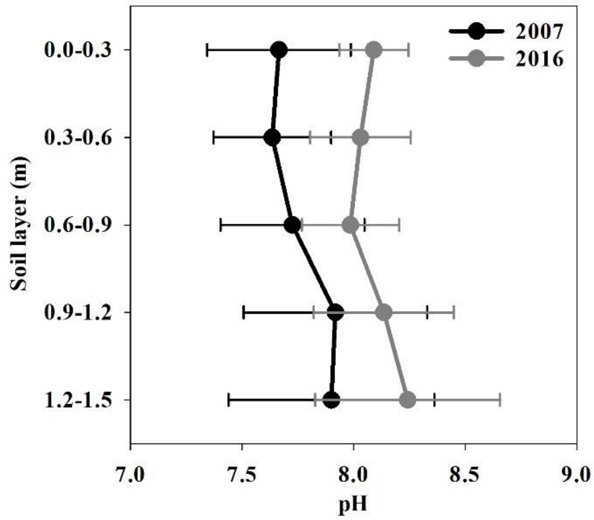

Figure 2.2. Mean pH for each soil layer at the two sampling dates in 2007 and 2016. Error bars represent one standard deviation from the mean.

Boron

Boron (B) is an essential micronutrient for crop production. However, it has a narrow range of optimum concentration in soil solution before it becomes deficient at lower levels or toxic at higher levels than the desired range (Wallender and Tanji 2011). In the case of cotton, B deficiency is a main limiting factor in the US (Ayers, 1985), especially since cotton is very tolerant to B toxicity (Maas 1984). In this study, B concentrations had a similar range (0.0-1.9E-4 mol L^{-1}) and average (3.6E-5 mol L^{-1}) in both 2007 and 2016. The profiles of B at each sampling date are presented in Figure 2.3. When considering B profiles, concentrations were smallest for the top two layers and then increased with depth on both sampling dates. Boron exists in the neutral boric acid form, so it is not adsorbed by charged soil colloids. In 2007, the minimum B was observed at the top two layers of 0.0-0.3 and 0.3-0.6 m with average values of 1.8E-5 and 1.6E-5 mol

L^{-1}, respectively. The concentrations increased by depth to the maximum average of 5.6E-5 mol L^{-1} at the deepest layer (Figure 2.3). In 2016, the average B was 1.6E-5 and 2.1E-5 mol L^{-1} for the top two layers, respectively. These values increased with depth to the maximum average of 6.1E-5 mol L^{-1} at the deepest soil layer.

The amount of B in the soil was adequate for most plants and was not toxic even with the highest amount in the lower part of the soil profile. The ANOVA revealed that some of the layers had statistically significant differences. Based on the Tukey's pairwise comparison, the two layers of 0.0-0.3 m and 0.3-0.6 m were significantly different from the two layers 0.9-1.2 and 1.2-1.5 m at 0.05 level on both sampling dates. The change in B over time was not significant at any depth, suggesting that the wet and dry cycles during the study period did not have any considerable impact on B concentrations.

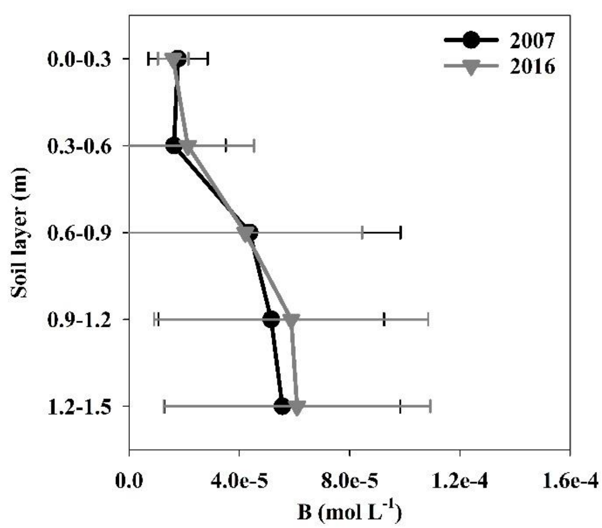

Figure 2.3. Mean B for each soil layer at the two sampling dates in 2007 and 2016. Error bars represent one standard deviation from the mean.

Sodium

In 2007, average sodium (Na) concentration increased with depth to a maximum of 0.049 mol L^{-1} at the 0.6-0.9 m soil layer, and then declined to the minimum of 0.031 mol L^{-1} at the deepest soil layer (Figure 2.4). However, Tukey's pairwise comparison showed that the differences in Na among soil layers were not statistically significant. The profile of Na had a similar pattern in 2016, but the minimum average was observed at the topmost layer with the value of 0.018 mol L^{-1}. It then increased gradually by depth until the maximum of 0.055 mol L^{-1} at the 0.6-0.9 m depth and decreased slightly for the two deeper layers. Based on the Tukey's pairwise comparison for data obtained in 2016, the first layer was significantly different from the third and fourth layers at 0.05 level, but the remaining layers were not significantly different.

When considering all locations/depths, the average Na was 0.042 and 0.040 mol L^{-1} in 2007 and 2016, respectively, demonstrating no significant difference. The change in Na during the study period was strongly depth dependent. The top two soil layers experienced a decrease while the bottom three layers showed an increase in Na concentrations. The maximum reduction was at the top layer at 0.021 mol L^{-1}, which is about 53% reduction based on the 2007 level. This difference was statistically significant ($p = 0.02$). The maximum increase in Na was at the bottom layer at 0.010 mol L^{-1}, or about 34% of the 2007 levels. However, this difference was not statistically significant. The magnitudes of total decreases and increases in Na concentration were similar, resulting in a value of zero when differences were summed for all soil layers. These findings suggest that the top 0.6 m of the soil experienced leaching of Na during the study period, but the net change for the top 1.5 m of the soil was negligible as transported Na was deposited in the layers below.

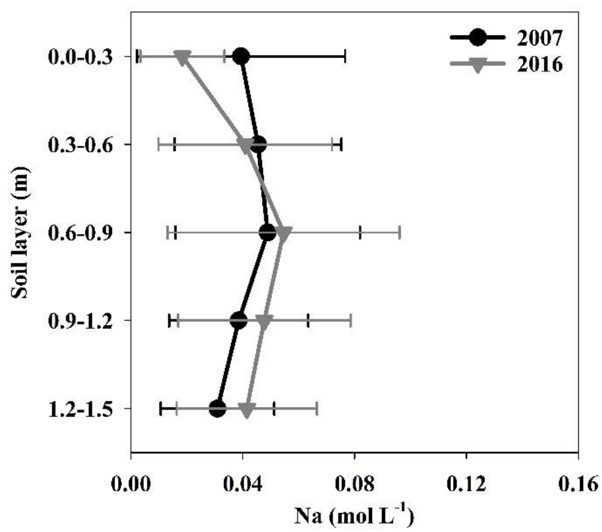

Figure 2.4. Mean Na for each soil layer at the two sampling dates in 2007 and 2016. Error bars represent one standard deviation from the mean.

Calcium

The average Calcium (Ca) significantly decreased from 1.3E-2 mol L^{-1} in 2007 to 8.9E-3 mol L^{-1} in 2016 ($p = 0.008$). Higher levels of Ca concentration in soil have been found to help minimize adverse impacts of salinity on cotton growth (Kent and Lauchli 1985) since Ca flocculates clay particles and builds better soil structure. Similar to Na findings, the reduction in Ca with time was strongly depth dependent, being largest at the top layer (0.0-0.3 m) with a decline of 6.2E-3 mol L^{-1} on average and smallest at 0.9-1.2 m layer with a decline of 6.0E-4 mol L^{-1}. The change in Ca over the study period was statistically significant only for the top layer ($p = 0.002$).

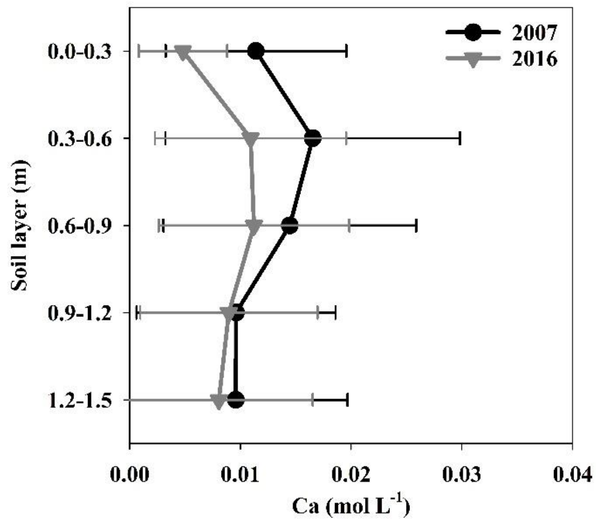

Figure 2.5. Mean Ca for each soil layer at the two sampling dates in 2007 and 2016. Error bars represent one standard deviation from the mean.

Little or no salt input during the drought and salt movement downward during the wet period might have contributed to the decrease in Ca in the surface soil. The increase in pH may have also resulted in downward movement of soluble Ca. When considering each sampling date separately, Ca concentrations first increased and then decreased with depth, but these changes were not statistically significant (Figure 2.5).

Magnesium

The average Magnesium (Mg) concentration decreased from 7.3E-3 mol L^{-1} in 2007 to 6.3E-3 mol L^{-1} in 2016, but this decrease was not statistically significant. The reduction in Mg with time was depth dependent, with the largest decrease observed at the top layer (2.1E-3 mol L^{-1}) and the smallest decrease of near zero at the bottom two layers. These changes, however, were not statistically significant at any layer. The change in Mg over time was expected to be significant at least for the top layer (similar to Na and Ca). A possible reason for not observing a significant change may be the impact of increase in pH on decreasing Mg solubility due to precipitation with carbonate.

Variations in Mg with depth were similar to those observed in Ca, with concentrations increasing gradually from the top layer to maximum levels at 0.6-0.9 m and then decreasing at deeper layers (Figure 2.6). The differences in Mg among soil layers were not statistically significant in 2007, and in 2016 only the two layers of 0.0-0.3 and 0.6-0.9 had a statistically significant difference ($p = 0.0008$).

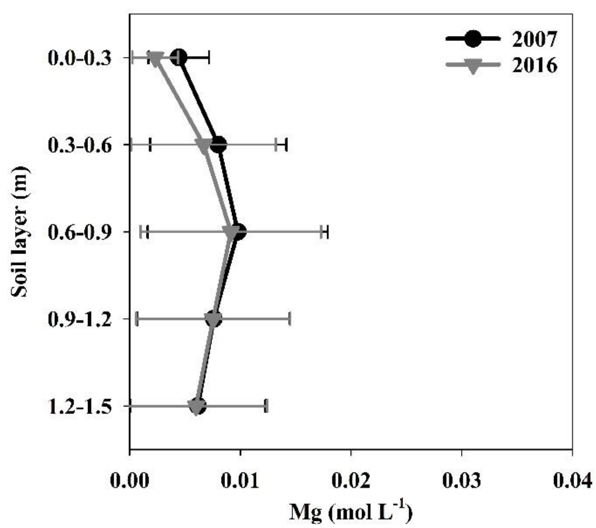

Figure 2.6. Mean Mg for each soil layer at the two sampling dates in 2007 and 2016. Error bars represent one standard deviation from the mean.

Electrical Conductivity

The sub-cores extracted from different locations/depths at two sampling times had a wide range of electrical conductivity (EC), varying from 0.3 to 26.0 dS m^{-1}. In 2007, the average EC of all sampling locations was smallest at the top layer with a value of 5.7 dS m^{-1} (Figure 2.7). Below this layer, EC increased to 11.0 dS m^{-1} at 0.6-0.9 m, then decreased and reached EC of 7.1 dS m^{-1} at the deepest level. The differences in EC between the first and the second layers and the first and the third layers were statistically significant according to Tukey's pairwise test ($p < 0.05$). In 2016, similar to 2007, the lowest average EC was observed at the shallowest layer at 4.3 dS m^{-1}. It then increased by depth to the maximum value of 11.1 dS m^{-1} at the depth of 0.6-0.9 m, followed by a decrease to 8.4 dS m^{-1} at the deepest soil layer. The differences in average EC were only statistically significant between the top soil layer and the layers 0.6-0.9 m and 0.9-1.2 m ($p < 0.05$).

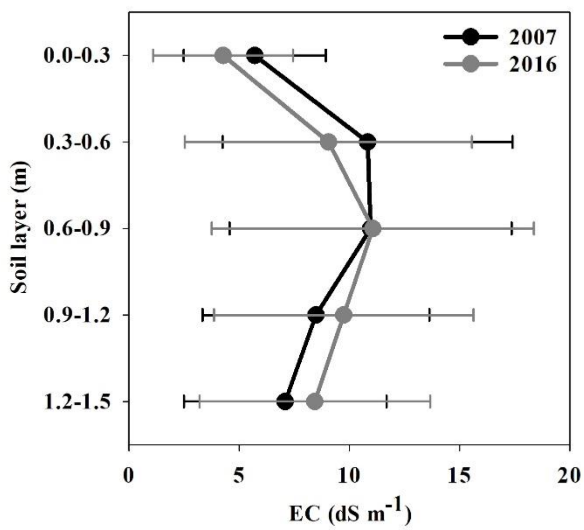

Figure 2.7. Mean EC for each soil layer at the two sampling dates in 2007 and 2016. Error bars represent one standard deviation from the mean.

The results from the two sampling times provided valuable information on how salinity changed with depth and time during the study period. When considering changes with depth, the lowest salinity observed was in the shallowest soil layer at both sampling times. This could be attributed to the effect of infiltrating water quality in terms of irrigation application and rainfall. The top soil salinity is more strongly impacted by the salinity of the irrigation water compared to deeper soil layers, where other factors such as soil type, salt type, and the method of water application play greater roles (Grattan 1994). The EC of irrigation water measured at different times during the study period at a central location within LAID was 2.0 dS m^{-1} on average, smaller than the average EC at all soil layers/dates which may attribute slightly to soil salinity particularly at the top soil layers. Another contributing factor could be soil texture, which is coarser for the top layer and heavier for deeper layers in the study area. The change in soil texture affects permeability and thus the leaching potential.

In addition to observing the lowest EC at the shallowest layer, both sampling dates had similar EC patterns that first increased and then decreased with depth. This profile distribution of EC could be linked to the texture profile. Salts can accumulate in the B-horizon, which is typically associated with elevated clay content (the case with the Hollister soil profile). As a result, the salts tend to accumulate in this layer because they cannot be leached further down. A similar pattern of EC variations by depth was observed by Feikema and Baker (2011) under irrigation with low-salinity water, where EC increased with depth to mid-layers of 0.8-1.0 m and then decreased to deepest sampling layer of 1.8 m. Under high-salinity irrigation application, however, they reported continuous increase in EC with depth (Feikema and Baker 2011). The increasing EC pattern has been reported by several other researchers (e.g. Ayars *et al*. 1993; Armstrong *et al*. 1996), especially under the presence of shallow groundwater (Moreno *et al*. 1995; Goyal *et al*. 1999).

The changes in EC during the study period varied largely among sampling locations/depths, from a decrease of 12.7 to an increase of 11.5 dS m^{-1}. The magnitude and direction of EC change was impacted by depth. The average EC decreased by time for the top two layers, remained unchanged for the third layer, and increased for the bottom two layers. This is similar to the temporal change in Na and indicates some downward movement of salts (leaching) during the study period. When considering all data points, EC decreased 2.8 dS m^{-1} from 2007 to 2016 on average. However, this difference was not statistically significant; suggesting that the net impact of dry and wet cycles on the salinity of soil profile was negligible at the district level.

Despite finding no significant change in EC at the large scale, the high level of variability in salinity changes among studied locations deserves further investigation. Out of the 20 locations sampled in this study, 12 (60%) had experienced some level of leaching from 2007 to 2016. The remaining locations were divided equally between no significant change and increase in EC over time. Since these locations were similar in many characteristics (e.g. climate, soil type, irrigation source, crop type), different responses in soil EC is most likely caused by on-farm factors such as irrigation management and the effectiveness of removing excess water from the root zone using surface and subsurface drains.

Cotton is the dominant crop in the study area and plays a vital role in the local economy. Hence, it is of great importance to investigate potential impacts of soil salinity on cotton performance. In conducting such analysis, however, it should be taken into account that crop productivity is affected by the average salinity of the soil profile and elevated salinity in certain depths may not significantly affect productivity if other depths had low salt concentration. Cotton is considered a salt tolerant crop with the threshold EC of 7.7 dS m^{-1} (Bernstein 1956). Above this threshold, cotton yields start to decline at a rate of 5.2% per unit increase in EC (Maas 1984). When averaged over the entire sampling profile (1.5 m), the EC estimates for the 20 locations had a range of 3.4-14.0 dS m^{-1} in 2007 and 2.8-14.2 dS m^{-1} in 2016. At locations with the highest profile EC, yield declines of 33 and 34% in cotton are expected in 2007 and 2016, respectively. The average profile EC for all sampling locations was 8.8 dS m^{-1} in 2007 and 8.5 dS m^{-1} in 2016 in this study, which is about one unit larger than the threshold and may have a small potential impact on reducing cotton yield (5 and 4%, respectively) at the district scale. For the top two layers (0.0-0.6 m), the average EC was 8.3 dS m^{-1} in 2007 (above the threshold) and 6.6 dS m^{-1} (below the threshold) in 2016.

Exchangeable Sodium Percentage

High levels of Exchangeable Sodium Percentage (ESP) can have detrimental effects on soil physical and hydraulic properties such as aggregate stability, permeability, and hydraulic conductivity (Wallender and Tanji 2011). In addition, elevated ESP levels have been linked to lower cotton yield and fiber quality in studies conducted in Australia (Dodd *et al*. 2013) and India (Choudhary *et al*. 2001). In this study, ESP had a range of 2-65% in 2007 and 3-64% in 2016 when considering all samples. Despite observing large ESP levels in some soil samples, 80% of them had an ESP level less than 15% in both years. According to Abrol *et al*. (1988), the sodicity hazard is none to slight when ESP is smaller than 15%. About 18% of samples in both years had ESP levels larger than 15% but smaller than 30%, which is classified as light to moderate sodicity hazard. The average ESP was 12% and 13% in 2007 and 2016, respectively. However, this difference was not statistically significant.

The trend of ESP profiles was similar to those of Ca, Mg, Na, and EC. In 2007, the minimum ESP observed was in the top soil layer with the average (median) of 9.9% (9.5%). This value increased by depth and reached the maximum of 13.5% (10.1%) at the soil layer 0.6-0.9 m, then decreased and reached 10.6% (10.9%) at the deepest layer (Figure 2.8). In 2016, the minimum ESP also was observed at the shallowest soil layer with the value of 7.7% (7.5%). ESP increased by depth below the top layer and reached the maximum of 16.4% (11.1%) at the deepest soil layer. Although these estimates show a decrease in ESP for the top soil layers and an increase for the bottom layers over time, the differences among layers were not statistically significant based on the Tukey's pairwise comparison.

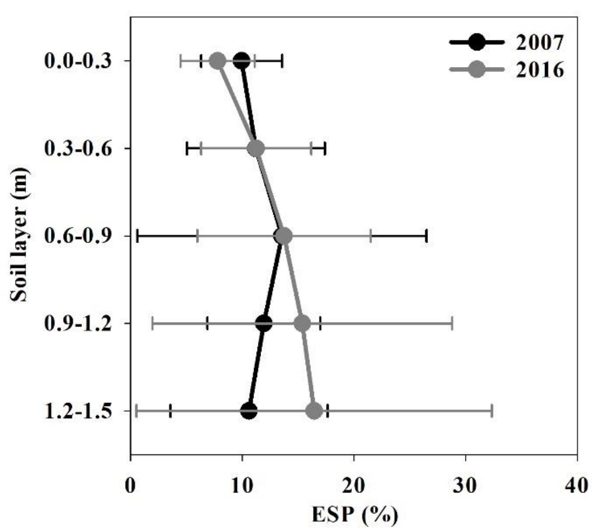

Figure 2.8. Mean ESP for each soil layer at the two sampling dates in 2007 and 2016. Error bars represent one standard deviation from the mean.

Soil salinity classification

Several classification systems have been proposed in the past for dividing soils into different categories for variable purposes. A classification commonly used in irrigated agriculture is based on soil salinity and sodicity (Abrol *et al*. 1988). In this system, an EC threshold of 4.0 dS m^{-1} is considered for salinity and an ESP of 15% is used for sodicity. Combining the two thresholds allows for classifying soil samples into four groups. Normal soils have both EC and ESP values less than the thresholds. Saline soils are identified with EC of greater than 4 dS m^{-1} and ESP of less than 15%. Nonsaline-sodic soils have EC levels of less than 4 dS m^{-1} and ESP of greater than 15%. If both thresholds are exceeded the soil is classified as saline-sodic. Figure 2.9 shows a scatterplot of EC vs. ESP for all collected samples in 2007 and 2016. The horizontal and vertical lines also represent the thresholds. The majority of the soils were either normal or saline (see Table 2.2 for details). This classification is beneficial if reclamation is implemented. For saline soils, adequate water is needed to leach excess salts out of the soil profile, but gypsum and organic matter are required in addition to water to remove the excess sodium first by exchanging Na with Ca (Zhang 2013).

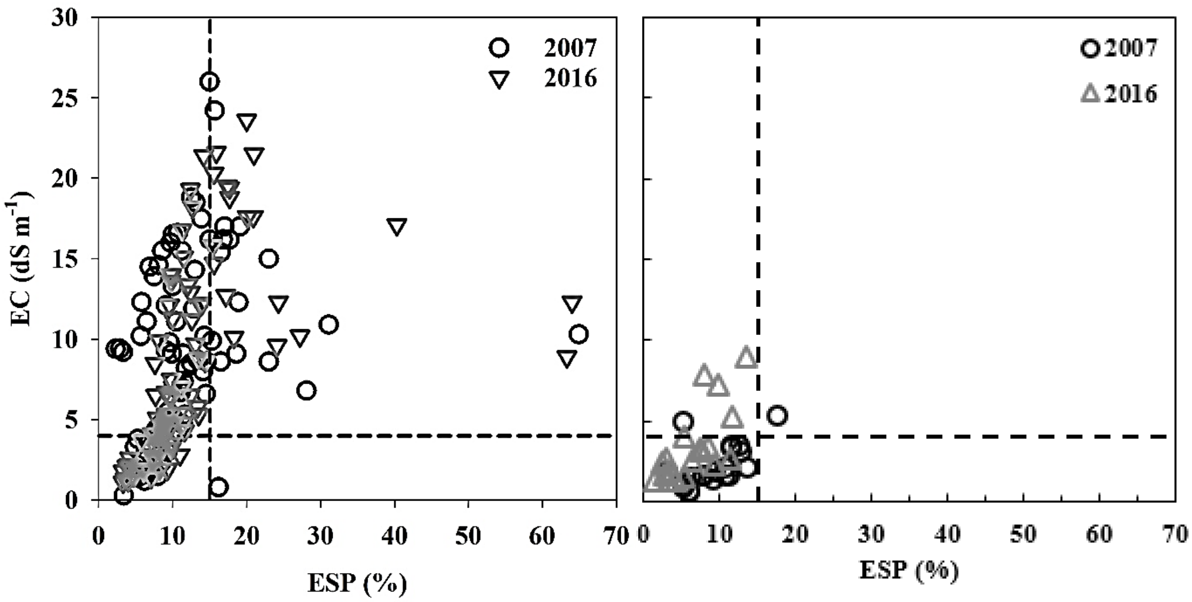

Figure 2.9. Distribution of all collected soil samples across four soil classes of normal (lower left), saline (upper left), saline-sodic (upper right), and sodic (lower right).

When considering the change in soil classification over the study period for all collected samples (profile), the largest difference observed was in the saline class. The percentage of samples belonging to this class decreased from 60% in 2007 to 50% in 2016 (Table 2.2).

The majority of the points that were no longer classified as saline had moved to the normal class, represented by a frequency increase in this class from 22% in 2007 to 29% in 2016. Although this a considerable change in the right direction, it still shows that a significant portion (half) of all samples had salinity issues after the eight years of dry/wet cycles. In addition, the percentage of samples classified as saline-sodic increased from 17% in 2007 to 21% in 2016. The increase in saline-sodic soils deserves further investigation and continued monitoring since high sodicity results in aggregates dispersion and decrease in soil permeability to air and water, especially if salts are leached out of saline-sodic soils. Soil classification comparison of the top 15 cm of the soil revealed that saline-sodic sampling points decreased from 5% in 2007 to 0% in 2016. Whereas, saline soils increased from 10% in 2007 to 25% in 2016. Although there was an increase in soil salinity of the top 15 cm of the soil, the root zone salinity and sodicity has a significant impact on root development and its ability for water uptake.

Table 2.2.

Percentage of samples in each soil class in 2007 and 2016 reported for the entire soil profile and each of the five sub-layers.

Class	EC	ESP	Profile		0.0-0.3		0.3-0.6		0.6-0.9		0.9-1.2		1.2-1.5	
			2007	2016	2007	2016	2007	2016	2007	2016	2007	2016	2007	2016
Normal	<=4	<=15	22	29	35	60	20	20	15	20	18	18	25	25
sodic	<=4	>15	1	0	0	0	0	0	0	0	1	0	0	0
Saline	>4	<=15	60	50	55	35	60	60	60	45	65	59	58	50
Saline-sodic	>4	>15	17	21	10	5	20	20	25	35	12	24	17	25

When studying each soil layer, the largest change was observed in the top layer (0.0-0.3), with an increase in the normal class from 35% of samples in 2007 to 60% in 2016. The largest decline of the saline class was observed in the same layer, from 55% in 2007 to 35% in 2016. The distribution of samples among classes remained unchanged over time for the second layer (0.3-0.6 m). The bottom three layers experienced a reduction in saline samples and an increase in saline-sodic ones.

Temporal stability

To investigate the effect of wet and dry cycles experienced during the study period (2007-2016) on temporal stability of soil salinity, the nonparametric Spearman's test was performed on all studied soil parameters, averaged over the sampled soil profile (1.5 m).

Table 2.3.

Spearman's rank correlation coefficients (r_s) for studied soil parameters.

Parameter	r_s
B	0.761
Ca	0.687
Mg	0.877
Na	0.814
pH	0.781
EC	0.875
ESP	0.935

The Spearmen's rank correlation coefficient (r_s) varied from 0.687 for boron (B) to 0.935 for ESP as observed in Table 2.3. All r_s values were close to unity and larger than the critical r_s values of 0.447 and 0.570, which represent 0.05 and 0.01 significance levels, respectively. This suggests the presence of a strong and statistically significant time stability in the ranks of sampling locations during the study period. The r_s for EC in this study was 0.875. Douaik *et al*. (2006) reported similar r_s values for EC estimated at twenty sampling locations across a grassland in eastern Hungary over a span of about seven years.

The static-dynamic nature of temporal changes in spatial patterns of the soil properties was evaluated by developing linear regression models for each measured property where the independent and dependent variables were the soil property of interest on the first (2007) and the second (2016) sampling dates, respectively. The intercept (I), slope (S), and the coefficient of determination (r^2) of developed regression models are presented in Table 2.4. All developed linear regression models were statistically significant at 0.01 level, according to the F-test. Except for Ca, the r^2 values were larger than 0.67, suggesting that more than two-thirds of the spatial variance observed at the end of the study can be explained by the variations at the beginning of the experiment.

Table 2.4.

Temporal and spatial pattern of electrical conductivity between the two sampling dates of 2007 and 2016 for different soil layers.

Parameter	**I**	**P (H_0: I = 0)**	**S**	**P (H_0: S = 1)**	**r^2**
B	0.02	0.589	1.00	0.499	0.86
Ca	113.6	0.235	0.46	<u>0.001</u>	0.34
Mg	-14.1	0.580	0.93	0.292	0.78
Na	79.7	0.619	0.85	0.149	0.67
pH	4.11	<u>0.000</u>	0.51	<u>0.000</u>	0.72
EC	0.10	0.944	0.93	0.310	0.71
ESP	0.69	0.396	0.97	0.299	0.93

Note: the underlined P-values are less than the 0.01 significance level, indicating that the corresponding null hypotheses can be rejected.

Statistical analyses were performed on the coefficients of the regression models (I and S) to identify the represented static-dynamic category among the four categories mentioned in the methodology section. The null hypotheses were that intercepts are equal to zero and slopes are equal to unity. In the case of Ca, the null hypothesis was rejected only for S. This suggests that the changes in Ca concentration were not uniform across sampling sites (the spatial patterns were dynamic), but the mean Ca concentration did not change over the study period. In the case of pH, both null hypotheses were rejected, meaning that the average pH changed with time and the magnitude of change between two dates was not uniform and differed from one location to the other. The means and the spatial patterns of all other parameters were static. The largest slope belonged to B at the value of unity, followed by the slopes of ESP, EC and Mg. Figure 2.10 depicts the scatterplots and regression lines for EC and ESP between the two sampling dates.

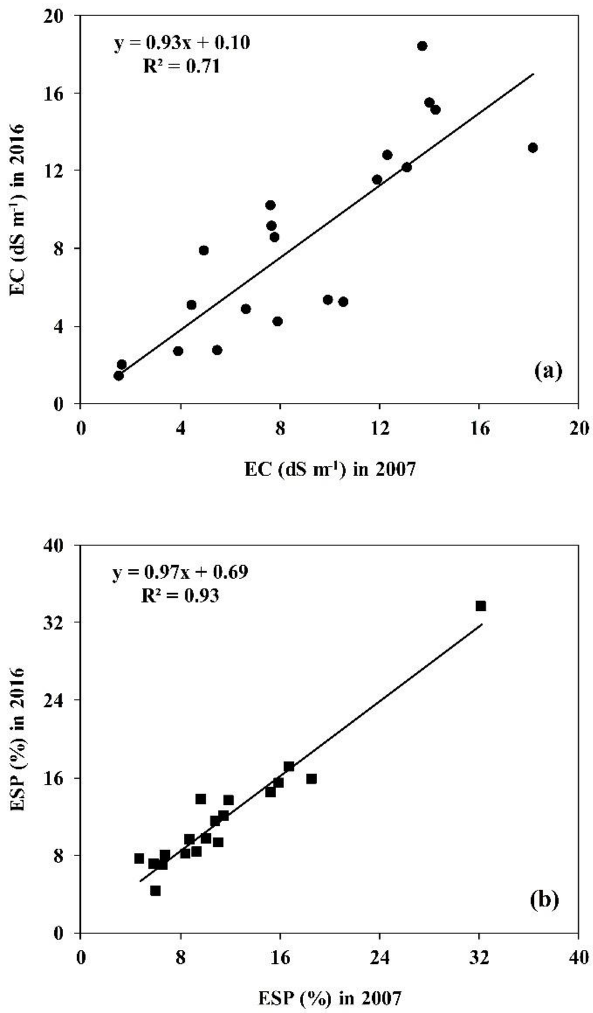

Figure 2.10. Linear regression models developed for EC (a) and ESP (b) on the two sampling dates in 2007 and 2016.

Conclusion

The top soil across an irrigation district in southwest Oklahoma was sampled before and after a period spanning eight growing seasons. This period was characterized by four years of exceptional drought when irrigation deliveries were terminated due to water scarcity, followed by a period of record precipitation. Except for pH and Ca, the district-wide mean of studied parameters did not experience a statistically significant change. The mean pH increased significantly from 7.7 to 8.1 and the mean Ca decreased from $1.3E^{-2}$ to 8.9E-3 mol L^{-1} over the study period. When investigating temporal changes at five sub-layers, a decrease in Na and EC for top layers and an increase for bottom layers were observed. Although this observation is an indication of shallow leaching, the net impact for the entire sampled profile was negligible. Analysis of time stability revealed that except for the same two parameters (pH and Ca), spatial patterns of other parameters were static. Overall, four years of extreme drought with no irrigation application succeeded by a period of intensive rainfall reduced soil salinity in the surface layer but moved salts downward to the middle section of soil profiles. This reduction in surface soil salinity is beneficial for seedling establishment. However, levels of pH, EC, and ESP appear to be high enough to cause yield loss, especially at some of the sampling locations.

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CHAPTER III

STUDY OF THE EFFECTS OF IRRIGATION WATER QUALITY AND LEACHING ON SOIL SALINITY AT THE GREENHOUSE SCALE USING MONITORING AND MODELLING

Abstract

To deal with fresh water shortage, saline water is increasingly being used in irrigated agriculture. However, saline irrigation can lead to secondary salinization, and consequently, lead to soil degradation, abnormal crop growth, and decreased productivity. Leaching is suggested as a simple management practice to wash the salts below the root zone and control salinization. Using saline water in leaching applications can lead to excess accumulation of salts, particularly in arid and semiarid areas along with and downstream water contamination. There is a gap in knowledge on best management practices that facilitate using saline irrigation water. In this respect, utilization of good quality water for leaching applications, to control rootzone salinization without excessive application of salts, is investigated in the current study.

In this regard, a monitoring and modeling approach was undertaken in this study, which involved a greenhouse study followed by simulation study using HYDRUS 1-D model.

The greenhouse study consisted of six treatments of saline irrigation water of 0.5, 3, 6, and 9 dS m^{-1} with and without fresh water leaching. Based on the results of the greenhouse study, increases in irrigation water salinity lead to accumulation of salts at the soil surface, and a sharp decrease in salinity at the top 10 cm of the soil for all treatments. Although, increase in irrigation water salinity, lead to an increase in soil profile salinity, leachings were able to reduce soil salinity throughout the soil profile. Fresh water leaching lead to 23% and 29% reductions in average soil profile salinity for soil columns irrigated by 6 and 9 dS m^{-1}, respectively. The reduction values were even higher at the top soil layer, representing 28% and 46% reductions in top soil salinity as a result of fresh water leaching at the end of the experiment. The greenhouse experiment revealed that fresh water leaching enables utilization of saline irrigation water, which are traditionally considered unsuitable for irrigation.

Based on HYDRUS 1-D simulation results, the modeled electrical conductivity of the saturated paste (EC_e) for all treatments were in good agreement with the greenhouse observations, as indicated by RMSE, MBE, MAE, and r values. The agreements between the modeled and observed values were stronger for lower salinities with RMSE of 0.32 dS m^{-1} for EC of 0.47 dS m^{-1} and increased with increases in irrigation water salinity to 3.14 dS m^{-1} for EC of 9 dS m^{-1}.

HYDRUS 1-D was able to predict the general trend of increases in soil salinity with increasing irrigation water salinity with RMSE greater than MAE for all treatments. Similar agreement between the observed and simulated results were detected in other statistical parameters. We concluded that application of fresh water leaching is a promising method to facilitate saline water irrigation application without leading to soil salinization.

Keywords

Saline irrigation, greenhouse study, leaching, rootzone salinity, HYDRUS.

Introduction

Irrigated agriculture has contributed significantly to supplying the food and fiber needs of the fast-growing population (Rhoades et al., 1992); however, it increasingly competes for the limited available water supplies with other sectors such as municipalities and industries (World Health Organization, 2004). Alternatively, widespread availability of saline water around the world has made it a potential irrigation water supply and thus significantly expanded the available irrigation water resources (Rhoades et al., 1992; SouDakouré et al., 2013). This sustainable approach has helped fill the ever-increasing gap between the supply of and demand for fresh water (Qadir et al., 2009). Even under conventional farming practices, water generally classified as saline for irrigation may successfully be used for irrigation (Rhoades et al., 1992). Utilization of saline water for irrigation has been facilitated by adapting suitable management practices to further expand irrigated agriculture (Allen et al., 1998). Saline water irrigation will not only satisfy the main goal of agricultural production, which is to maximize productivity, but it also facilitates the efficient use of fresh water resources and promotes long-term sustainability of agriculture production (Ayars et al., 1993). However, there are challenges in the way saline water is used for irrigation purposes as it can lead to secondary salinization, which results in the loss of land, reduced rates of crop growth, and eventually total crop failure (Crescimanno & Garofalo, 2006). Farmers are facing more environmental constraints on their management decisions and have to continuously update their management practices to adapt to new environmental patterns (Hanson and Grattan, 2006). In this respect, scientists should take the lead in providing technical information to guide farmers and policymakers in making the best decisions regarding how to manage limited water resources, maintain soil health, and prevent environmental degradation so that making high crop productivity can be achieved (Ghassemi et al., 1995). One basic management tool that farmers have long been using to control salinity is leaching, which includes the application of extra water beyond crop needs for evapotranspiration, in order to flush salts from the root zone (Corwin et al., 2007; Yurtseven et al., 2014). This can become complicated if irrigation water resources are limited especially since irrigation water can add extra salts to the soil (Crescimanno & Garofalo, 2006; Ayers & Wescot, 1985). Therefore, it is very critical to understand the factors in the optimization of irrigation quality and quantity to keep soil salinity at desired levels while best utilizing saline water resources. Extensive research has been carried out in the past to study the impacts of irrigating with saline water on different parameters in experimental soil columns (Fujimaki et al., 2008; Yurtseven et al., 2013) and field scales (Zeng et al., 2014; Haj-Amor et al., 2016). Studying salt dynamics at field scales is not always feasible as it requires extended soil sampling, which is expensive and time-consuming. Application of experimental soil columns in the greenhouse has the advantage of allowing the study of the plant development and drainage water quality and quantity simultaneously without disturbing the soil profile (Chang and Silva, 2013). Furthermore, the controlled environment in the greenhouse enables the study of the effect of each individual parameter and thus simplifies the modeling (He et al., 2017). Yurtseven et al. (2013) conducted soil column experiments to analyze the water flow and solute transport processes after irrigation with different water quality and leaching rates. They found that salts, which had accumulated due to irrigation with saline waters could be leached from the soil columns. Although column studies are cost beneficial to study crop response to a variety of stress functions, they may not suffice in the study of crop growth and tolerance to salinity as they may restrict the root development as compared to freely growing crops in the field (Fujimaki et al., 2008). Applicability of such experiments for various experimental conditions including local climates, soil types, topography, and crop types should be applied. In this regard, conducting greenhouse studies and using measured parameters to validate and calibrate computer models for optimization of management parameters is the best starting point to assess practice applicability and cost.

Computer models play an important role for analysis of irrigation, soil salinization, solute transport in the soil solution, and their effects on crop productivity (Zeng et al., 2014; Tafteh and Sepaskhah, 2012). Many studies have used the HYDRUS model to compare the observed variables versus the model predicted ones. In this respect, some studies have been conducted at field scale (Goncalves et al., 2001), while others have focused on greenhouse studies in the controlled environment (Fujimaki et al., 2008; Yurtseven et al., 2013). Wang et al. (2017) used the HYDRUS-1D model, calibrated and validated with field data to evaluate the effects of the application of four levels of irrigation amounts and water salinity on soil salinity dynamics. This model was later used to investigate the effect of long-term use of saline water for irrigation on salt accumulation and grain yields. They observed good agreement between measured and simulated soil water content, salt accumulation, evapotranspiration (ET), and grain yield data. Increased irrigation volume represented a decrease in the quantity of salt accumulated in the soil. In another study Haj-Amor et al., (2016) conducted a two-year field study and used HYDRUS 1-D simulation to study the effect of various irrigation regimes with different frequency, amount of added water, and irrigation water salinity on the soil salinity. Based on the results of this study, the HYDRUS 1-D simulation was able to estimate the salt and soil dynamics of the soil profile at an acceptable level. The simulation demonstrated that frequent year-round irrigations with small amounts of water were able to keep the soil salinity and water content at the acceptable levels in terms of its EC and soil water content. Zeng et al. (2014) conducted field scale irrigation experiments and concluded that irrigation amount did not affect the soil water storage significantly but had a direct effect on the salt leaching rate. While some of the studies mentioned above employed only simulations (e.g., Ebrahimian et al., 2013), a number of investigations used experimental data to calibrate and test HYDRUS predictions (e.g. Haj-Amor et al., 2016; Zeng et al., 2014), which provided confidence that the simulation could describe the complex soil salt and water dynamics adequately.

Although there are numerous studies on the effect of leaching applications with saline water on soil salinity profiles, far too little attention has been paid to the use of good quality water for leaching applications after irrigation with saline water. This practice has the potential to allow for utilization of more saline irrigation water without adversely affecting vegetation growth. This gap in the previous literature on the utilization of good quality water leaching practices to control soil salinity was the main focus of this experiment. The objective of this study was to investigate the effect of fresh water leaching on soil salinity profile. This objective was investigated first by conducting a greenhouse column study and measuring soil salinity profile in receiving soils of various qualities of saline irrigation water with and without fresh water leaching application. Later, the soil salinity dynamics for the greenhouse study were simulated using the HYDRUS model.

Materials and Methods

Experimental Setup

The experiment was conducted in the Oklahoma State University experimental greenhouse. Twenty columns made of 40 cm height and 10.16 cm diameter PVC tubing were used along with a jar for excess water collection at the bottom. Out of the twenty columns, three were used to test the soil's initial salinity and water content. Bermuda grass seeds were planted in the columns at a rate of 0.06 g per pot, totaling approximately 80 seeds per column. Fertilizer application was initiated after plant germination. The macro and micronutrient requirements of the plant were calculated based on the recommended amounts for fertilization (i.e. 2.2 gram of nitrogen per 93 m^2 per month). A mixture of 0.755 grams of 20-20-20 fertilizer with 0.21 grams of micronutrients was added to 604 ml of water and applied every three days. The plants were trimmed regularly to keep their height uniformly around 7.5 cm to promote root lateral growth and occasionally sprayed with Bifen and Tourney to repel insects and fungus. To ensure regular, uniform water quantity, misters were initially used for irrigation water application. However, it was discovered that mister applied water caused fungus growth and extensive evaporation. Therefore, during the plant growth phase and before salinity application started, the irrigation method was changed to manual application. The columns were irrigated every evening under the assumption that plants do not uptake water during the night (Fujimaki et al., 2008; Homaee et al., 2002). The percolation collection jars and columns were weighed before and after each irrigation. The irrigation quantity was calculated based on the crop ET using the weight loss of the between two consequent days. This approach allows salt built up while avoiding drought stress (Fujimaki et al., 2008). The controlled environment in the greenhouse provided stable conditions in terms of wind speed, temperature, and relative humidity; therefore, their effect on increased evaporation was negligible. Before the salinity stress experiment started, some columns were dismantled for analysis of initial soil moisture content and initial soil salinity. Three replicates were used for each parameter. The total duration of experiment consisted of two parts. The preparation period was around 78 days to prepare crop for salinity stress test. Once the maximum water holding capacity of each of the columns was determined and the grass had sprouted, reached its full growth potential, it was ready for the salt stress experiment. The stress period which started at sunset on November 9, 2017 and continued for 42 days until December 23, 2017. The synthetic saline water was prepared by mixing different salts of NaCl, $MgCl_2$, KCl with the amounts of 0.07, 0.12, and 0.13 respectively in 1 liter of distilled water to produce solutions with targeted EC value of 3 dS m^{-1} (Rahman et al., 2015). The other targeted salinities were prepared similarly. Six salinity stress treatments were defined based on salinity levels of irrigation water as displayed in Table 3.1.

Table 3.1.

Various treatments conducted with the corresponding initial water quality and irrigation water applied.

Treatment	**Treatment Explanation**	**EC (dS m^{-1})**	**Leaching application**
CTRL	Control with tap water	0.5	no
EC3N	slightly saline water	3	no
EC6L	moderately saline water	6	yes
EC6N	moderately saline water	6	no
EC9L	highly saline water	9	yes
EC9N	highly saline water	9	no

Maas and Hoffman (1977) reported salt tolerance of Bermuda grass to be around 6.9 dS m^{-1}. In order to avoid osmotic shock to the plants, salts were gradually added to the soil and irrigation was carried out with lower salinity at first and reached the designated salinity values in 3-4 days. To keep the parameters low and just account for the irrigation water quality, all treatments were irrigated with 130% of plant ET on the last day of the experiment, so water quality was the only variable considered in any treatment. The extra irrigation water was the same quality as previous irrigation in the treatments without leaching experiment, however, in treatments with fresh water leaching application (EC6L and EC9L) the last irrigation water had salinity of 0.47 dS m^{-1}. This experiment would allow for investigation of effect of leaching water quality on soil salinization.

After the salinity stress experiments were completed, each column was dismantled into eight 5-centimeter ring sections and samples from each ring were fully mixed, put in bags and sent to the Oklahoma State University Soil Testing Lab for further analysis for their salinity and ion concentration.

Measurements

The average meteorological parameters in the greenhouse during 42 days of experiment is presented in Table 3.2.

Table 3.2.

Meteorological parameters in the greenhouse during 42 days of experiment.

Parameter	**Value**
Mean R_s^2 (W/m^2)	129.9
Min T_{air}^3 (°C)	10.3
Max T_{air} (°C)	38.1
Mean T_{air} (°C)	24.35
Min RH^4 (%)	5.5
Mean RH (%)	36.6
Mean VP^5 (kpa)	98.53

[1] Precipitation

[2] Total daily accumulation of solar radiation

[3] Air temperature

[4] Relative humidity

[5] Vapor Pressure

Prior to the salinity stress experiment, soil samples were taken to determine their texture, initial salt concentration, and water content in the study column. These samples were kept at over 150 ˚C for several hours to remove soil moisture. Based on the results of texture analysis, the soil was classified as sandy loam with sand, silt, and clay percentages of 77.5%, 12.5%, and 10%, respectively (Gee and Bauder, 1996). The soil parameters were calculated at the soil physics laboratory at OSU.

Table 3.3.

Soil hydraulic parameters of the samples.

Bulk D. [a] **(gr cm^{-3})**	**θ_r**[b] **(cm^3 cm^{-3})**	**θ_s**[c] **(cm^3 cm^{-3})**	**α**[d] **(cm^{-1})**	**n**[e] **(-)**	**K_s**[f] **(cm day^{-1})**	**L**[g] **(-)**
1.57	0.02	0.36	0.02	1.37	38.31	0.79

a. Bulk density

b. residual water contents

c. saturated water contents

d. inverse of the air entry suction

e. pore-size distribution

f. saturated hydraulic conductivity

g. empirical parameter

The gravimetric method was employed to measure the water content of the soil samples. Soil samples were initially weighed, then oven-dried at a temperature of 105 ˚C to a constant weight before its oven-dry weight was determined. The gravimetric soil moisture content was converted to volumetric soil water content by multiplying the gravimetric soil water content by the bulk density of the soil. From each soil layer, a core sample was taken and mixed until homogeneous. A sample from each layer was sent to Soil and Forage Laboratory and analyzed for its chemical composition. Soil samples were oven dried overnight at the temperature of 65 ºC and ground to pass through a 2-mm sieve. The EC, pH, and buffer index of the samples were measured in a 1:1 soil to water suspension and Sikora buffer suspension (Sims, 1996; Sikora, 2006), where 100 g of oven dried soil was mixed with 100 mL of deionized water (Richards, 1954). After reaching equilibrium within 4 hours, the suspension was extracted using a low-pressure filter press apparatus. A spectrometer (SPECTROBLUE, SPECTRO Analytical Instruments GmbH, Germany) was employed to quantify the amounts of phosphorus (P), potassium (K), calcium (Ca), and Magnesium (Mg) in the extract (Soltanpour et al., 1996). Exchangeable Sodium Percentage (ESP) and Sodium Adsorption Ratio (SAR) were calculated using the equations provided in the USDA protocol (1954). Total dissolved solids (TDS) was expressed as parts per million (ppm) (USDA, 1954). All parameters were measured at 1:1 ratio before they were converted to saturated paste equivalent based on the conversion factors described by Richards (1954). The results were used to calculate the initial soil water content and salinity of the soil solution.

HYDRUS Model

The HYDRUS-1 D model was employed to simulate one-dimensional water flow and solute transport in the variably saturated column (He et al., 2017). This model combines Richard’s equation to describe water flow in microspores as the mobile water region, and a mass balance equation to numerically predict the solute and moisture dynamics in the matrix as the immobile water region (Šimunek, 2013). The unsaturated soil hydraulic properties are described in the Van Genuchten-Mualem equation (Šimůnek et al., 2013; Van Genuchten and Cleary, 1979). This model has been used successfully in the past to predict soil salinity accumulation after brackish water irrigation (Al-Busaidi et al., 2007; He et al., 2017). Assuming that soil water flow and salt transport are mainly vertically downward in the experimental columns, a uniform equilibrium, water flow in a partially porous medium can be described using the modified form of Richards equation (Šimůnek et al., 2013).

Solute transport in HYDRUS is described by the partial differential equation governing one-dimensional solute transport under transient flow in a variably saturated medium (Šimůnek et al., 2013):

$$\frac{\partial \theta C}{\partial t} = \frac{\partial}{\partial z}\left(\theta D \frac{\partial C}{\partial z}\right) - \frac{\partial \upsilon \theta C}{\partial z} \qquad (3.1)$$

where D is the dispersion coefficient ($L^2 \cdot T^{-1}$), C is the solute concentration ($M.\ L^{-3}$), and υ is the average pore water velocity ($q.\ \theta^{-1}$) in the flow direction ($L \cdot T^{-1}$). The dispersion coefficient is defined as (ignoring molecular diffusion) (Šimůnek et al., 2013):

$$D = \lambda . \upsilon \qquad (3.2)$$

where λ is dispersivity (L). The dispersivity is a material dependent constant which is a factor of the flow rate. Furthermore, it is the only solute transport parameter that is required for numerically solving the CDE equation of the water flow model and was assigned the value of 4 to represent the column condition (Li et al., 2015).

To study the water and solute uptake by roots, the crop root distribution was estimated to be 86% for top 5cm and the remaining 14% distributed linearly at the depth of 5-10 cm. The simulated soil profile was 40 cm with one soil layer and eight 5-cm depth subregions. The minimum and maximum time steps were the HYDRUS default values of 10^{-6} d and 10 days, respectively. The initial soil water and salinity contents were measured from 5-cm sampled soil rings prior to the salinity stress tests. In the graphics editor of the soil profile, 100 nodes in a single soil type were selected.

The van Genuchten-Mualem hydraulic model with no hysteresis was selected as the soil hydraulic model. The top water flow boundary consisted of a time variable flux and the bottom boundary condition was set to seepage face with zero cm height as recommended to be used in laboratory soil columns when the bottom of the soil column is exposed to the atmosphere (gravity drainage) (Šimůnek et al., 2008). The irrigation volume and irrigation water quality were introduced in the graphic soil columns time variable boundary conditions.

The solute transport upper and bottom boundary conditions were set to concentration flux and zero concentration gradient, respectively. The initial concentration was the mass solute per soil volume and ranged between 0.28 and 0.82 dS m^{-1}. The initial water content of the column varied in the range of 0.244 to 0.354 cm^3 cm^{-3}. The distribution coefficient of chemical species was set to zero assuming EC_{sw} to be an independent solute which was available only in the liquid phase (Ramos et al., 2011).

The Feddes root water uptake reduction model (Feddes et al.,1988) was selected with no salinity stress as the actual evapotranspiration values including all the stresses were manually introduced to the HYDRUS model. The resulting water stress reduction was parameterized using the following Feddes parameters, which were slightly modified from the values of HYDRUS database to adopt for no water stress conditions, as:
h_1 = -10, h_2 = -25, h_3 = −1500, h_4 = −8000 cm. The critical index for water uptake of less than 1 was considered to have no compensation root water uptake.

The initial soil solution salinity was determined in terms of EC of saturated extract before it was converted to EC of soil solution which was used in HYDRUS based on the following equation (Corwin and Lesch, 2003):

$$EC_{sw} = \frac{BD\,.SP\,.EC_e}{100.\theta} = \frac{\theta_s\,.EC_e}{\theta} \tag{3.3}$$

where EC_{sw} is the electrical conductivity of soil solution, EC_e was the electrical conductivity of saturated paste extract, *SP* was the saturation percentage (the water content of the saturated soil-paste expressed in terms of dry-weight basis), *BD* is the bulk density (gr cm^{-3}) and θ_s and θ are the saturated soil water content and soil water content (L^3. L^{-3}). The values of soil specific parameters of longitudinal dispersivity, D_L [L], dimensionless fraction of adsorption sites (Frac.) and Immobile water content (Thlm) were selected as 8.5, 1, and 0 based on the model recommendations (Ramos et al., 2011). Solute specific parameters of the molecular diffusion coefficient in free water, D_w (L^2T^{-1}) and molecular diffusion coefficient in soil air, Da (L^2T^{-1}) were set to two and zero respectively by default.

Evaluation of the Model Performance

Statistical analysis was used to evaluate the agreement between the data measured in the greenhouse and the results obtained using the HYDRUS-1D model. In this analysis the mean bias error (MBE), mean absolute error (MAE), root mean square error (RMSE), correlation coefficient (r), and Nash-Sutcliffe modelling efficiency (NSE) were used to compare the observed versus model predicted variables using the formulas below:

$$MBE = \frac{1}{N}\sum_{i=1}^{N}(O_i - P_i) \tag{3.4}$$

$$MAE = \frac{1}{N}\sum_{i=1}^{N}|O_i - P_i| \tag{3.5}$$

$$RMSE = \frac{1}{N}\sqrt{\sum(P_i - O_i)^2} \tag{3.6}$$

$$r = \frac{\sum(\mathrm{O}-\bar{\mathrm{O}})(P-\bar{P})}{\sqrt{\sum(O-\bar{O})^2\sum(P-\bar{P})^2}} \tag{3.7}$$

$$NSE = 1 - \frac{\sum_{i=1}^{n}(O_i - P_i)}{\sum_{i=1}^{n}(O_i - O_{mean})} \tag{3.8}$$

Whereas, P_i is the predicted value corresponding to the observed value of O_i and n is the number of data points which is eight in this study. MBE and MAE represent the difference between the predicted and measured data, the closer they are to zero, the better is the simulation performance. RMSE represents the degree of overestimation and is generally greater than or equal to MAE (Ramos et al., 2011). The closer the RMSE is to zero, the better is the model performance. Correlation coefficient ranges between zero to one and the value of one represents great performance. NSE range is -∞ to 1. A perfect match between the simulated and experimental is achieved when NSE is equal to 1. NSE of zero indicates that the model predictions are as accurate as the mean of the observed data. NSE of less than zero indicates that the observed mean is a better indicator than the model. Statistical analysis for significant differences were conducted using the General Linear Model procedure in *Minitab V.13* (Minitab Inc., Pen., USA). The analysis was based on One-way ANOVA along with Tukey's pairwise comparison test at the family error of 0.05 (95% confidence interval). General Linear Model is a flexible and useful statistical model as it assumes an exponential family model for the response. Once a significant difference between groups was determined, then the Tukey's test was used to determine overall accuracy of the tests.

Results and Discussion

The cumulative daily irrigation and evapotranspiration (ET) amounts are represented in Figure 3.1. It is observed in the Figure 3.1 that cumulative amounts of irrigation and ET varied in the range of 0-27.7 and 0-26.2, respectively. Which represents that the until the last day of experiment, irrigation was applied in the quantity to meet the crop ET requirements.

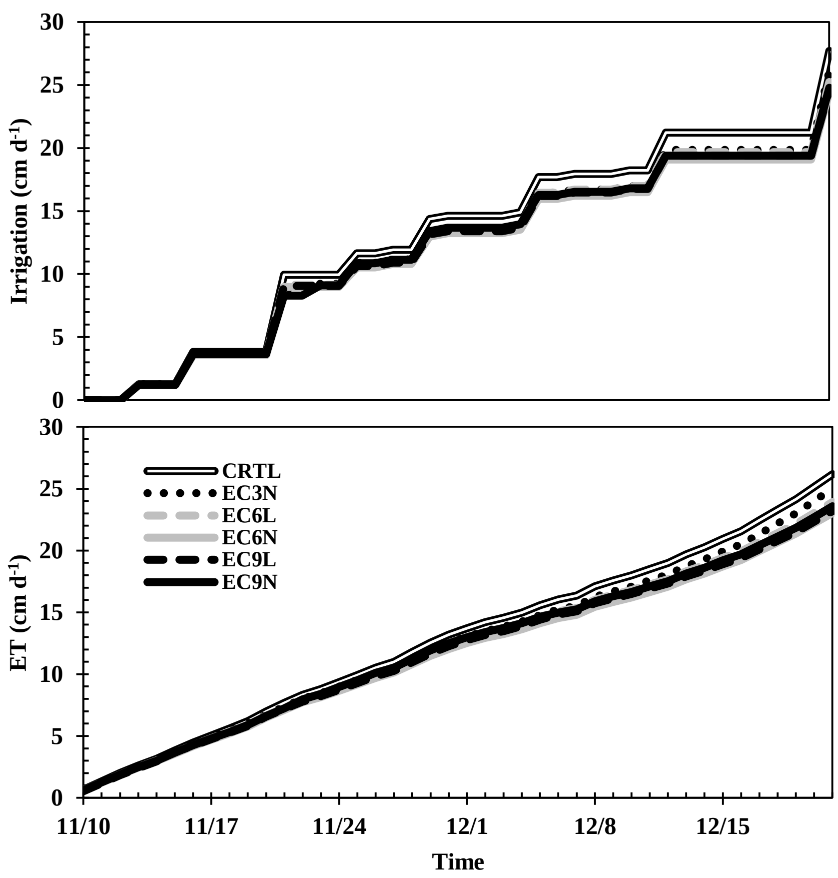

Figure 3.1. Cumulative Irrigation application and evapotranspiration amounts for all treatments.

Soil Salinity

Figure 3.2 illustrates the effect of various saline irrigation and application of leaching on soil salt distribution for soil irrigated with various saline water.

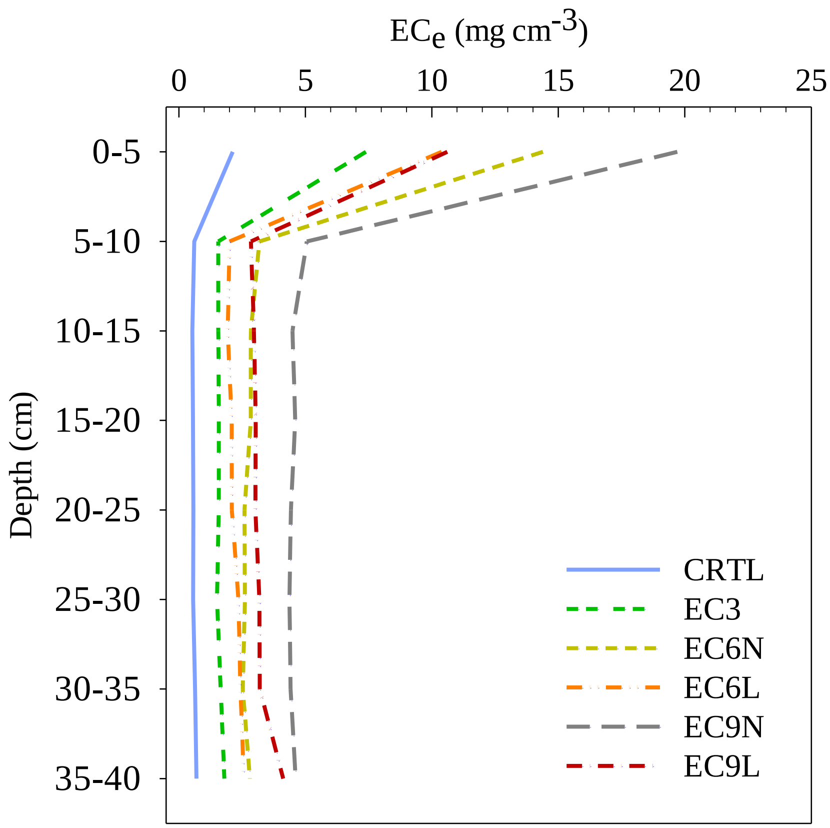

Figure 3.2. Soil salinity profile for various treatments of saline water irrigation with target EC values of 0.47 (CRTL), 3, 6, and 9 dS m^{-1} with leaching (L) and without leaching (N).

It appears that leaching application did not affect the salinity profile trend, however, it decreased soil salinity throughout the soil column. Therefore, application of leaching was able to decrease soil salinity irrigated with 9 dS m^{-1} to less than soil irrigated with 6 dS m^{-1} without leaching application.

Table 3.4 displays the values of maximum and average soil analysis parameters for various treatments of saline water irrigation with (L) and without (N) leaching. The maximum values were observed at the shallowest soil layers of zero to 5.1 cm in all treatments. It appears that soil solution salinity increased with increasing irrigation water salinity. However, leaching application decreased the soil solution salinity to lower levels than columns irrigated with less saline water. For example, although without leaching application soils irrigated with 9 dS m^{-1} were greater than soils irrigated with 6 dS m^{-1}. However, soils irrigated with 9 dS m^{-1} with leaching application contained less soil solution salinity than soils irrigated with 6 dS m^{-1} without leaching application.

Blanco and Folegatti, (2002) observed a similar trend of increasing soil salinity with increasing irrigation water salinity in a greenhouse study with irrigation water salinities of 1.54, 3.10, and 5.20 dS m^{-1}. They also reported that application of a leaching fraction of 0.2 with the same water quality as irrigation water was not able to prevent soil salt build up. Soil solution salinity generally ranges from salinities close to irrigation water salinity and increases by the depth. However, steady-state conditions are not reached in irrigated agriculture (Rhoades et al., 1992).

The comparison between various soil parameters measured at the top layer versus the average values throughout the soil profile is presented in Table 3.4. For all parameters, the top layer values were much greater than profile average values, likely due to the fact that the values decreased significantly in the deeper layers. Statistical significance of three replicates for each parameter and each soil layer was evaluated using Tukey’s pairwise comparison at 95% confidence intervals using *MINITAB V13*. The grouping results are presented with alphabetic characters as displayed in Table 3.4-b.

Table 3.4.

Soil solution analysis of irrigation with various water quality after the last irrigation application with leaching (L) and without leaching (N).

0-5 cm			(a)				
Treatment	**ECe ($dS\ m^{-1}$)**	**SAR %**	**Na^+ (ppm)**	**Ca^{2+} (ppm)**	**Mg^{2+} (ppm)**	**K^+ (ppm)**	**pH**
CTRL	3.23[a]	8.53[a,b]	346[a]	70[a]	33[a]	24[a]	7.1[a]
EC3N	9.24[c]	7.30[b]	650[c]	354[c]	147[c,d]	58[b,c]	7.0[a]
EC 6L	12.98[c]	7.63[b]	816[c]	508[c]	216[c]	76[d,b,c]	6.8[a]
EC 6N	17.99[b]	8.30[a,b]	1067[b]	745[b]	312[b]	77[d,b]	6.7[a]
EC 9N	24.63[d]	6.67[a]	1373[d]	1103[d]	430[e]	88[d]	6.6[a]
EC 9L	13.27[c]	7.60[a,b]	802[c]	498[c]	214[c]	61[b,c]	6.9[a]
Average 0-40 cm			**(b)**				
Treatment	**ECe ($dS\ m^{-1}$)**	**SAR %**	**Na^+ (ppm)**	**Ca^{2+} (ppm)**	**Mg^{2+} (ppm)**	**K^+ (ppm)**	**pH**
CTRL	1.2 [a]	5.5[a]	129.1 [a]	24.7 [a]	10.1 [a]	12.0 [a]	6.7 [a]
EC 3N	3.3 [b]	5.0 [a]	239.4 [a,b]	105.5 [b]	42.5 [b]	25.7 [b]	6.4 [b]
EC 6L	4.6 [c]	5.0 [a]	299.8 [b,c]	158.8 [c]	64.6 [c]	31.7 [c]	6.3 [b,c]
EC 6N	5.9 [c]	5.2 [a]	371.1 [b,c]	227.7 [c]	90.0 [c]	33.0 [c]	6.3 [b,c]
EC 9N	8.1 [d]	5.5 [a]	475.1 [b,c]	334.1 [d]	125.4 [d]	40.5 [d]	6.1 [b,c]
EC 9L	5.8 [c]	5.3 [a]	357.6 [c]	206.5 [c]	80.4 [c]	32.1 [c]	6.2 [c]

[a, b, c,d] Grouping information in Tukey's pairwise comparison for all replicates for each layer. Treatments with different letters are significantly different for each parameter.

Tukey's pairwise comparison demonstrated that for both EC_e of topsoil and average of soil profile, there were no significant differences between EC3N, EC6L, and EC9L. However, there were statistically significant differences between CTRL, EC6N, and EC9N. This clearly demonstrates the effectiveness of fresh water leaching application which was able to reduce salinity of the soil profile irrigated with 9 dS m^{-1} to those irrigated by 3 dS m^{-1}. Similar statistical differences were observed for Na^{+}, Mg^{2+}, and Ca^{2+} for topsoil and Na^{+} and Mg^{2+} for the soil profile. There was no significant difference between SAR values throughout the soil profile for the various treatments.

The impact of applying various water qualities of irrigation water together with the effects of good quality water leaching application on the distribution of Na^{+}, Mg^{2+}, K^{+}, and, Ca^{2+} are noticeable in Figure 3.3. Similar to the trend observed in Figure 3.1, the maximum ion concentration for all treatments were observed in the top 5 cm of the soil. All treatments showed a sharp decreasing trend with depth from 5 to 10 cm and showed very minor changes at the deeper soil levels. The increase in irrigation water salinity lead to increases in concentrations of selected cations throughout the soil profile. Wang et al. (2017) observed a similar trend of salinity with the maximum salinity at the topsoil when saline irrigation water was applied to compensate evaporation.

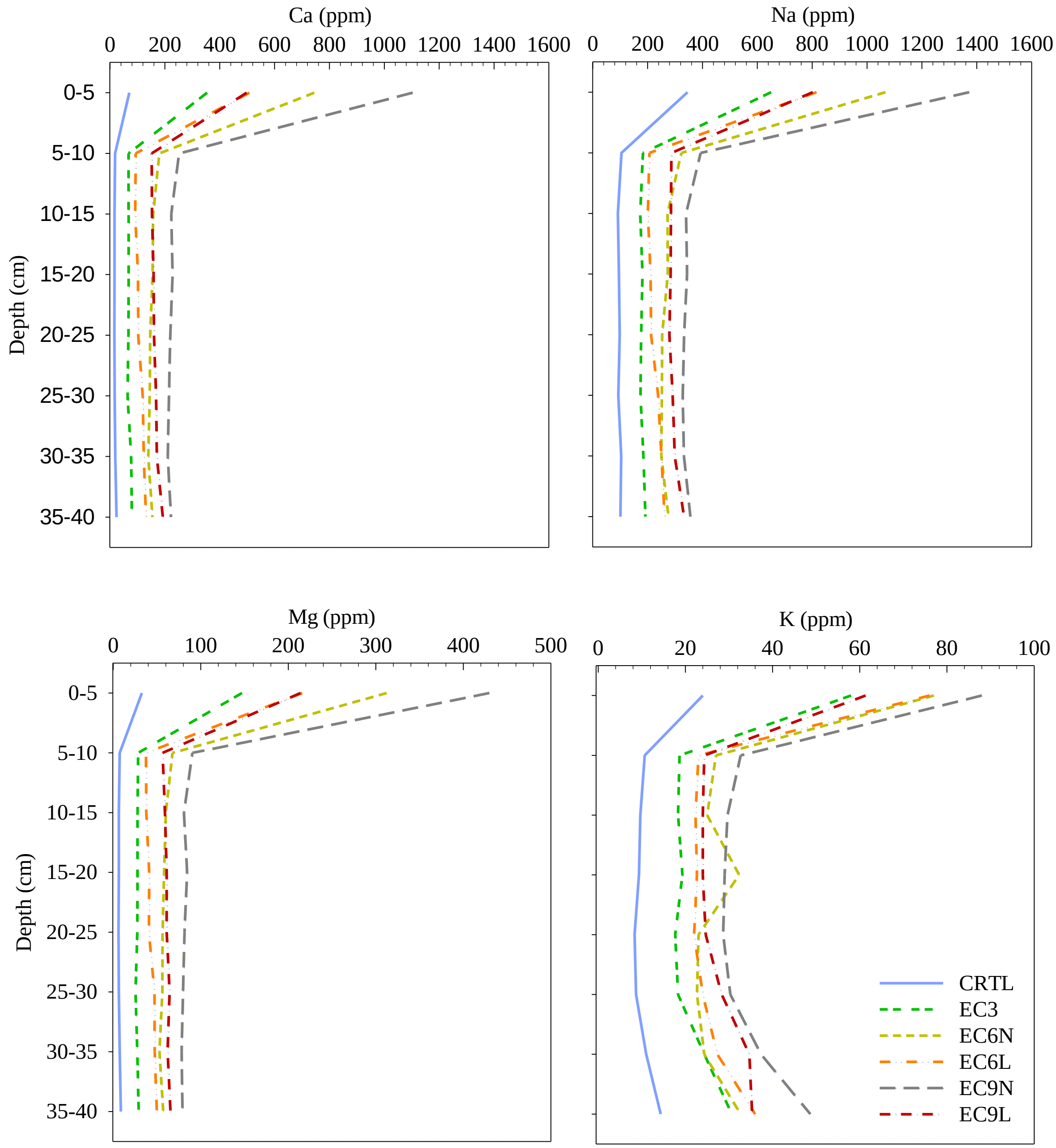

Figure 3.3. Distribution of major ions of Ca, Mg, Na and K throughout the soil profile for various treatments of saline water irrigation with ctrl (0.47), 3, 6, and 9 dS m^{-1} with leaching (L) and without leaching (N).

Leaching application decreased cation concentrations to levels below the ones irrigated with more saline irrigation water. In this respect, sodium concentrations in the topsoil layer for columns irrigated with 6 dS m^{-1} was 1067 ppm which was decreased to 816 and 802 ppm for the EC6L and EC9L treatments. Comparison of EC6N and EC6L treatments showed that leaching application lead to 28% and 23% decreases in soil salinity for the topsoil and average of the soil profile, respectively. It also lead to 19%, 30%, and 28% decreases in sodium, calcium, and magnesium concentrations in the soil profile.

Comparison of the effect of leaching application of good quality water was specifically useful in comparison of soil salinity between various irrigation water qualities after leaching application. In this respect, comparison of EC6L and EC9L treatments showed 2% and 20% increases in soil salinity of topsoil and the average of soil profile by an increase in irrigation water salinity. A very similar trend was observed for K^+, Mg^{2+}, and Na^+ for which values are not presented, however, K^+ showed a slightly different pattern as the concentration of all treatments increased at soil depths of greater than 30 cm. Furthermore, leaching application had a negligible effect on soil salinity between treatments EC6N and EC6L similar to what was observed between treatments of EC9L and EC6L. In this respect, increases in salinity increased K^+ levels even when leaching was applied as there were a 24% increase and 1% decrease at the topsoil and average of soil profile treatments 6L and 9L.

As illustrated in Figure 3.4., sodium adsorption ratio (SAR) and pH showed different trends than EC and cations. The pH values showed that saline water irrigation increases soil acidity throughout the soil profile and acidity increases by depth, particularly for more saline waters. The acidity at the top 10 cm of the soil which most of the root is located is however negligible. The pH values were slightly greater for CRTL treatments than the rest of treatments and the rest of treatments did not vary significantly. The SAR levels decreased sharply by depth for the top 10 cm of the soil and increased slightly by depth at the deepest soil layers. The treatments of EC9L and CRTL had slightly greater values than the rest of the treatments but the differences between the treatments were negligible.

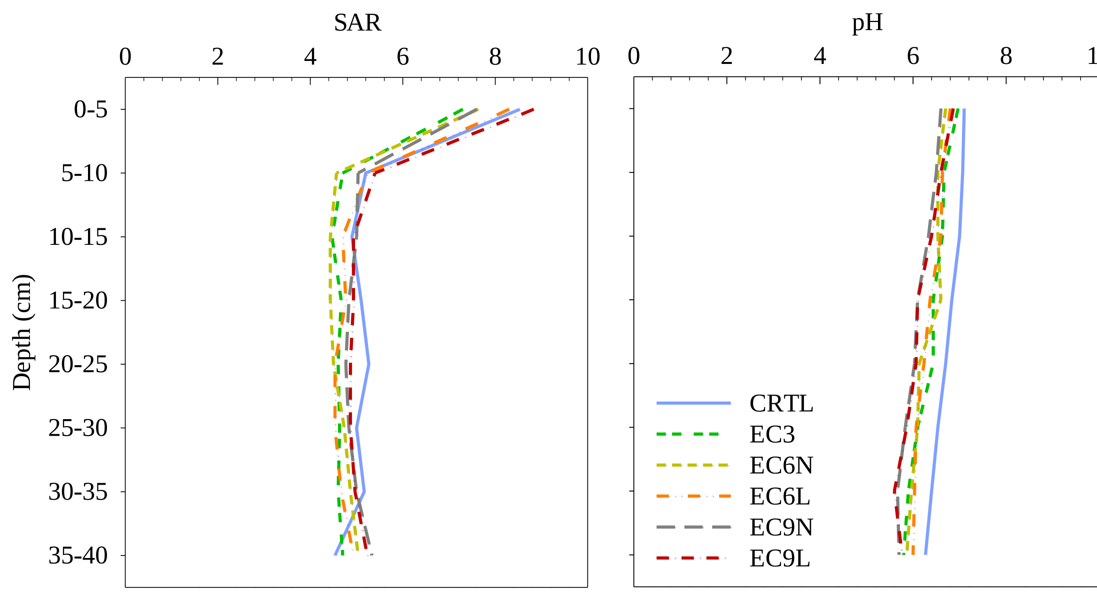

Figure 3.4. Distribution of pH and SAR throughout the soil profile for various treatments of saline water irrigation with CRTL, EC3, EC6, and EC9 with leaching (L) and without leaching (N).

Simulation Results

The comparison of the soil profile salinity distribution between the observed and simulated average salinity values represents that HYDRUS over-estimated average soil profile salinity for all treatments (Figure 3.5). However, the over-estimation was negligible as its minimum value was 0.2 dS m^{-1} for CRTL treatment and increased to a maximum value of 1.44 dS m^{-1} for EC6N treatment and decreased to value of 0.98 dS m^{-1} in EC9N treatment. Furthermore, salt accumulations were more concentrated at the top 10 cm of the soil and were predicted to be more distributed in the top 15 cm of the soil. Similar to soil column experience, simulation results showed that soil salinity increased in the soil columns with an increase in irrigation water salinity. Moreover, both simulated and observed soil profiles demonstrated a sharp decrease by depth in the top soil layers after which they showed a mild or no reduction in salinity by depth.

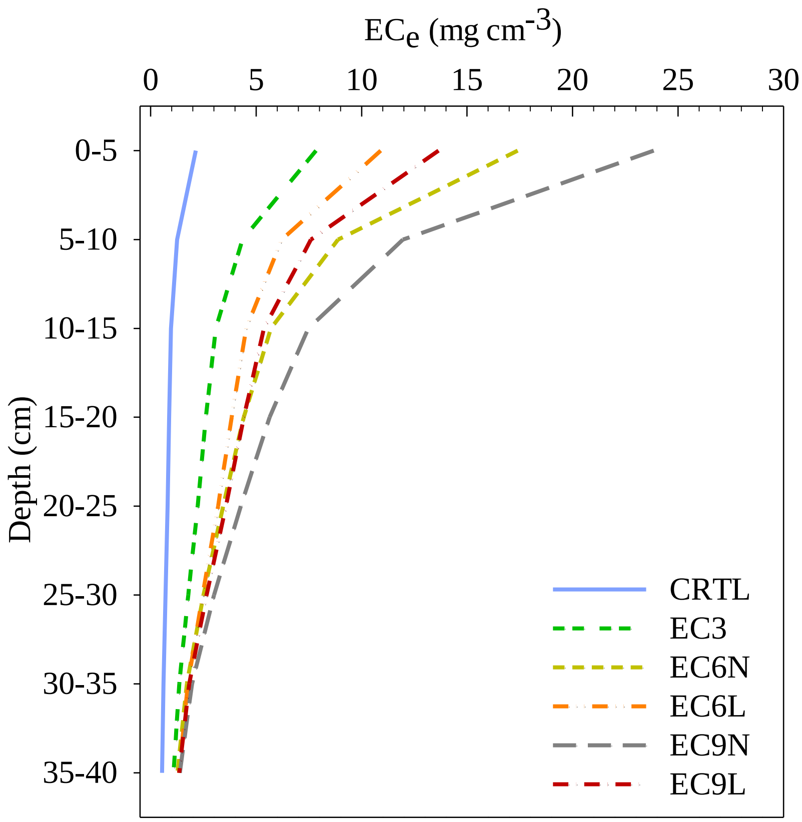

Figure 3.5. Distribution of soil electrical conductivity by depth for various treatments of saline water irrigation namely CRTL, EC3, EC6, EC9 with electrical conductivity of 0.47, 3, 6, and 9 dS m^{-1} with leaching (L) and without leaching (N).

Figures 3.5 and 3.6 represent simulated soil profile salinity and comparison of the observed and predicted average soil profile salinity. HYDRUS overestimated the average EC in all treatments and this overestimation increased by increase in irrigation water salinity as represented in Figure 3.6.

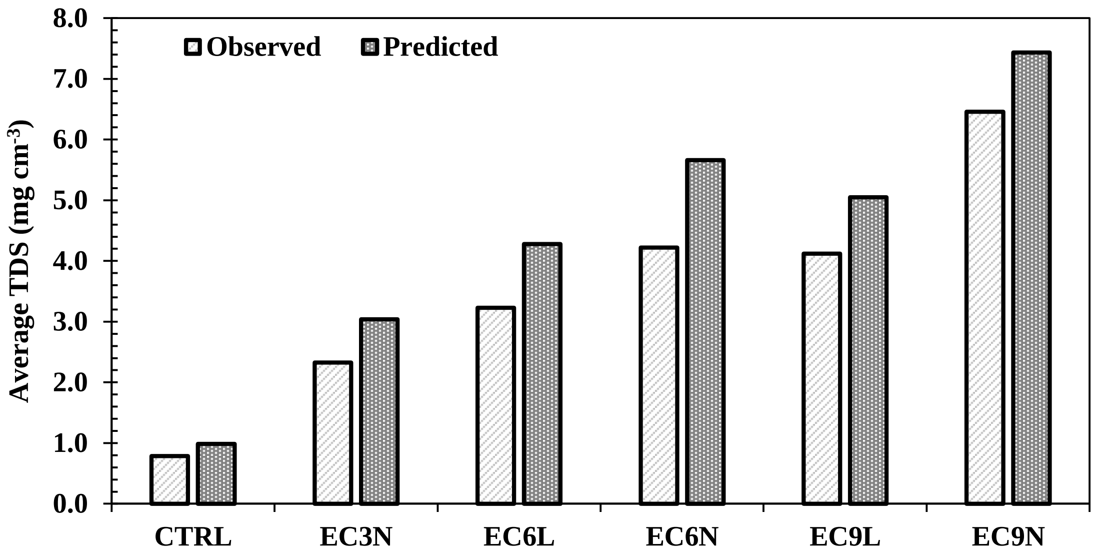

Figure 3.6. Distribution of average soil electrical conductivity by depth for various treatments of saline water irrigation namely CRTL, EC3, EC6, EC9 with electrical conductivity of 0.47, 3, 6, and 9 dS m^{-1} with leaching (L) and without leaching (N).

To evaluate the performance of the HYDRUS 1-D model in predicting the average soil profile salinity, further statistical analysis was performed. Table 3.5 represents the key summary statistics for each of the estimated regression functions. MAE and MBE values closer to zero represent a better performance of HYDRUS in predicting lower soil salinities. MAE values increased by increase in irrigation water salinity from the value of 0.31 to 3.48 mg cm^{-3} for CRTL and EC9L, respectively. Similar trend was observed in MBE with the difference that the maximum MBE was observed in treatment EC6N with the value of 1.8 mg cm^{-3} and was decreased to 1.16 mg cm^{-3} for the treatment EC9N. RMSE values ranged between 0.39 to 4.26 mg cm^{-3} for CRTL to EC9L treatments. RMSE of closer to zero indicated a great agreement between the predicted and simulated values. Correlation coefficient ranges between zero to one, the value of one representing a strong performance of the model. In all treatments r was greater than 0.8 which represents the good performance of HYDRUS 1-D.

Table 3.5.

Summary Statistics for the root zone average ECe between HYDRUS simulation and observed values.

Treatment	**MBE**	**RMSE**	**MAE**	**r**	**NSE**
CTRL	0.25	0.39	0.31	0.88	0.62
EC3N	0.89	1.56	1.21	0.87	0.58
EC 6L	1.31	2.46	1.90	0.83	0.47
EC 6N	1.80	3.30	2.57	0.91	0.53
EC 9N	1.16	3.11	2.62	0.80	0.00
EC 9L	1.22	4.26	3.48	0.90	0.54

Ramos et al., (2011) reported values of 1.6 and 2.04 for RMSE and MAE respectively, for a field study with 244 sampling points. Although other researchers reported a smaller value for RMSE. Li et al. (2015) reported a RMSE value of 0.037 and 0.046 dS m^{-1}. The values are a factor of the irrigation water salinity. Ramos et al. (2011) reported that the degree of RMSE exceeding MAE is a good indicator of the outliers or the variance of differences between the observed and simulated data. As observed in Table 3.5, in all treatments RMSE is greater than MAE and the values of these statistical parameters increase with an increase in soil salinity. The NSE indicated how well the plot of observed and predicted data fits the 1:1 line. NSE of one indicates a perfect match between the observed and simulated data and a value of zero indicates that model predictions are as accurate as the mean of the measured data (Nash and Sutcliffe, 1970). The NSE values in Table 3.5 conform a better prediction of average salinity in lower irrigation salinities. HYDRUS overestimated EC values and comparison of statistical parameters (Table 3.5) and average EC values (Figure 3.6) confirm that HYDRUS better estimated the lower saline irrigation treatments which is in alignment with what is observed in past studies (Li et al., 2017).

There are many limitations that might have affected the differences between the measured and predicted soil profiles. The solute transport parameter (dispersivity) which is a factor of particle size and uniform distribution of soil that affects how salts move in the soil need to be measured in the lab. For an accurate soil salt and water dynamics, standard HYDRUS solute transport model requires a variety of input data that need to be readily available. Using a distribution coefficient (K_d) equal to zero is under the assumption that cations are transported independently from each other through complex processes of adsorption and cation exchange using linear and non-linear adsorption isotherms. Based on the results, despite the considerable demand in accurate input data, HYDRUS 1-D is an effective tool to study the soil solution solute distribution and accumulation in various scenarios.

Conclusions

In this study, a monitoring and modeling approach was undertaken to study the effect of using saline irrigation water, and fresh water leaching application on soil salinity profile. Various treatments were studied in the experimental study using irrigation water with salinities of 0.5, 3.0, 6.0, and 9.0 dS m^{-1} in soil columns of Bermudagrass. The effect of leaching application was studied by comparing the soil salinity profile in treatments with and without final fresh water leaching application. Increasing salinity of irrigation water lead to an increase in soil profile salinity, particularly at the top 10 cm of the soil. However, application of good-quality leaching reduced soil salinity. Soil salinity decreased sharply in the top 10 cm of the soil and stayed at the same levels for the deeper soil layers. Although the soil salinity increased with increasing salinity of irrigation water, good-quality leaching reduced the soil salinity levels to those irrigated by less saline water. A similar trend was observed for major cations of Ca, Mg, Na, and K. There was a gradual increase in K concentrations at the deeper soil layers. Fresh water leaching application reduced average soil profile salinity by 23% and 29% for the soils irrigated by 6 and 9 dSm^{-1}, respectively. These reductions were greater at the top soil, which were 28 and 46% for the soils irrigated by 6 and 9 dSm^{-1}.

The performance of HYDRUS 1-D was tested by comparing the soil salinity profile in greenhouse for all treatments. Based on the results of the model performance, there was a good agreement between the simulated and measured soil profile salinities. HYDRUS slightly over-estimated the salinity values. The correlation coefficient for al treatments were greater than 0.8 and RMSE values ranged between 0.2 and 1.44 dS m^{-1}. HYDRUS estimated soil salinities better for treatments under lower irrigation water salinities.

Regardless of the great demand for input data, HYDRUS can be used for choosing the most suitable management practices, particularly in arid and semi-arid irrigated fields which have scarce fresh water resources.

The above results contribute to our understanding of the impacts of utilizing low-quality irrigation waters in agricultural soils. In this study, potential utilization of various qualities of saline water for irrigation was studied along with the effect of leaching on the soil profile salinity after saline water irrigation.

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CHAPTER IV

SIMULATING THE IMPACTS OF SALINE WATER APPLICATION ON SOIL UNDER VARIABLE IRRIGATION LEVELS IN THREE OKLAHOMA CLIMATIC DISTRICTS

Abstract

Water scarcity and soil salinization are among the major concerns and wide-spread environmental problems of irrigated agriculture. Regarding the competition for fresh water resources, reuse of low-quality water for irrigation is gaining interest. Produced water is the largest by-product of oil and gas fracturing in Oklahoma. It is crucial to manage the locally available produced water in Oklahoma, because of large volumes of saline water produced and environmental problems associated with their disposal. Furthermore, utilization of these saltwaters could have positive impacts on irrigated agriculture especially during prolonged drought that would halt agricultural production.

Prior to field-scale experiments with saline water as irrigation source, it is beneficial to use widely used numerical simulations such as HYDRUS to investigate the distribution of solutes and water fluxes in a long-term saline irrigation district. In this study, potential surface application of locally produced water in various parts of Oklahoma was investigated using 36 scenarios. Various irrigation water salinities of 6, 44, 70, and 340

dS m^{-1} under various water quantity applications of under, full, and over irrigations were simulated using the climatic and soil parameters of central, southwest, and panhandle area in Oklahoma.

Based on the results of this study, over-irrigation in low saline water was able to leach salts and increase relative root water uptake (RRWU), however, in more saline waters increasing irrigation water quantity decreased RRWU as a result of excess salt accumulation in the soil root zone and thus salinity stress. Moreover, increase in precipitation and thus decrease in irrigation requirements lead to a decreased relative solute accumulation in Central area as compared to Southwest and Panhandle. The relative solute amounts reached the peak value of 1201, 632, and 470 times its initial value for panhandle, southwest, and central Oklahoma, respectively which received at average annual rainfall of 643, 492, and 280 mm of precipitation. Based on the results of this study, PW needs to be diluted to the crops' salinity tolerance threshold levels and then it can be used as irrigation water. Coupling this study with various management practices can be a starting point for understanding the long-term effects of saline water irrigation with management practices without adversely affecting environment.

Keywords

Saline irrigation, produced water, numerical simulation, water and salt dynamics

Introduction

The global urban population is increasing continuously and is projected to reach 7.3 billion by 2050 (Cumberlege, 1993). A substantial portion of this population's food and fiber needs is provided from irrigated land (Yurtseven et al., 2014). The area under severe water stress is predicted to expand from 36.4 to 38.6 million km^2 from 1995 to 2025 (Alcamo, et al., 2017). Water shortage is predicted to be one of the major challenges of the human civilization in the 21st century, moreover, dissolved salts are one of the major pollutants that degrade the quality of water (Qadir et al., 2009). In the past, wastewater resources were considered liability due to the cost of the disposal and the fact that when released to the environment, they caused "salt scars" (Zhang, et al., 2005). However, nowadays, they are increasingly viewed as potential alternative water resources due to the increased demand for fresh water resources, increased drought occurrence, and the ever-increasing production of the low-quality water from different resources (Gleick, 2000; Wang et al., 2017).

The inevitable change in the direction of reusing low-quality water, such as saline water, requires an understanding of its potential long-term environmental impacts and sustainability (Tilman et al., 2002). Maintaining soil health and crop productivity are the main concerns in the utilization of saline water for agricultural production (Verma et al., 2012). Numerous research studies have been conducted on agricultural reuse and land application of the low-quality water (Oster & Grattan, 2002; Dudley et al., 2008). Reuse of saline water is a sustainable approach in arid and semi-arid areas like Oklahoma that could potentially decrease the dependence of irrigation of non-food crops on fresh water resources (O'Connor et al., 2008). The main sources of saline water in Oklahoma include urban wastewater, naturally occurring saline groundwater, and produced water (PW) from oil and gas exploration. (Hagstrom, et al., 2016). PW has recently been gaining more attention due to its being the largest volume industrial waste stream and the problems with its reinjection for underground disposal (Veil, 2015).

Produced water is the emerging source of low-quality water in Oklahoma. In 2012, over 2.3 billion barrels of water were produced in Oklahoma which accounted for 11% of all US PW (OWRB, 2015), from which 47% of the PW was reinjected for enhanced recovery, 47% was injected for disposal, 6% was managed through offsite commercial disposal, and no PW was managed for beneficial use (Veil, 2015). Nationally, only around 5% of the PW was surface discharged in 2012, and the major trends in produced water management has not made major changes (Veil, 2015). Most of the water produced in Oklahoma is located in the central and northern counties and production decreases in the eastern and western counties in Oklahoma as observed in Figure 4.1. A very similar pattern is observed for the distribution of salinity of the PW although the PW salinity data is not available for many wells especially on the western and eastern parts of the state.

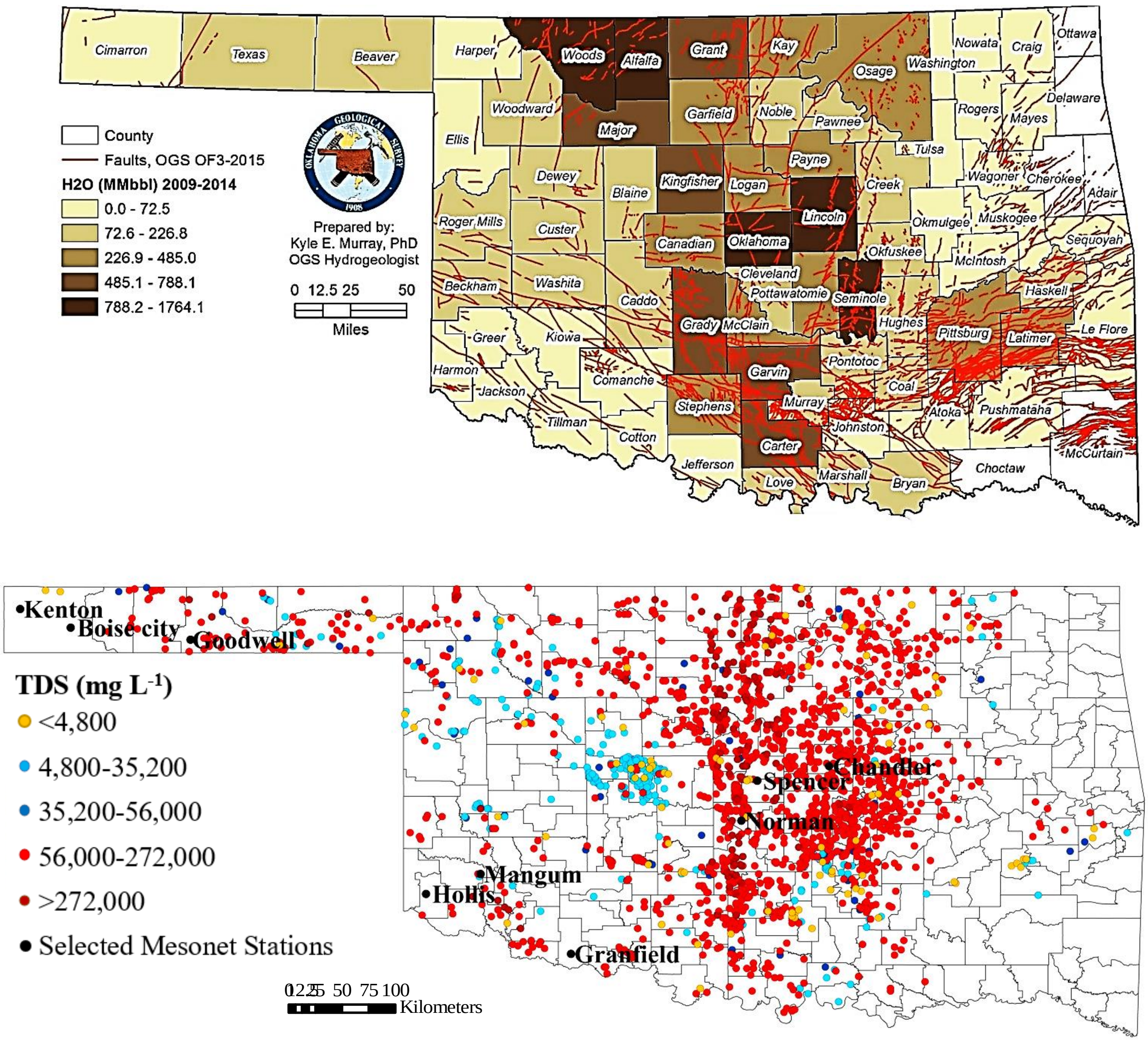

Figure 4.1. Total Produced water by County (top) and salinity distribution of PW wells (bottom) in the State of Oklahoma (Murray, 2016; USGS, 2018).

Regarding the crucial need for water resources and potential hazards of reinjecting PW, there have been studies on the application of PW among which agricultural reuse and land application have been debated in few studies (OWRB, 2015). However, there seem to be very few experimental studies to use PW and study the effect of its land application on soil salinity. Lewis (2015) conducted a field experiment at Lubbock, Texas comparing the effect of irrigation with a blend of desalinated PW on cotton yield. They used a blend of one-part local PW with four parts well water and evaluated the effect of the potential plant toxicity and soil salinity on cotton yield and lint quality. The EC of soil irrigated with the blended water was lower than that of irrigated with well water throughout the soil profile. Based on the field experiments, irrigation with desalinated PW blended with well water (ratio of 1:4) did not reduce the cotton production yield or lint quality and was able to reduce soil salinity parameters. Lewis (2015) claimed that blending higher amounts of desalinated PW could improve soil physical and chemical properties by reducing the salt load. It would also help conserve fresh water resources and enhance the longevity of agricultural production in arid and semi-arid areas. In another study, Al-Haddabi & Ahmad (2007) conducted an experiment and compared the effect of treated PW and fresh water on soil physical and chemical properties. Based on the results of their study, the land application of the treated PW even for short period adversely affected the chemical and physical properties of the soil and increased soil salinity and sodicity. They also reported that increasing the rate of water application was not able to reduce soil salinity. Such conflicts in the literature suggest that land application of PW should be studied locally and is a factor of site-specific parameters such as climate, soil type, and the quality of applied PW. Therefore, it is very important to conduct local studies using the meteorological, soil, and PW information for feasibility analysis of using PW in the region.

One specific concern regarding the land application of PW is its long-term sustainability issue (Qadir et al., 2009). The interaction between the hyper-saline PW and irrigation practices can be very complex and evaluating this interaction in the experimental field studies can be time-consuming, expensive, and lead to environmental hazards. Moreover, the magnitude of salt accumulation in the soil root zone, hazards to the crop production, and downstream contamination due to PW application may not be well-reflected in short-term studies given the climatic conditions such as precipitation patterns of the study field (Wang, 2017). Therefore, modeling long-term soil and water interactions under local climate, soil type, and available PW is a great starting point to elucidate the spatial and temporal soil water and salt dynamics and the water fluxes in the flow domain. HYDRUS-1 D has been widely used due to its good performance in predicting salt and water dynamics in variably saturated media (Šimůnek et al., 2013). Few studies have investigated the effect of irrigation water salinity on root water uptake and long-term relative solute accumulation as a result of saline irrigation (Li et al., 2015; Mguidiche et al., 2015; He et al., 2017).

Saline water irrigation management is traditionally based on the application of extra irrigation water to maintain root zone salinity at acceptable levels (Ayers & Wescot, 1985; Crescimanno & Garofalo, 2006). However, the application of extra irrigation water volume results in the accumulation of salts in the soil rootzone, particularly in the case of inadequate leaching, shallow water tables, and high evapotranspiration rates (Gonçalves et al., 2006). Moreover, over-irrigation does not necessarily correspond to increased water use efficiency and has many adverse effects such as accumulation of salts in the soil root zone and contamination of downstream water resources (Amer, 2010). Therefore, it is important to investigate the optimal saline irrigation quantity and quality that would maintain the root-zone salinity at the acceptable levels and control the amount of salt and chemicals released to the groundwater and downstream waters (Mguidiche et al, 2015).

In this study, the HYDRUS-1 D model (Šimůnek et al., 2013) was used to simulate the soil water fluxes, solute transport, and root water uptake in various locations in Oklahoma. Our objectives were to conduct scenario analysis using HYDRUS-1 D model to study: 1) effects of long-term irrigation water application with various salinities on root water uptake, and water and salt fluxes in the selected flow domain, 2) to evaluate the optimal irrigation water quantity with under, full, and over irrigation using various qualities of saline water, 3) to analyze the impact of climatic and soil conditions in various part of Oklahoma on long-term root zone salt accumulation under saline irrigation.

Materials and Methods

Numerical model

We simulated water and salute dynamics using HYDRUS-1D model (Šimunek et al, 1999). Employing this model we simulated a one-dimensional water flow and solute transport in the variably saturated column (He et al., 2017). This model combines Richard's equation to describe water flow in the microspores as the mobile water region, and a mass balance equation to numerically predict the solute and moisture dynamics in the matrix as immobile water region (Šimunek, 2013). The dispersion length of 10 was selected in the inverse model with the minimum and maximum iteration of 5 and 20 based on the manual guidelines (Radcliffe & Šimůnek, 2010).

Simulation scenarios

Various simulation scenarios were carried out using HYDRUS 1-D to evaluate the long-term impact of irrigation practices on the temporal variations of the soil water and salinity profile. In this regard, we selected three parameters to monitor their effects on soil profile salinity. These parameters are namely the water quality, quantity, and climatic conditions. Table 4.1 represents a summary of the selected parameters and simulated treatments performed in this study. In this study, a comprehensive study of the combination of 36 different scenarios in terms of the effect of different irrigation quality and quantity was conducted.

Table 4.1.

Simulated scenarios.

Treatment	**Location**	**Irrigation quantity**	**EC ($dS\ m^{-1}$)**
C-U-EC1	Central	Under	6
C-U-EC2	Central	Under	44
C-U-EC3	Central	Under	70
C-U-EC4	Central	Under	340
C-F-EC1	Central	Full	6
C-F-EC2	Central	Full	44
C-F-EC3	Central	Full	70
C-F-EC4	Central	Full	340
C-O-EC1	Central	Over	6
C-O-EC2	Central	Over	44
C-O-EC3	Central	Over	70
C-O-EC4	Central	Over	340
S-U-EC1	Southwest	Under	6
S-U-EC2	Southwest	Under	44
S-U-EC3	Southwest	Under	70
S-U-EC4	Southwest	Under	340
S-F-EC1	Southwest	Full	6
S-F-EC2	Southwest	Full	44
S-F-EC3	Southwest	Full	70
S-F-EC4	Southwest	Full	340
S-O-EC1	Southwest	Over	6
S-O-EC2	Southwest	Over	44
S-O-EC3	Southwest	Over	70
S-O-EC4	Southwest	Over	340
P-U-EC1	Panhandle	Under	6
P-U-EC2	Panhandle	Under	44
P-U-EC3	Panhandle	Under	70
P-U-EC4	Panhandle	Under	340
P-F-EC1	Panhandle	Full	6
P-F-EC2	Panhandle	Full	44
P-F-EC3	Panhandle	Full	70
S-F-EC4	Panhandle	Full	340
P-O-EC1	Panhandle	Over	6
P-O-EC2	Panhandle	Over	44
P-O-EC3	Panhandle	Over	70
P-O-EC4	Panhandle	Over	340

Water quality, in terms of electrical conductivity of irrigation water, was selected because the PW composition and salinity varies by location and the fracturing processes used and it was assumed that PW was blended with good-quality irrigation water to reach designated salinities. In this regards, various water qualities were selected to represent the mid-range brackish, high brackish, saline, and brine water categories with EC values of 6, 44, 70, and 340 dS m^{-1} equivalent to TDS values of 4800, 35200, 56000, 272000 ppm. The next selected parameter was water quantity and various irrigation volumes were selected to represent 0.7, 1.0, and 1.3 times the evapotranspiration considering the climatic conditions of the selected locations. Climate was selected as another parameter as it affects the precipitation and evapotranspiration. Therefore, various climatic locations in Oklahoma were selected to incorporate climatic factors in this study. The selected sites covered the major districts of Oklahoma namely Panhandle, Central, and Southwest Oklahoma. In each site, three Mesonet station with the least missing climate data points were selected which were Boise city, Kenton, and Goodwell in Panhandle area, Chandler, Norman, Spencer in central Oklahoma, and Granfield, Hollis, Mangum stations in Southwest. The climatic data such as average monthly precipitation and evapotranspiration of these Mesonet stations were taken for the time frame of 1998 to 2017 (20-year). A summary of important meteorological data is presented in Table 4.2.

Table 4.2.

Meteorological parameters in the selected sites for the 20-year average of 1998-2017.

Parameter	Central	Southwest	Panhandle
Mean R_s[1] (KWh/m^2)	4.65	4.99	5.22
Min T_{air}[2] (°C)	10.32	9.34	4.88
Max T_{air} (°C)	22.31	24.31	21.38
Mean T_{air} (°C)	16.16	16.58	12.95
Mean Wind 2m (kph)	10.29	11.05	13.80
Total Precip. (mm)	420.9	900.5	655.4
Mean RH[4] (%)	65.98	62.63	55.65
Mean VD[5] (kpa)	0.84	1.05	1.01

[1] Total daily accumulation of solar radiation

[2] Air temperature

[3] Average Wind Speed at 2m

[4] Relative humidity

[5] Vapor Deficit

Input parameters to HYDRUS

Soil parameters

Applying the HYDRUS model for field condition requires numerous soils, crop, water, and weather parameters. The major soil type for Panhandle was clay loam with soil class of fine, mixed, superactive, mesic Aridic Paleustolls, for Central Oklahoma the dominant soil type was soil loam with soil class of fine-silty, mixed, superactive, thermic Pachic Haplustolls, and for southwest Oklahoma it was fine-loamy, mixed, superactive, thermic

Pachic Argiustolls. A more detailed soil information for each soil layer is presented in Table 4.3.

Soil data was retrieved from USDA soil survey for the dominant soil type in the selected Mesonet stations (WSS, 2018). For each soil layers, hydraulics parameters were calculated using ROSETTA based on the percent sand, silt, clay, and soil bulk density as observed in Table 4.3. The initial soil water capacity and soil solution salinity were also retrieved form the USDA soil survey. Based on the soil type, the soil field capacity and wilting point were estimated, and the initial water content was assumed to be at its field capacity. Soil information for all the locations was available for the top 2 meters with various soil types of different depths as illustrated in Table 4.3.

Table 4.3.

Soil characteristics and hydraulic parameters of van Genuchten-Mualem functions

Station	**Soil depth (cm)**		**Sand**	**Silt**	**Clay**	**Bulk Density**	**Salinity EC_{sw}**	θ_s	θ_{fc}	θ_r	α	**n**	K_s
	from	to	%	%	%	$g.cm^{-3}$	$dS.m^{-1}$	$cm^3.cm^{-3}$	$cm^3.cm^{-3}$	$cm^3.cm^{-3}$	cm^{-1}	[-]	$cm.day^{-1}$
Panhandle	0	18	35	34	31	1.4	0.1	0.43	0.35	0.08	0.01	1.45	9.99
	18	132	30	32	38	1.45	0.1	0.44	0.35	0.09	0.01	1.38	8.01
	132	203	30	32	38	1.45	0.1	0.44	0.35	0.09	0.01	1.38	8.01
Central	0	18	12	68	20	1.41	0.0	0.43	0.29	0.07	0.01	1.65	16.61
	18	53	11	68	21	1.34	0.0	0.45	0.29	0.08	0.01	1.65	21.67
	53	102	7	69	25	1.38	0.0	0.45	0.29	0.08	0.01	1.61	13.87
	102	201	7	69	24	1.39	1.5	0.45	0.29	0.08	0.01	1.61	13.77
SW	0	33	43	38	19	1.42	0.5	0.40	0.25	0.06	0.01	1.51	14.35
	33	107	34	36	30	1.42	0.5	0.43	0.25	0.08	0.01	1.46	9.04
	107	137	34	37	29	1.52	0.5	0.40	0.25	0.07	0.01	1.43	5.54
	137	183	35	37	28	1.52	1	0.40	0.25	0.07	0.01	1.44	5.66
	183	203	42	38	20	1.53	1	0.38	0.25	0.06	0.01	1.46	8.63

Weather variables

For each of the selected locations, daily meteorological data of three Mesonet stations were averaged to better represent climatic conditions of the regions. Although the stations with least missed meteorological data were selected, the missing data had to be filled to have no missing data for the final analysis.

Water fluxes

HYDRUS requires daily values of evaporation and transpiration as input parameters so that it can calculate the actual values of evapotranspiration in each soil, crop type, irrigation quality and quantity. In this study, we used the Bushland ET calculator (BETC) to calculated grass reference evapotranspiration (ET_0, mm) from daily meteorological data such as maxim and minimum air temperature (˚C), solar radiation (MJ $m^{-2}day^{-1}$). Crop evapotranspiration rates (ET_c) were calculated by multiplying ET_0 and K_c, where K_c is the crop coefficient which accounts for both transpiration (T) and Evaporation (E). We used the modified FAO 56 approach to calculate the daily estimates of E and T for each location (Allen, 1998). The values of growth stages were modified to represent Bermudagrass cultivated in Oklahoma. The initial, development, middle, and late stages of Bermudagrass were modified from the initial values of 10,15, 75, and 35 to 13, 19, 94, and 44 days as recommended by Allen (1998). The total length of growth stages was increased from 135 days to 170 days to account for Bermudagrass cultivation dates which are typically from May 3rd to October 19th. The crop coefficient and crop basal coefficients were modified to account for the climatic conditions of each location using the equation below:

$$\mathrm{K}_{c,adj} = K_c + (004U_2 - 2) + (RH_{min} - 45)\left(\frac{h_{max}}{3}\right)^{03} \tag{4.1}$$

Based on the method described by Allen et al. (1998), we used the average of daily adjusted coefficients to adjust the final $K_{c,\ mid}$ and $K_{c,\ end}$ values to account for climatic conditions of each location. The crop evapotranspiration (ET_c) and transpiration (T) values were calculated using the K_c and $K_{c,\ b}$ values and multiplying them by ET_0. The difference between the adjusted K_c and $K_{c,\ b}$ values were used to calculate the total potential evaporation (E) values. The total monthly E, T and precipitation values for each month were calculated and used as HYDRUS input values in terms of cm day^{-1}. For simplification purposes, the monthly values were used instead of daily values for each of the twenty years studied in this research which accounted for 240 input values for all parameters for each scenario and location.

Required irrigation amounts were calculated for days that the crop evapotranspiration was greater than precipitation. Irrigation was added to compensate evapotranspiration assuming the irrigation efficiency was equal to one. Time variable boundary condition was selected and the infiltrating water was calculated by adding up the infiltrating water from precipitation and irrigation.

The salinity of infiltrating water was calculated based on the concentration and volume of irrigation water with designated qualities as mentioned in Table 4.1 and precipitation water with zero salinity. Based on the evapotranspiration requirements of Bermudagrass and monthly precipitation amounts, there were 280, 492, and 643 mm of irrigation in Central, Southwest, and Panhandle Oklahoma respectively.

Other input parameters

Infiltrating water to the topsoil was either rain, irrigation water, or a combination of rain and irrigation which was introduced in terms of infiltrating water (cm month^{-1}) as HYDRUS input parameter. Monthly ET_c values were divided into monthly evaporation (E) and transpiration (T) and were the other input parameters. Simulations were carried out for a period of 20 years from 1998 to 2017, with monthly time steps. The initial water content of all three districts were assumed to be similar and we neglected any potential effect of various irrigation water quality and quantity on root growth and the root distribution was set to be distributed 50% at top 5 cm, 20% from 5 to 10 cm, 10% from 10 to 15 cm, and the remaining 20% distributed linearly at deeper soil layers (Duble, 1996) . The initial soil salinity was uniformly 0.18 dS m^{-1} and initial water content 0.35 cm3 cm^{-3} which were close to the field capacity.

The solutes in the standard HYDRUS model were assumed to be present only in the dissolved phase and thus the distribution coefficient was set to zero (K_d = 0 cm^3 g^{-1}). Feddes model was used to account for water stress (Feddes et al., 1988).

In order to include multiplicative water and osmotic stress, the method proposed by Feddes et al. (1988) was applied for the determination of the actual RWU. The resulting water stress reduction was parameterized using the following Feddes parameters, which were slightly modified from the values of HYDRUS database to adopt for no water stress conditions, as: h1 = 4, h2 = 0, h3 = −1500, h4 = −8000 cm. The reduction of root water uptake due to the salinity stress was evaluated using the S-shaped response function, which is calculated using the equation below developed by (Van Genuchten, 1980):

$$\alpha(h, h_{\varphi}) = \frac{1}{1+(\frac{h+h_{\varphi}}{h_{50}})^{p}} \tag{4.2}$$

Where The exponent, p, in the root water uptake response function associated with salinity stress (-). The recommended value was 3. The coefficient, h_{50}, in the root water uptake response function associated with salinity stress (L). Root water uptake at this osmotic head was reduced by 50%. The parameters in HYDRUS database (Maas and Hoffman, 1977) for Bermuda grass were used as h_{50} of 29.425 and P_3 of 3.

Boundary conditions

The water flow boundary conditions were set to atmospheric boundary condition with surface runoff and free drainage as top and bottom boundary conditions. We assumed that precipitation was free of salts. Atmospheric boundary conditions were specified using meteorological data for three Mesonet stations at each location. The upper and bottom solute transport boundary conditions were set to concentration flux at the top and zero concentration gradient at the bottom.

Results and Discussion

The monthly values of the inflow and outflow amounts of various water flux parameters are displayed in Fig. 4.2. As observed in this figure, the parameters of precipitation (P) and irrigation (I) enter the flow domain and the values of transpiration (T) and evaporation (E) leave the domain. The values of T and E are maximum in the growing stage of Bermudagrass, particularly in the summer. Hence the irrigation application is maximum in these months based on the precipitation received. The precipitation received was decreased from Central to Panhandle.

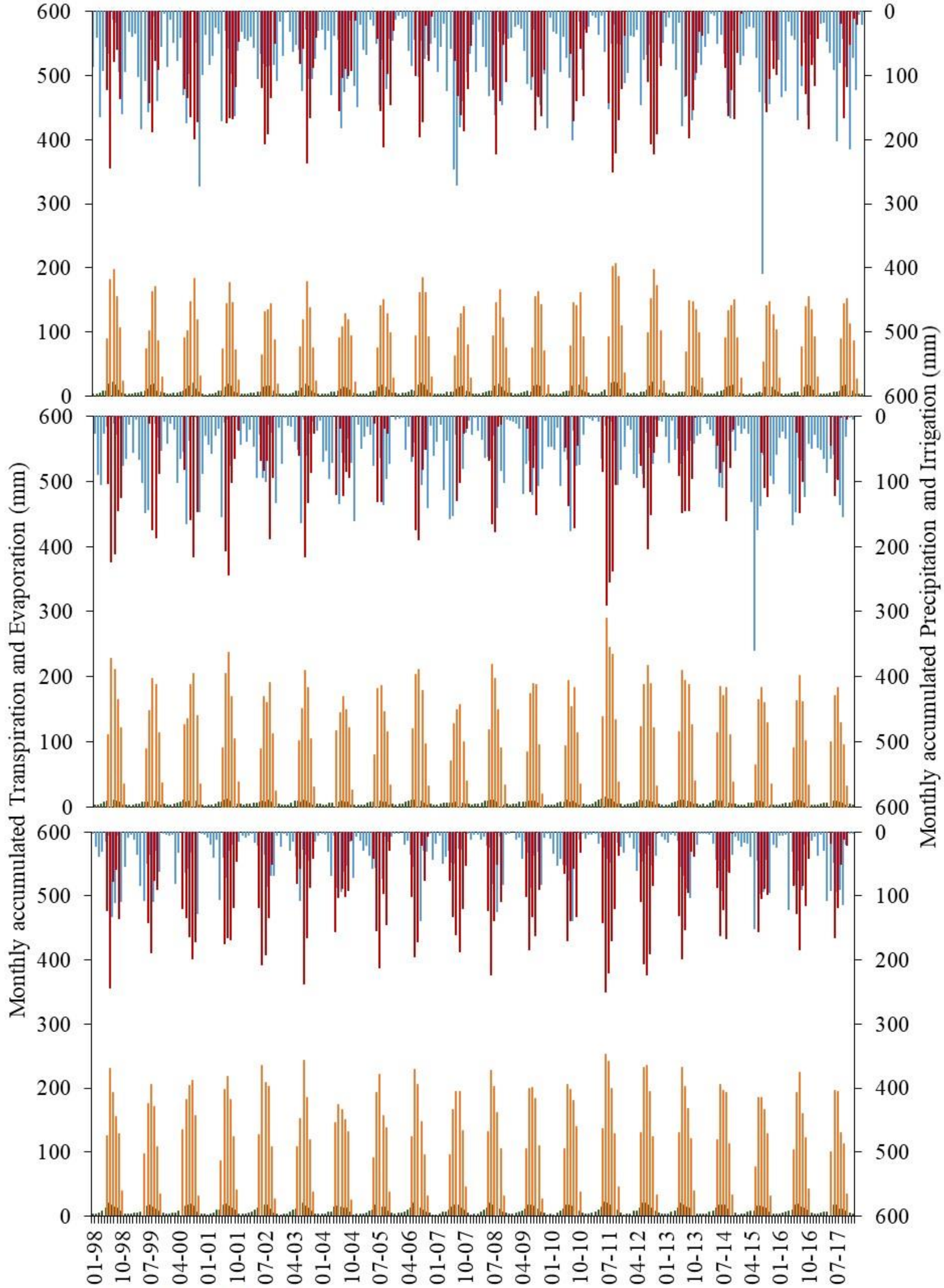

Figure 4.2. Monthly Evaporation (E), transpiration (T), precipiration (P), and irrigation (I) for

bermudagrass at Central (top), Southwest (middle), and Panhandle (bottom) regions during 1998-2017.

Lower amounts of precipitation in P area, leads to increase in irrigation requirement, hence increases salt buildup in this area. As precipitation exceeded the ET needs in most of the winter months, the irrigation requirements in theses months were zero which is an important factor in salt leaching below rootzone (Li., et al., 2015).

Root Water Uptake

Water consumption efficiency can be used as an important parameter to investigate the plant response to salinity and is useful for reflecting the effects of climatic conditions on crop growth. In this regard, the effect of irrigation water quality and quantity on root water uptake (RWU) is studied by evaluating the variations in the ratio of actual to potential RWU as demonstrated in Figure 4.3. Although water salinity of 6 dS m^{-1} (EC1) is still considered saline for irrigation, it is observed that the actual RWU was equal to its potential value when irrigated by irrigation quantities of 0.7, 1, and 1.3 times its evapotranspiration needs, namely under (U), full (F), and over (O) irrigation. The relative RWU, however, dropped significantly for all treatments under irrigation with 340 dS m^{-1} (EC4). This ratio was 0.38, 0.2, and 0.28 for under-irrigated Central (C), Southwest (S), and Panhandle (P) area and was further decreased by an increase in irrigation water quantity. Furthermore, it was apparent that the location of land application of saline water plays an important role in RRWU, as RWU under C-EC3 were greater than those of S-EC2 with less saline irrigation water under same irrigation water quantity application.

The inclusion of the depressing effect of salinity on plant water consumption could help avoid excess water application in saline irrigation. As the irrigation water salinity increases, salts accumulate the soil rootzone leading to salinity stress to the crop. The salinity stress restrains crop root from root uptake, which eventually restricts crop growth and root development. The crop growth and its water consumption are decreased by increase in irrigation water salinity, therefore, it is more probable that fields under saline water irrigation are over-irrigated. Over-irrigation of saline water has many negative outcomes associated with it and adds more salts to the root zone as observed in the Fig. 4.3. Moreover, saline irrigation is generally considered in arid and semi-arid regions which deal with water shortage.

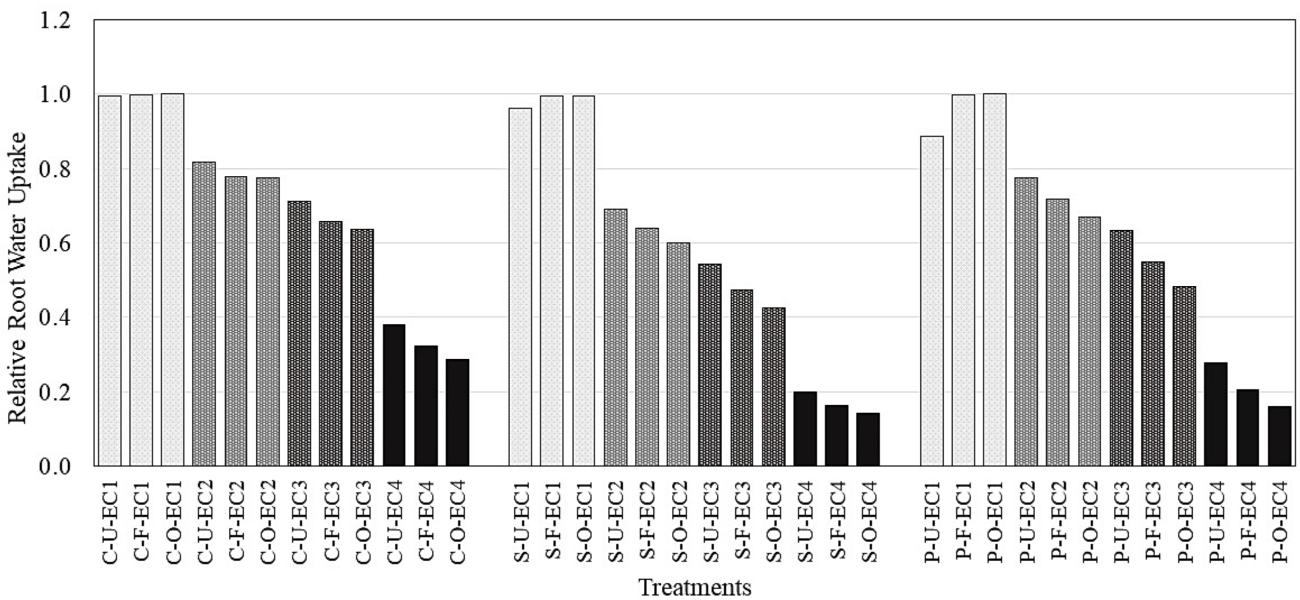

Figure 4.3. The ratio of cumulative actual over potential root water uptake under irrigation water salinities of 6 and 340 ds m^{-1} and irrigation water quantities of under (U), Full (F), and Over (O) irrigation based on crop evapotranspiration requirements.

Comparison of RRWU of treatments under EC1 reveals that under-irrigation did not adversely affect RRWU in location C, and decreased RWU in locations P and S by 4 and 11%, respectively. This aligns well with some of the previous works that claim over-irrigation decreased salt accumulation as a result of leaching of salts below the root zone (Haj-Amor et al., 2016; Zeng et al., 2014; Wang et al., 2017). In more saline irrigation scenarios of EC2, EC3, and EC4, however, increasing irrigation water quantity from U to O decreased RWU in all locations. This is in contrast with the previous claim and can be justified by the fact that increasing irrigation water quantity of highly saline waters introduces more salts to the root zone and restrictions roots to uptake water and thus produces drought-like conditions despite the availability of water (Haj-Amor et al., 2016). Therefore, the effect of irrigation water quantity on soil salt accumulation is a factor of many parameters such as salinity of the irrigation water, frequency of irrigation, climatic conditions, the depth of water table, the availability and cost of irrigation, and most importantly the quality of leachate (Haj-Amor et al., 2016; Zeng et al., 2014). Unlukara et al. (2015) reported that water consumption of the plant decreased exponentially, and fruit yield decreased linearly by an increase in unit soil salinity after the threshold salinity of the plant was reached.

The location of land application of saline water plays an important role in RRWU as observed in Fig. 2. RWU was maximum in all treatments in location C and minimum for location S. In this regard, RWU under C-EC3 were greater than those of S-EC2 with less saline irrigation water under same irrigation water quantity application. Therefore, it can be concluded that the climatic conditions play an important role in terms of crop water loss to evaporation and precipitation amount and its pattern as it is able to leach salts and minimize the damaging impact of saline irrigation on crop roots (Armstrong et al., 1996; Haj-Amor et al., 2016; Zeng et al., 2014).

Water Balances

The effect of irrigation water quantity is studied by comparison of application of various scenarios of under, full, and over irrigation. Generally, there was an increasing trend in the cumulative value of the actual surface flux (vTop) and the cumulative value of the bottom boundary flux (vBot) by an increase in irrigation water quantity in irrigation under EC1. Increasing irrigation water quantity did not increase the cumulative value of the actual transpiration (vRoot). The values of vTop and vBot for all treatments were negative which means that the top flux is infiltrating, and the bottom flux is the outflow to the domain boundary. For a better comparison between scenarios, the least irrigation water salinity namely EC1 (left) is compared to the maximum water salinity EC4 (right) as displayed in Figure 4.4. Based on the results, increasing irrigation water salinity decreased transpiration values by 67, 79, and 83% for full irrigation at C, P, and S, respectively.

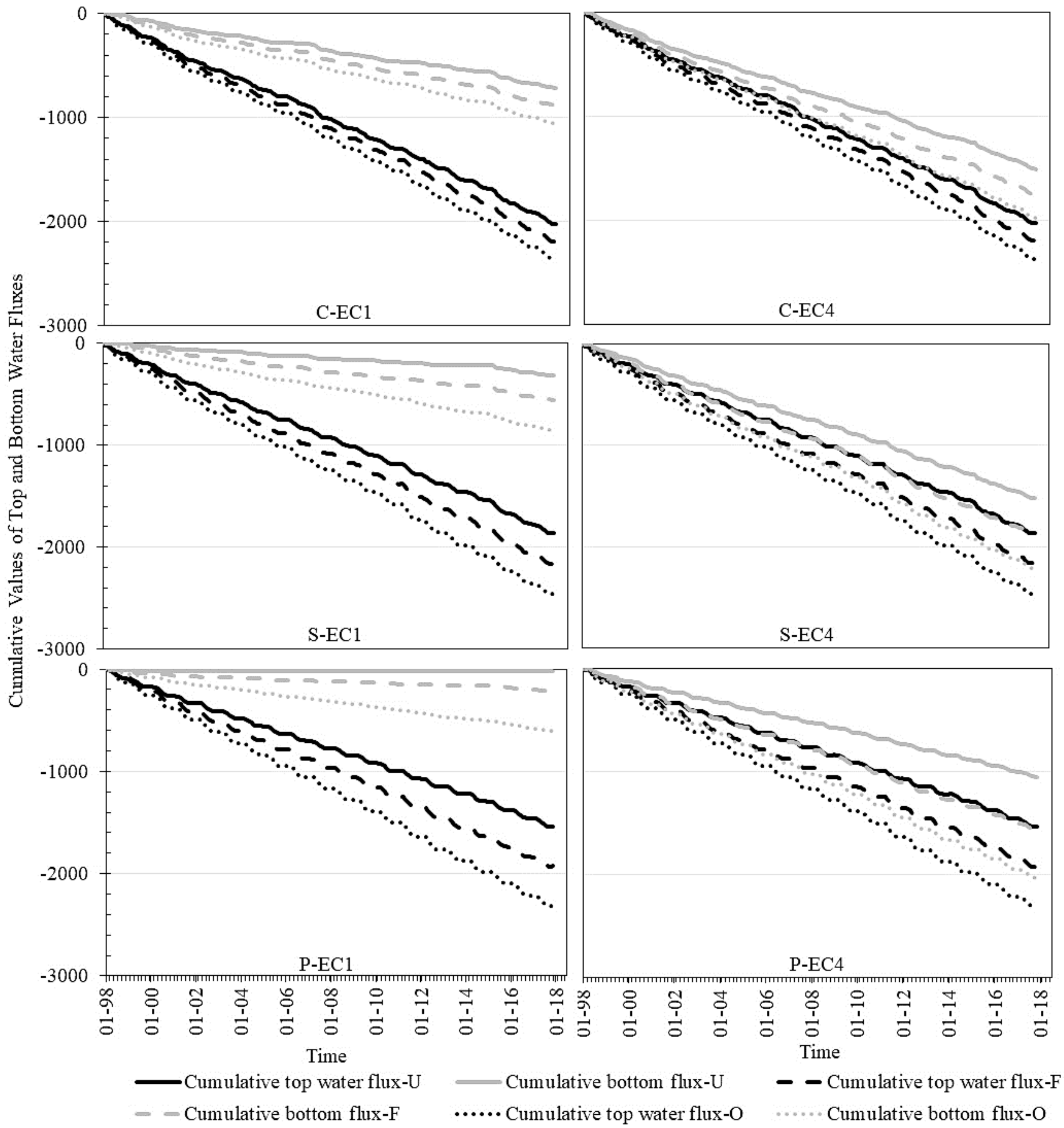

Figure 4.4. Comparison of the cumulative value of the actual surface flux (vTop), the cumulative value of the bottom boundary flux (vBot) of various locations of Southwest, central, and panhandle irrigated with 6 and 340 dS m^{-1} (EC6 and EC340).

This impressive reduction in transpiration rates as a result of increase in irrigation water salinity can be justified by crop salinity stress that limits the availability of water for root uptake and makes drought-like conditions for roots despite the availability of adequate water levels. The comparison between EC1 and EC4 also revealed that as a result of the increase in irrigation water salinity, the cumulative value of the bottom boundary flux increased significantly. This implies that increasing the salinity of irrigation water, both confines the crop root from transpiring the applied water due to salinity stress, and it introduces a significant amount of water to the bottom boundary flux, which can potentially contaminate the downstream water resources (Hagstrom et al., 2016; O'Connor et al., 2008). Similarly, Wang et al. (2017) also reported a decreased transpiration as a result of salinity stress. However, they claimed that water use efficiency was a factor of precipitation and suggested that over-irrigation during low-rainfall growing seasons could effectively maintain water use efficiency. Once the amount of fluxes inflow and outflow to the flux domain is evaluated, it is important to estimate the amount of salts that are leaving the flux domain.

In order to explain the increased bottom fluxes as a result of more saline irrigation application in EC340 as compared to EC6, the relative water consumption was studied in terms of percentage of irrigation water consumed by crop in transpiration process. In full irrigation of location C, 23% of the irrigation water was consumed in transpiration, this ratio however, decreased to 18%, 15%, and 7% for EC44, EC70, and EC340, respectfully. This ratio decreased, by increase in soil salinity from EC6 to EC340, from 16% to 3% in S area, and from 13% to 3% for P area. The percentage of irrigation water consumption through transpiration further decreased in over-irrigation and was only 2%

for S-O-EC340 and P-O-EC340 and 5% for S-U-EC340 and P-U-EC340. It was decreased from 13% to 5% for C-EC340 as irrigation water quantity was increased from U to O. This means a higher percentage of irrigation water was consumed in under-irrigated transpiration. This results in higher over-irrigation in saline waster irrigation, which has the adverse effects of elevated salt accumulation in the soil and declined crop productivity. Moreover, over-irrigation results in the loss of water in arid and semiarid areas and the saline bottom flux may contaminate the downstream waterbodies. The adverse effect of saline water over-irrigation is even more severe in areas with limited precipitation. As observed in Fig. 4.4, the bottom flux is greater in P area as compared to C area. This is due to greater precipitation in C area, as precipitation is able to balance out the severe effects of saline irrigation and leach salts below the root zone. These results are in favor of reuse of PW in central Oklahoma, as displayed in Fig. 4.1, the PW availability is greater in C area, which has greater precipitation. These parameters make C area more suitable for PW discharge, especially that limited transportation requirements are in favor of this selection. However, regarding the population density of C area, further studies regarding the long-term effects of PW land application close to populated areas needs to be conducted.

Long-term soil salinity trends

To assess the long-term salinity trends in various locations of Oklahoma irrigated under various irrigation water quality and quantities, the amount of monthly accumulation of solute in the entire flow domain is represented in Fig. 4.5, Fig. 4.6, and Fig. 4.7. Accumulation of solutes were maximum in Panhandle area followed by Southwest and Central Oklahoma. This trend can be justified by the amount of precipitation received by these areas (Wang et al., 2017). In all locations, salinity increased by an increase in irrigation water salinity and the peak salinity was generally observed in June, July and August months which received the least amounts of precipitation and thus required more irrigation. For a better comparison, the ratio of the solute amount in the flow domain in respect to its initial value before January 1998 is represented in Fig. 4.5, Fig. 4.6, and Fig. 4.7.

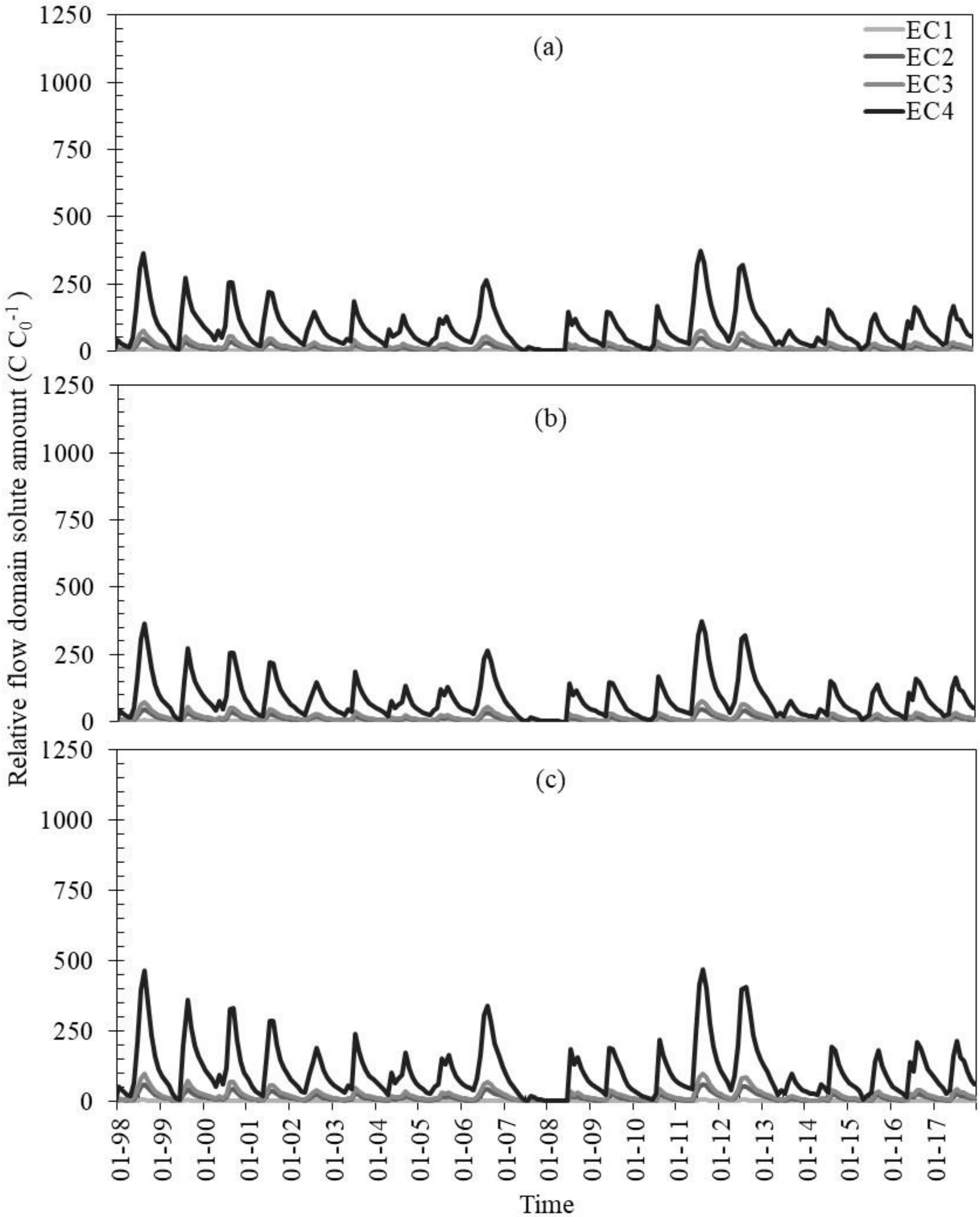

Figure 4.5. The ratio of the flow domain solute amount to its initial value in treatments with irrigation water quality of U (a), F (b), and O (c) in Central Oklahoma.

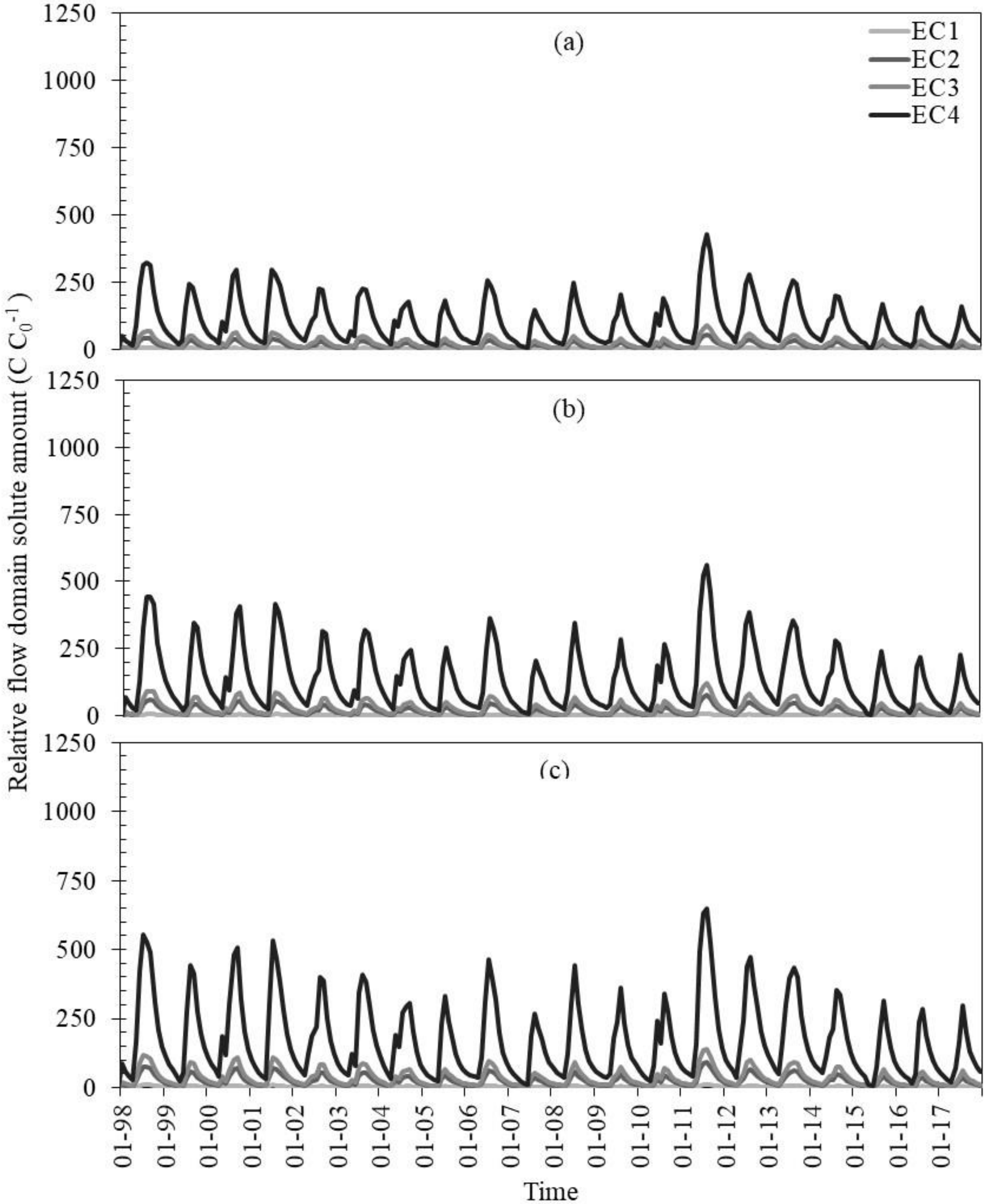

Figure 4.6. The ratio of the flow domain solute amount to its initial value in treatments with irrigation water quality of U (a), F (b), and O (c) in Southwest Oklahoma.

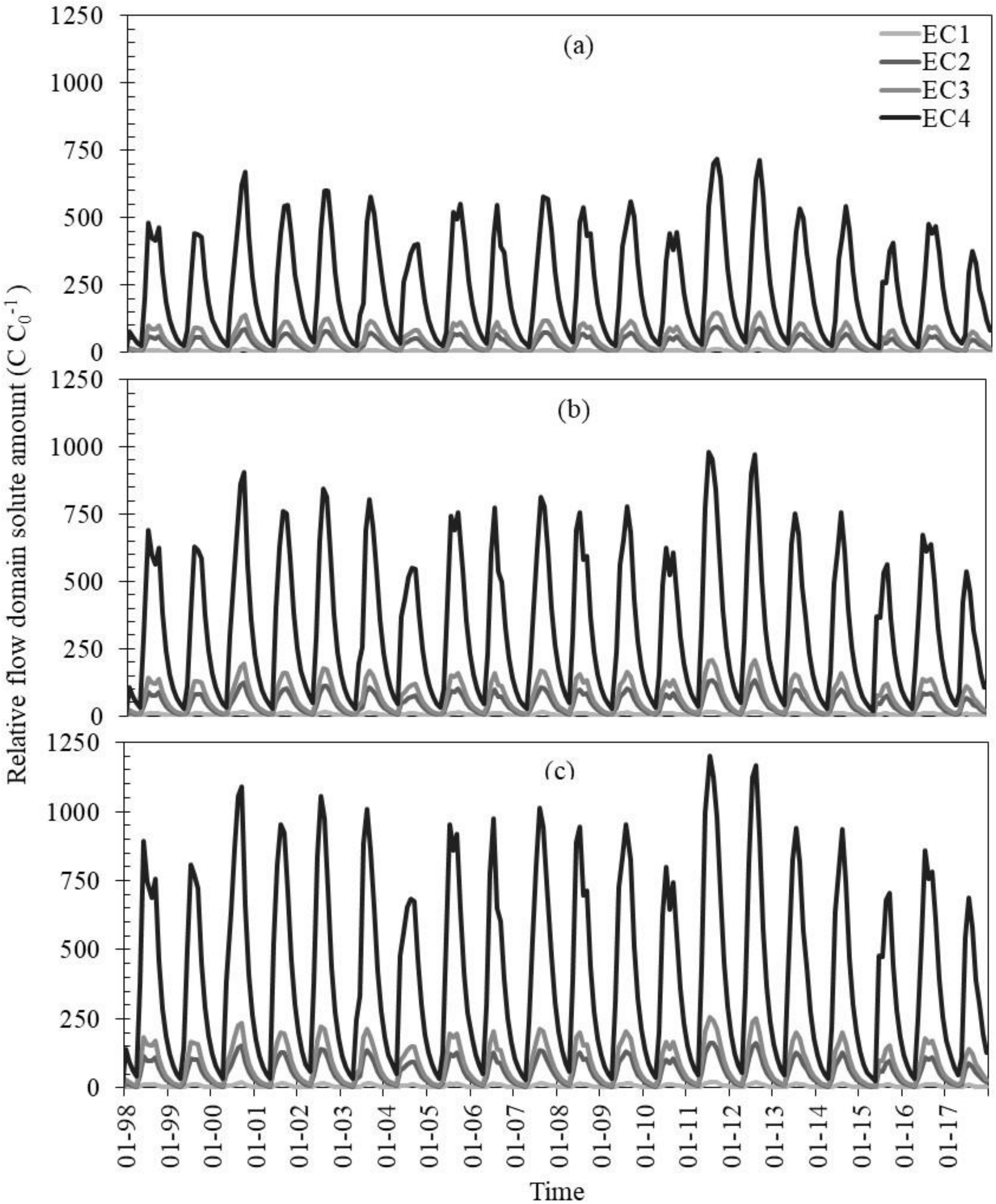

Figure 4.7. The ratio of the flow domain solute amount to its initial value in treatments with irrigation water quality of U (a), F(b), and O (c) in Panhandle Oklahoma.

Increasing irrigation water application amounts would generally leach salts out of the flow domain, however, application of excess saline water increased the soil solution salinity and added more salts to the soil. In Panhandle area, with irrigation water salinity of EC4, over-irrigation increased maximum salinity amounts to 1201 times its initial value. This peak in salinity occurred during a period of high irrigating requirement of 250 and 220 cm in June and July of 2011 in this area. On the other hand, the minimum solute amount peak for EC4 was observed in June 2007 in the under-irrigated Central area. The solute amount ratio to its initial value was only 338 times. This was due to the high precipitation amounts in May, June, and July of that year which were 245, 271, and 180 cm, respectively. Although the maximum precipitation in Central area occurred in May 2015 with the amount of 408 cm, lower precipitations in the proceeding months of August to October prevented this month from being associated with the smallest solute amount peaks. This could indicate that consecutive application events of good-quality water are more effective in reducing soil salinity accumulation as compared to the one-time application of a great amount of good quality water followed by events of low-quality irrigation water (Haj-Amor et al., 2016; Wang et al., 2017).

Based on the results of relative flow domain solute amounts, solute accumulations are minimum in location C and decreases to location S, and is minimum at location P (Fig. 4.5, Fig. 4.6, and Fig. 4.7). This trend aligns well with the precipitation amounts at these locations which were 900, 655, and 421 for C, S, and P, respectively. Elevated precipitation at P area was able to not only leach salts below the root zone, but also increase transpirations of root by minimizing the adverse effects of salinity stress on crop. In all locations, increase in irrigation amounts lead to an increase in solute amounts which is contrary to the trend observed by Wang et al. (2017). As they observed decreased salinity by increase irrigation amounts when irrigated with water with salinity of 5 dS m^{-1}.

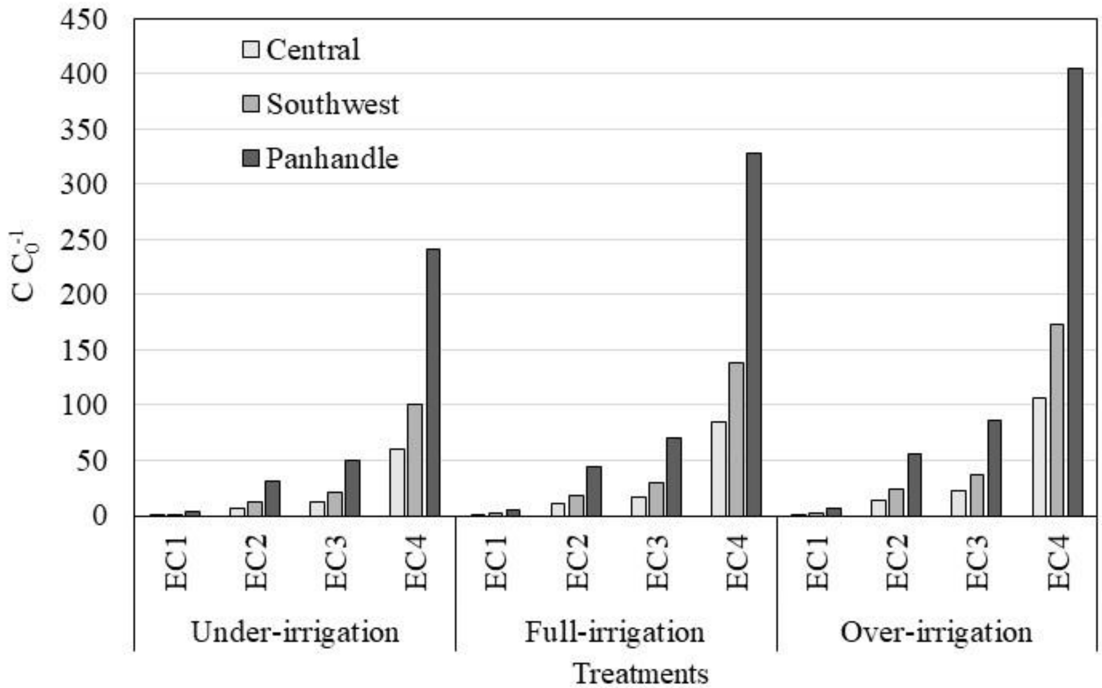

Figure 4.8. The 20-year average (1998-2017) ratio of solute amounts to their initial values ($C.C_0^{-1}$) for treatments irrigated with 6, 44, 70, and 340 dSm^{-1} for Panhandle, Southwest, and central Oklahoma.

The relative solute amount, which is the 20-year average flow domain solute amount changes in respect to its initial values ($C\ C_0^{-1}$), increases by an increase in irrigation water salinity as presented in Fig. 4.8. The maximum values of solute ratio were observed at irrigation water salinity of EC340 in Panhandle with values of 404, 329, and 241 for irrigation quantity of O, F, and U. The ratio of solute amounts to their initial values decreased to 173, 139, and 101 for southwest and 107, 84, and 60 times its initial value for central Oklahoma with irrigation quantities of U, F, and O in EC4 treatments.

Simulation limitations

Some parameters limit the span of simulations projection of water content and solute distribution in the soil profile. It is not easily possible to identify these parameters; however, it is important to include the effects of such a parameter in the simulation analysis. Some of the major parameters affecting the water and salt distribution in soil are preferential flow caused by cracks and macropores (Garg et al., 2009), spatial heterogeneity (Vazifedoust et al., 2008), and precipitation and dissolution reactions in various layers of the soil (Li et al., 2015).

The RWU parameters used in HYDRUS model require to be adjusted to match the parameters of the local conditions. The sensitivity analysis of HYDRUS model for RWU under various irrigation water salinity should be conducted to verify the ranges of salinity that HYDRUS maintains acceptable prediction accuracy.

Conclusions

In order to understand the impacts of saline irrigation on soil salinity dynamics in various parts of Oklahoma under Bermudagrass cultivations, 36 various simulation scenarios were conducted. The HYDRUS-1 D numerical model was used to simulate the soil water and salt dynamics in a 20-year study from 1998 to 2017 using the local soil and meteorological data.

Produced water is a major byproduct of fracking process and the water produced in Oklahoma accounts for 11% of that of US. Regarding the potential hazard of reinjection of produced water and the crucial need for irrigation water resources, the potential land application of produced water in various locations of Oklahoma under Bermudagrass cultivation is suggested in this study. Various qualities of irrigation water with 6, 44, 70, and 340 dS m^{-1} were used under various water quantities of 0.7, 1, and 1.3 times crop evapotranspiration needs to represent under, full, and over-irrigation. As predicted, the average ratio of solute amounts compared to their initial values increased by the increase in irrigation water salinity. This increase, however, was maximum for Panhandle and minimum in Central areas. Similarly, the comparison of the ration of actual root water uptake (RWU) to its potential value revealed that there was no reduction in RWU in treatments under 6 dS m^{-1} irrigation. These values however dropped as irrigation water salinity and its quantity increased. The minimum actual to potential RWU ratio was observed at the over-irrigated treatments under 340 dS m^{-1} for all locations with the values of 0.29, 0.16, and 0.14 in Central, Panhandle, and Southwest respectively. Comparison of precipitation pattern and that of the accumulated solute flux at the soil surface revealed that although precipitation can leach salts below the rootzone and lessen the severity of saline irrigation, a consequent pattern of normal precipitation is more effective than one-time extreme precipitation event in reducing the salt load in the flow domain.

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CHAPTER V

CONCLUSIONS

Major Findings

Three different research projects were conducted to combine monitoring and modeling techniques to assess the soil profile salinity in Oklahoma, its response to elevated dry and wet conditions, and potential land application of produced water (PW) from oil and gas companies in various locations in Oklahoma.

In the first study, the top soil across an irrigation district in southwest Oklahoma was sampled before and after a period spanning eight growing seasons from 2011 to 2015. This period was characterized by four years of exceptional drought when irrigation deliveries were terminated due to water scarcity, followed by a period of record precipitation. Except for pH and Calcium (Ca), the district-wide mean of studied parameters did not experience a statistically significant change. Overall, four years of extreme drought with no irrigation application succeeded by a period of intensive rainfall reduced soil salinity in the surface layer but moved salts downward to the middle section of soil profiles. This reduction in surface soil salinity is beneficial for seedling establishment. However, levels of pH, EC, and ESP appear to be high enough to cause yield loss, especially at some of the sampling locations.

In the second study a combined monitoring and modeling approach was undertaken to evaluate the effect of irrigation water salinity on soil profile salinity, effect of fresh water leaching application on receiving soils of saline irrigation, and testing performance of HYDRUS 1-D model in predicting the soil and solute interactions. The results of the greenhouse study were used to validate a simulation model. The results of greenhouse study revealed that soil solution salinity increased by increase in irrigation water salinity particularly at top soil layer, furthermore, fresh water leaching application significantly decreased soil profile salinity. Statistical analysis showed that there was a good agreement between model predicted and measured values of soil solution electrical conductivity (EC_{sw}).

The validated model was later used in the third research study to investigate the effect of land application of the greatest volume of industrial wastewater in Oklahoma, which is produced water (PW), in various locations of Oklahoma. In this study 36 simulation scenarios were defined to study the effects of using saline water qualities of 6, 44, 70, and 340 in various volumes of under-, full-, and over-irrigated in three different locations of Panhandle, Southwest, and Central Oklahoma. Based on the results of this study, similar to the second study, soil solution salinity increased by increase in irrigation water salinity. Root water uptake was severely decreased in hyper-saline irrigation water; thus, greater amounts of saline water were released from the bottom of the flow domain (2 meter), potentially contaminating the downstream water resources (Rameshwaran et al., 2016). Precipitation had a positive impact in leaching salts and hence neutralize the harmful effects of saline irrigation in salinities close to crop salinity tolerance threshold. As confirmed by the results of this study, previous researches also reported that the annual hydrological characteristics of the study period is the most significant determining factor in evaluating the effects of saline water irrigation on rootzone soil salinization (He et al., 2017; Li et al., 2015).

Future Research Recommendations

Optimization of saline irrigation parameters and various irrigation scheduling in response to precipitation patterns requires in depth study of solute and soil moisture interactions. Effects of saline irrigation on soil characteristics particularly its permeability should be taken into accounts as it is an important parameter in successful leaching. Not only the performance of simulated models decrease in hyper-saline irrigation studies, there are numerous environmental concerns regarding their utilization. A comparison between various management practices can be simulated with irrigation was salinities close to salinity tolerance threshold of the crop in order to maintain the sustainability of saline irrigation. The most efficient modeled management practices can be tested in greenhouse and larger scales. It is crucial to study long-term sustainability of practices in regards to marginal-quality irrigation application before implementing them in real-life and large scales.

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APPENDICES

HYDRUS INPUT PARAMETERS

The input parameters for the HYDRUS model are described in the third and fourth chapters. Table A.1. and A.2. summarize the major input parameters that were used in this study. A more detailed soil hydraulic parameters are represented in Table 4.3.

Table A.1.

HYDRUS input parameters for project 2.

Input Category	Input section	Value	Unit
Geometry Information	Number of soil materials	1	
	Number of mass balance layers	8	
	Depth of soil profile	40	cm
Time information	Time duration	50	days
Soil hydraulic models	van Genuchten-Mualem		
	No hysteresis		
	Residual soil water content	0.025	$cm^3\ cm^{-3}$
	Saturated soil water content	0.356	$cm^3\ cm^{-3}$
	Parameter *n* in the soil water retention function	0.2	
	Saturated hydraulic conductivity	38.311	$cm\ day^{-1}$
	Tortuosity parameter in the conductivity function	0.5	
Water flow boundary condition	Atmospheric upper-BC with surface layer		
	Seepage face lower-BC		
Solute transport parameters	Bulk density	1.5	$gr\ cm^{-3}$
	Dispersivity	4	
	Molecular diffusion coefficient in free water	2	
	Adsorption isotherm coefficient	0	
Solute transport BC	Concentration flux Upper-BC		
	Zero concentration gradient lower-BC		
Root water uptake model	Feddes water uptake reduction model		
Root water uptake parameters	h1	-10	cm
	h2	-20	cm
	h3	-1500	cm
	h4	-8000	cm
Root distribution	Linearly distributed depth	10	cm

Table A.2.

HYDRUS input parameters for project 3.

Input Category	Input section	Value	Unit
Geometry Information	Number of soil materials	4	
	Number of mass balance layers	1	
	Depth of soil profile	200	cm
Time information	Time duration	240	days
Soil hydraulic models	van Genuchten-Mualem		
	No hysteresis		
	Saturated hydraulic conductivity	300	cm day^{-1}
	Tortuosity parameter in the conductivity function	0.79	
Water flow boundary condition	Atmospheric upper-BC with run off		
	Free drainage lower-BC		
Solute transport parameters	Dispersivity	10	
	Molecular diffusion coefficient in free water	2	
	Adsorption isotherm coefficient	0	
Solute transport BC	Concentration flux Upper-BC		
	Zero concentration gradient lower-BC		
Root water uptake model	Feddes water uptake reduction model		
Root water uptake parameters	h1	4	cm
	h2	0	cm
	h3	-1500	cm
	h4	-8000	cm
Solute stress model	Multiplicative Model S-Shape		
	The coefficient, *h50*, in the root water uptake response function	29.425	cm
	root water uptake response function associated with salinity stress	3	(-)
Root distribution	80% top 15 cm		
	20% deeper layers		

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