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Evaluating abiotic influences on soil salinity of inland managed wetlands and agricultural fields in a semi-arid environment

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EVALUATING ABIOTIC INFLUENCES ON SOIL SALINITY OF INLAND MANAGED WETLANDS AND AGRICULTURAL FIELDS IN A SEMI-ARID ENVIRONMENT

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The Lord says: Let not the wise man boast in his wisdom, let not the mighty man boast in his might, let not the rich man boast in his riches, but let him who boasts boast in this, that he understands

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ABSTRACT

Agriculture and moist-soil management are important management techniques used on wildlife refuges to provide adequate energy for migrant and wintering waterbirds. However, in arid systems, the presence and accumulation of soluble salts throughout the soil profile can limit total biomass production of wetland plants and agronomic crops and thus jeopardize meeting waterbird energy needs. It is unknown how moist-soil management and traditional agriculture practices influence the accumulation and distribution of soluble salts of soil profiles. In this study of an arid wetland ecosystem, I determine: 1) the effect of long-term, distinct surface hydrologic regimes associated with moist-soil management and agricultural production on salt accumulation; and 2) the specific effects of rototillage and irrigation frequency on salinity concentrations and plant biomass in moist-soil impoundments. My study was conducted at Bosque del Apache National Wildlife Refuge near San Antonio, New Mexico. In May 2012, prior to the growing season, I collected one meter deep soil cores from both moist-soil impoundments and agricultural fields; cores were analyzed in 10 cm segments for soluble salt concentrations. I implemented a split-plot experiment to evaluate salinity concentrations in moist-soil impoundments between rototilled and no-till soils under a 9 and 14 day irrigation frequency. Soil salinity was measured in May and August of 2011 and 2012 and plant biomass in August. My findings suggest that agricultural fields contain significantly higher concentrations of soluble salts in deeper portions of the profile. This may be attributed to the lack of leaching afforded by summer agricultural irrigations as little connectivity to the groundwater and groundwater salinity was detected during groundwater monitoring. In contrast, periodic flooding in winter and summer flood irrigations in moist-soil impoundments may serve as leaching events and created a more dynamic groundwater hydrograph. This seasonal wetland hydroperiod may facilitate lower soil profile salinities but further research is needed to evaluate its successful use in agriculture fields to lower soil salinities. Few differences in soil salinity were detected between tillage and irrigation treatments within moist-soil

impoundments. However plant above ground biomass of annual wetland grasses was greater in rototilled soils. This is most likely attributed to the effects of physical disturbance that stimulates germination rather than differences in soil salinity, however greater aboveground biomass does not necessarily equate to higher seed or tuber production.

CHAPTER 1
INTRODUCTION

Salt-affected soils can be defined as soil where soluble salts adversely affect the growth of most crop plants, with or without high amounts of exchangeable sodium (SSSA 2008). Over 932 million hectares of the world's soils are classified as salt affected soils (FAO 2000). Many valuable agricultural crops and seasonal wetland plants favored for wildlife are sensitive to elevated concentrations of salts in the soil. High concentrations of soluble salts can have adverse effects on plant productivity by lowering osmotic water potential in the soil so that water is inhibited from being absorbed by plant roots (Mass and Hoffman 1977). This inability to access soil water, for plants ill-adapted to salinity, results in drought like symptoms leading to plant stunting and reductions in overall biomass. Although the accumulation of soluble salts is a natural phenomenon in arid and semi-arid environments, anthropogenic modifications to hydrologic events can exacerbate the salinization of soil. For example, the advent of irrigated agriculture in systems of high evapotranspiration and minimal precipitation has degraded land productivity resulting from the use of marginal quality irrigation water that encourages the accumulation of soluble salts in the soil profile (Hillel 2000). This type of human induced soil degradation has contributed to an additional 76.3 million hectares worldwide of non-natural salt affected soils (Oldeman et al. 1991). As a result, soil salinization is one of the major causes of declining agricultural productivity in arid and semi-arid regions of the world (Qadir et al. 2000). However, the products and value of both agricultural lands and seasonal wetlands are an ever-increasing demand and production is likely to continue despite the potential to jeopardize land quality.

The processes, movements, and concentrations of soil salinity throughout a soil profile are dynamic and efforts to manage saline soils for sustainable use require an understanding of the abiotic variables that influences them.

1.1 Origin of Salts

The presence and concentrations of salts at both the soil surface and in the soil mantle are controlled by geologic, geomorphic, climatic, and hydrologic factors (Metternicht and Zinck 2009). Salts enter the soil system as a result of the weathering of primary minerals contained in crystalline and sedimentary rock. Upon weathering, the primary minerals release cations and anions representative of the earth's crust elemental concentration (Table 1).

Table 1 - Percentages of common elements in the earth's crust. Derived from Chhabra, 1996.

Common element content (%) of the earth's crust			
Element	% Content	Element	% Content
Oxygen (O)	49.13	Hydrogen (H)	1.00
Silicon (Si)	26.00	Titanium (Ti)	0.61
Aluminum (Al)	7.45	Carbon (C)	0.35
Iron (Fe)	4.20	Chlorine (Cl)	0.20
Calcium (Ca)	3.25	Phosphorus (P)	0.12
Sodium (Na)	2.40	Sulphur (S)	0.10
Magnesium (Mg)	2.35	Manganese (Mn)	0.10
Potassium (K)	2.35		

The geochemical mobility of these elements, due to their individual weathering processes (hydrolysis, hydration, oxidation, and carbonation), are not all equal and the most common salts in soil systems are those that have high leaching capabilities (Chhabra 1996). These include compounds of Sodium (Na^+), Calcium (Ca^{2+}), Magnesium (Mg^{2+}), Potassium (k^+), Ammonium (NH_4^+), Chloride (Cl^-), Sulfate (SO_4^{2-}), Nitrate (NO_3^-), and Carbonate (CO_3^-).

Parameters have been established to classify salt-affected soils and discriminate between the effects of certain cations and anions. Traditionally soil properties such as pH, electrical conductivity (EC), and exchangeable sodium percentage (ESP) have been used to create these boundaries (Richards 1954). Saline soils are those soils whose electrical conductivity is > 4 decisiemens (dS m^{-1}). Alkaline soils, or sodic soils, are distinguished by an ESP > 15% and Saline-Sodic soils are soils that exhibit both properties (Table 2). However, it is important to note that these crisp classifications are only generalizations and some species of plants can experience adverse effects of soil salinity in non-saline defined soils. (Mass and Hoffman 1977).

Table 2: Chemical and Physical Parameters of Salt-Affected Soils. Derived from Richards (1954).

Chemical and Physical Parameters of Salt-Affected Soils			
Soil Types	Chemical Indicators	Soil Properties	Effects on Soil Properties
Saline soils	EC > 4 dS/m ESP<15% pH < 8.5	Clay particles generally remain flocculated; Presence of white crust on the soil surface	Higher osmotic pressure;
Alkaline (sodic)soils	EC < 4 dS/m ESP >15% pH > 8.5	Organic matter dispersion and dissolution; Clay deflocculation; Columnar or prismatic structure	Changes in structure; Decrease in permeability and porosity;
Saline-alkaline (sodic) soils	EC > 4 dS/m ESP > 15% pH: variable	Combination of both soil types as mentioned above	

While the effects of saline soils are largely the restriction of water availability to plants due to the high osmotic potential in soils, alkaline/sodic soils affect the integrity of the soil structure. In

alkaline soils, monovalent sodium cations displace divalent calcium cations which encourage the deflocculation of clay particles. This dispersion of clays reduces soils structure, decreasing the porosity of the soil and water permeability (Abu-Sharar et al. 1987).

1.2 Soil Salinization

All soils contain salts, but the extent and redistribution from initial points of weathering is a result of the solubility of salts and the climatic, topographic, hydrologic, and land management practices that the soil is subjected to. In addition to salts originating from weathering parent materials, salts can be deposited to non-native landforms through a series of dispersion phenomena. In its simplest form this is the natural process of soil salinization whereby water soluble salts are disseminated from a salt source to an area originally free of salt. Volcanic eruptions can deposit ash containing salt bearing minerals over large areas (Chaun 1994). Low-lying coastal interfaces receive salt deposition from ocean high tides and hurricanes that push sea water into marshes that would otherwise be non-saline (Gardner et al. 1992). Coastal winds may also carry salts from sea spray further inland. However, detrimental salinization occurring in humid regions is more likely to be temporary due to the available precipitation that leaches salts down the profile or into the groundwater, restoring previous conditions. In contrast, semi-arid environments are more likely to accumulate and retain salts as a result from the limited precipitation. Here, low lying depressional landforms can either accumulate large quantities of salts from runoff waters or receive quantities of salts from upwelling saline groundwater. In most semi-arid cases evapotranspiration exceeds precipitation and evaporated water can leave behind salt precipitates at the surface of depressional basins (Domingo et al. 2001). In turn these precipitates can be carried away by winds and return to the global soluble salt cycle.

Human induced salinization, or secondary salinization, is the processes by which the salt concentration in salt-affected soils is increased, or salt free soils are contaminated by inadequate water

and land management (Hillel 2000). Secondary salinization can come about through a variety of ways but all are related to environmental variables, water supply and infrastructure, and land management procedures.

1.3 Environmental Variables to Secondary Salinization

In rigorously irrigated landscapes that have poor natural drainage or no artificial way to remove excess water, groundwater levels can rise. An increase in the groundwater table can serve as an adverse risk that brings saline water to the surface by means of capillary action (Northey 2006). Capillary action is the upward movement of groundwater through the soil caused by surface tension of water in the soil pores. Capillary rise by no means occurs at a steady rate and can vary by the actual depth to groundwater and the texture and structure of the soil. Generally, soils with groundwater tables shallower than 1.5-2 m are susceptible to capillary rise and salinization in semi-arid environments (Yang et al. 2011). Soils that have good porosity from the groundwater table to the soil surface provide a readily accessible transport pathway for water and soluble salts to assimilate upward. However, the soil medium in which capillary rise occurs strongly affects the extent of upwelling groundwater. Clayey soils will exhibit more capillarity due to their increased surface area which allows for more adhesion of water to the soil pores. In turn if a soil became sodic, deflocculated clays could plug up macropores but increase micropore surface area, exacerbating capillarity. In contrast soil composed primarily of sands of large texture will have reduced capillarity properties.

1.4 Effects of Water Supply on Secondary Salinization

Marginal quality water used for irrigation can also contribute to the process of soil salinization (Costa et al. 1991). Both groundwater and surface waters are commonly used for flood irrigation. In both cases, applied water with concentrations of solutes contributes to overall accumulation in the soil.

Even if concentrations of solutes are low, the long term use of this water can quickly accumulate salts to detrimental levels if soil leaching is insufficient.

The fraction of water needed to move down the soil profile to effectively remove salts from the rhizosphere is described as the *leaching requirement* (Letey et al. 2011; Ayers and Wescott 1985); the leaching requirement does not include water absorbed by plants or that is evaporated. However the successful use of this parameter is difficult to manage because excessive leaching can contribute to the rise of the groundwater table (Jolly et al. 2008). Depending on the salinity of the irrigation water and the quantity of salts needed to be moved from the profile, groundwater concentrations can increase as a result of leaching. In the case of irrigation water that is significantly dilute in concentrations compared to that of the groundwater, leaching events can serve to temporarily decrease groundwater salinities (Jolly et al. 2008).

1.5 Effects of Land Management Practices on Secondary Salinization

In many semi-arid environments, flood irrigation is a preferred water delivery method for crops and managed wetlands because it is generally cheaper and requires less maintenance than center-pivot groundwater wells. However the combination of flood irrigation with certain land management practices can exacerbate the extent of soil salinization. In agricultural settings, using flood irrigation on furrowed, row crop fields can serve to intensify the accumulation of salts on the row ridges (Figure 1). Applied water can mobilize both salts currently in the soil and in solution of the water and salinize the tops of rows through the same mechanism of capillary rise as it does in groundwater (Bernstein et al. 1955; FAO 1988). Many agricultural crops are most affected by salinity in their germination and seedling stages (Mass and Hoffman 1977). The salinization of rows during this time period can have adverse effects on the overall biomass of the crop.

Figure 1- Salt precipitates accumulating on the tops of rows before (left) and after (right) germination at Bosque del Apache National Wildlife Refuge, April 2012.

1.6 Project Objectives

The Bosque del Apache NWR, located within the floodplain of the Rio Grande River, uses an integrated management scheme of moist-soil management (the creation of exposed, saturated soils in wetlands by irrigation or drawdown during the growing season to promote germination, growth, and seed production of high energy wetland plants on mudflats) (Haukos and Smith 1993) and traditional flood irrigated agriculture to meet the energetic needs of wintering waterbirds. Field corn (*Zea mays*) is heavily relied upon to provide a rich carbohydrate food source to supplement moist-soil seed production. Production though of field corn has proven difficult at the Bosque del Apache NWR. While successful corn production is the result of a variety of properly managed agronomic procedures, the buildup of soil salinity in these agricultural fields may be a contributing factor to their low yields.

Field corn is a crop relatively intolerant to elevated concentrations of soil salinity. Its threshold to withstand salinity before succumbing to biomass loss is 1.7 dS/m (Mass and Hoffman 1977) and is most susceptible to saline conditions during the seedling and vegetative stages (Mass and Hoffman 1982).

The difficulty to grow field corn under traditional practices has caused refuge managers to consider alternative methods for the successful and sustainable production of corn. One possible

alternative is to incorporate a rotation of moist-soil management practices with traditional agriculture under the premise that the hydroperiod of moist-soil management may be better suited to control soil salinization.

Soils in the fields that support agriculture at the Bosque del Apache have been in a continuous rotation of alfalfa and corn production for 20 years. During this time period the above ground hydrologic regime is limited only to the irrigation of these crops and minimal precipitation that occurs is the late summer to early fall. The water supplied in flood irrigations for agriculture is largely lost through evapotranspiration and a leaching fraction may not exist to sufficiently move salts through the profile resulting in an accumulation of salts over time. In contrast, the surface hydrological regime of moist-soil management is much more seasonally dynamic and of greater magnitude than irrigations in agricultural production. Periods of sustained flooding and periodic flash floods may serve as leaching events that prohibit salts from accumulating in the upper portions of the soil profile.

However, soils at the Bosque del Apache NWR may also be susceptible to capillary rise of groundwater under shallow groundwater conditions. If groundwater is saline, then salinization could be occurring from surface hydrologic events and/or sub-surface groundwater interactions. Anecdotal evidence prior to this study suggests this may be occurring on select moist-soil impoundments and agricultural fields. It is unclear though what role the surface hydroperiods of moist-soil management and agricultural management have on their belowground watertables.

The effects of long term agricultural irrigation practices compared to the use of moist-soil management flooding on salinity in soils has not been evaluated on the refuge. Research is needed to evaluate whether or not implementing a wetland type hydroperiod into the rotation of fields under agricultural production would assist in lowering soil salinity levels and contribute to the overall sustainability of field corn production. Therefore the first set of objectives of my research is to:

1) To evaluate the effect of moist-soil management in comparison to agricultural production on salt accumulation in soil profiles

and

2) Evaluate how the hydrologic regimes associated with moist-soil management and agricultural production serve as leaching fractions that influence groundwater and salt removal.

The effects of specific management practices such as tillage and frequency of irrigation on soil salinity in units currently under moist-soil management have not been quantified on the refuge and therefore leave managers without any clear understanding to the effects of theses commonly used practices.

Rototilling to a depth of 12 cm is a successful tool for controlling the successional shift in wetland plants and creating a soil surface that is suitable for the germination of annual wetland plants. However, rototilling has not been evaluated at BdANWR for its effect on influencing the movement of salts within the root zone. When these tillage disturbances do occur, modifications of soil structure can increase or decrease hydraulic conductivity and permeability for water, heat, and air flow, as well as solute transport in soils and their spatial distribution (El Titi 2003). In moist-soil impoundments with shallow groundwater tables capillarity may be affected by rototilling. Some research has shown that tillage can serve to disconnect soil micropores and reduce the capillary rise of saline water. On a silt loam used for rice production, Wilson et al. (2000) demonstrated that no-till soils had higher salt concentrations in the root zone than did three variations of tillage. This suggests that conservative tillage practices paired with proper water management may be beneficial for stimulating the germination of wetland plants in low to non-saline conditions.

In many semi-arid environments where moist-soil management occurs, water availability may be limited and efforts to conserve freshwater inputs are a primary concern. However, it is unknown

how more conservative irrigation regimes might affect plant production. Increasing the time in between irrigation events may allow for the greater concentration of salts and increase osmotic stress (Chhabra 1996). Therefore there is a need to determine irrigation frequencies that conserve water yet does not negatively influence plant productivity by increasing osmotic stress.

The objectives of this study are to 1) evaluate the effects of rototilling on surface soil salinities in moist-soil units and to 2) evaluating the effects of frequency of irrigation on plant productivity in moist-soil units.

CHAPTER 2

EVALUATING THE INFLUENCE OF DISTINCT HYDROLOGIC REGIMES ON THE CONCENTRATION AND ACCUMULATION OF SOLUBLE SALTS IN MANAGED SOILS OF SEMI-ARID FLOODPLAIN ENVIRONMENTS

2.1 Introduction

The accumulation of soluble salts in the soil profile is defined as soil salinization and can occur through natural processes (Domingo et al. 2001; Rengasamy 2006) or through human modifications (e.g., secondary salinization; Chhabra 1996; Qadir et al. 2000). In many semi-arid environments, the active floodplain of rivers have been narrowed as a result of dykes and levees for the control of flooding and access to fertile soils for agriculture (Crawford et al. 1993). As a result, historic sections of the floodplain no longer receive surface flooding and have become more prone to secondary salinization (Morway and Gates 2012; references to salinization herein will refer to secondary salinization). In general, salinization occurs in the presence of soluble salts, a shallow water table that promotes capillary wicking (Northey 2006), and evapotranspiration demands that exceed fresh water inputs (Metternicht and Zinck 2009). Climatic variations and geomorphic characteristics (Jolly et al. 2008) are influential in the extent of salinization but surface hydrologic management such as the quantity and quality of irrigation water and the timing of application are also important (Ayers and Wescott 1984; Corwin and Rhoades 2007; Isidoro and Grattan 2011). Applied water that contains solutes can contribute to the net accumulation of salts in the profile. Even if concentrations of solutes are low, the long term use of water with low solute concentrations can rapidly increase salt to detrimental levels in the soil if soil leaching is insufficient (Ayers 1977). The quantity and quality of applied water determines whether an irrigation event leaches salts out of the root zone (Qadir et. al 2000). For mobilization of salts and leaching to occur, the amount of water applied must be greater than the evapotranspirational demand.

In semi-arid floodplain ecosystems, water management and salt accumulation in the rooting zone have important implications to migratory bird management. Wildlife refuges in these regions

commonly use irrigation to support moist-soil management and traditional agriculture to produce food resources for migratory birds (Kang et al. 2000; Taylor and Smith 2003, 2005). Moist-soil management is the creation of exposed, saturated soils in wetland impoundments by irrigation or drawdown during the growing season to promote germination, growth, and seed production of high energy wetland plants on mudflats (Haukos and Smith 1993). However, the success of both practices in semi-arid regions is partially dependent on the control of soluble salt accumulations in the rooting zone. Excessive accumulations can reduce osmotic potential and make plant water uptake more difficult, thus restricting the growth of vegetation and ultimately lowering biomass production (Mass and Hoffman 1977).

Although both management techniques use flood irrigation infrastructure, they have different hydrologic regimes and possibly different capacities to accumulate or remove salts. Traditional agricultural practices, such as an alfalfa (*Medicago sativa*) and field corn (*Zea mays*) rotation, irrigate to meet the transipirational needs of the crop. As a result, a smaller volume of water is applied relative to moist-soil impoundments over the course of the growing season. Moist-soil management practices use intermittent flood pulses during the summer that inundate the impoundment for 12 hours to 3 days during the summer and extended inundation (i.e, up to three months) during the winter (Fredrickson and Taylor 1982; Taylor and Smith 2005). Mean evapotranspiration rates diminish during the winter and likely decreases electrical conductivity of applied river water. Thus the impoundments are flooded for long periods with water possessing low solute concentrations. In contrast, agricultural fields are not flooded during winter.

Much of the work regarding the remediation and regulation of saline soils has been in the context of agriculture and little information is available on soil salinity under moist-soil management practices in semi-arid environments. The objectives of this study are: 1) To evaluate the effect of moist-soil management in comparison to agricultural production on salt accumulation in soil profiles and 2)

Evaluate how the hydrologic regimes associated with moist-soil management and agricultural production serve as leaching fractions that influence groundwater and salt removal.

I hypothesize that the hydrologic regime of moist-soil management will leach salts more thoroughly than soils exposed to the more modest growing season application of water found in traditional agricultural production.

2.2 Study Site and Management Practices

<u>2.21 Location</u>

The Bosque del Apache National Wildlife (33° 48", 106^{o} 53) Refuge is located south of San Antonio, New Mexico and lies within the Middle Rio Grande Basin, straddling the Rio Grande River. The river itself is a consequence of a continental rift valley dating back to the Paleogene (Crawford et al. 1993) and originates in the mountains of the Rio Grande National Forest in southern Colorado and flows south into New Mexico. The basin is bounded by mountain ranges rising 2,000 m to the west and 1,600 m to the east while the valley floor elevations average 1, 470 m (Crawford et al. 1993).

Regional climatic conditions are characterized by high light intensity, low relative humidity, and an average Class A pan evaporation of 250 cm per year (Johnson 1988). The annual average precipitation is approximately 23 cm, primarily occurring during the months of July through October (WRCC 2005).

Soils within the Rio Grande basin are derived from alluvial and clastic sediments (Crawford et al. 1993). For the purpose of my research, I limited my study sites to those that were mapped as Gila clay loam soil series to isolate the potential variability of soil salinity between different soil series. This series is used in both moist-soil management and agricultural production.

2.22 Management Practices

The refuge is divided into a series of management units. Within each unit are numerous fields that are managed similarly for either agricultural production or moist-soil management. I selected fields within 2 units for intensive study. Unit 6 (Figure 2) was selected for fields managed for moist-soil plant production and unit 9 (Figure 2) was selected for fields managed for agricultural production. Unit 6 was converted from agricultural production in 1993 and was done so because of previous problems with high soil salinities (Taylor 2000 and Vradenburg, personal communication). Unit 9 has been under agricultural production since at least 1990.

Irrigation water used for units 6 and 9 is diverted from the Rio Grande at San Acacia, New Mexico approximately 40 km north of the refuge and delivered to moist-soil impoundments and agricultural fields through a complex system of irrigation canals and drains. Fields under moist-soil management are impounded and are served with an interior feeder canal and feeder drain that provides independent field irrigation capability. Interior feeder canals lead to a single inflow gate and an equivalent outflow gate controls water that leaves the unit via the feeder drain.

Unit 9 is subdivided into individual fields which are capable of independent flooding but differ in that agricultural sub-units are equipped with a series of vertical lift gates that are distributed equally along the length of a field in order to deliver water in a more homogenous manner. Vertical lift gates on fields in unit 9 run along the north and south boundaries of a unit. In general, the basic irrigation structure of unit 9 is common to all agricultural fields on the refuge.

2.23 Moist-soil Management Practices

Annual moist-soil management begins in late April or early May with a shallow (20 cm) flood that is sustained for three days. On the fourth day, water is slowly released from the impoundment as stop logs are removed individually from the outflow gate over a period of three days. This slow draw

down of the water level creates a mudflat that stimulates the germination of annual wetland vegetation. Impoundments are subsequently flash flooded throughout the summer approximately every 14 days or as determined by managers. These flash floods inundate the units to 20 cm; as soon as this is accomplished all stop logs are removed from the gates to maximize drainage. Irrigations cease once vegetation reaches maturity and senesces in mid-August to mid-September. Wintering and migratory waterbirds can arrive as early as September, peak in mid to late December and remain until February; initial fall/winter flood dates and flood duration of moist soil impoundments varies among units to facilitate the provision of moist soil foods for waterbirds throughout the winter. Thus, fall and winter flooding of impoundments can last 2 weeks to 2 months depending on the size of the impoundment and use by waterbirds. Impoundments are usually drained in the January- March and left fallow until they are re-flooded in late April and early May to begin the next moist soil cycle. However, continued use of this cycle eventually encourages the propagation of perennial and woody vegetation that has lower food values for waterbirds. As a result, about every four years managers use sustained flooding, fire, and/or soil tillage to disturb the units to set back perennial vegetation and promote the production of desired annuals. Although the frequency of disturbance varies depending on vegetation response, soil tillage is the most commonly used disturbance and its variations include heavy and light disking and rototillage.

2.24 Agricultural Practices

Fields under traditional agriculture at the refuge rotate between alfalfa and field corn. Typically alfalfa is grown for four to five years in a field and then field corn is grown for two years. In April before the first year of corn, managers disc in alfalfa residue, laser level the field, and then create furrows. Ridges are irrigated prior to planting the corn to create a moist seed bed. Once soil moisture conditions are ideal, corn is sown into the ridge. After germination, managers irrigate approximately every 9 days or as determined necessary. Traditional management practices such as cultivation, macro-nutrient

fertilization, and application of herbicides and pesticides are all included in the current growing regime. Field corn typically reaches maturity in mid-October and is left standing in the fields. By the end of February, extensive waterbird use results in little remaining corn. Fields remain fallow until April when mangers either begin to prepare for a second year of corn or rotate the field back into alfalfa.

2.3 Methods

Three moist-soil impoundments and three agricultural fields described as the Gila soil series were selected for intensive study (Figure 2). Four sites were randomly selected within each impoundment and field. To determine initial soil salinities entering the growing season, a hydraulic Gidding's probe was used to extract one meter deep soil cores from each site on May 10^{th}, 2012. Upon removal from the ground each core was placed in a halved, four centimeter diameter, PVC pipe and wrapped with flexible plastic wrap. Paper towels were placed on both ends of the soil core to ensure that the contents did not fall out.

Each soil core was segmented into 10 cm portions and dried and ground to pass through a 2 mm sieve. Samples were sent to the LSU AgCenter Soil and Plant Laboratory for analysis. Samples were prepared in a 1:2 soil to water ratio, shaken for one hour and then filtered through a #42 Whatman filter paper screen. Extracts were then analyzed for water soluble elemental ions of calcium, chlorine, magnesium, and sodium using Inductively Coupled Plasma Spectroscopy (ICP) and reported in parts per million (ppm). Extracts were additionally used to calculate electrical conductivity and total salts from a temperature-compensating conductivity electrode standardized to 25 C. Plant salinity tolerances are typically given in the electrical conductivity of the soil saturated paste extract (Mass and Hoffman 1977), however, soil laboratories more commonly measure electrical conductivity from a soil to water ratio because it is less time consuming. Hogg and Henry (1984) found strong correlations between saturated paste extracts and soil to water preparations. Therefore saturated paste electrical conductivities were

estimated from 1:2 soil to water extracts using the regressions equations calculated by Hogg and Henry (1984).

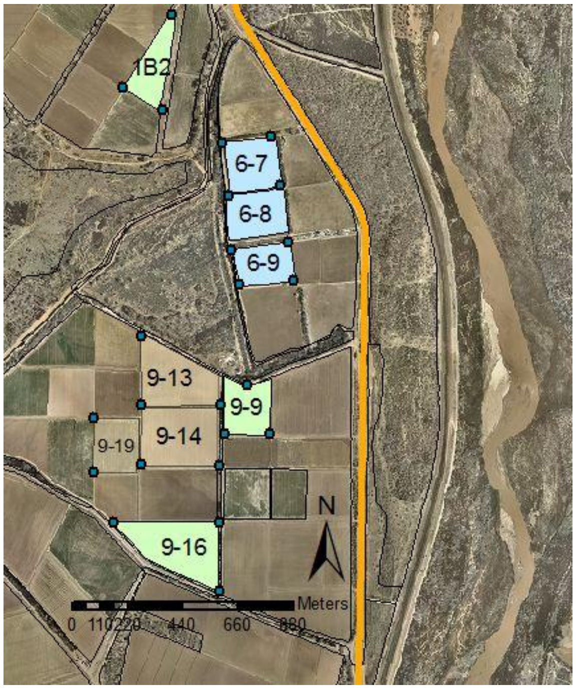

Figure 2 – Selected study sites at Bosque del Apache National Wildlife Refuge. Green polygons represent agricultural fields and blue polygons represent moist-soil impoundments. Brown polygons are the additional agricultural fields monitored for changes in depth to groundwater. The canal used to divert water for irrigation purposes is highlighted in orange.

Figure 3 - Moist-soil impoundments units 17A7, 17A4, and 17A1 were selected to evaluate the effects of flood irrigations on depth to groundwater and groundwater conductivity. These moist-soil impoundments are located south of Unit 6 and 9 on the refuge.

Particle size analysis by the hydrometer method as described by Gee and Bauder (1986) was performed and used to identify the proper regression equation for each soil texture (Table 3). In order to determine the effect of surface hydrologic regimes on depth to groundwater and groundwater electrical conductivity, groundwater wells were installed alongside the perimeter of the intensively selected moist-soil impoundments and agricultural fields. In addition, to increase sample size, I selected three additional agricultural fields (Figure 2) and three moist-soil impoundments (Figure 3).

Table 3 - Regression equations for converting $EC_{1:2}$ to EC_{SE} calculated by Hogg and Henry (1984) where EC_{SE} is equal to the estimated electrical conductivity of the saturated paste extract and $EC_{1:2}$ is the measured electrical conductivity of the 1:2 soil to water extract.

Soil Texture	Regression Equation
Coarse (Sand –Loamy Sand)	$EC_{SE} = 2.79 (EC_{1:2}) + 0.17$ [1]
Medium (Sandy loam – Silty Clay Loam)	$EC_{SE} = 2.35 (EC_{1:2}) – 0.36$ [2]
Fine (Sandy Clay – Clay)	$EC_{SE} = 2.16 (EC_{1:2}) + 0.03$ [3]

Groundwater wells were constructed of 4 cm diameter solid PVC pipe that was 1.5 m long and connected to a 1.5 m long slotted PVC well screen with a drainable end-cap. Wells were dug with a 20 cm diameter diesel powered auger bit, back-filled with sediment and tamped. Groundwater wells on Unit 17 (Figure 4) were installed in May 2011 to support a previous study while groundwater wells on Unit 9 and Unit 6 were installed on 28 May 2012 and 7 June 2012, respectively. Well measurements for Unit 17 begin on 15 May 2012, Unit 9 on May 30 2012, and Unit 6 on 8 June 2012. Measurements were taken three times a week until 1 August 2012. Depth to groundwater was collected using an in-situ electric dip tape. Electrical conductivity of the groundwater was measured using a portable temperature-compensating electrode standardized to 25 C.

Data collected from a United States Geological Survery (USGS 08355490) stream guage was used to monitor changes in electical conducivity of river water from the Rio Grande River. The stream gauge was located approximately 13 kilometer north of the refuge in San Antonio, New Mexico. Data was available for 2011 and 2012. Weekly electrical conductivity measurements from 10 May 2012 to 1 August 2012 were additionally taken from the the irrigation canal within the refuge used to specifically irrigate the units in my study (Figure 2). Measurements were taken with a portable temperature-compensating electrode standardized to 25 C.

Figure 4 – Drilling groundwater wells (left) and installment (right) in moist-soil units at Bosque del Apache National Wildlife Refuge, June 2011.

2.4 Statistical Analysis

SAS 9.3 software (SAS Institute, Inc., Cary, NC) was used for all statistical analyses. Differences in measured variables of each core sample were evaluated using a nested analysis of variance to evaluate changes in soluble salts with depth between treatments and by depth within treatments. Treatment and depth were assigned as fixed effects. Previous studies indicate strong relationships between high clay content and electrical conductivity. Therefore the percentage of clay in the soil at each depth was assigned as a random effect in the model. For purposes of this study, means were considered different when P-values where < 0.05.

Depth to groundwater and groundwater conductivity measurements were evaluated for differences between moist-soil impoundments and agricultural fields. The amount of variability in each treatment was determined by calculating the total range (Δ) in values throughout the measured period, where:

$$\Delta = \text{Maximum value} - \text{Minimum value}$$

Calculated Δs were then analyzed in an analysis of variance model where treatment was assigned as a fixed effect. Mean depth to groundwater and groundwater conductivity for the measured periods were calculated for groundwater wells in each treatment. Means were analyzed in an analysis of variance model where treatment was assigned as a fixed effect. Means were considered different when P-values where <0.05.

2.5 Results

Annual rainfall during the study period was 18.54 cm and 11.02 cm in 2011 and 2012, respectively (Table 4). On December 1, 2010, the Natural Resources Conservation Service assisted in describing a more recent profile of the structural components of the Gila clay loam (Table 5).

2.51 Soil Cores

Type 3 tests of fixed effects indicated that mean values of several soil parameters differed across management type, depths, and tillage treatment. The soil profiles were divided into 10 cm segments and depths mentioned herein refer to the midpoint depth of a 10 cm segment (i.e, depth at 15 cm represents the 10 to 20 cm segment of the soil profile). Mean values in percent clay for cores from agricultural fields ranged from 20% to 34%; however, no differences were detected in the percentage of clay by depth throughout the profile. Mean values in percent clay for cores from moist-soil impoundments ranged from 4% to 25%. The percentage of clay differed by depth in moist-soil impoundments at three depths throughout the profile. Mean values at 55 cm, 85 cm, and 95 cm were lower than all other depths but did not differ from each other. Clay concentration in the sampled profile cores of moist-soil impoundments and agricultural fields (Figure 5) differed only at depths of 85 cm and 95 cm.

Table 4 – Mean maximum and minimum monthly temperature and precipitation for 2011 and 2012 collected from the National Oceanic and Atmospheric Administration weather station located at Bosque del Apache National Wildlife Refuge . (-) Represents missing data.

Station: BOSQUE DEL APACHE, NM US 2012									
Elev: 4511 ft. Lat: 33.804° N Lon: 106.891° W									
2011					2012				
Date	Temperature (°C)		Precipitation (cm)		Date	Temperature (°C)		Precipitation (cm)	
Month	Mean Max.	Mean Min.	Total	Greatest Observed Event	Month	Mean Max.	Mean Min.	Total	Greatest Observed Event
January	14.50	-8.89	0	0	January	15.78	-5.61	0	0
February	15.50	-7.50	0.61	0.61	February	16.28	-4.61	0.08	0.05
March	24.89	0.22	0	0	March	22.39	-1.00	0.25	0.25
April	27.89	4.50	0	0	April	27.61	5.11	0.43	0.41
May	29.89	5.39	0	0	May	30.89	8.39	1.32	0.76
June	-	-	-	-	June	37.78	11.61	0.05	0.05
July	38.00	17.39	3.78	1.96	July	36.00	16.61	1.73	1.17
August	36.78	16.61	2.08	0.66	August	36.22	15.11	2.31	1.55
September	32.50	10.39	1.35	0.61	September	32.22	10.39	2.69	1.78
October	27.22	2.00	1.27	0.89	October	28.11	2.22	0	0
November	18.39	-3.78	0.30	0.30	November	21.61	-3.22	0.28	0.25
December	9.22	-8.28	9.14	2.79	December	14.28	-6.78	1.88	1.60
Summary	25.00	2.56	18.54	2.79	Summary	26.61	4.00	11.02	1.78

Table 5 –Physical soil profile description of the Gila series taken by the Natural Resource Conservation Service at the Bosque del Apache National Wildlife Refuge on December 1, 2010.

Gila Series: Coarse-loamy, mixed, superactive, calcareous, thermic Typic Torrifluvents							
Depth (cm)	Horizon	Matrix Color (Moist)	Texture	Clay %	Rupture Resistance		
					Resistance (Moist)	Stickiness	Plasticity
0-10	Ap	7.5 YR (3/2)	clay loam	34	Friable	Moderately Sticky	Moderately Plastic
10-35	C1	7.5 YR(4/3)	silty clay loam	36	Friable	Moderately Sticky	Very Plastic
35-60	C2	7.5 YR (4/2)	silty clay loam	34	Firm	Moderately Sticky	Very Plastic
60-75	C3	5 YR (4/2)	silty loam	26	Friable	Very Sticky	Moderately Plastic
75-100+	C4	7.5 YR (4/4)	loamy sand	3	Loose	Non-sticky	Non-plastic

Covariance estimates of clay when used as a random variable were smaller than the total residual in all measured variables (Table 6). These results suggest that clay concentration was not a dominate influence on soluble salt concentrations. Therefore, % clay was excluded as a covariate in the model and all results reported are without % clay as a covariate.

Table 6 – Covariance Parameter Estimates for % clay as a random effect in analysis of variance to evaluate mean differences in soil soluble salts from moist-soil units and agricultural fields by depth at Bosque del Apache National Wildlife Refuge, 10 May 2012.

Covariance Parameter Estimates		
Variable	Parameter	Estimate
Total Soluble Salts	Clay	0.000188
	Residual	0.1644
Conductivity	Clay	0.000192
	Residual	0.2996
Calcium	Clay	0.000287
	Residual	0.2917
Chlorine	Clay	0.000112
	Residual	0.3329
Magnesium	Clay	0.00025
	Residual	0.1818
Sodium	Clay	0.000231
	Residual	0.1273
Sodium Absorption Ratio	Clay	0.000044
	Residual	0.1294
Sulfur	Clay	0.00042
	Residual	0.2248

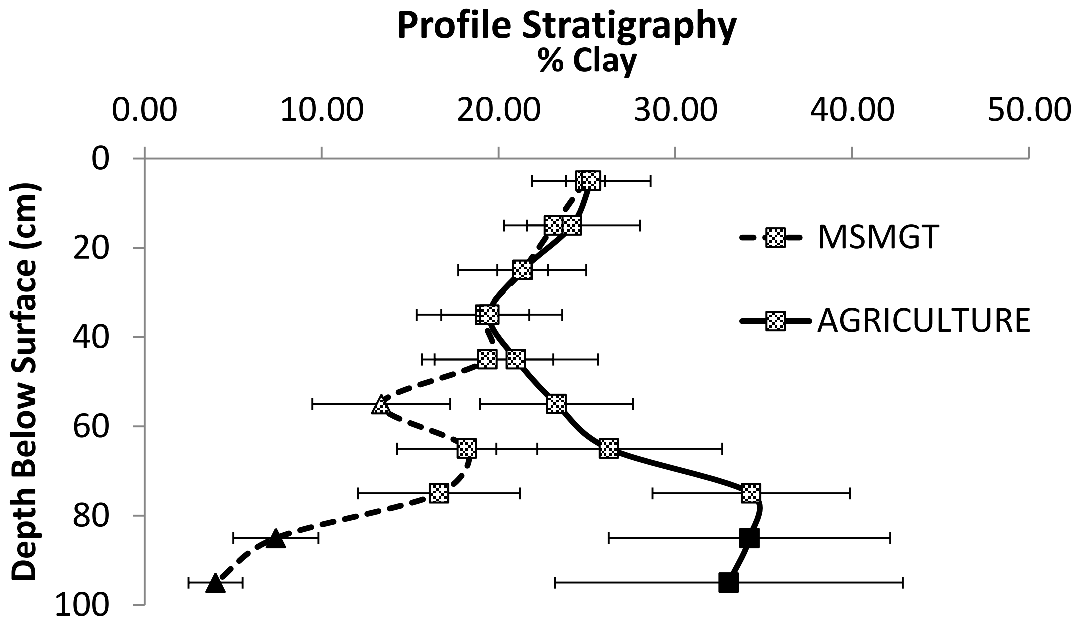

Figure 5 – Mean values for percent clay concentrations in sampled profiles from 0 – 100 cm taken at Bosque del Apache National Wildlife Refuge, 10 May 2012. Soil profiles were divided into 10 cm segments and point markers represent the midpoint of each section. Point markers that share the same shape within treatments represent similar groups. Depths with checkered point markers represent no difference in values between treatments. Depths with solid fill point markers represent differences in values between treatments.

Table 7 - Type III Test of Fixed Effects from Nested Analysis of Variance of soluble salts, electrical conductivity, and pH measured in moist-soil units and agricultural fields from Bosque del Apache National Wildlife Refuge, 10 May 2012. (*) represents differences in means at the alpha =0.05 level.

Type 3 Tests of Fixed Effects					
Variable	Effect	Num DF	Den DF	F Value	Pr > F
Total Soluble Salts	trt	1	196	200.53	<.0001*
	depth	9	196	1.16	0.325
	trt*depth	9	196	4.87	<.0001*
Electrical Conductivity	trt	1	196	200.3	<.0001*
	depth	9	196	1.16	0.3226
	trt*depth	9	196	4.87	<.0001*

(Table 7 Continued)

Variable	Effect	Num DF	Den DF	F Value	Pr > F
Calcium	trt	1	196	2.86	0.0922
	depth	9	196	6.21	<.0001*
	trt*depth	9	196	1.3	0.2388
Chlorine	trt	1	195	126.51	<.0001*
	depth	9	195	6.46	<.0001*
	trt*depth	9	195	8.39	<.0001*
Magnesium	trt	1	196	60.15	<.0001*
	depth	9	196	2.08	0.0329*
	trt*depth	9	196	1.79	0.0721
Sodium	trt	1	196	407.84	<.0001*
	depth	9	196	1.96	0.0461*
	trt*depth	9	196	10.92	<.0001*
Sodium Absorption Ratio	trt	1	196	386.95	<.0001*
	depth	9	196	8.03	<.0001*
	trt*depth	9	196	8.68	<.0001*
Sulfur	trt	1	196	247.97	<.0001*
	depth	9	196	2.05	0.036*
	trt*depth	9	196	11.72	<.0001*
pH	trt	1	196	0.8	0.3713
	depth	9	196	8.39	<.0001*
	trt*depth	9	196	0.59	0.8012

Within agricultural fields, the range of mean values in electrical conductivity and total soluble salts in cores of agricultural fields ranged from 1.21 dS/m to 2.93 dS/m and 776ppm to 1875 ppm, respectively. Although mean values of electrical conductivity and total soluble salts increased with depth, only the 95 cm depth differed statistically from the remaining profile. Within moist-soil units, the range of mean values in total soluble salts (Figure 6) and electrical conductivity (Figure 7) ranged from

385ppm to 545 ppm and 0.6 dS/m to .85 dS/, respectively. No differences in mean values were detected by depth. Between moist-soil management and agricultural treatments, a treatment*depth interaction was revealed for electrical conductivity and total soluble salts (Table 7). Agricultural fields had greater concentrations of total soluble salts and electrical conductivity than did moist-soil units in the 45 cm to 95 cm portion of the profile.

Within agricultural fields, pH (Figure 8), chlorine (Figure 10), sodium (Figure 12), Sodium Absorption Ratio (Figure 13) and sulfate (Figure 14) differed by depth, whereas in moist-soil units only pH, calcium (Figure 9), and sodium differed by depth. Among treatments, a depth*treatment interaction was observed for chlorine, magnesium (Figure 11), sodium, Sodium Absorption Ratio, and sulfate.

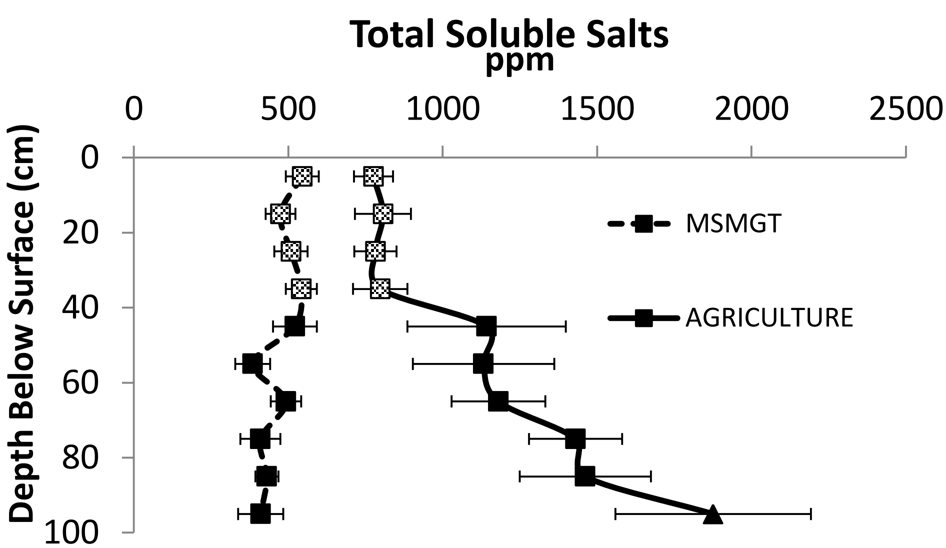

Figure 6- Mean values for total soluble salt concentrations in sampled profiles from 0 – 100 cm taken at Bosque del Apache National Wildlife Refuge, 10 May 2012. Soil profiles were divided into 10 cm segments and point markers represent the midpoint of each section. Point markers that share the same shape within treatments represent similar groups. Depths with checkered point markers represent no difference in values between treatments. Depths with solid fill point markers represent differences in values between treatments.

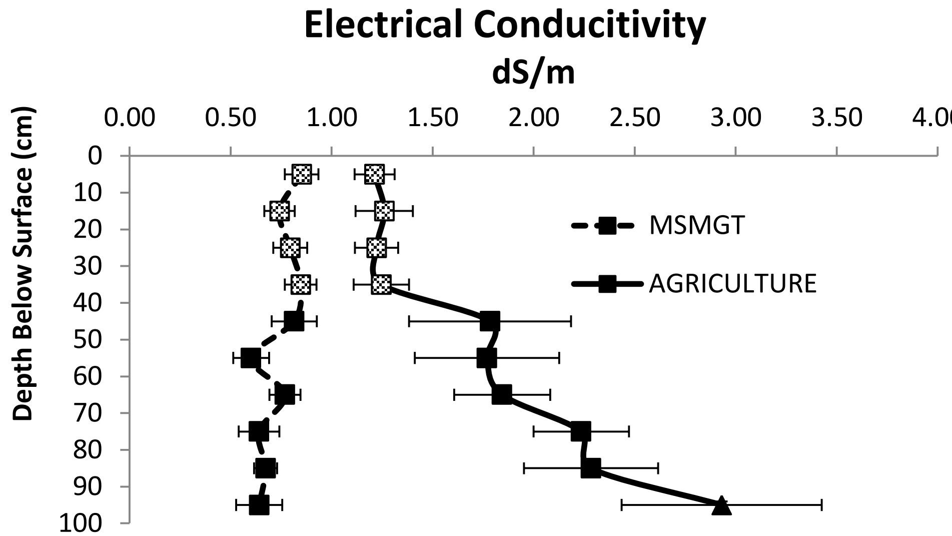

Figure 7 - Mean values for saturated paste electrical conductivity in sampled profiles from 0 – 100 cm taken at Bosque del Apache National Wildlife Refuge, 10 May 2012. Soil profiles were divided into 10 cm segments and point markers represent the midpoint of each section. Point markers that share the same shape within treatments represent similar groups. Depths with checkered point markers represent no difference in values between treatments. Depths with solid fill point markers represent differences in values between treatments.

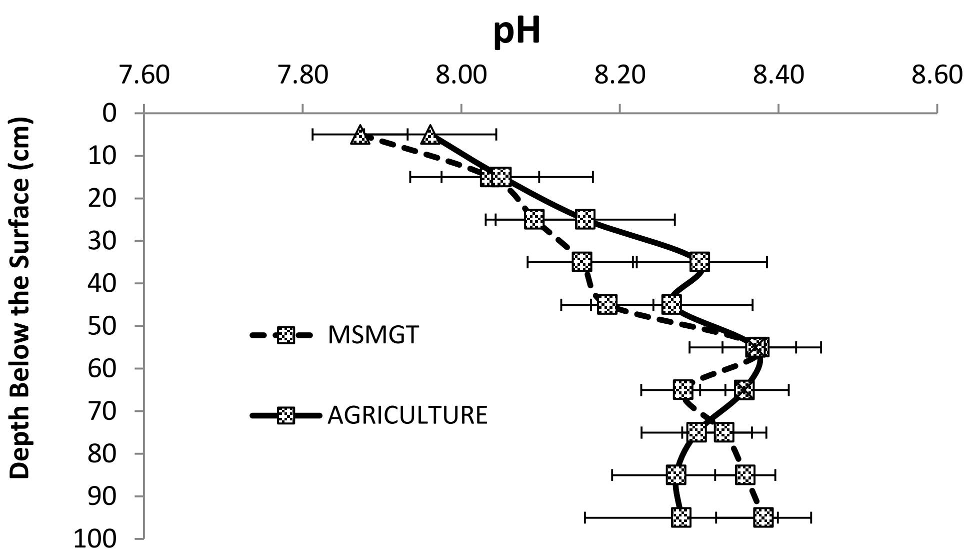

Figure 8 - Mean values for pH in sampled profiles from 0 – 100 cm taken at Bosque del Apache National Wildlife Refuge, 10 May 2012. Soil profiles were divided into 10 cm segments and point markers represent the midpoint of each section. Point markers that share the same shape within treatments represent similar groups. Depths with checkered point markers represent no difference in values between treatments. Depths with solid fill point markers represent differences in values between treatments.

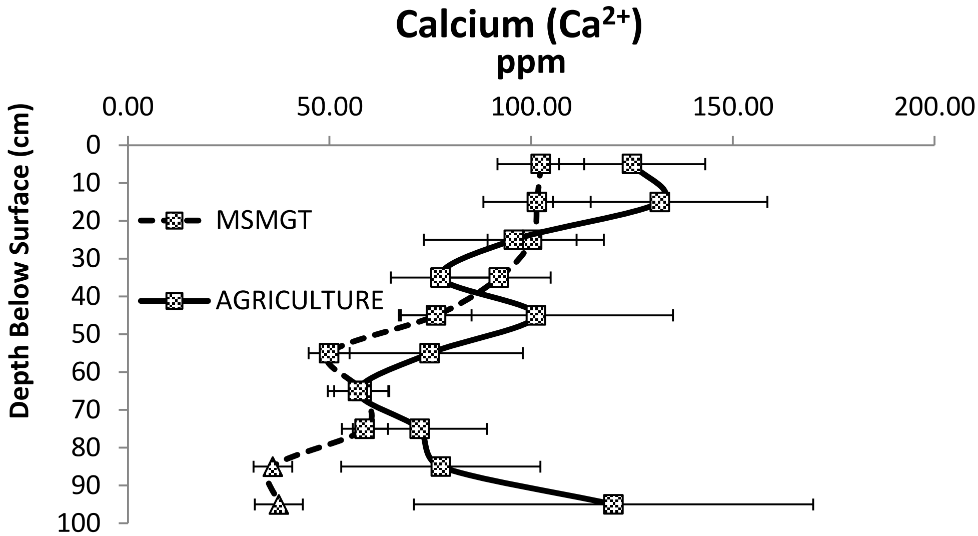

Figure 9 - Mean values for calcium concentrations in sampled profiles from 0 – 100 cm taken at Bosque del Apache National Wildlife Refuge, 10 May 2012. Soil profiles were divided into 10 cm segments and point markers represent the midpoint of each section. Point markers that share the same shape within treatments represent similar groups. Depths with checkered point markers represent no difference in values between treatments. Depths with solid fill point markers represent differences in values between treatments.

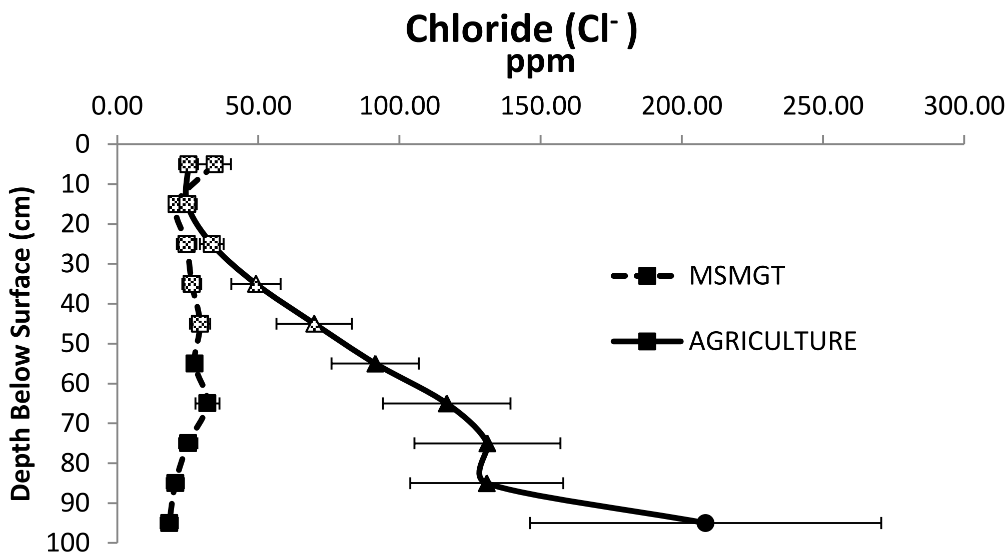

Figure 10- Mean values for chloride concentrations in sampled profiles from 0 – 100 cm taken at Bosque del Apache National Wildlife Refuge, 10 May 2012. Soil profiles were divided into 10 cm segments and point markers represent the midpoint of each section. Point markers that share the same shape within treatments represent similar groups. Depths with checkered point markers represent no difference in values between treatments. Depths with solid fill point markers represent differences in values between treatments.

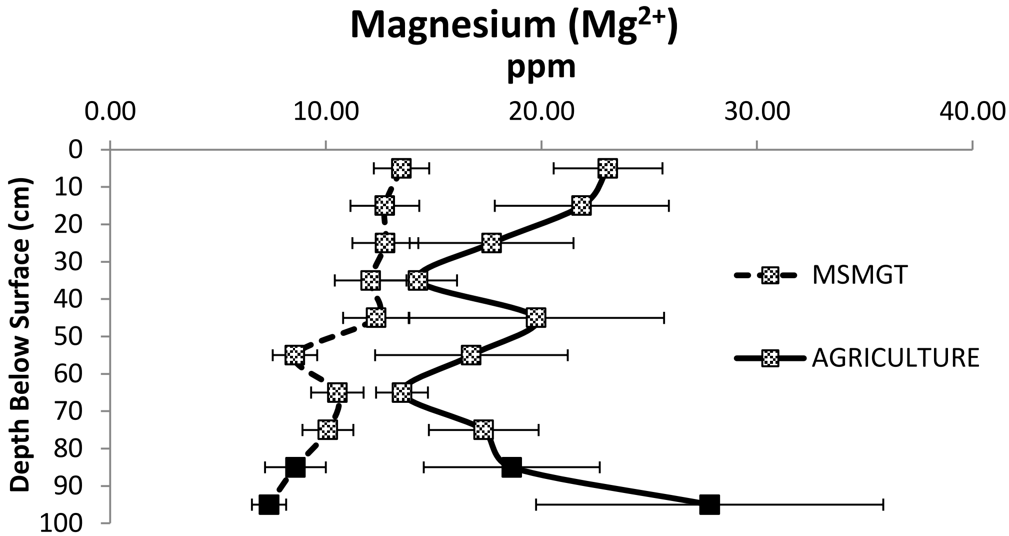

Figure 11- Mean values for magnesium concentrations in sampled profiles from 0 – 100 cm taken at Bosque del Apache National Wildlife Refuge, 10 May 2012. Soil profiles were divided into 10 cm segments and point markers represent the midpoint of each section. Point markers that share the same shape within treatments represent similar groups. Depths with checkered point markers represent no difference in values between treatments. Depths with solid fill point markers represent differences in values between treatments.

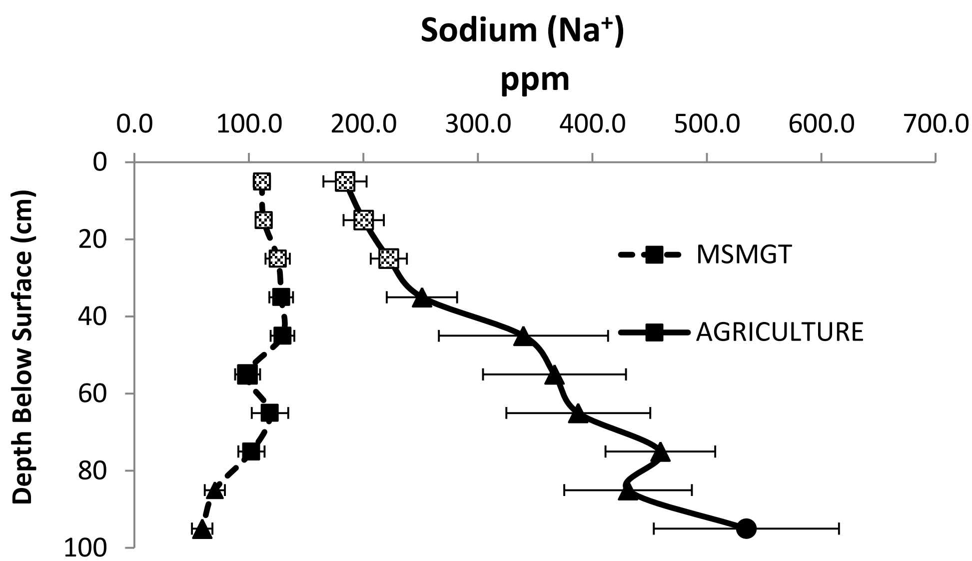

Figure 12 - Mean values for sodium concentrations in sampled profiles from 0 – 100 cm taken at Bosque del Apache National Wildlife Refuge, 10 May 2012. Soil profiles were divided into 10 cm segments and point markers represent the midpoint of each section. Point markers that share the same shape within treatments represent similar groups. Depths with checkered point markers represent no difference in values between treatments. Depths with solid fill point markers represent differences in values between treatments.

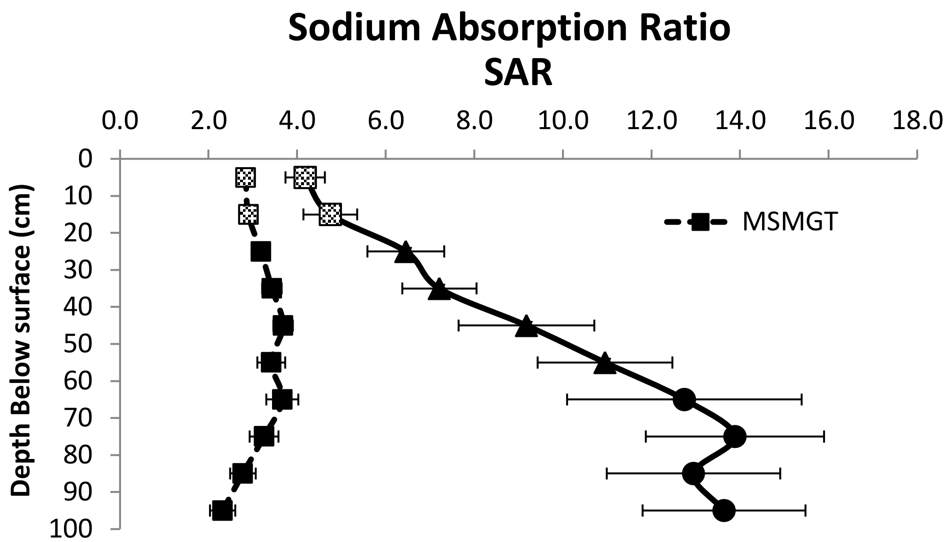

Figure 13 - Mean values for sodium absorption ratios in sampled profiles from 0 – 100 cm taken at Bosque del Apache National Wildlife Refuge, 10 May 2012. Soil profiles were divided into 10 cm segments and point markers represent the midpoint of each section. Point markers that share the same shape within treatments represent similar groups. Depths with checkered point markers represent no difference in values between treatments. Depths with solid fill point markers represent differences in values between treatments.

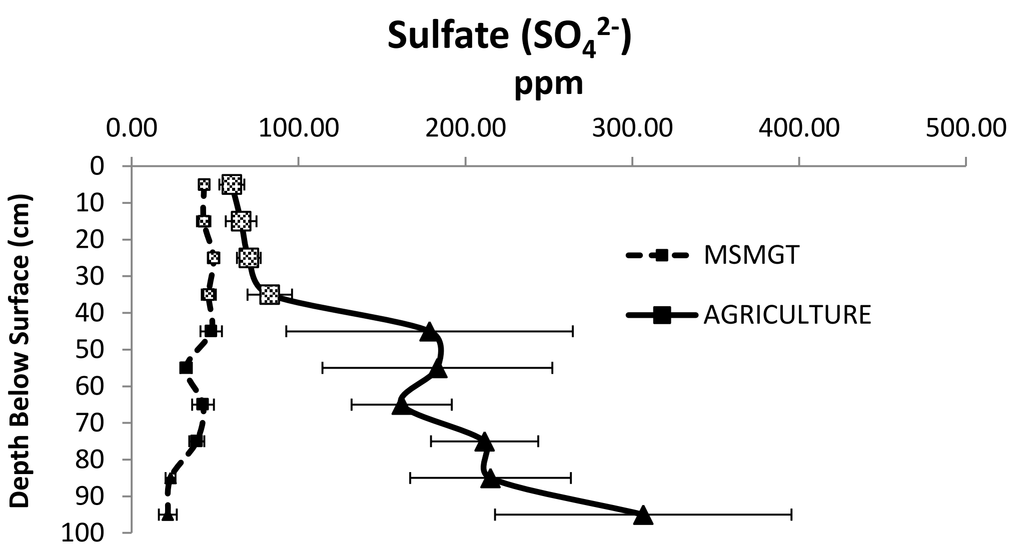

Figure 14 - Mean values for sulfate concentrations in sampled profiles from 0 – 100 cm taken at Bosque del Apache National Wildlife Refuge, 10 May 2012. Soil profiles were divided into 10 cm segments and point markers represent the midpoint of each section. Point markers that share the same shape within treatments represent similar groups. Depths with checkered point markers represent no difference in values between treatments. Depths with solid fill point markers represent differences in values between treatments.

2.52 Water Quality of Applied Irrigation Water

Electrical conductivity of river water varied from 420 uS/cm to 1080 uS/cm in 2011 (Figure 15), and 335 uS/m to 2950 uS/cm in 2012 (Figure 16). In both 2011 and 2012 electrical conductivity was highest during the summer months and lowest during the winter. The seven year daily discharge mean (Figure 17) depicts a peak in discharge in spring and a period of drying throughout the summer with return flows during the winter.

Increases in conductivity during the summer months most likely are a result of greater rates of evapotranspiration that results in high water efflux and concentrated salts. In addition, high water demand for agricultural irrigation within the river valley can further reduce the amount of water in the river. In 2012, a severe spike in river water electrical conductivity corresponded to the drying up of the river. The river was dry for the majority of the time during July to November. When flow returned, electrical conductivity subsequently returned closer to average levels.

Within the refuge, electrical conductivity of water taken from the Riverside irrigation canal during the growing season had a mean value of 896 uS/cm ± 36 uS/cm and ranged from 813 uS/cm to 1210 uS/cm (Figure 18). In late May, 2012, the refuge was faced with a temporary shortage in water supply causing water in the irrigation canal to almost completely dry up. A spike in electrical conductivity during this time period is likely a result of concentrated salts in the remaining water. Normal water flow returned within a week and electrical conductivity values returned closer to the mean.

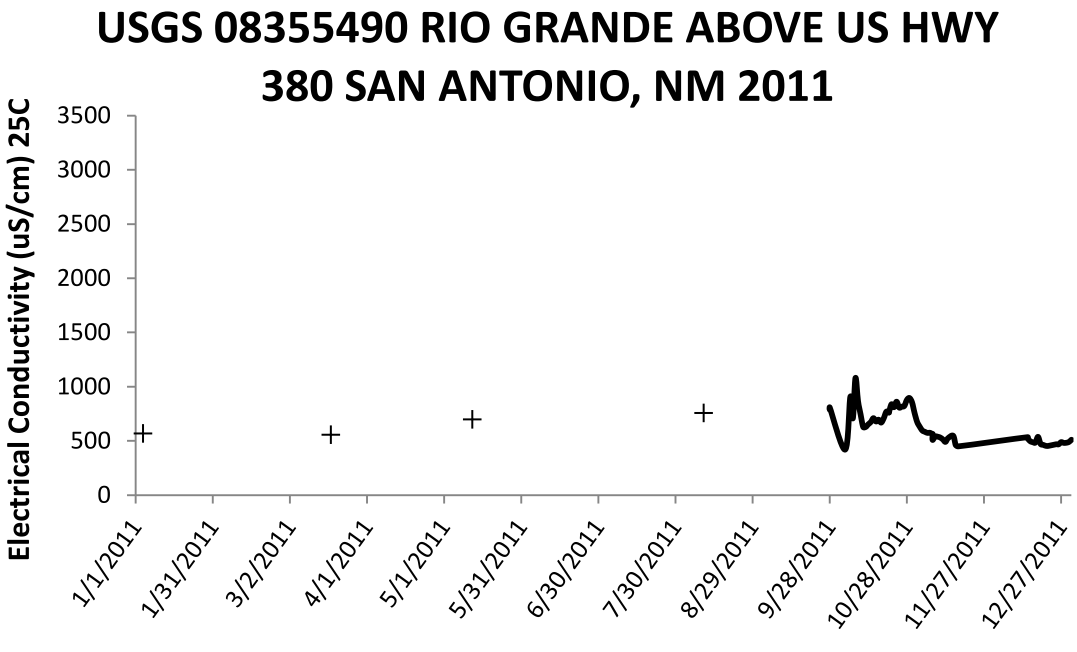

Figure 15 - 2011 electrical conductivity data collected from the USGS monitoring station on the Rio Grande River located approximately 8 miles north of the Bosque del Apache NWR. + point markers represent individual observations prior to continuous recording beginning in early October 2011.

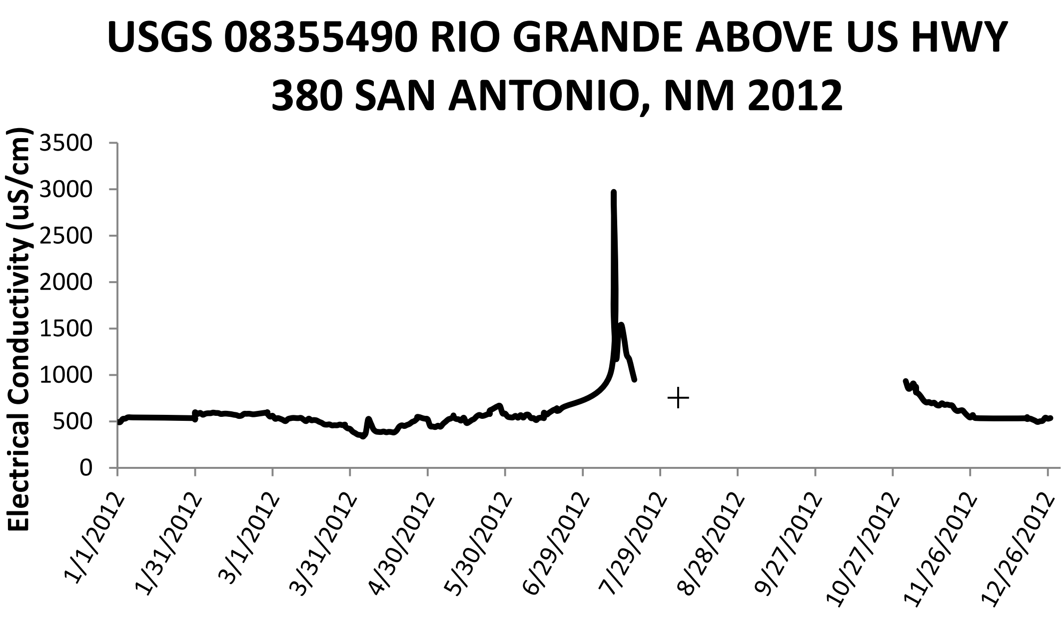

Figure 16 –2012 electrical conductivity data collected from the USGS monitoring station on the Rio Grande River located approximately 8 miles north of the Bosque del Apache NWR. + point markers represent an individual observation recorded during a period when data was otherwise not collected by the data logger. The extreme peak in mid-July is likely a result of the drying out of the river. This peak

and subsequent period of uncollected data in August, September, and October corresponds with the no to little observed discharge during those months in Figure 17.

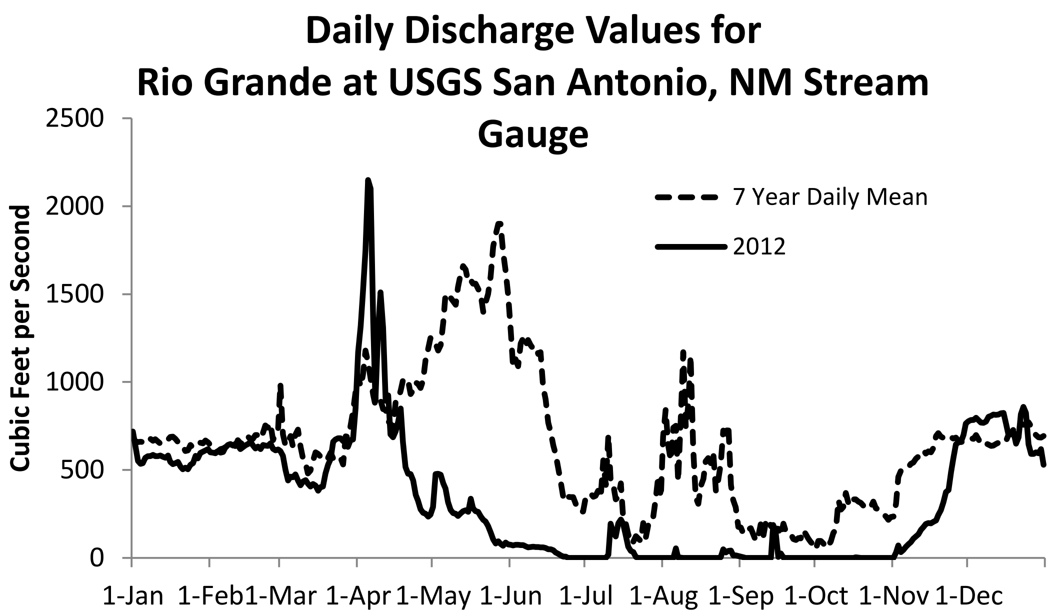

Figure 17 - Daily discharge for the seven year mean and 2012 at USGS stream gauge 08355490 in San Antonio, New Mexico on the Rio Grande River.

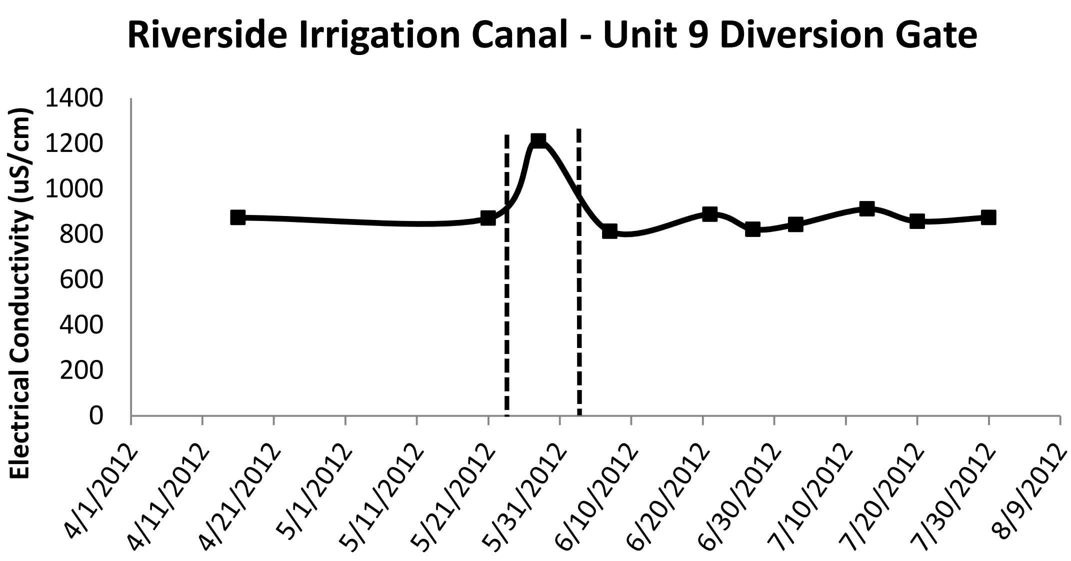

Figure 18 – Electrical conductivity of water used to irrigate study plots in Unit 6, 9, and 17 in 2012 at Bosque del Apache National Wildlife Refuge. Shaded point markers represent dates that a grab sample was collected. Values in between the dashed line represent the time period when water was in short supply. It is likely to have caused the observed temporary increase in electrical conductivity.

2.53 Effect of Hydroperiod on Depth to Groundwater and Groundwater Electrical Conductivity

Irrigation events in moist-soil impoundments resulted in a temporary decrease in depth to groundwater. Initial irrigations caused a temporary increase in groundwater conductivity, but subsequent irrigations tended to cause a temporary dilution in groundwater (Figures 20-25). Upon installation, three groundwater wells quickly filled in with silt and became ineffective for measuring depth to groundwater. As a result, moist-soil units 6-7 and 6-8 had only 2 and 3 wells, respectively, instead of 4.

No data are available on dates of irrigation in agricultural fields although irrigations are known to have occurred during the monitoring period. Irrigation events in agricultural fields tended to have a less pronounced effect on depth to groundwater and groundwater conductivity based on the low variability in groundwater depths throughout the growing season (Figures 26- 31). Depth to groundwater was generally deeper than it was in moist-soil impoundments and tended to deepen over the course of the growing season.

Type 3 test of fixed effects showed that moist-soil impoundments and agricultural fields differed in overall variability of depth to groundwater during the course of the growing season (Table 8). Differences also existed in variability of groundwater conductivity among treatments (Figure 19). Mean values in depth to groundwater and groundwater conductivity in agricultural fields were greater than in moist-soil impoundments (Figure 19).

Table 8 – Type three test of fixed effects for ANOVA of variability and mean values of depth to groundwater and groundwater conductivity in two treatments: moist-soil impoundments and agricultural fields at Bosque del Apache National Wildlife Refuge, May –August 2012. Treatment type was used as a fixed effect.

Type III Tests of Fixed Effects					
Variable	Effect	Num DF	Den DF	F Value	Pr > F
Δ DTGW	Treatment	1	39	169.75	<.0001
Mean DTGW	Treatment	1	39	151.71	<.0001
Δ GW EC	Treatment	1	39	818.65	<.0001
Mean GW EC	Treatment	1	39	1295.32	<.0001

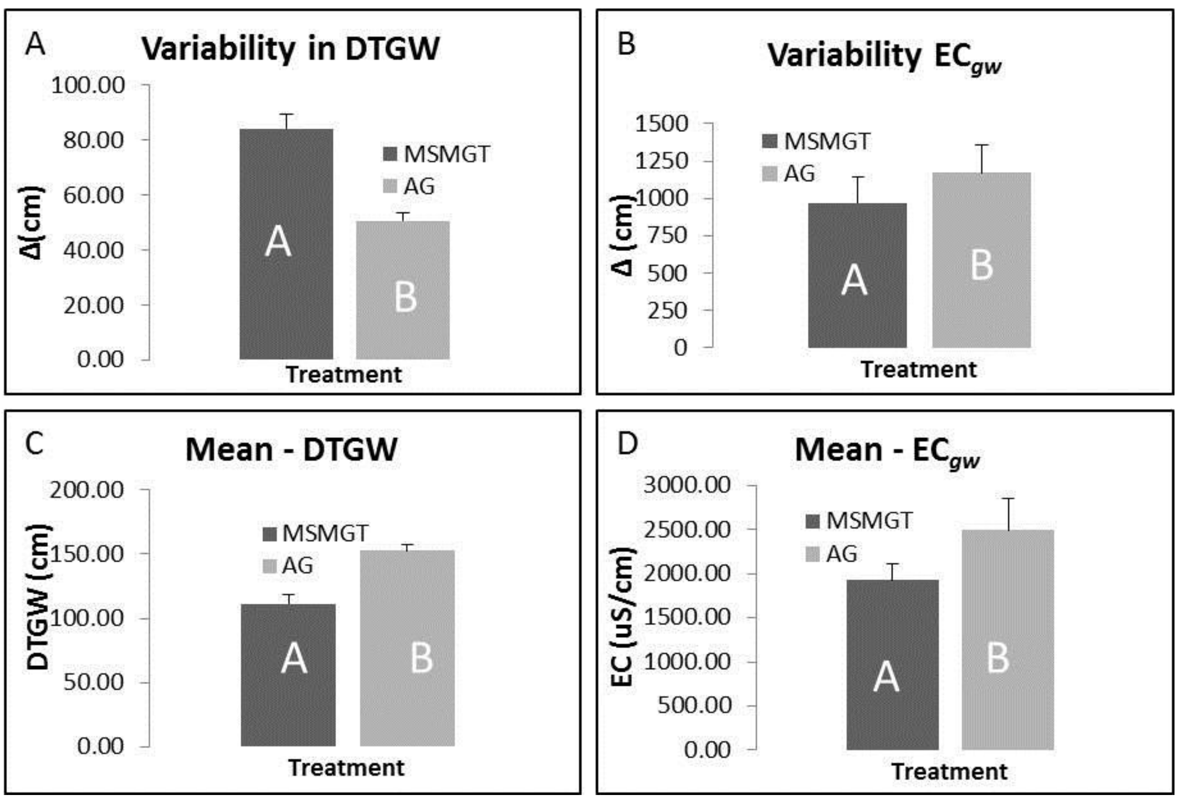

Figure 19 –Mean values between a) variability in depth to groundwater, b) variability in groundwater conductivity, c) depth to ground water, and d) groundwater electrical conductivity within moist-soil units and agricultural fields at Bosque del Apache National Wildlife Refuge, from 15 May 2012 to 1 August 2012. Means sharing a letter do not differ (P > 0.05).

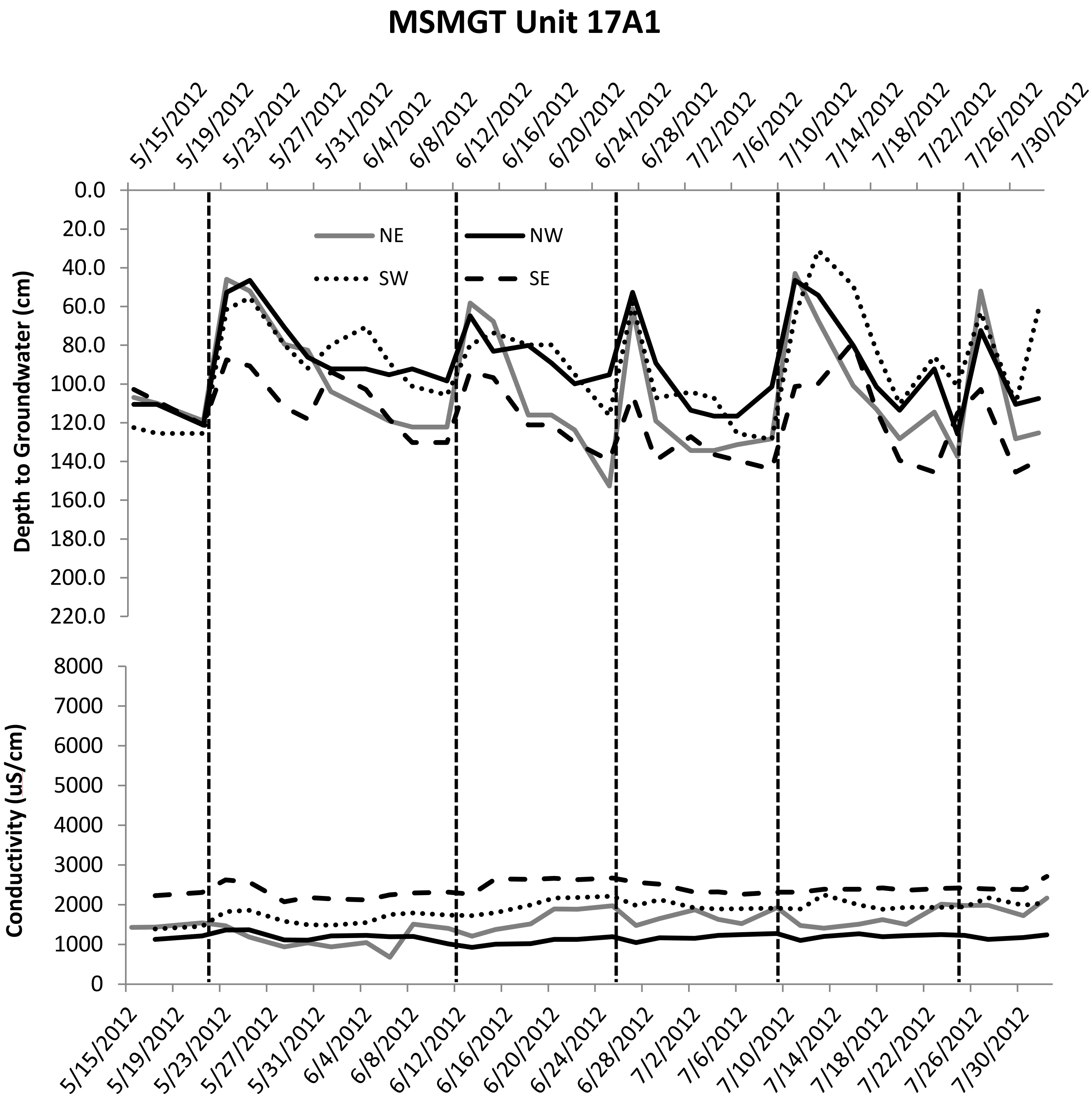

Figure 20 – Changes in depth to groundwater (upper panel) and groundwater electrical conductivity (lower panel) in moist-soil management site 17A1 during the growing season at Bosque del Apache National Wildlife Refuge, 2012. Dashed vertical lines represent recorded flash flood irrigations. Corresponding patterned lines represent the depth to groundwater and groundwater electrical conductivity in response to irrigation events of individual groundwater monitoring wells located on the northeast (NE), northwest (NW), southeast (SE), and southwest (SW) corners of the moist-soil unit.

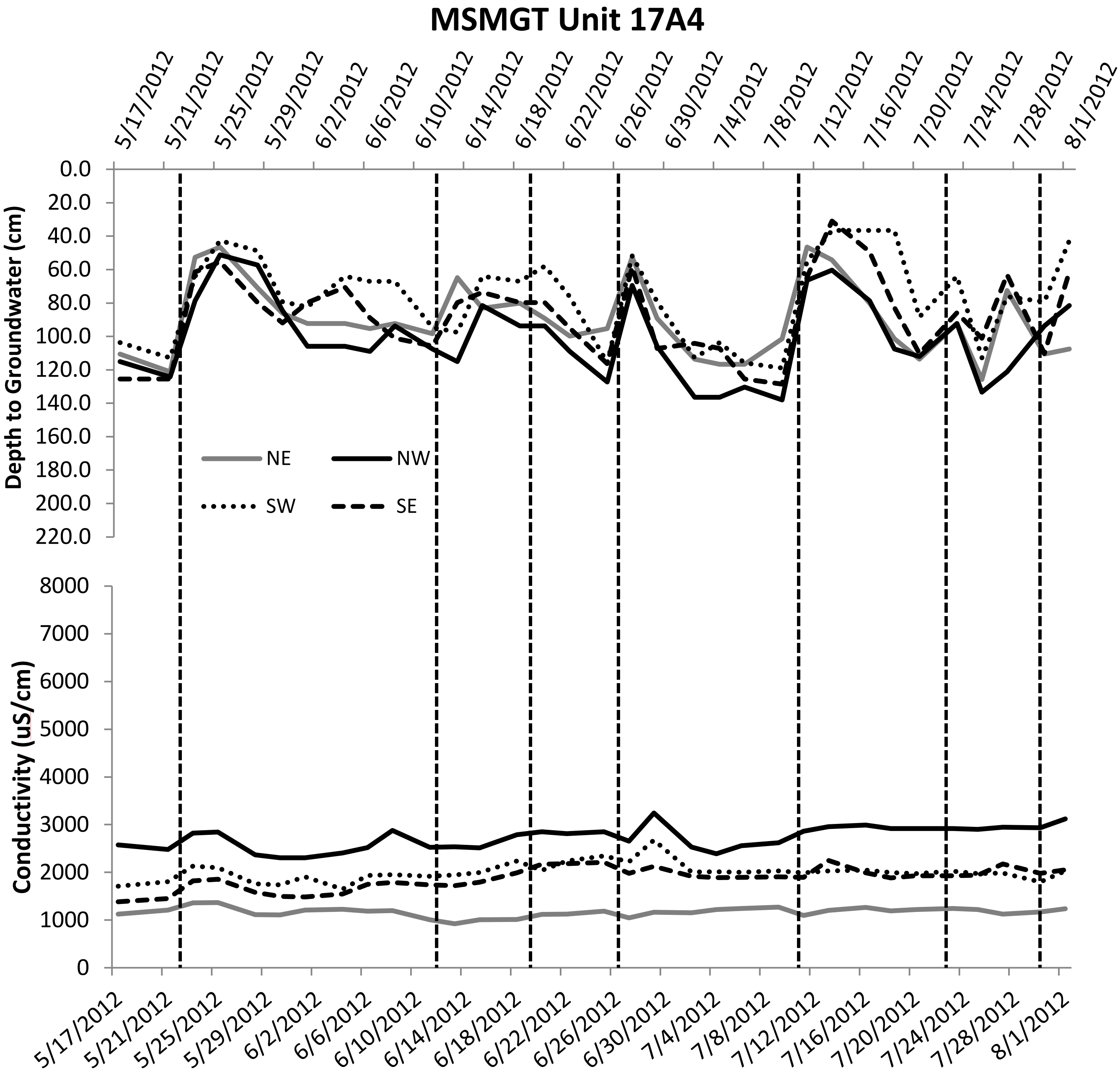

Figure 21 - Changes in depth to groundwater (upper panel) and groundwater electrical conductivity (lower panel) in moist-soil management site 17A4 during the growing season at Bosque del Apache National Wildlife Refuge, 2012. Dashed vertical lines represent recorded flash flood irrigations. Corresponding patterned lines represent the depth to groundwater and groundwater electrical conductivity in response to irrigation events of individual groundwater monitoring wells located on the northeast (NE), northwest (NW), southeast (SE), and southwest (SW) corners of the moist-soil unit.

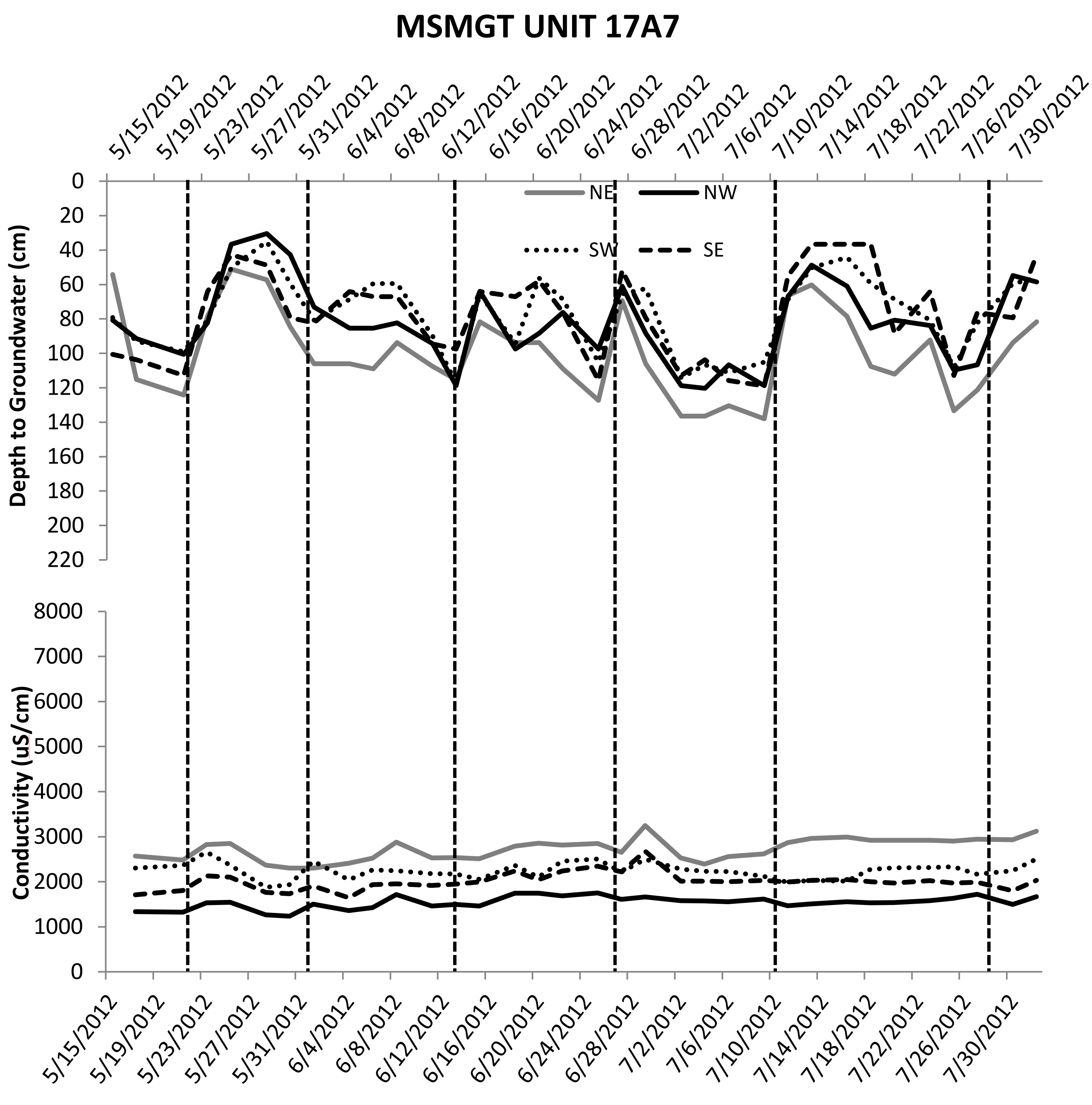

Figure 22 - Changes in depth to groundwater (upper panel) and groundwater electrical conductivity (lower panel) in moist-soil management site 17A7 during the growing season at Bosque del Apache National Wildlife Refuge, 2012. Dashed vertical lines represent recorded flash flood irrigations. Corresponding patterned lines represent the depth to groundwater and groundwater electrical conductivity in response to irrigation events of individual groundwater monitoring wells located on the northeast (NE), northwest (NW), southeast (SE), and southwest (SW) corners of the moist-soil unit.

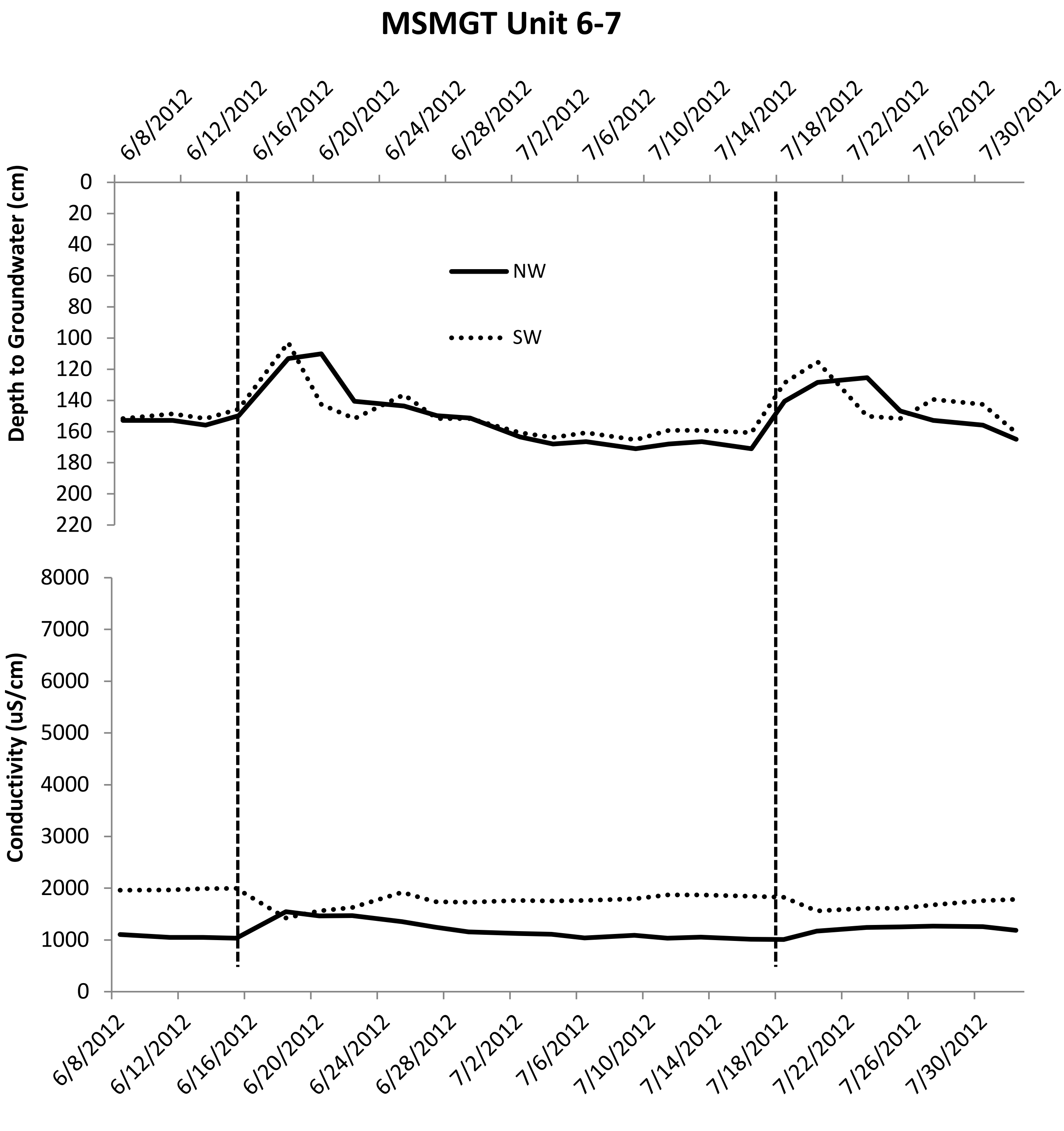

Figure 23 - Changes in depth to groundwater (upper panel) and groundwater electrical conductivity (lower panel) in moist-soil management site 6-7 during the growing season at Bosque del Apache National Wildlife Refuge, 2012. Dashed vertical lines represent recorded flash flood irrigations. Corresponding patterned lines represent the depth to groundwater and groundwater electrical conductivity in response to irrigation events of individual groundwater monitoring wells located on the northwest (NW) and southwest (SW) corners of the moist-soil unit. Fewer irrigation events in Unit 6 as compared to Unit 17 was a result of limited water availability during the growing season.

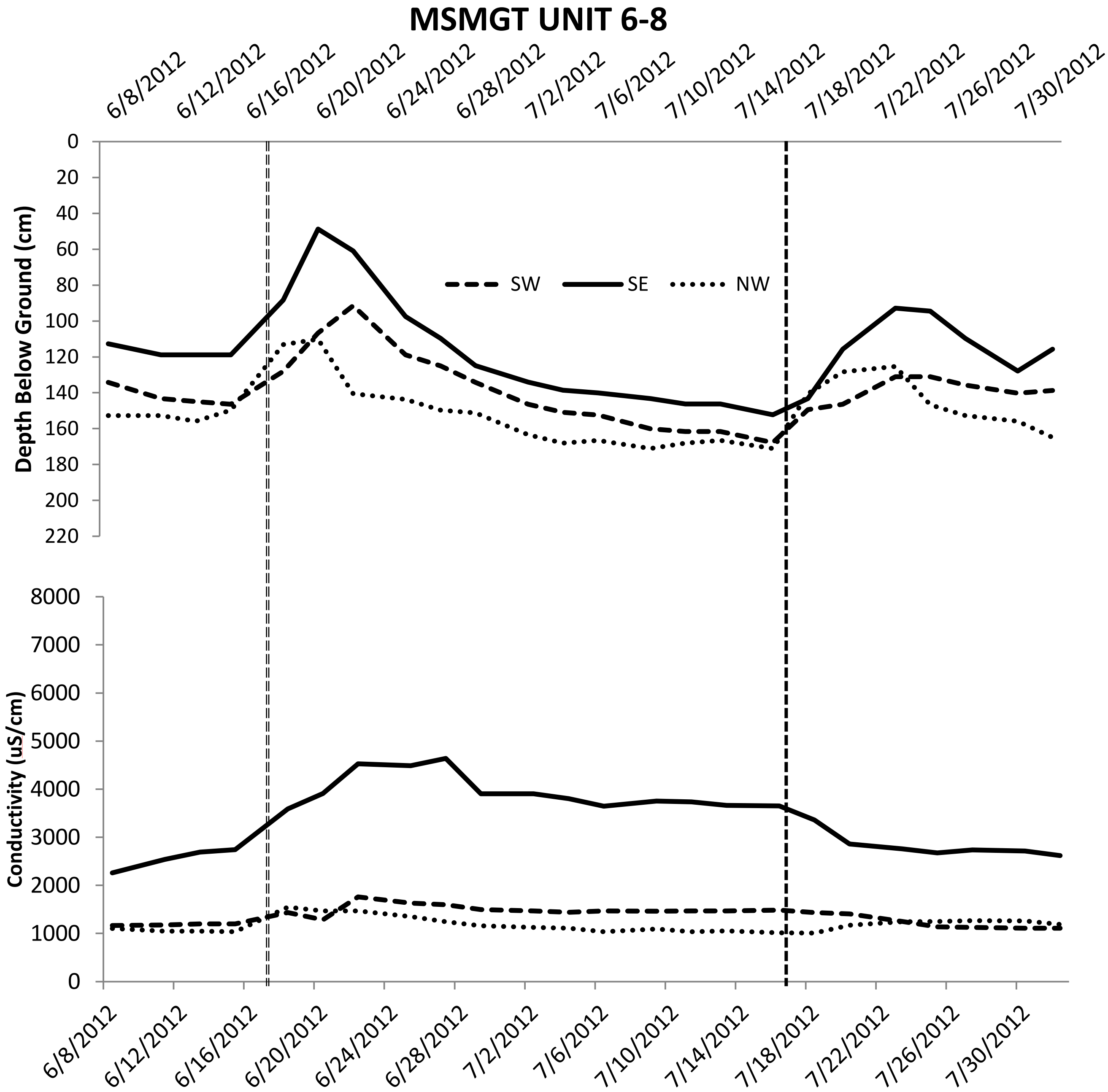

Figure 24- Changes in depth to groundwater (upper panel) and groundwater electrical conductivity (lower panel) in moist-soil management site 6-8 during the growing season at Bosque del Apache National Wildlife Refuge, 2012. Dashed vertical lines represent recorded flash flood irrigations. Corresponding patterned lines represent the depth to groundwater and groundwater electrical conductivity in response to irrigation events of individual groundwater monitoring wells located on the northwest (NW), southeast (SE), and southwest (SW) corners of the moist-soil unit. Fewer irrigation events in Unit 6 as compared to Unit 17 was a result of limited water availability during the growing season.

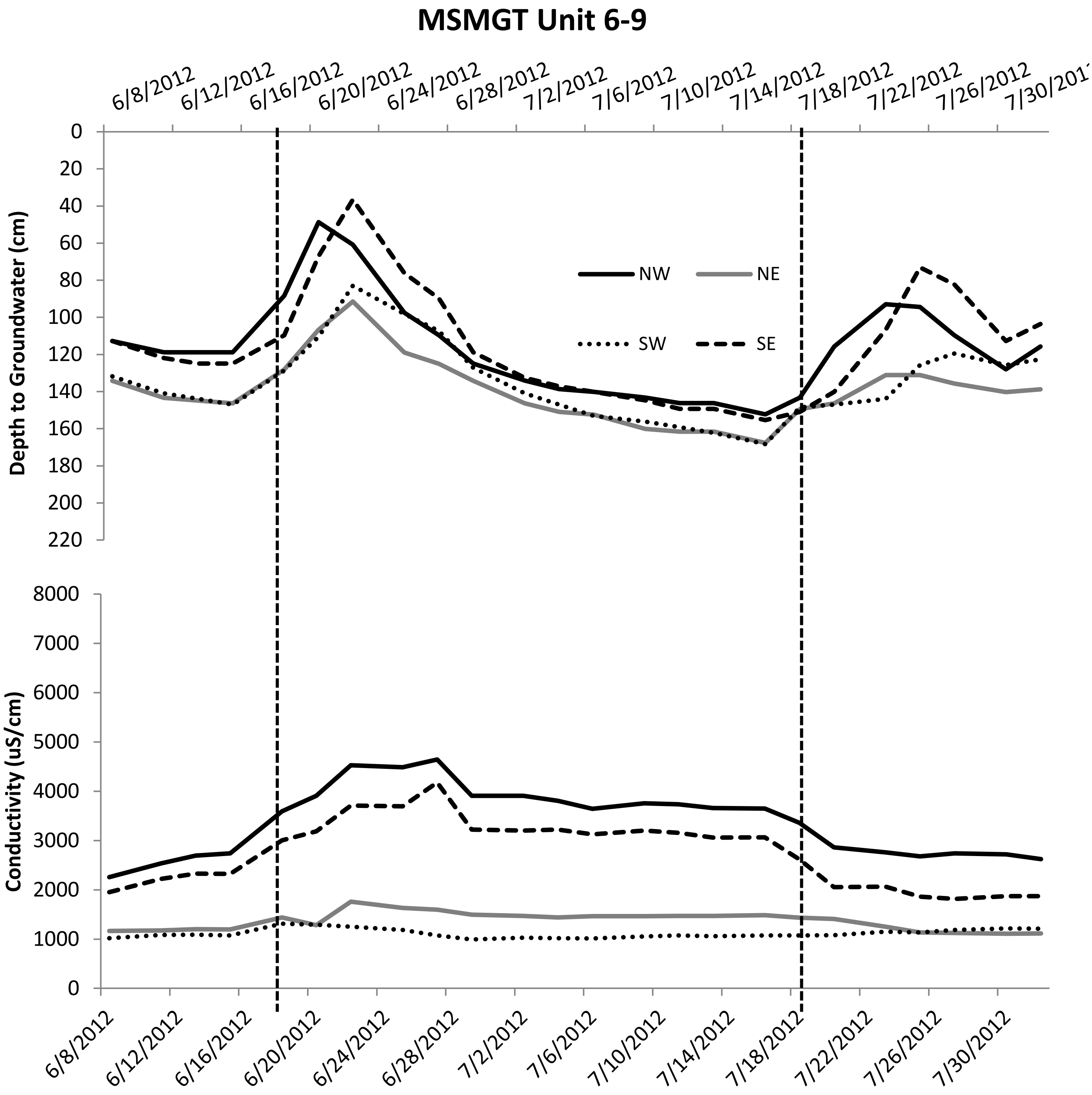

Figure 25 - Changes in depth to groundwater (upper panel) and groundwater electrical conductivity (lower panel) in moist-soil management site 6-9 during the growing season at Bosque del Apache National Wildlife Refuge, 2012. Dashed vertical lines represent recorded flash flood irrigations. Corresponding patterned lines represent the depth to groundwater and groundwater electrical conductivity in response to irrigation events of individual groundwater monitoring wells located on the northeast (NE), northwest (NW), southeast (SE), and southwest (SW) corners of the moist-soil unit. Fewer irrigation events in Unit 6 as compared to Unit 17 was a result of limited water availability during the growing season.

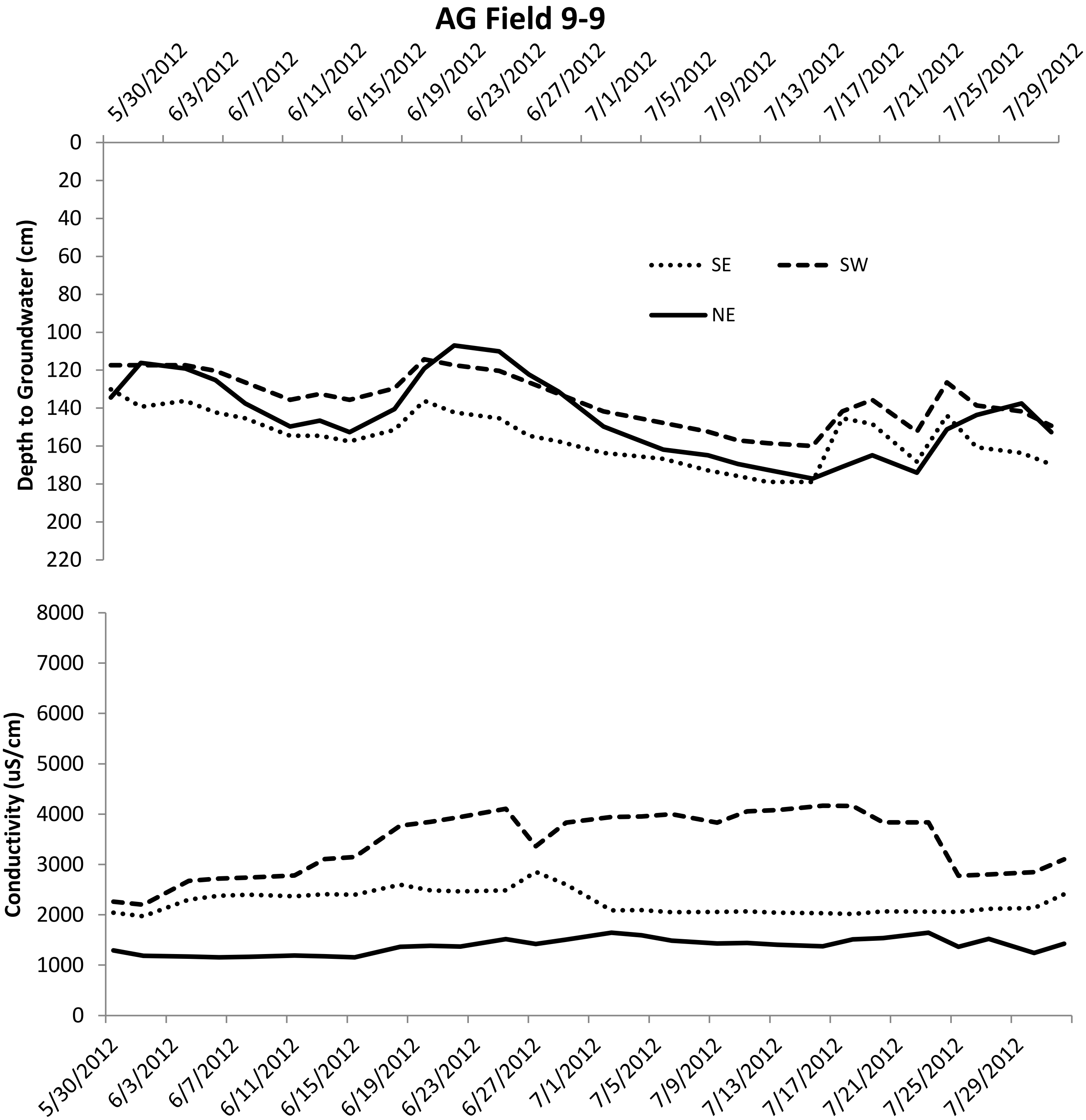

Figure 26 – Changes in depth to groundwater (upper panel) and groundwater electrical conductivity (lower panel) in agricultural field 9-9 during the growing season at Bosque del Apache National Wildlife Refuge, 2012. Data on the timing of irrigation events was not able to be collected; however irrigations did occur throughout the observed time period. Corresponding patterned lines represent the depth to groundwater and groundwater electrical conductivity of individual groundwater monitoring wells located on the northeast (NE), southeast (SE), and southwest (SW) corners of the moist-soil unit.

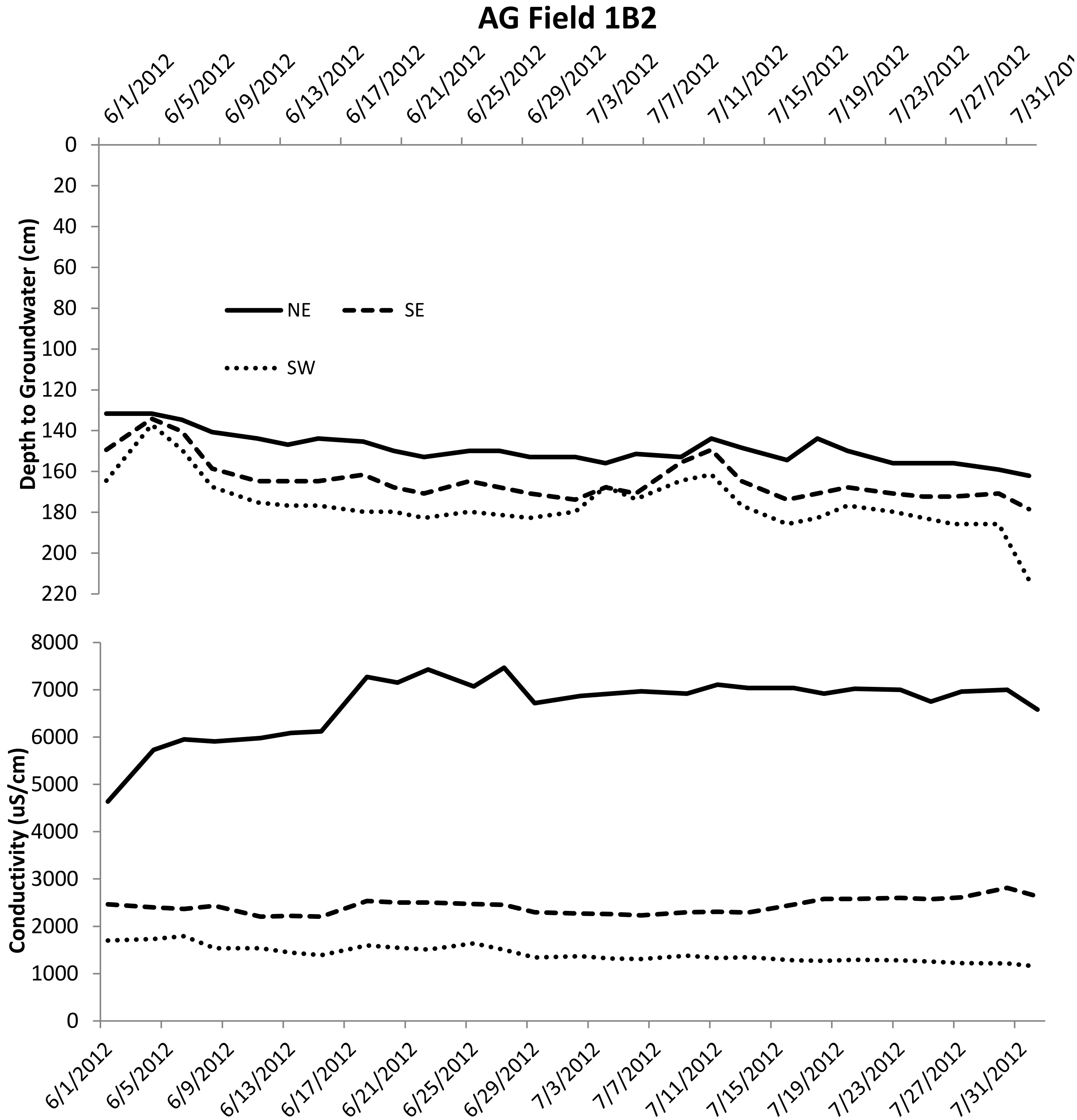

Figure 27 - Changes in depth to groundwater (upper panel) and groundwater electrical conductivity (lower panel) in agricultural field 1B2 during the growing season at Bosque del Apache National Wildlife Refuge, 2012. Data on the timing of irrigation events was not able to be collected; however irrigations did occur throughout the observed time period. Corresponding patterned lines represent the depth to groundwater and groundwater electrical conductivity of individual groundwater monitoring wells located on the northeast (NE), southeast (SE), and southwest (SW) corners of the moist-soil unit.

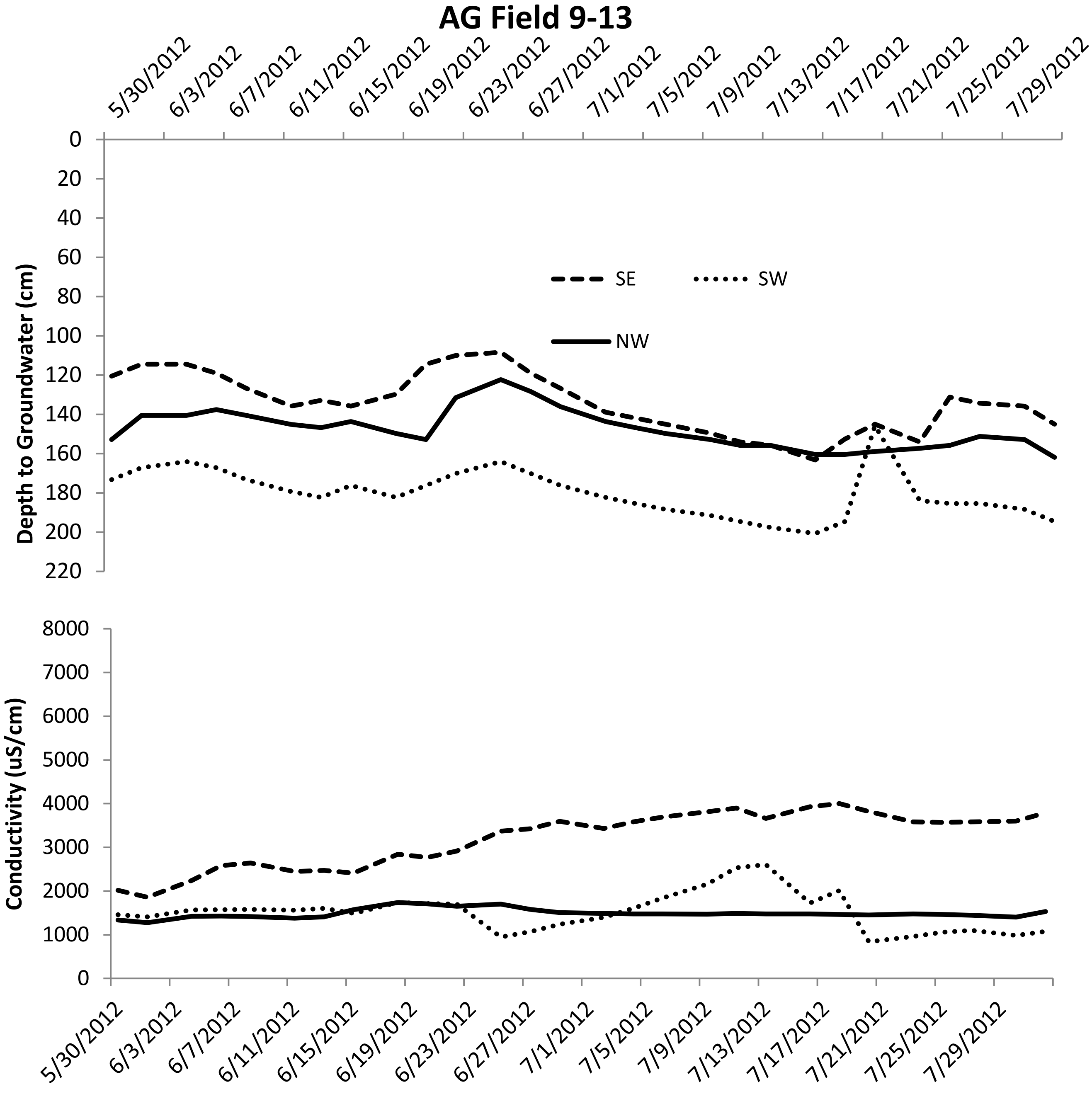

Figure 28 - Changes in depth to groundwater (upper panel) and groundwater electrical conductivity (lower panel) in agricultural field 9-13 during the growing season at Bosque del Apache National Wildlife Refuge, 2012. Data on the timing of irrigation events was not able to be collected; however irrigations did occur throughout the observed time period. Corresponding patterned lines represent the depth to groundwater and groundwater electrical conductivity of individual groundwater monitoring wells located on the northwest (NW), southeast (SE), and southwest (SW) corners of the moist-soil unit.

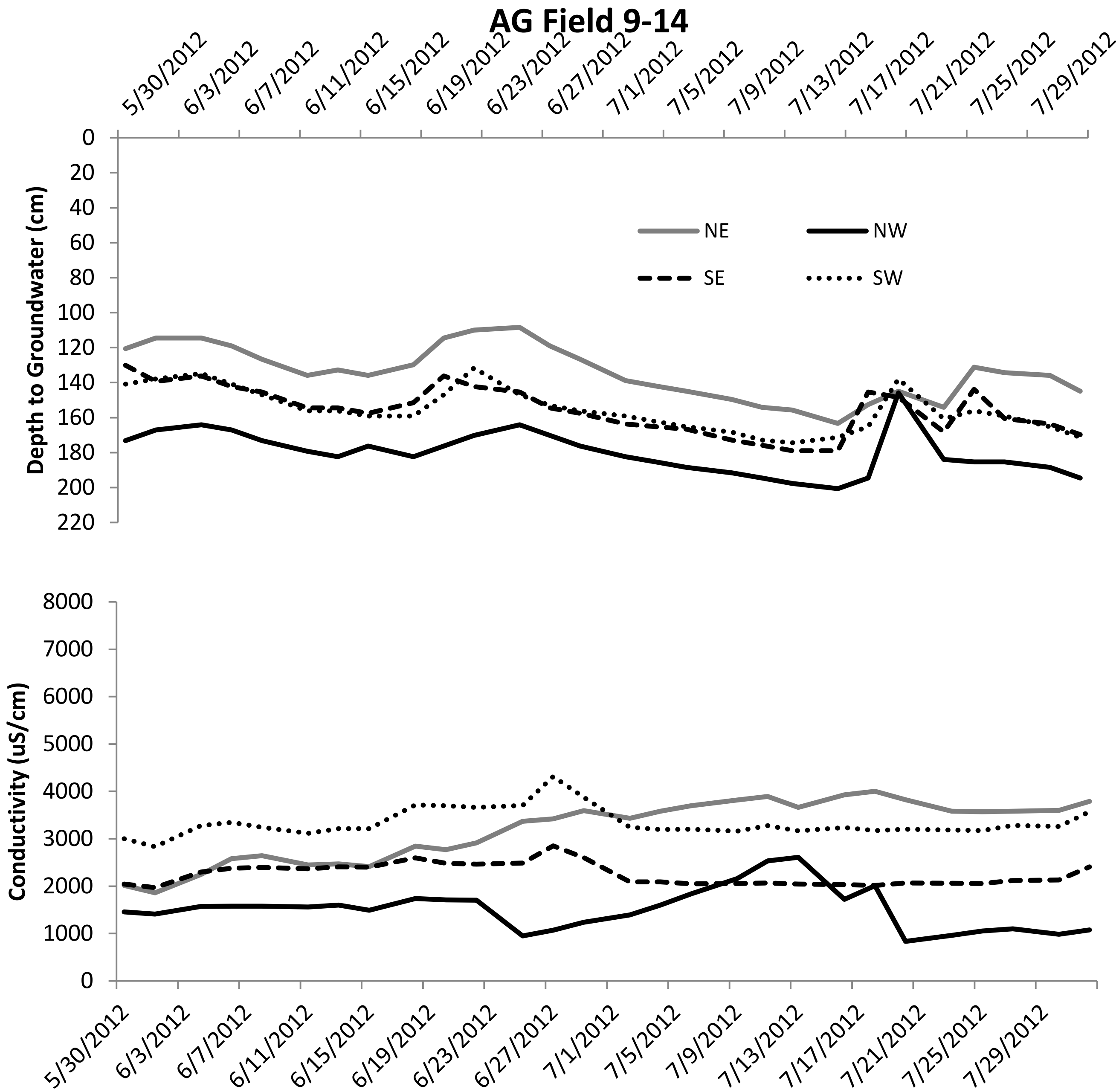

Figure 29- Changes in depth to groundwater (upper panel) and groundwater electrical conductivity (lower panel) in agricultural field 9-14 during the growing season at Bosque del Apache National Wildlife Refuge, 2012. Data on the timing of irrigation events was not able to be collected; however irrigations did occur throughout the observed time period. Corresponding patterned lines represent the depth to groundwater and groundwater electrical conductivity of individual groundwater monitoring wells located on the northeast (NE), northwest (NW), southeast (SE), and southwest (SW) corners of the moist-soil unit.

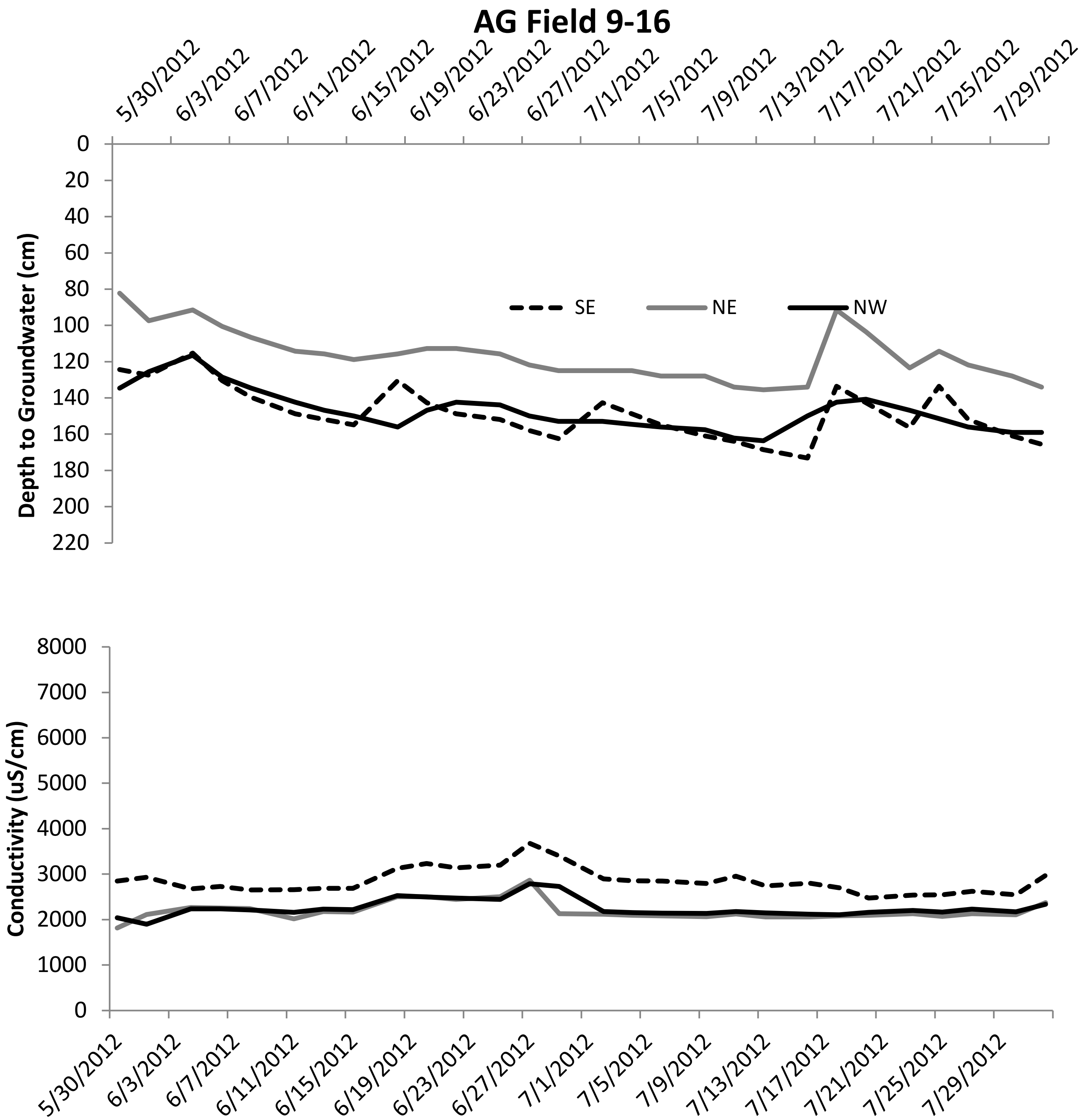

Figure 30- Changes in depth to groundwater (upper panel) and groundwater electrical conductivity (lower panel) in agricultural field 9-16 during the growing season at Bosque del Apache National Wildlife Refuge, 2012. Data on the timing of irrigation events was not able to be collected; however irrigations did occur throughout the observed time period. Corresponding patterned lines represent the depth to groundwater and groundwater electrical conductivity of individual groundwater monitoring wells located on the northeast (NE), northwest (NW), and southeast (SE) corners of the moist-soil unit.

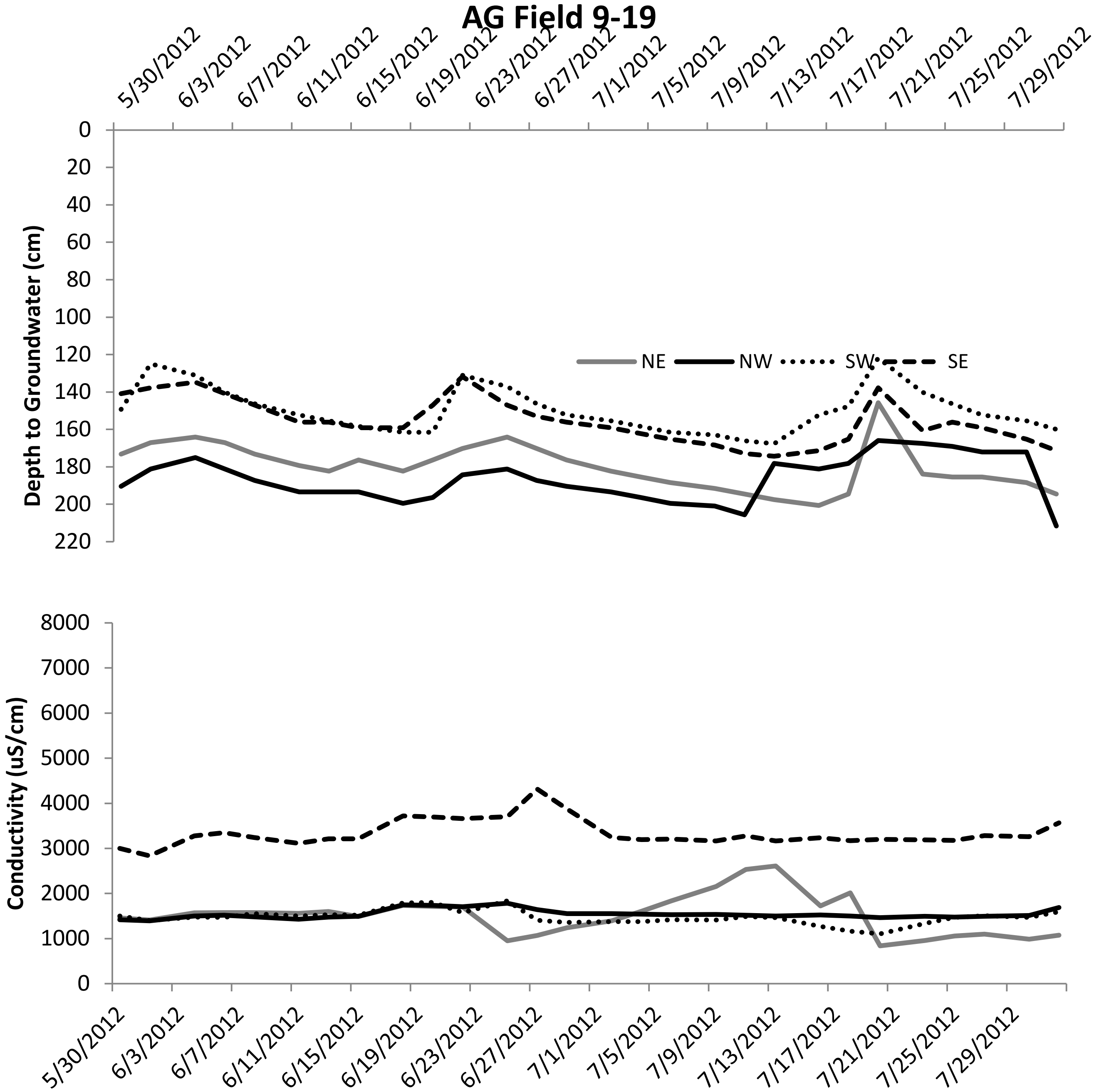

Figure 31 - Changes in depth to groundwater (upper panel) and groundwater electrical conductivity (lower panel) in agricultural field 9-19 during the growing season at Bosque del Apache National Wildlife Refuge, 2012. Data on the timing of irrigation events was not able to be collected; however irrigations did occur throughout the observed time period. Corresponding patterned lines represent the depth to groundwater and groundwater electrical conductivity of individual groundwater monitoring wells located on the northeast (NE), northwest (NW), southeast (SE), and southwest (SW) corners of the moist-soil unit.

2.6 Discussion

The results of this study indicate that moist-soil management practices and traditional agricultural production have different effects on the accumulation of soluble salts in soils of semi-arid floodplain environments. The interaction of surface and subsurface hydrologic regimes create complex soil salinity regimes. Differences in overall soil salinities and variability in depth to groundwater between treatments are likely related to differences in the volume and quality of water applied in irrigation, as well as the season of application. Moist-soil impoundments had lower overall soil salinities than did agricultural fields at the time of sampling. Summer irrigations in moist-soil management had a larger impact on variability in depth to groundwater than did agricultural irrigations. In moist-soil management, summer irrigation events coincided with a rise in groundwater level accompanied either by a temporary increase in groundwater electrical conductivity in initial irrigation events or a temporary decrease in groundwater electrical conductivity in subsequent later irrigations. This suggests that summer moist-soil management irrigations are recharge events that additionally serve as leaching mechanisms capable of flushing salts through the profile. In the case of my study, the conductivity of applied water (Figure 18) was lower than mean groundwater conductivity (Figure 19D). Therefore, it is likely that initial increases in groundwater conductivity are a result from salts flushed out of the soil profile and into the groundwater. However, subsequent irrigations tended to temporarily dilute groundwater conductivity as few soluble salts remain in the soil profile to be leached.

The seasonality and duration of flooding in hydrologic regimes likely plays a role in differences of soluble salt concentrations between treatments. Water used to irrigate moist-soil impoundments and agricultural fields varied in solute concentration throughout the year. Peaks in solute concentration occurred during the summer months when evapotranspiration and water demand was high and were lowest during winter months as a result of reduced evapotranspiration. Moist-soil impoundments additionally received a period of prolonged flooding during the winter with water containing lower

concentration of solutes (Figure 15 and 16). Previous studies have shown that ponding of water can be an effective tool to remove salts (Oster et al. 1984). Mean values in soluble salts analyzed from soil cores in moist-soil impoundments are likely a reflection of the effects of prolonged winter flooding. Salt concentrations measured prior to initial flood-up in moist-soil units showed no differences in concentration by depth throughout the entire profile. Additionally, concentrations were low enough to be considered non-saline (Chhabra 1996) and would be expected to have no biological impact on common wetland plants found in moist-soil management production.

In contrast, soils under agricultural production received irrigations only during those months when crops were cultivated (April-September). Hydrographs in agricultural fields revealed less variability and both mean depth to groundwater and groundwater conductivity were larger in agricultural fields (Figure 19). Because of growing season water limitations, agricultural managers try to minimize the amount of water that moves through the root zone and is reflected in common calculated leaching requirements (Corwin and Rhoades 2008). The limited leaching fractions, and the high solute concentrations of applied water, result in increasing concentrations of salt with depth in agricultural fields. As a result, applied water in agricultural irrigations is likely mostly evapotranspired and serves to deposit solutes rather than leach them.

Although solute deposition likely occurred in the agricultural fields, soil salinity concentrations, at least at the time measured, presented no limitations for the successful production of field corn. Salinity concentrations in the root zone (0-35 cm) prior to planting and pre-irrigation were below the threshold tolerance of 1.7 dS/m for field corn (Mass and Hoffman 1977). Low levels in the root zone could be explained by two potential possibilities. First, water solute concentrations used for irrigation at the refuge is relatively low compared to other studies (Amer 2010; Yazar et al. 2003) that experience loss of biomass production in field corn as a result of soil salinity. In a similar semi-arid floodplain used

for agricultural production Morway and Gates (2012) found that mean soil profiles had conductivities of 4.1 dS/m and 6.2 dS/m when irrigated with water at 1.3 dS/m and 3.0 dS/m, respectively. In my study, mean conductivity of applied water was .89 dS/m (±.036) (Figure 17). A second possibility that explains low salinities in my study was the amount of snowfall recorded during the 2011/2012 winter (Table 2). This snowfall likely served as an additional freshwater input that leached salts further down the profile.

While salinities were not adverse in the root zone, electrical conductivity increased with depth as sodium ions predominated soluble cations and approached sodic conditions at depths greater than 55 cm. Soils where sodium excessively outweighs the concentrations of magnesium and calcium are deemed *sodic* (Agassi et al., 1981). Sodicity can lead to negative effects on the soil structure as a result of clay deflocculation and thereby reduce soil air and water permeability (Rengasamy and Olsson, 1991). Sodic soils are defined by a Sodium Adsorption Ratio > 13, soil electrical conductivity < 4 dS/m, and a pH > 8.5; however it is important to note that deleterious effects can occur before these defined constructs (Chhabra, 1996). While profiles under moist-soil management had a relatively consistent SAR of 3 throughout the one meter section measured, lower portions (>55 cm) of agricultural profiles are closely approaching sodic conditions (Figure 12) because of low calcium and magnesium but high sodium. In turn, these conditions may contribute to further salinization through water logging of poorly permeable soils or enhanced capillarity of saline groundwater.

<u>2.61 Limitations and Trade-offs</u>

While the results from my study suggest that moist-soil management may have a greater capacity to flush salts from soils relative to agricultural management, limitations may exist as large quantities of applied surface water may result in salinization from a rising saline groundwater table. Little research has been done on salinity in moist-soil management, however, the integrated relationships among surface flooding, groundwater, and salt accumulation in moist-soil units are similar

to those observed in semi-arid natural wetlands (Jolly et al. 2008). Flood pulses in natural wetlands in semi-arid environments can recharge groundwater and flush salts stored in the soil into the groundwater (Cramer and Hobbs 2002). However, flood events may contribute to a rise in the groundwater table shallow enough to result in an upward flux of saline groundwater (Hutmacher et al. 1996). Crosbie et al. (2009) demonstrated that wetting and drying cycles in semi-arid floodplain wetlands can alter the function of the wetland from a recharge system to a discharge system, respectively. During flood periods, flooding results in recharge, but in non-flooded periods with high groundwater the wetlands function as discharge systems. Similar processes could occur in moist soil management. My data are insufficient to unequivocally document these processes in my study. However, results from Chapter 3 indicate that soil salinity concentrations were greater near the surface relative to concentrations at 1 meter deep in the profile as a result of summer flood irrigations. This suggests the possibility that moist-soil units may at some periods function as discharge systems and display an inverted soil profile as a result of capillary upward flux of saline groundwater.

Results in agricultural fields did not reveal elevated levels of salinity that could be detrimental to corn survival or loss in biomass. However, my selected time period of sampling may have underestimated the adverse effects of soil salinity on agricultural production. Samples were extracted in early May while fields were laser-leveled. After sampling, rows were created and then irrigated prior to planting. Anecdotal evidence suggest that salts brought in from applied water as well as salts pre-existing in the soil were mobilized and concentrated on the tops of the rows as a results of capillarity (Figure 32). These observations are consistent with previous literature (Bernstein et al. 1955; FAO 1988) that documents this effect in furrow irrigated agricultural systems. Elevated salinity levels prior to planting could subject seedlings to osmotic stress and incur injury during its most sensitive stage to soluble salts.

Figure 32 - Salt precipitates accumulating on the tops of rows before (left) and after (right) germination at Bosque del Apache National Wildlife Refuge, 2012.

While my study determined differences in salt concentrations among treatments as a result of differences in hydrologic regimes, it is important to note the variability that did occur within treatments of the same soil type. Within treatments, depth to groundwater and groundwater conductivity varied in magnitude among my installed groundwater wells. These variations can be most likely attributed to the alluvial floodplain environment in which my study was conducted. Floodplain soil environments are highly variable in nature due to the geomorphic processes from which they are derived (Jacobson et al. 2011). Historic depositional events and shifting meandering channels can create preferential pathways such as sand lenses for the movement of subsurface water (Makaske 2001) or impermeable clay layers. Therefore the effect of an implemented hydrologic regime is likely to have differential impacts on soil and groundwater salinity throughout the spatial landscape. In areas that have poor drainage, applied surface water may infiltrate slowly and contribute little to influencing groundwater or leaching. In contrast, the successive applications of large quantities of water on highly porous soils may serve to permanently raise the groundwater table and encourage alternative processes of salinization such as capillary upward flux. Therefore, management procedures based upon the findings of this research need to take into consideration site-specific characteristics.

2.7 Conclusions

Differences in the timing, volume, and quality of artificial hydrologic regimes influence the degree of salt accumulation in semi-arid environments. Soils under moist-soil management appear to be better regulated in their accumulation of soluble salts. Inundation during the winter, when applied water has its lowest annual concentration of solutes, enables a large portion of salts to be removed from the soil prior to the growing season. Flash floods in the summer growing season tend to serve as leaching and recharge events that may keep soil salinity accumulation to a minimum. In contrast, soils under long term agricultural production seem to lack a fraction of water capable of moving salts out of the profile and this has led to the greater accumulation of salts, particularly in the lower portions of the profile (>55 cm) that have sodic like conditions. While salinity levels measured in the root zone (0-35 cm) of agricultural profiles were below salt tolerance thresholds for field corn, over winter flooding may be a technique utilized if root zone salinities are high. While the remediation of sodic soils likely requires the addition of chemical amendments such as gypsum, the incorporation of a seasonal leaching fraction similar to that found in moist-soil management may be a solution that discourages the further accumulation of soluble salts and soil degradation. Careful management of these soils is recommended as a tradeoff exists between the repetitive flushing of salts into the groundwater that can lead to water table rise and enhanced salinization via capillary rise versus the buildup of high levels of salts in the profile that could jeopardize the success of desired crops and wetland vegetation.

CHAPTER 3
EVALUATING THE EFFECT OF ROTOTILLAGE AND FREQUENCY OF IRRIGATION ON SURFACE SALINITIES OF MOIST-SOIL MANAGEMENT IN A SEMI-ARID ENVIRONMENT

3.1 Introduction

Moist-soil management (MSMGT) can be defined as the manipulation of soils, hydrology, and vegetation to create mudflats for the germination and production of high energy wetland plants in provision to meet the energetic demands of waterbirds (Fredrickson and Taylor 1982; Haukos and Smith 1993). In response to continued wetland loss (Dahl 2000), moist-soil management has become a common practice to re-establish waterbird habitat along migratory flyways of North America. Studies of moist-soil management practices have focused on the ability to maximize seed production for waterbirds (Fredrickson and Taylor 1982; Kross et al. 2008). Naylor (2002) evaluated the effects of drawdown date, drawdown rate, summer irrigation, and soil tillage on moist-soil seed production and found that while all four variables influences seed production; summer irrigation and soil tillage had the greatest positive impact. These management practices are common amongst moist-soil managers along the Mississippi flyway (Reinecke and Loesch 1996) as well as in more semi-arid environments along the Central (Taylor and Smith 2005) and Pacific (Naylor et al. 2005) flyways.

However, moist-soil management in semi-arid environments differ from moist-soil management in more humid settings in that higher rates of evapotranspiration can increase levels of soil salinity and may have a negative impact on plant productivity by lowering the osmotic water potential within the soil. This can result in drought like symptoms leading to plant stunting and reductions in overall biomass (Mass and Hoffman 1983). Soil salinization can occur as both a natural process (Domingo et al. 2001; Rengasamy 2006) or through human modifications, (e.g., secondary salinization; Chhabra 1996; Qadir 2004) but in general, salinization occurs in the presence of soluble salts, a shallow water table that promotes capillary wicking (Northey 2006), and evapotranspiration demands that exceed fresh water inputs (Metternicht and Zinck 2009).

Soil salinization in semi-arid and arid environments has been heavily researched in the context of agricultural production (U.S Salinity Laboratory Staff 1954; Morway and Gates 2008) and is considered one of the primary management concerns in soils under intensive agricultural practices (UNEP 1991). In many cases the quality and quantity of water applied has been a focus point for managing sustainable agricultural practices to allow for adequate flushing of salts from the root zone while minimizing the contribution of excess waters that might contribute to a rise in water table levels and capillary upward flux of saline groundwater (Oster et al. 1984; Qadir 2004; Corwin and Rhoades 2007). However, the active management of non-agricultural lands in semi-arid environments has received less attention and little is known on how moist-soil management practices initiated to promote desired wetland plants affect soil salinity and subsequently, plant productivity.

Rototilling is a successful tool for setting back wetland plant succession and creating a soil surface that is suitable for the germination of annual wetland plants. However, rototilling has not been evaluated in moist-soil units for its effect on influencing the movement of salts within the root zone. When tillage disturbances occur, modifications of soil structure can increase or decrease hydraulic conductivity and permeability for water as well as solute transport in soils (El Titi 2003). In semi-arid and arid environments, one source of potential salinization occurs in the presence of shallow groundwater table that encourage the upward flux of saline water. In moist-soil impoundments with shallow groundwater tables, this upward flux may be affected by rototilling. Previous research has shown that tillage can serve to disconnect soil micropores and restrict the capillary rise of saline water into the tilled zone. On a silt loam used for rice production, Wilson et al. (2000) demonstrated that no-till soils had higher salt concentrations in the root zone than did three variations of tillage. This suggests that conservative tillage practices paired with proper water management may be beneficial for stimulating the germination of wetland plants in low to non-saline conditions.

In many semi-arid environments where moist-soil management occurs, water availability may be limited and efforts to conserve freshwater inputs are a primary concern. However, it is unknown how more conservative irrigation regimes might affect plant production. Increasing the time in between irrigation events may allow for the greater concentration of salts and increase osmotic stress (Chhabra 1996). Therefore there is a need to determine irrigation frequencies that conserve water yet does not negatively influence plant productivity by increasing osmotic stress.

The objectives of this study are to 1) evaluate the effects of rototilling on surface soil salinities in moist-soil units and to 2) evaluating the effects of frequency of irrigation on plant productivity in moist-soil units.

3.2 Study Site and Management Practices

3.21 Location

The Bosque del Apache National Wildlife Refuge (33° 48", 106º 53) is located south of San Antonio, New Mexico and lies within the Middle Rio Grande Basin, straddling the Rio Grande River. The river itself is a consequence of a continental rift valley dating back to the Paleogene (Crawford et al. 1993) and originates in the mountains of the Rio Grande National Forest in southern Colorado and flows south into New Mexico. The basin is bounded by mountain ranges rising 2,000 m to the west and 1,600 m to the east while the valley floor elevations average 1, 470 m (Crawford et al. 1993).

Regional climatic conditions are characterized by high light intensity, low relative humidity, and an average Class A pan evaporation of 250 cm per year (Johnson 1988). The annual average precipitation is approximately 23 cm, primarily occurring during the months of July through October (WRCC 2005).

Soils within the Rio Grande basin are derived from alluvial and clastic sediments (Crawford et al. 1993). For the purpose of my research, I limited my study sites to those that were mapped as Gila clay

loam soil series to isolate the potential variability of soil salinity between different soil series. The Gila series is described as Coarse-loamy, mixed, superactive, calcareous, thermic Typic Torrifluvents (Soil Survey Staff, USDA).

The refuge is divided into a series of management units. Within each unit are numerous fields that are managed similarly for either agricultural production or moist-soil management. I selected fields within 1 unit for intensive study. Unit 17 (Figure 33) was originally used for corn and alfalfa production, however, a high water table encouraged the upward flux of saline water and caused increased soil salinities that made corn production difficult. Managers installed tile drains in an attempt to keep the water table stable at a depth that would decrease salinization but the tile drains soon filled with silt and were ineffective. In 1993 agricultural production was abandoned and unit 17 was converted to moist-soil production (John Vradenburg, USFWS, personal communication).

Irrigation water used for unit 17 is diverted from the Rio Grande at San Acacia, New Mexico approximately 40 km north of the refuge and is delivered to moist-soil units through a complex system of irrigation canals and drains. Fields under moist-soil management are impounded structures and are served with an interior feeder canal and feeder drain that provides independent field irrigation capability. Interior feeder canals lead to a single inflow gate and an equivalent outflow gate controls water that leaves the unit via the feeder drain.

3.22 Moist-soil Management Practices

Annual moist-soil management begins in late April or early May with a shallow (20 cm) flood that is sustained for three days. On the fourth day, water is slowly released from the impoundment as stop logs are removed individually from the outflow gate over a period of three days. This slow draw down of the water level creates a mudflat that stimulates the germination of annual wetland vegetation. Impoundments are subsequently flash flooded throughout the summer approximately every 14 days or

as determined needed by managers. These flash floods inundate the units to 20 cm; as soon as this is accomplished all stop logs are removed from the gates to maximize drainage. Irrigations cease once vegetation reaches maturity and senesces in mid-August to mid-September. Wintering and migratory waterbirds can arrive as early as September, peak in mid to late December and remain until February; initial fall/winter flood dates and flood duration of moist soil impoundments varies among units to facilitate the provision of moist soil foods for waterbirds throughout the winter. Thus, fall and winter flooding of impoundments can last 2 weeks to 2 months depending on the size of the impoundment and use by waterbirds. Impoundments are usually drained in the January- March and left fallow until they are re-flooded in late April and early May to begin the next moist soil cycle. However, continued use of this cycle eventually encourages the propagation of perennial and woody vegetation that has lower food values for waterbirds. As a result, about every four years managers use sustained flooding, fire, and/or soil tillage to disturb the units to set back perennial vegetation and promote the production of desired annuals. Although the frequency of disturbance varies depending on vegetation response, soil tillage is the most commonly used disturbance and its variations include heavy and light disking and rototillage.

3.3 Methods

This study took place in the summer of 2011 and 2012. In 2011, six independent moist-soil impoundments were selected from unit 17 (Figure 33). Prior to this study moist-soil impoundments selected had not received a soil disturbance in at least two years. A split-plot experiment was designed with irrigation frequency (9 or 14 day frequency) as the main plot and tillage regime (rototillage or no-tillage) as the sub-plot. Independent impoundments were randomly assigned a 9 or 14 day irrigation and the

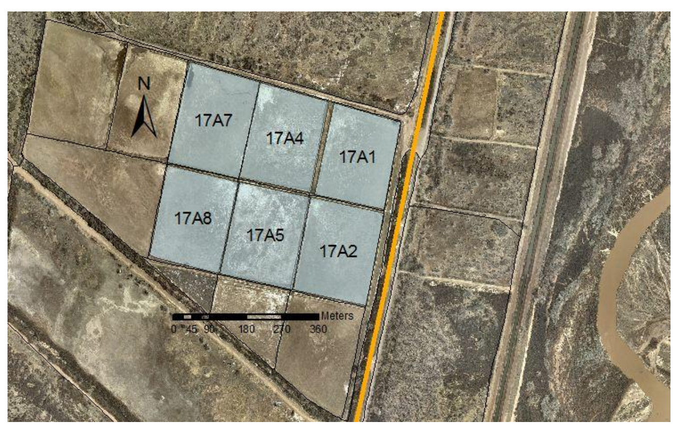

Figure 33 - Six selected moist-soil impoundments in Unit 17 at Bosque del Apache NWR, May 2011. Highlighted yellow line denotes the interior feeder canal that supplies Unit 17 with irrigation water from the Rio Grande River.

east or west side of the impoundment was randomly selected for the rototillage treatment. Each main and sub-plot treatment was replicated three times (Figure 34).

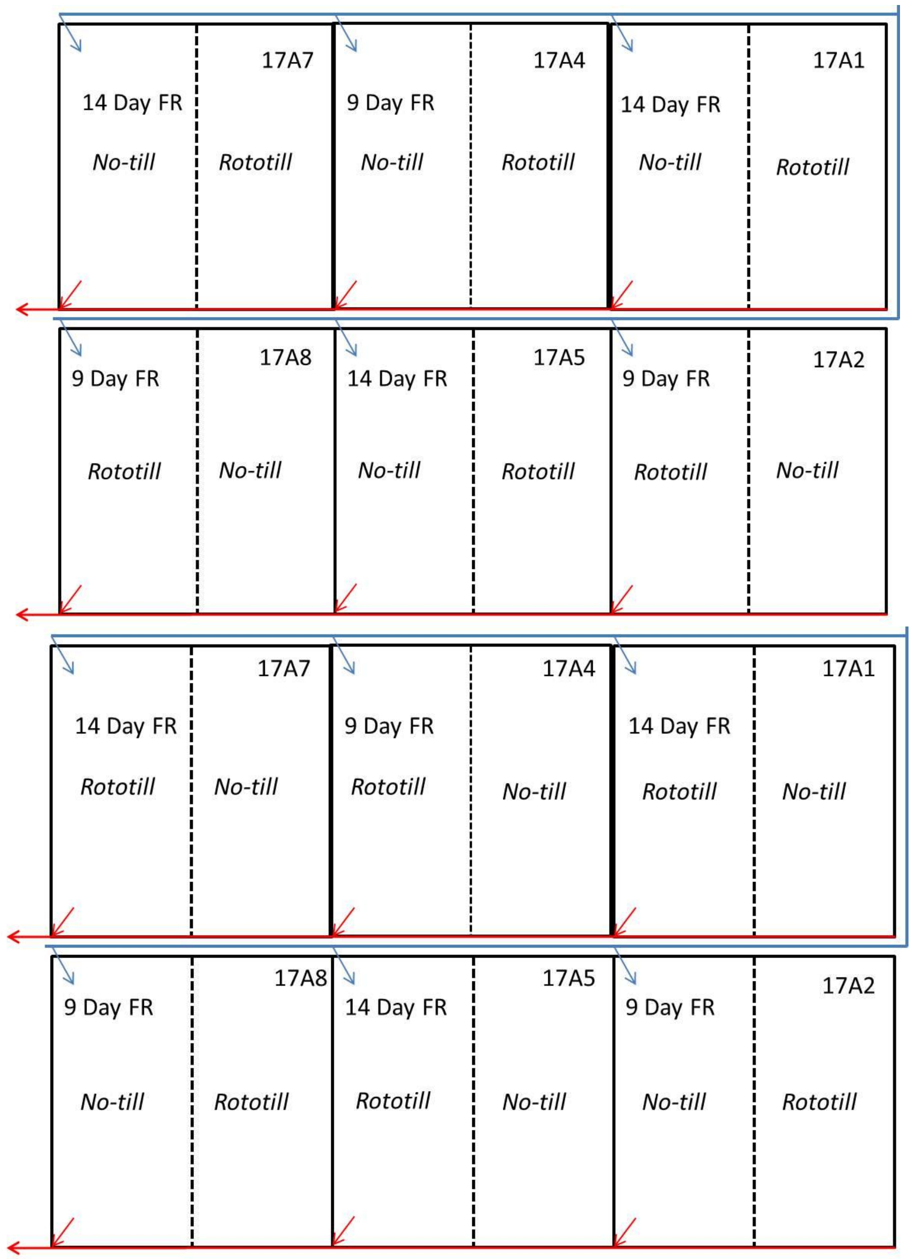

Figure 34 – Randomly selected treatment assignments of irrigation frequency (main plot) and tillage type (sub-plot) for moist-soil units in 2011 (top) and 2012 (bottom). Blue lines represent the irrigation canals that feed water into the top of moist-soil unit via the blue arrows. Red arrows exiting the impoundments represent drainage gates to remove water into drainage ditches (red line). Only sub-plot treatments were switched in individual moist-soil impoundments in between years. Irrigation main plot treatments remained the same.

In 2011, soil rototillage took place during 9 May to 11 May to an approximate depth of 15 cm. Subsequently, all moist-soil impoundments were flooded on 12 May and 13 May and were inundated for three days. The fourth day initiated the slow draining of the impoundments over the course of three days.

After impoundments were determined dry enough to support agricultural machinery (approximately 7 days after water was completely removed from the impoundment), the Geonics EM-38 (Geonics Inc, Canada) was used to non-invasively measure apparent bulk soil electrical conductivity (EC_a)(Diaz and Herrero 1992). The EM-38 is an electromagnetic induction instrument with a transmitting and receiving coil at opposite ends of the instrument (Figure 35). The transmitting coils uses alternating current to create a primary magnetic field in the soil. This magnetic field induces currents in the soil, which generates a secondary magnetic field (Sudduth et al. 2002). The receiving coil responds to both the primary and secondary magnetic field and the ratio between the two is a linear function of conductivity (McNeil 1992). Bulk apparent conductivity is influenced by soil temperature, soil moisture, percent clay concentration, cation exchange capacity, and levels of soluble salts (Rhoades et al. 1999). Williams and Baker (1982) found that in salt affected soils, the largest variation in measurements was attributed to differences in concentrations of soluble salts. Software developments have created the ability to connect the EM-38 to mobile data loggers with GPS capabilities allowing for geo-referenced conductivity values (Geonics Inc.).

The EM-38 was field calibrated (Geonics Limited 2003) and connected to a geo-referenced data logger DAS70-AR Data Acquisition System (Archer Field PC). The EM-38 and data logger were attached to a homemade non-magnetic sled (Figure 35) pulled behind an agricultural tractor and surveyed across each treatment in approximately 10 m spaced transects that ran north to south.

Figure 35 – Preparing the EM-38 for a bulk apparent conductivity survey at Bosque del Apache NWR, August 2011. The EM-38 was placed in a homemade non-magnetic sled to eliminate interferences with the instrument.

At the beginning of each survey, soil temperature was measured using a digital thermometer at 23 cm deep. Regression was used to standardize conductivity readings to 25 C with the following equation:

$$EC_{25} = ECa\ (0.4779+1.3801e^{(-T/25.64)})$$

where EC_{25} = temperature standardized at T = temperature measured at 23 cm (Reddy and Scanlon 2003).

After surveys were completed, data were uploaded into the ESAP-RSSD software program (USDA Riverside Salinity Laboratory 2000) which is a statistical program that generates optimal soil sampling designs representative of the differences in conductivity from the bulk apparent electrical conductivity survey information (Lesch et al. 2000). Within each impoundment, twelve geo-referenced conductivity sites for each treatment were selected from the total EM-38 survey for soil sampling (Figure 36). Soil samples were collected either on the same day as the survey or the day after. At each selected site, soil samples were extracted at three depths (0-10cm, 20-30 cm, and 80-90 cm) and stored individually in sealed plastic bags.

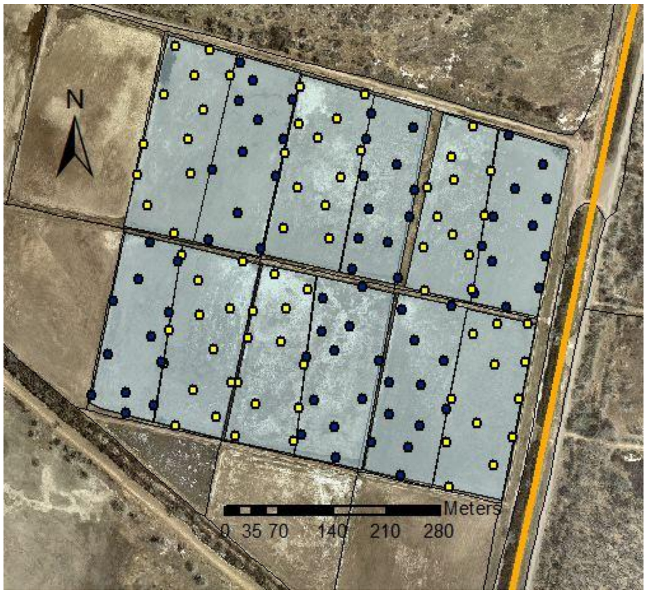

Figure 36- Soil sampling design derived from EM-38 apparent bulk soil conductivity (EC_a) survey with the ESSAP-RSSD software at Bosque del Apache NWR, May 2011. Blue dots represent soil collected in no-till treatments while yellow dots represent soils collected in no-till treatments.

Each soil sample was measured for gravimetric soil moisture (Dane and Topp 2002), particle size distribution by hydrometer (Gee and Bauder 1986), and 1:1 (soil to water) electrical conductivity using a temperature compensating electrode standardized to 25 C. 1:1 electrical conductivities ($EC_{1:1}$) were then converted to estimate saturated paste extract electrical conductivities (EC_{SE}) using the regression equation derived by Hogg and Henry (1984) for each soil textural class (Table 9).

Table 9 - Regression equations for converting $EC_{1:1}$ to EC_{SP} calculated by Hogg and Henry (1984) where EC_{SP} is equal to the estimated electrical conductivity of the saturated paste extract and $EC_{1:1}$ is the measured electrical conductivity of the 1:1 soil to water extract.

Soil Texture	Regression Equation
Coarse (Sand –Loamy Sand)	EC_{SP} =3.01 ($EC_{1:1}$) -0.06 [1]
Medium (Sandy loam – Silty Clay Loam)	EC_{SP} =3.01 ($EC_{1:2}$) – 0.77 [2]
Fine (Sandy Clay – Clay)	EC_{SP} =2.66 ($EC_{1:2}$) + 0.97 [3]

Following sampling, impoundments were irrigated according to their assigned frequency. As part of standard moist-soil management practices at the refuge, all impoundments in the study were

mowed during the first week of July and immediately flooded for three days to reduce the growth of cocklebur (*Xanthium strumarium*). After three days of flooding, impoundments were drained and experimental irrigation frequencies resumed.

Irrigations ceased in early August 2011, once plants began to senesce. Following the last irrigation, each impoundment was surveyed again using the EM-38 and a new sampling design was created to assess soil salinity. Soil sampling was identical to methods described previously.

The experiment was replicated in 2012. All independent impoundments maintained their assigned main plot irrigation frequencies, however, sub-plot tillage regimes were reversed in 2012 (e.g., if the west of an impoundment was tilled in 2011, in 2012 in was not disturbed while the east side was rototilled) (Figure 34). Weekly grab samples of irrigation water were collected in 2012 and measured for electrical conductivity (EC_{IW}) using a temperature compensating electrode. An EM-38 survey occurred after the initial summer flood up and the same procedures listed above were used to generate a sampling design and collect soil samples. A second EM-38 survey at the end of the growing season did not occur due to time constraints, however, soil samples were re-collected based upon the sampling design generated from the first 2012 survey.

In August 2012, three sites were randomly selected in each sub-plot treatment within individual moist-soil impoundments to survey plant biomass. A .25 m^2 PVC quadrat was arbitrarily thrown into the air at the selected site. Upon landing all above-ground vegetation within the quadrat was clipped with pruning shears and placed into a paper sack. Each sack was subsequently dried at 105 C for 24 hours in a drying oven. After drying, vegetation within each bagged was sorted by species, dried again for 24 hours and then weighed.

3.4 Statistical Analyses

A split-plot Analysis of Variance was performed in SAS 9.3 (SAS Institute Inc. 2011) on the differences in bulk apparent electrical conductivity between treatments based upon the twelve conductivity sites generated by the ESAP-RSSD program for the May 2011 and August 2011 survey. Irrigation frequency, tillage regime, and time of survey were treated as fixed effects using the PROC GLIMMIX model:

Conductivity = (Irrigation) (Tillage) (Visit) (Irrigation (Tillage)) (Visit)*(Irrigation (Tillage))

To determine if there were differences between years in initial conductivity, the June 2012 (1^{st} survey) survey was compared to the May 2011 survey using the PROC GLIMMIX model:

Conductivity = (Irrigation) (Tillage) (Year) (Irrigation (Tillage)) (Year)*(Irrigation (Tillage))

EC_{SP} and gravimetric soil moisture from soil samples collected in 2011 and 2012 were evaluated for differences among treatment types and between sampling periods. The PROC GLIMMIX models were:

EC_{SP} = (Visit) (Irrigation) (Tillage) (Irrigation (Tillage)) (Depth) (Depth)*(Irrigation (Tillage)) (Depth)*(Visit)*(Irrigation (treatment))

and

Soil Moisture = (Visit) (Irrigation) (Tillage) (Irrigation (Tillage)) (Depth) (Depth)*(Irrigation (Tillage)) (Depth)*(Visit)*(Irrigation (treatment))

In both models, year (2011 and 2012) was treated as a random effect. Akaike's Information Criterion (AIC) was used to evaluate the best fit distribution. A post-hoc test was performed using a Tukey's adjustment to identify specific treatment differences. An analysis of variance was used to evaluate differences of total dry-weight plant biomass between treatments. The PROC GLIMMIX model was used:

Dry Weight = (Irrigation) (Tillage) (Irrigation (Tillage))

3.5 Results

Analysis of Variance revealed that means in bulk apparent conductivity (Figure 37) did not differ in any treatment between visits in 2011 (p = *0.7521;* Table 10). Comparisons of mean bulk apparent conductivity between initial surveys in 2011 and 2012 (Figure 38) indicated that densities did not differ (P= 0.0582; Table 11).

Mean values in EC_{SP} across all treatments sampled after the initial flood-up (May) ranged from 2.04 – 2.23 dS/m at the .05 m depth, 1.65 -1.79 dS/m at the .25 m depth, and 1.43-1.75 dS/m at the .86 m depth. EC_{SP} measured at the end of the growing season (August) ranged from 2.01 -2.54 dS/m at the .05 m depth, 1.65 -1.90 dS/m at the .25 m depth, and 1.38 – 1.59 dS/m at the .86 m depth across all treatment types (Figures 39-41).

Type 3 Test of Fixed Effects revealed that EC_{SP} was influenced by tillage treatment ($P \leq .0001$ and depth (P = *0.0041)* but not by irrigation frequency (P = *0.8766;* Table 12). However differences occurred only in soil rototillage paired with a 9 day irrigation frequency with lower EC_{SP} at the .86 m depth than the .05 and .025 depth in both the May and August sampling periods (Figures 39 and 40).

At the end of the growing season (August sampling) EC_{SP} was greater in the .05 m depth than the .25 m by .25 dS/m in rototilled soil under a 14 day irrigation frequency (Figure 40). In the same treatment EC_{sp} was greater in August than at sampling in the beginning of the season (May) (Figure 39).

Mean values in gravimetric soil moisture (GSM) across all treatment sampled after the initial flood-up ranged from 18.20-24.54% at the .05 m depth, 24.38-25.66% at the .25 m depth, and 31.68-33.33% at the .86 m depth. GSM measured at the end of the growing season ranged from 18.60-24.93%

at the .05m depth, 22.18-24.73% at the .25 m depth, and 29.14-31.98% at the. 86 m depth (Figures 42-44).

Type 3 Test of Fixed Effects revealed that GSM was influenced by tillage treatment (P = .0001), depth (P < .0001), and time of visit (P = .0001) but not by irrigation frequency (P = *0.948*; Table 13). Depth was the only fixed effect that influenced GSM across all treatments (Figure 42-44). GSM was greater at the .86 m depth than at shallower depths (P < .0001). During the May sampling period, GSM was greater in rototilled treatments than no-tilled treatments at the .05 m depth under a 14 day irrigation frequency (Figure 42).

Type 3 Test of Fixed Effects revealed that dry-weight plant biomass was influenced by tillage type but not irrigation frequency (Table 14). Mean values in total dry-weight biomass in rototilled soils under a 9 day irrigation frequency was 103.75 g (±12.83), versus 80.08 g (±5.02) in no-till soils. Dry-weight biomass in rototilled soils under a 14 day irrigation frequency was 87.21 g (±39.27), versus 63.12 g (±26.46) in no-till soils (Figure 45). Means and standard errors for the top three dominant species found in all treatments are found in Table 15 and Figure 46.

Table 10 – Type 3 Test of Fixed Effects of Analysis of Variance for bulk apparent electrical conductivity (EC_a) in moist-soil impoundments between visit in treatment types in 2011 at Bosque del Apache NWR. Effects were considered different when P ≤ 0.05.

Type III Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
irrigation	1	279	1.05	0.3067
tillage	1	279	0.02	0.8871
visit	1	279	0.10	0.7521
irrigation(tillage)	1	279	1.11	0.2927
visit*irrigation(tillage)	3	279	0.21	0.8898

Table 11 - Type 3 Test of Fixed Effects of Analysis of Variance between initial EM-38 surveys in 2011 and 2012 for bulk apparent electrical conductivity (EC_a) in moist-soil impoundments at Bosque del Apache NWR. Effects were considered different when P ≤ 0.05.

Type III Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
irrigation	1	207	0.02	0.8803
tillage	1	207	0.09	0.7643
year	1	207	3.63	0.0582
irrigation(tillage)	1	207	0.19	0.6649
year*irrigation(tillage)	3	207	0.80	0.4952

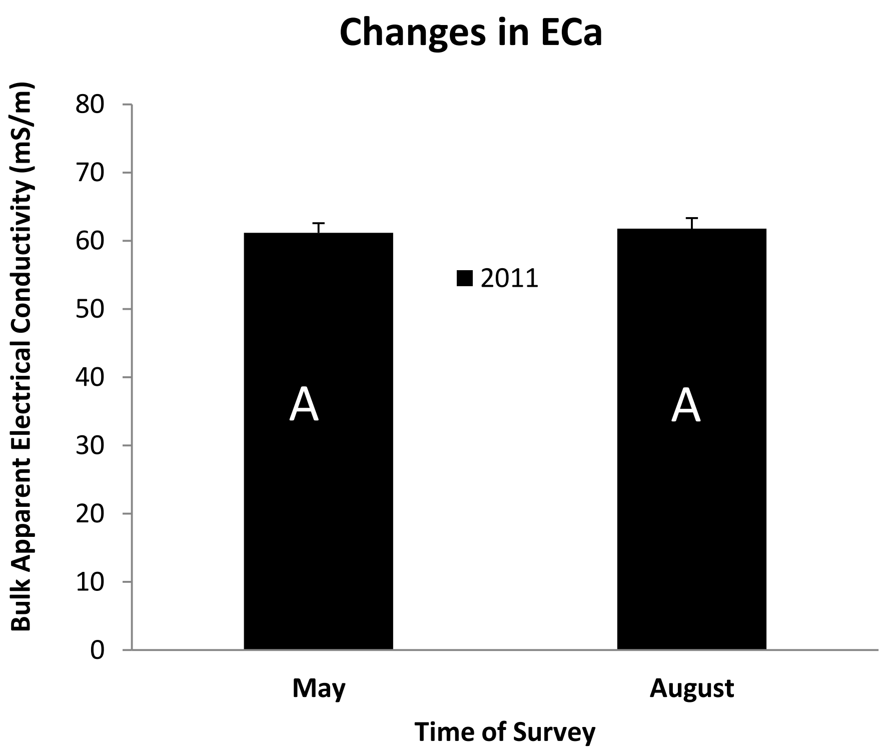

Figure 37 – Mean values in bulk apparent electrical conductivity between two survey periods in moist-soil impoundments at Bosque del Apache NWR, 2011. Means sharing a letter do not differ (P > 0.05).

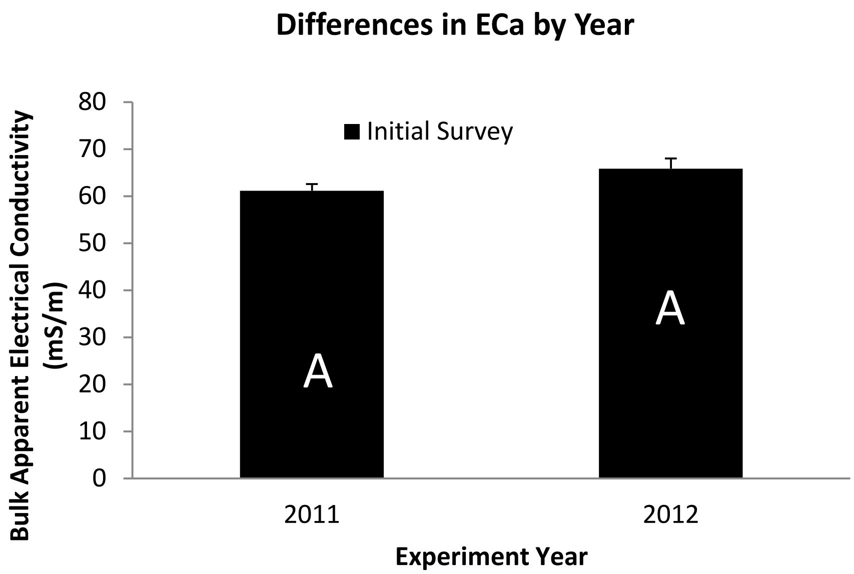

Figure 38 – Mean values in bulk apparent electrical conductivity between survey after initial summer flood-ups in moist-soil impoundments at Bosque del Apache NWR in 2011 and 2012. Means sharing a letter do not differ ($P > 0.05$).

Table 12 - Type 3 Tests of Fixed Effects for Analysis of Variance in soil saturated paste electrical conductivity (EC_{SP}) collected in 2011 and 2012 among visits and treatment type at Bosque del Apache NWR. Effects were considered different when $P \leq 0.05$.

Type III Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
visit	1	1246	0.02	0.8824
irrigation	1	1246	0.02	0.8766
tillage	1	1246	112.32	<.0001
irrigation(tillage)	1	1246	0.15	0.7031
depth	2	1246	5.52	0.0041
irrigation*depth(tillage)	6	1246	70.36	<.0001
visit*irrigation* depth(tillage)	11	1246	71.82	<.0001

Table 13 - Type 3 Tests of Fixed Effects for Analysis of Variance in gravimetric soil moisture collected in 2011 and 2012 among visits and treatment type at Bosque del Apache NWR. Effects were considered different when $P \leq 0.05$.

Type III Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
visit	1	1248	14.72	0.0001
irrigation	1	1248	0	0.948
tillage	1	1248	14.69	0.0001
irrigation(tillage)	1	1248	1.78	0.1827
depth	2	1248	302.93	<.0001
irrigation*depth(tillage)	6	1248	3.33	0.0029
visit*irrigation*depth(tillage)	11	1248	9.59	<.0001

3.6 Discussion

The results of this study indicate that rototillage has no effect on initial root zone salinities after flooding compared to no-till soils and that treatments of tillage and irrigation frequency in moist-soil management have little influence on the net accumulation of salts over the course of the growing season. Overall plant biomass was not influenced by irrigation frequency but did increase by the application of rototillage. However, few differences in soil salinities between rototilled soils and no-till soils suggests that the increases in plant-biomass is most likely related to the physical disturbance of the soil that promotes germination (Gray et al. 1999), rather than benefiting from any differences in salinity.

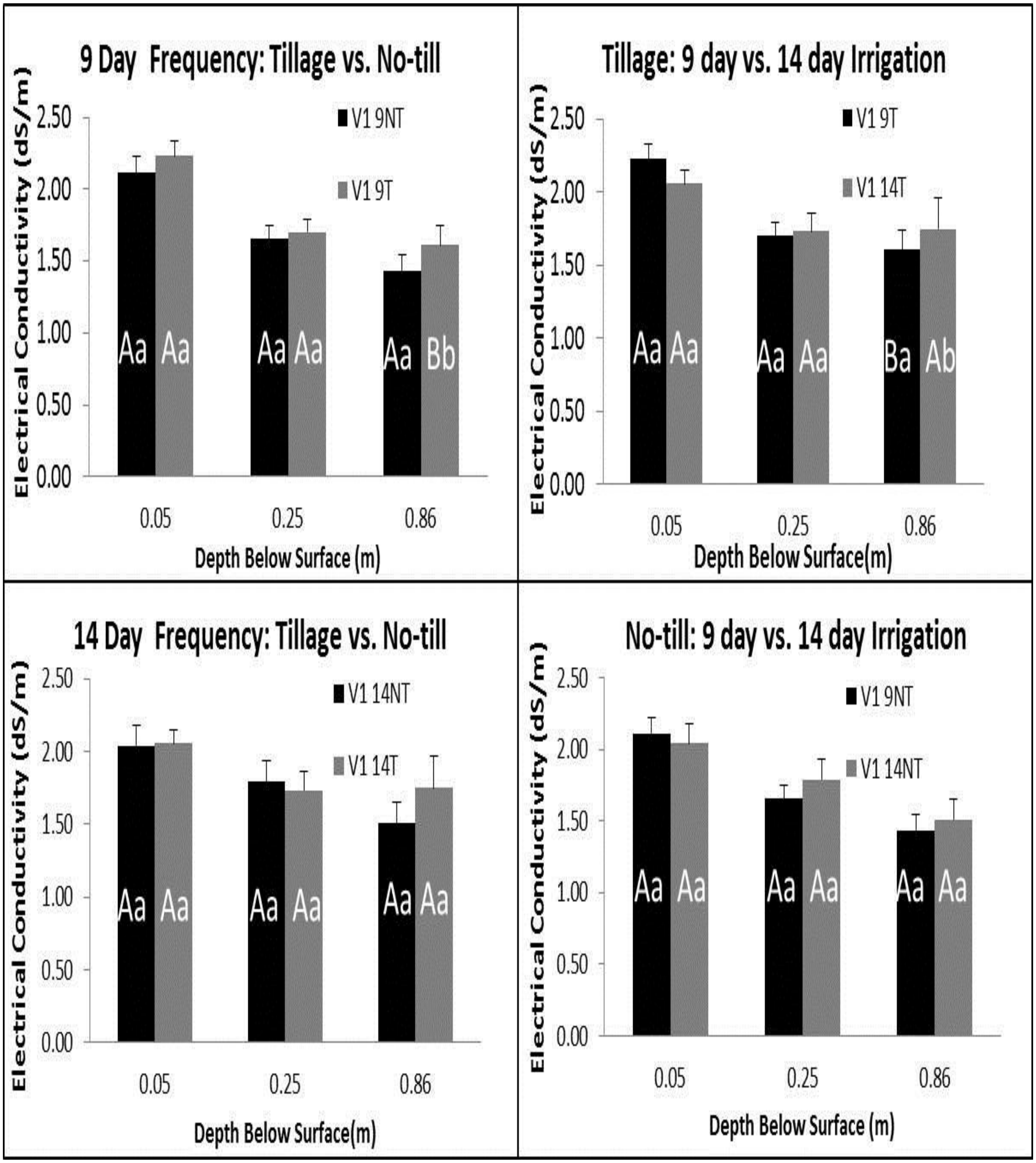

Figure 39 – Mean values in saturated paste electrical conductivity (EC_{SP}) collected from soils after initial summer flood-up (Visit 1, V1) in 2011 and 2012 among assigned treatments in moist-soil impoundments at Bosque del Apache NWR. Main plot treatments were 9 day irrigation frequency (top left) and 14 day irrigation frequency (bottom left). Sub-plot treatments were rototillage (top right) and no-tillage (bottom right). At the time of V1 sampling all treatments had received only one irrigation, the initial flood-up. Means sharing a letter do not differ ($P > 0.05$). Capital letters represent groups within treatment type by depth. Lowercase letters represent groups among treatment type at one depth.

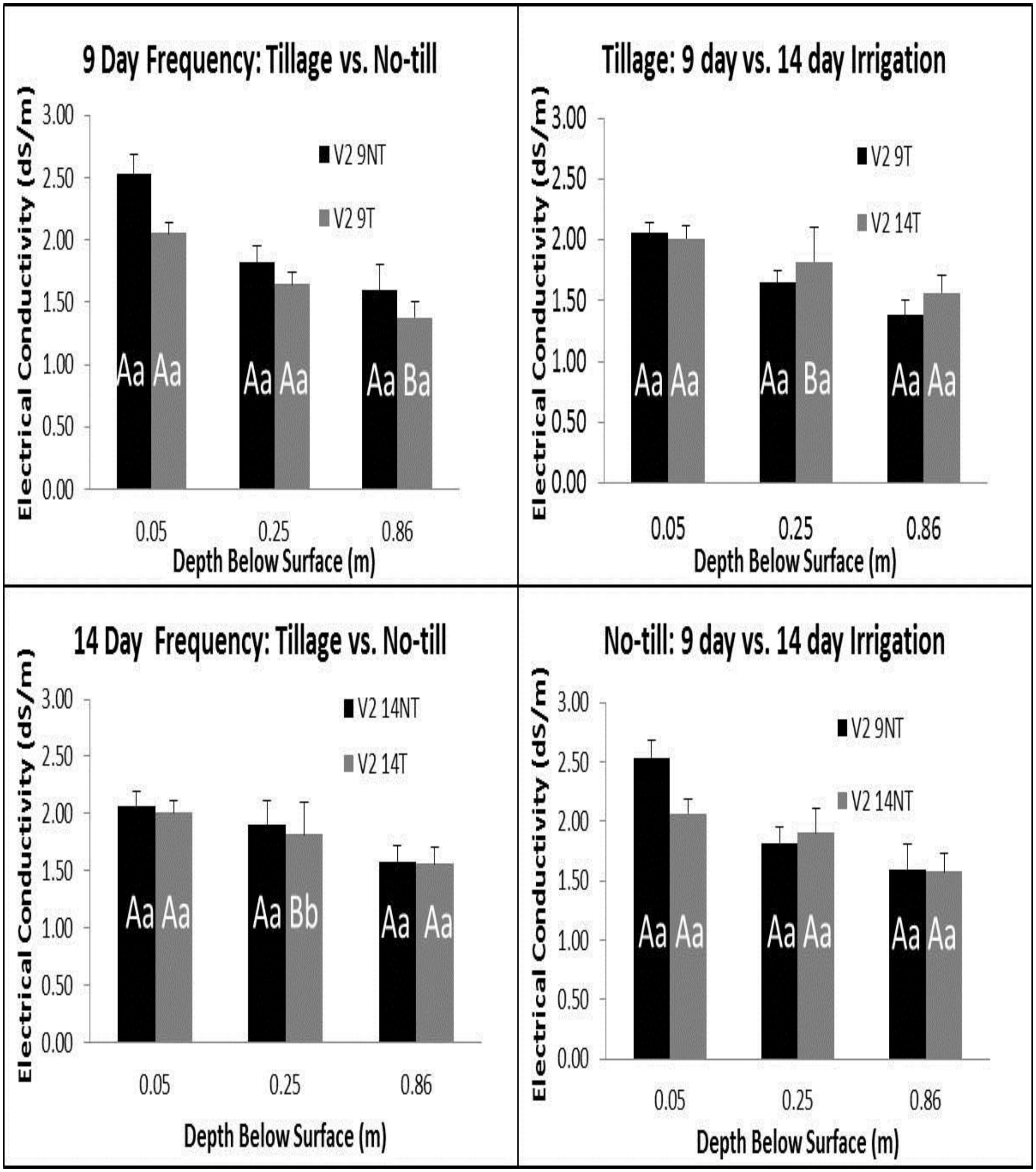

Figure 40 - Mean values in saturated paste electrical conductivity (EC_{SP}) collected from soils after the summer growing season (Visit 2 ,V2) in 2011 and 2012 among assigned treatments in moist-soil impoundments at Bosque del Apache NWR. Main plot treatments were 9 day irrigation frequency (top left) and 14 day irrigation frequency (bottom left). Sub-plot treatments were rototillage (top right) and no-tillage (bottom right). Means sharing a letter do not differ ($P > 0.05$). Capital letters represent groups within treatment type by depth. Lowercase letters represent groups among treatment type at one depth.

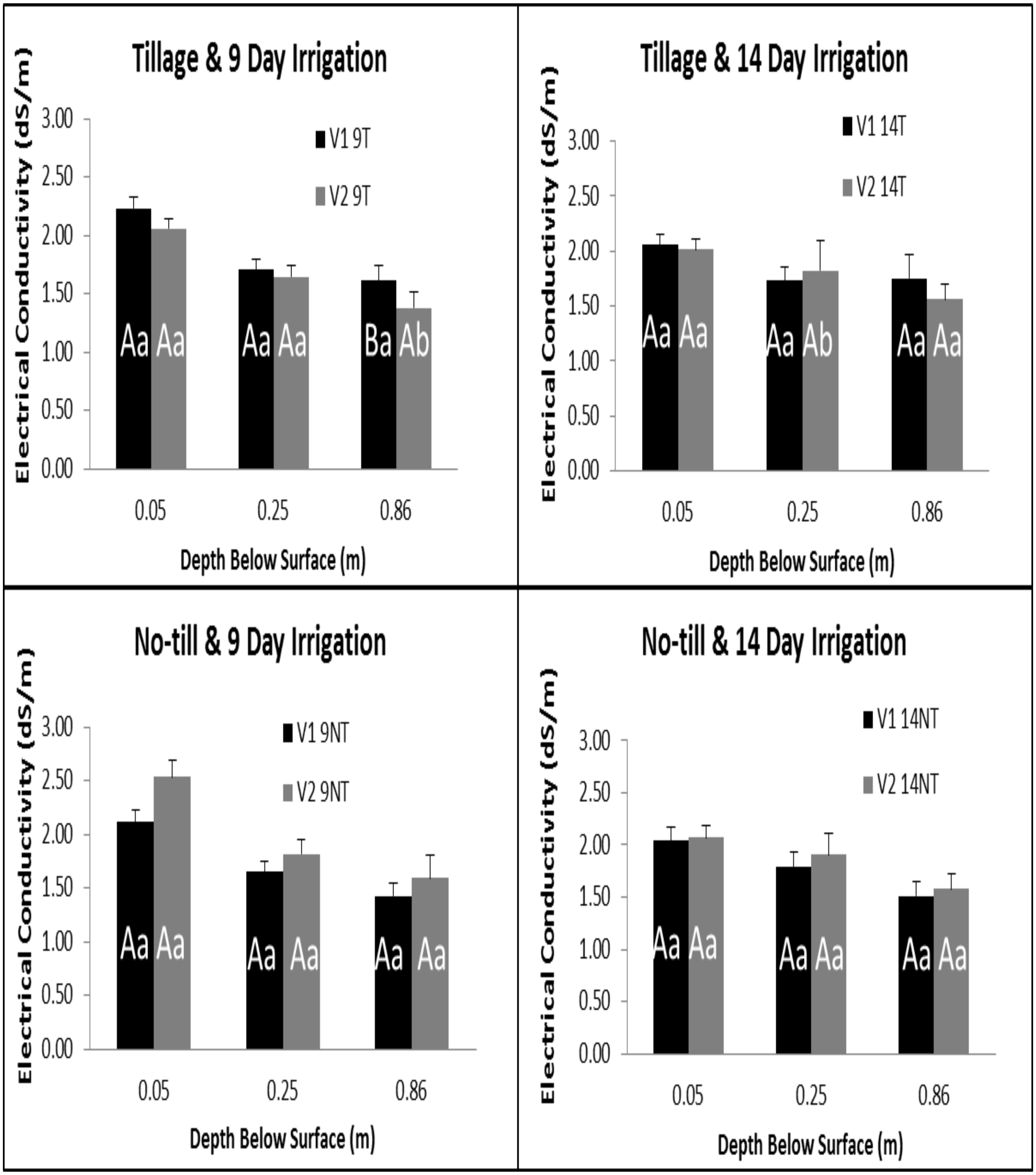

Figure 41- Differences in V1 and V2 mean values of saturated paste electrical conductivity (EC_{SP}) collected from soils in 2011 and 2012 among assigned treatments in moist-soil impoundments at Bosque del Apache NWR. Main plot treatments were 9 day irrigation frequency (top left) and 14 day irrigation frequency (bottom left). Sub-plot treatments were rototillage (top right) and no-tillage (bottom right). Means sharing a letter do not differ ($P > 0.05$). Capital letters represent groups within treatment type by depth. Lowercase letters represent groups among treatment type at one depth.

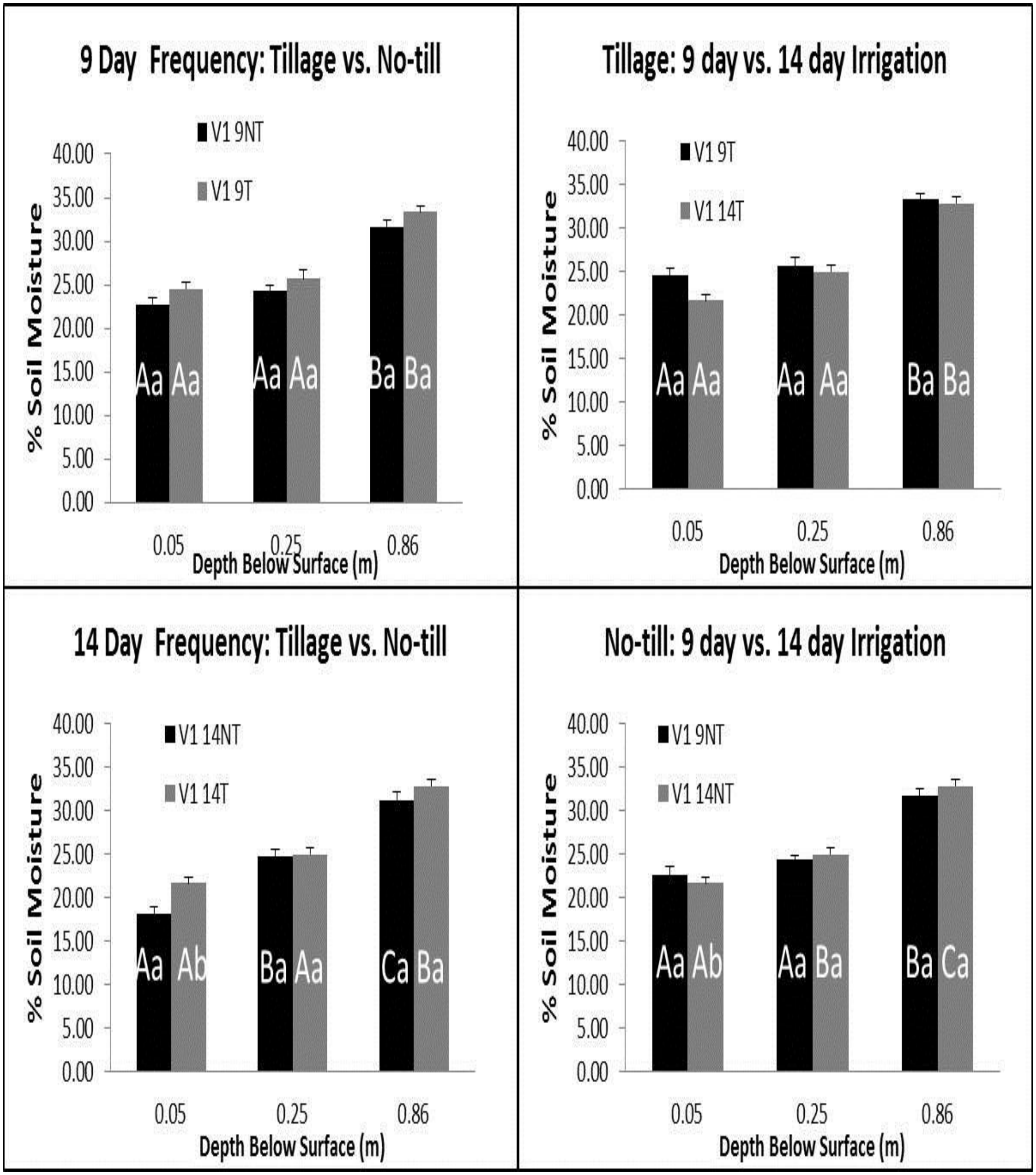

Figure 42 - Mean values in gravimetric soil moisture (GSM) collected from soils after initial summer flood-up (V1) in 2011 and 2012 among assigned treatments in moist-soil impoundments at Bosque del Apache NWR. Main plot treatments were 9 day irrigation frequency (top left) and 14 day irrigation frequency (bottom left). Sub-plot treatments were rototillage (top right) and no-tillage (bottom right). At the time of V1 sampling all treatments had received only one irrigation, the initial flood-up. Means sharing a letter do not differ (P > 0.05). Capital letters represent groups within treatment type by depth. Lowercase letters represent groups among treatment type at one depth.

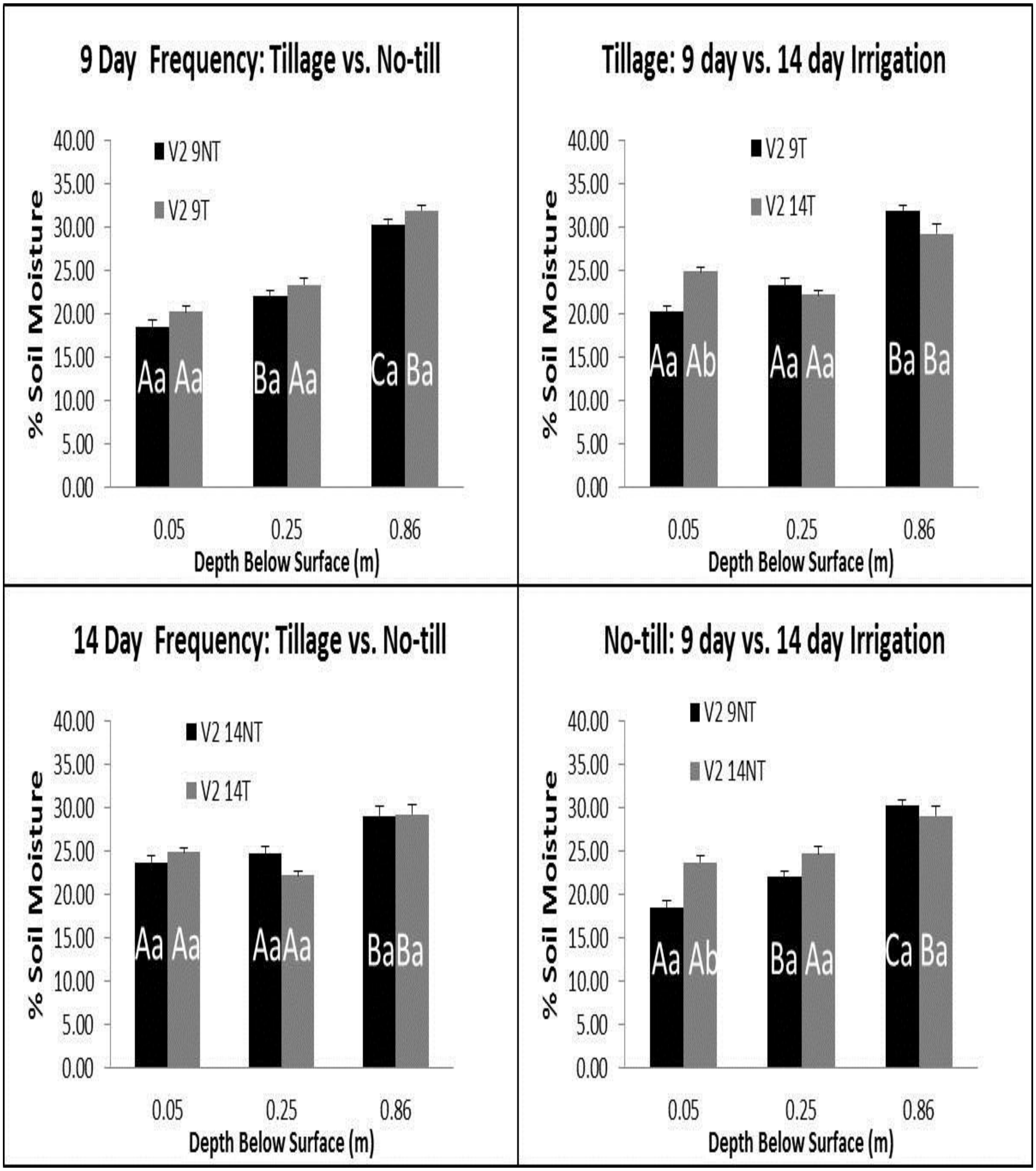

Figure 43 - Mean values in gravimetric soil moisture (GSM) collected from soils after the summer growing season (V2) in 2011 and 2012 among assigned treatments in moist-soil impoundments at Bosque del Apache NWR. Main plot treatments were 9 day irrigation frequency (top left) and 14 day irrigation frequency (bottom left). Sub-plot treatments were rototillage (top right) and no-tillage (bottom right). Means sharing a letter do not differ (P > 0.05). Capital letters represent groups within treatment type by depth. Lowercase letters represent groups among treatment type at one depth.

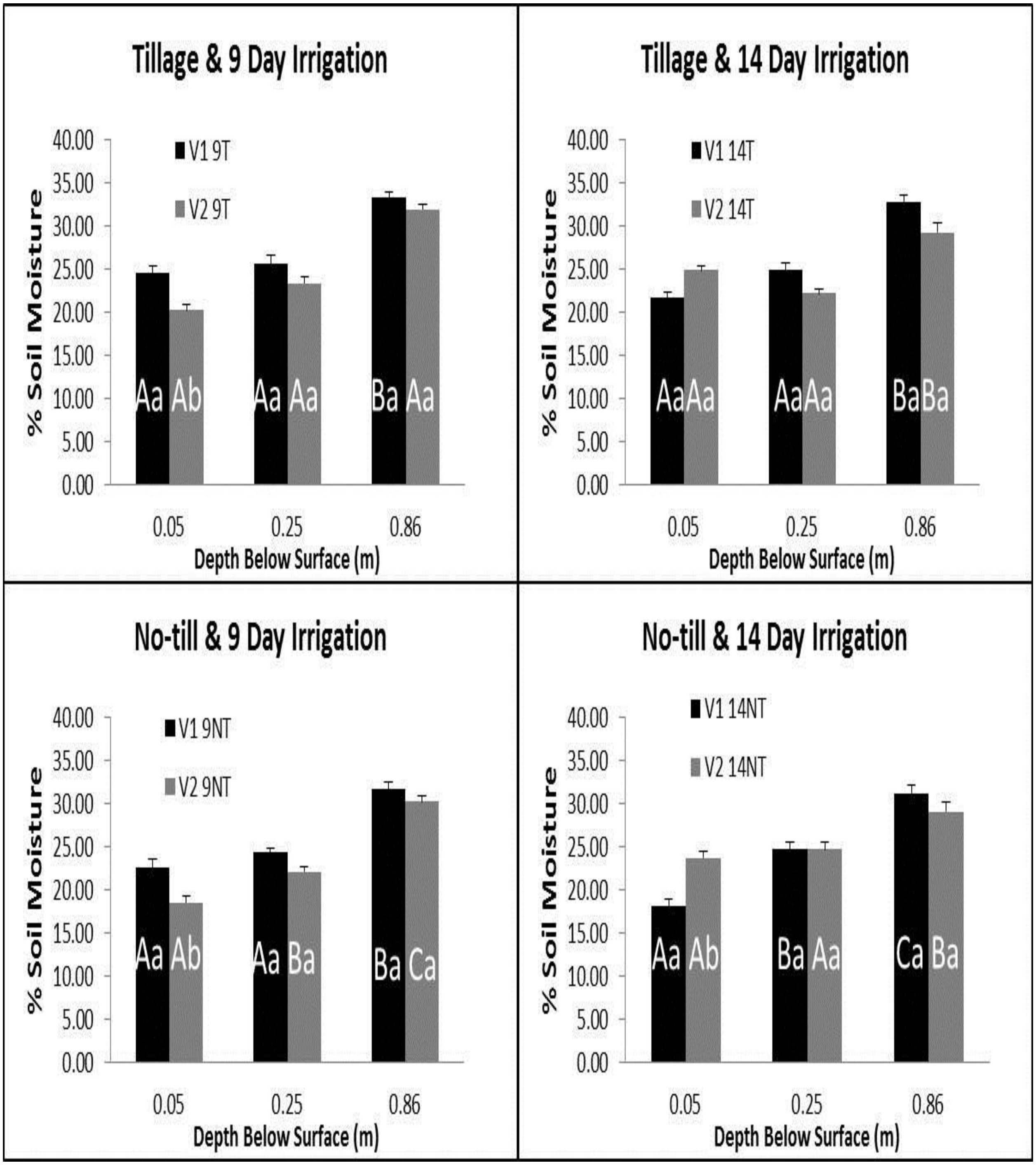

Figure 44 - Differences in V1 and V2 mean values of gravimetric soil moisture (GSM) collected from soils in 2011 and 2012 among assigned treatments in moist-soil impoundments at Bosque del Apache NWR. Main plot treatments were 9 day irrigation frequency (top left) and 14 day irrigation frequency (bottom left). Sub-plot treatments were rototillage (top right) and no-tillage (bottom right). Means sharing a letter do not differ (P > 0.05). Capital letters represent groups within treatment type by depth. Lowercase letters represent groups among treatment type at one depth.

Table 14- Type 3 Tests of Fixed Effects for Analysis of Variance for total dry-weight biomass and selected species collected in 2012 among treatment type at Bosque del Apache NWR. Effects were considered different when P ≤ 0.05.

	Type III Tests of Fixed Effects				
	Effect	Num DF	Den DF	F Value	Pr > F
Total Biomass	irrigation	1	32	1.97	0.1703
	treatment	1	32	17.83	0.0002
	irrigation*treatment	1	32	0.17	0.687
Millet	irrigation	1	28	0.01	0.9165
	treatment	1	28	4.68	0.0392
	irrigation*treatment	1	28	0.73	0.4006
Cupgrass	irrigation	1	22	0.49	0.4927
	treatment	1	22	14.61	0.0009
	irrigation*treatment	1	22	0.99	0.3299
Sprangletop	irrigation	1	31	0.07	0.7978
	treatment	1	31	38.11	<.0001
	irrigation*treatment	1	31	3.09	0.0885

Table 15 – Mean and Standard Error for the top three species found in moist-soil impoundments under no-till and rototillage, and 9 and 14 day irrigation frequency treatments at Bosque del Apache NWR, 2012.

	9 Day	14 Day
Sprangletop (No-till)	13.62 (±1.64) g	16.75 (±3.40) g
Sprangletop (Rototilled)	40.58 (±5.12) g	30.75 (±2.75) g
Millet(No-till)	14.85 (±2.56) g	13.18 (±2.05) g
Millet (Rototilled)	18.30 (±3.61) g	21.32 (±4.03) g
Cupgrass (No-till)	10.40 (±1.15) g	10.93 (±1.79) g
Cupgrass (Rototilled)	23.07 (±5.66) g	17.44 (± 5.55) g

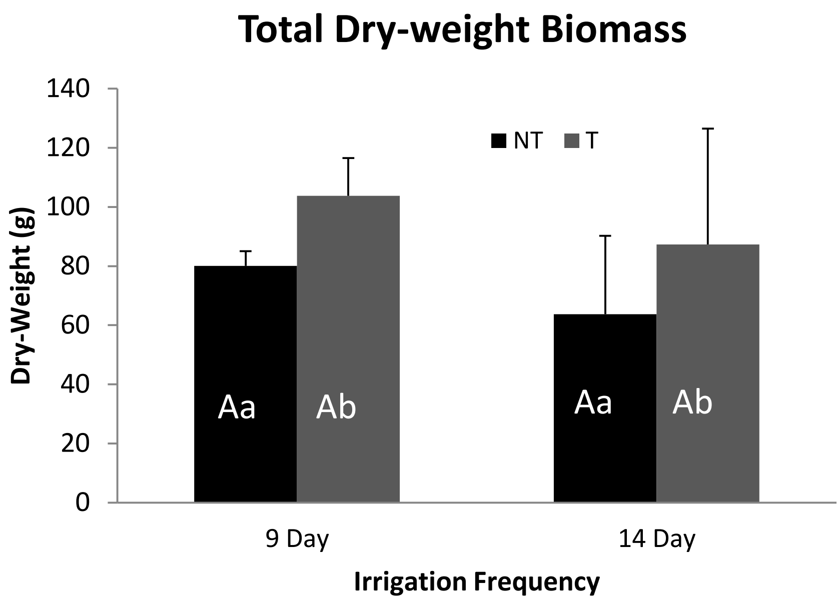

Figure 45 – Mean values of total dry-weight biomass for vegetation collected among treatments in moist-soil impoundments at Bosque del Apache NWR, 2012. Capital letters represent similar groups of tillage treatments among different irrigation treatments. Lower-case letters represent similar groups within irrigation treatments. Means sharing a letter do not differ (P > 0.05).

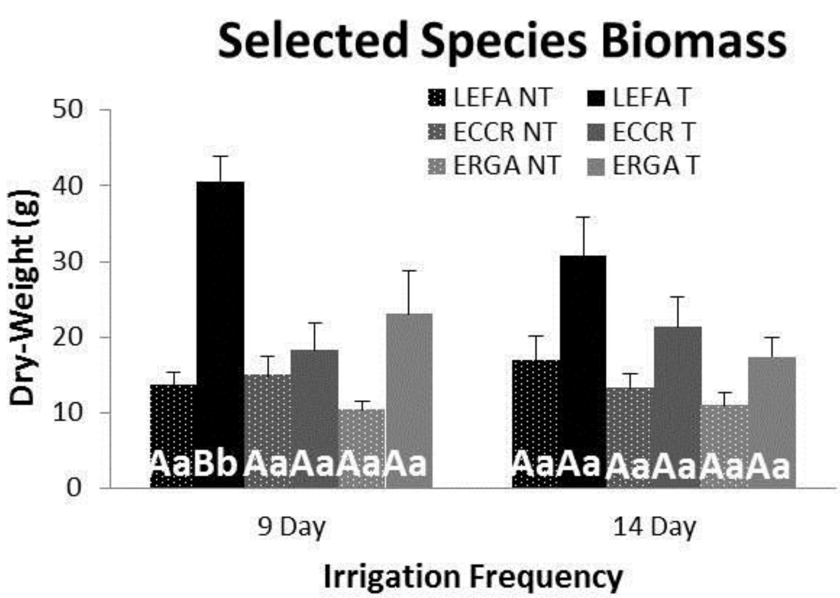

Figure 46 - Mean values of dry-weight biomass selected species (LEFA = Leptochloa fusca ssp. fascicularis; ECCR = Echinochloa crus galli; ERGA = Eriochloa gracilis) collected among treatments in moist-soil impoundments at Bosque del Apache NWR, 2012. Capital letters represent similar groups of tillage treatments among different irrigation treatments. Lower-case letters represent similar groups within irrigation treatments. Means sharing a letter do not differ (P > 0.05).

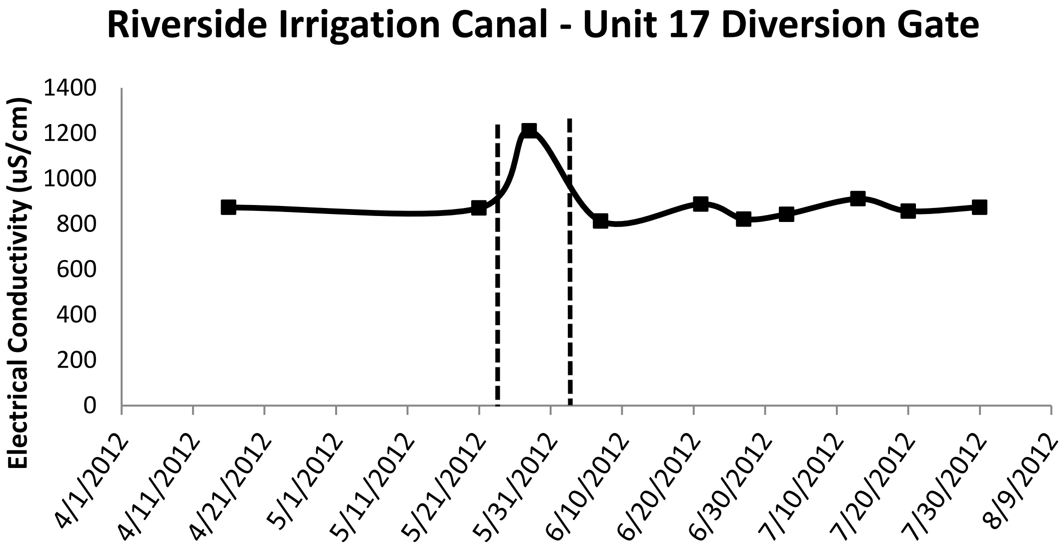

Figure 47 – Electrical conductivity of water used to irrigate study plots in Unit 17 in 2012 at Bosque del Apache National Wildlife Refuge. Shaded point markers represent dates that a grab sample was collected. Values in between the dashed line represent the time period when water was in short supply. It is likely to have caused the observed temporary increase in electrical conductivity.

3.61 Treatment Effects of Soil Salinity

The lack of influence of rototillage on salinity and gravimetric soil moisture was surprising. It is possible that rototillage simply does not affect these processes; however, another possibility is that the no-till and rototilled soils may be responding more to long-term soil management than the short-term treatment that I imposed. Moist-soil impoundments selected for study have been under a rotational soil tillage regime for at least two decades. The long term effects of this regime may have resulted in more similar soil physical characteristics than dissimilar characteristics as opposed to “native” soils that have never been managed or large periods of time un-disturbed. Park and Smucker (2005) evaluated saturated hydraulic conductivity (K_s), aggregate structure, and porosity at 0-5 cm deep in soils under conventional tillage regimes to adjacent native forest soils and found that K_s was reduced 50 and 80 fold in a silty clay loam and silt loam soil, respectively, in conventionally tilled soils. In addition to rototillage, common moist-soil management practices can include periods of more intensive soil disturbances such as heavy disking (approximately 20 cm deep) and root-raking (to a depth of 90cm) (Fredrickson and

Taylor, 1982; John Vradenburg, USFWS, personal communication) that may contribute to the homogeneity between soils actively disturbed one year and “no-till” soils that were not disturbed within the same particular year. I did not measure soil physical characteristics such as bulk density, porosity, or compaction, however, based on previous studies, if no-till soils reflected long-term management regimes then I would expect that rototilled soils would have greater soil moisture than no-till soils. Azooz and Arshad (1996) and Kargas et al. (2012) found higher soil moisture retention in tilled and rototilled than no-till soils as a result of reduced bulk density and porosity compared to long-term no-till soils. In my study, there was no definitive evidence to show that gravimetric soil moisture was greater in rototilled soils than in no-till soils after an irrigation event; nor was it greater at the .05 m depth compared to the .25 m depth within the same treatment. Although this evidence is clearly not conclusive, it does suggest that the potential impacts of long-term wetland management practices on soil structure and fertility should receive further study, as appropriate values of these soil characteristics are critical for long-term wetland and waterbird management goals.

3.62 Treatment Effects on Plant Biomass

No differences in soil salinity between treatments of tillage and irrigation frequency occurred. No information was available of the salinity tolerance of *Eriochloa gracilis* however literature on *Leptochloa fusca ssp. fascicularis* and *Echinochloa crus-gralli* suggest that salinity levels measured during my study were not to detrimental (Akhter et al. 2004; Wilson and Read 2006). This may be a result of the ability of most moist-soil species to tolerate the slightly brackish irrigation water (Gleason et al. 2009; 500-1100 uS/cm) supplied to moist-soil impoundments during the duration of the study (Figure 15). For example, Gleason et al. (2009) reported that Japanese Millet (*Echinochloa crus-grali*) can tolerate soil salinities up to 5.5 dS/m (Gleason et al. 2009) and Wilson and Read (2006) reported that a 50% reduction in biomass did not occur until irrigated with water of 13.9 dS/m. Maximum values

measured in my study were 1.21 dS/m (Figure 47). In my study, I did not evaluate changes in soil salinity concentrations in between irrigation events. In all treatment types, evapoconcentration of salts may have occurred during periods of drying after an irrigation event and led to the temporary increase in osmotic stress (Playan et al. 2008). A determining factor in the survivability and productivity of moist-soil plants in my study may have been consistent irrigations throughout the experiment, regardless of a 9 or 14 day frequency. Results from Chapter 2 suggest that flash flood irrigations in moist-soil impoundments serve as recharge events that are capable of flushing salts into the groundwater. This consistency likely serves to reduce the osmotic stress on moist-soil plants after periods on evapoconcentration. Future moist-soil management research would benefit from a more detailed analysis of changes in root zone salinities in between irrigation events with longer lag periods to determine osmotic thresholds that wetland managers can use to create more conservative water budgets.

While differences in plant dry-weight biomass were not detected as a result of irrigation frequency it is important to note that plant biomass is not a direct indication of seed production (Fredrickson and Taylor 1982); high seed production, not biomass, is the ultimate goal in most moist-soil management settings. Mushet et al. (1992) found that multiple irrigation events caused an increase in plant height and dry-weight but not in seed production. Haukos and Smith (2006) reported that in Pink smartweed (*Polygonum pensylvanicum*), shallowly flooded (0-5 cm) impoundments produced the greatest amount of biomass; however impoundments that maintained soil moisture at field capacity produced greater seed production. Thus, while a longer period between irrigation events could maintain plant biomass and conserve water; additional study on the impacts of irrigation on seed production is needed.

3.7 Conclusions

In conclusion, treatments of rototillage paired with irrigation events seem not to influence differences in soil salinity concentrations when sampled after a draw-down compared to soils that receive no tillage. At the time sampled soil salinity concentrations were not determined detrimental to common moist-soil plants, possibly because of acceptable quality of irrigation water. However, further research is necessary to determine how salinity might increase in between periods of irrigation as a result of evapoconcentration that might breach salinity thresholds. In my results, 9 and 14 day irrigation frequencies did not influence overall biomass levels. As water availability becomes more uncertain in semi-arid environments, additional studies could evaluate how more conservation irrigation frequencies might affect plant productivity in an effort to maximize water conservation without neglecting seed production.

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VITA

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