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Changes in spatial and temporal variability of SAR affected by shallow groundwater management of an irrigated field, California

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ABSTRACT

In the irrigated western U.S. disposal of drainage water has become a significant economic and environmental liability. Development of irrigation water management practices that reduce drainage water volumes is essential. One strategy combines restricted drainage outflow (by plugging the drains) with deficit irrigation to maximize shallow groundwater consumption by crops, thus reducing drainage that needs disposal. This approach is not without potential pitfalls; upward movement of groundwater in response to crop water uptake may increase salt and sodium concentrations in the root zone. The purposes for this study were: to observe changes in the spatial and temporal distributions of SAR (sodium adsorption ratio) and salt in a field managed to minimize drainage discharge; to determine if in situ drainage reduction strategy affects SAR distribution in the soil profile; and to identify soil or management factors that can help explain field wide variability. We measured SAR, soil salinity ($EC_{1:1}$) and soil texture over 3 years in a 60-ha irrigated field on the west side of the San Joaquin Valley, California. At the time we started our measurements, the field was beginning to be managed according to a shallow groundwater/drainage reduction strategy. Soil salinity and SAR were found to be highly correlated in the field. The observed spatial and temporal variability in SAR was largely a product of soil textural variations within the field and their associated variations in apparent leaching fraction. During the 3-year study period, the percentage of the field in which the lower profile (90–180 cm) depth averaged SAR was above 10, increased from 20 to 40%. Since salinity was increasing concomitantly with SAR, and because the soil contained gypsum, sodium hazard was not expected to become a limiting factor for long term shallow groundwater management by drain control. It is anticipated that the technology will be viable for future seasons.

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1. Introduction

Productive agriculture in semi-arid regions depends on the availability of irrigation water. Irrigated agriculture in the semi-arid western U.S. uses approximately 80% of the developed water resources and is in direct competition with urban, industrial, and environmental water users (Letey et al., 2002). To ensure sustainability, agricultural water users will need to adopt improved water conservation strategies, develop new sources of water, and learn to use lower quality water for irrigation. Water used for irrigation transfers salts to soils and also dissolves native salts, increasing soil salinity and decreasing crop yields. Excess irrigation water then must be applied to reduce salt build-up in the root zone (Ayars et al., 1993), further taxing precious water resources. Engineered subsurface drainage systems are usually installed to facilitate leaching and prevent water logging (Hoffman and Durnford, 1999). As well, drainage water collected from these systems is saline, may be sodic, and may contain agricultural chemicals or other contaminants (Banuelos, 1996), which makes disposal a significant environmental consideration (Letey et al., 2002; Schoups et al., 2005).

Agricultural drainage water has been discharged mainly into river systems in the irrigated western U.S. (Schoups et al., 2005). In the San Joaquin Valley of California during the 1970s and 1980s, irrigation water projects expanded, as did the need for drainage water disposal. This drainage water was used to replenish wetlands and other wildlife habitats (Schoups et al., 2005), but these waters contain significant concentrations of selenium, arsenic, boron, and other trace elements (Letey et al., 2002). These naturally occurring elements in the region's soils are dissolved and leached by irrigation ending up as drainage discharge. In wetlands receiving drainage, it was discovered that toxic trace elements

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were becoming bio-concentrated resulting in high exposure levels for animals near the top of the food chain (Fan et al., 1988). Reproductive failures and deformities in waterfowl were linked to the presence of toxic trace elements (Letey et al., 2002; Tanji et al., 1986).

The practice of discharging agricultural drainage water into wetland wildlife refuges was curtailed in the late 1980s, and alternative solutions have been under study. One proposed drainage water source reduction strategy is to restrict outflow of field drains thus maintaining a shallower water table. Certain crops are able to use this shallow groundwater to satisfy a portion of their evapotranspiration (ET) requirements (Ayars et al., 1996) thereby reducing drainage volume for disposal. Ayars et al. (1999), assert that deficit irrigation in combination with drain flow management is necessary to achieve optimal crop water use from shallow groundwater. By using proper irrigation management, cotton (Hutmacher et al., 1996) and alfalfa (Ayars et al., 2009) have been shown to take-up between 20 and 50% of their water requirement from shallow groundwater in column experiments.

A potential complication resulting from restricted drain flow with deficit irrigation, however, would be the development of a saline-sodic soil profile (Hornbuckle et al., 2005). Significant increases in sodium adsorption ratio ($\mathrm{SAR} = [\mathrm{Na}^+]/\left(\sqrt{([\mathrm{Ca}^{2+}] + [\mathrm{Mg}^{2+}])/2}\right)$, where concentrations are in meq/L) in the root zone may produce reductions in water transmission properties and the development of slowly permeable layers, leading to perched water tables and water logging. These conditions could decrease the effectiveness of the drainage reduction strategy by reducing crop productivity.

This issue was covered in a previous study in which we showed that salinity and boron occurred together in this irrigated field (Shouse et al., 2006), and that the shallow groundwater management scheme could be sustained without increased salinity and boron in the top meter, and without reduced cotton yield.

High SAR values also occur together with salinity and boron throughout the western San Joaquin Valley (Corwin et al., 2003), as well as in other arid and semi-arid regions of the world (Nicholaichuk et al., 1988; Ben-Gal and Shani, 2002; Ardahanlioglu et al., 2003; Nuttall et al., 2003). Soils along the west side of the Central Valley of California also contain significant amounts of gypsum and calcite which play a primary role in the "self-reclaiming" aspects of these soils (Oster et al., 1999) mitigating the deleterious effects of sodium on soil hydraulic properties.

Spatial and temporal distributions of SAR could change significantly during implementation of new management schemes that reduce irrigation amounts and increase water uptake directly from the shallow groundwater. Soil or management factors controlling sodium, calcium, and magnesium movement such as infiltration, gypsum and calcite content, clay content, plant water extraction rate, and depth to groundwater likely vary in space and time as well.

The objectives of our study were: first, to measure and describe the spatial and temporal variations in SAR of an irrigated field managed to reduce drainage water; second, to associate soil or management factors that help to explain the measured variability; and third, to determine if drainage minimization strategy affects SAR distribution in the soil profile.

2. Materials and methods

A detailed account of the materials and methods used can be found in our previous study (Shouse et al., 2006). We briefly describe essential information for these analyses here.

2.1. Field site

Our field site was located in the Broadview Water District (36°47.786′N, 120°29.955′W) on the west side of the San Joaquin Valley in California (Fig. 1). The field is a quarter-section with approximately 60 ha of drained land in production at the time of the experiment. The field soil belongs to two soil series, the Tranquillity clay (24% of the field) (Fine, smectitic, thermic Sodic Haploxererts) and the Calflax clay loam (76% of the field) (Fine-loamy, mixed, superactive, thermic Sodic Haplocambids) (Soil Survey Staff, 7/7/2009). In agreement with their sodic classification, the possibility of reductions in water permeability at SAR > 13 was indicated (Soil Survey Staff, 7/7/2009). The NRCS classification also indicated that the soils were saline, gypsiferous and calcareous. The solution calcium concentrations provided by the presence of gypsum and calcite assure adequate levels of electrolyte concentration to prevent significant reductions in hydraulic conductivity as either or both electrical conductivity and SAR of the soil water change with time as a result of irrigation (Quirk and Schofield, 1955; Quirk, 1986; Oster et al., 1999).

2.2. Cropping sequence

Cotton (*Gossypium hirsutum* L, cv MAXXA) was grown during the 1995 and 1997 seasons and tomato (*Lycopersium esculentum*, cv Heinz 3044 hybrid, *Lycopersium esculentum*, cv La Rosa, and *Lycopersium esculentum*, cv Apex 1000) was grown during the 1996 season. Agronomy and pest management were the responsibility of our farmer cooperator.

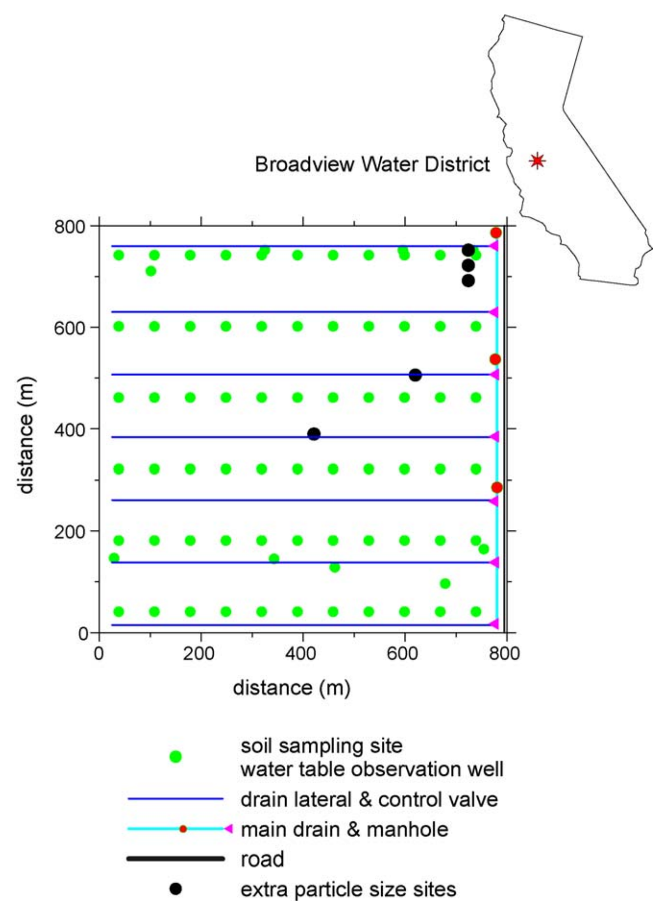

Fig. 1. Schematic map showing the relative location of the study site and the field infrastructure (reproduced in part from Shouse et al., 2006).

2.3. Irrigation and drainage

Tomatoes were sprinkler irrigated during the early part of the season, and furrow irrigated during the rest of the season; cotton was furrow irrigated. Irrigation scheduling was the responsibility of the cooperating farmer. Rainfall in the area was approximately 250 mm per year, and occurred during the fallow period between November and February.

Broadview Water District measured irrigation water quality during our study and electrical conductivity was approximately 0.9 dS m^{-1} with an SAR value of 3.78 (Wichelns et al., 2002).

Our field was drained by a subsurface drainage system consisting of seven laterals installed at a depth of 1.8 m perpendicular to the furrow direction (Fig. 1), making it possible to maintain a uniform water table depth of 1.2 m using control valves installed only on the east side where laterals met the submain. Ayars et al. (1996) give a more detailed account of the water table control apparatus. The water table would start the growing season at a depth of 1.2 m, and then decline as the season progressed and crops consumed groundwater. The drainage water electrical conductivity (5.3 dS/m) and SAR (10.2) reported for the area by Wichelns et al. (2002) indicated no permeability hazard for the drainage water. When drains were flowing, we periodically measured EC and SAR and obtained comparable values.

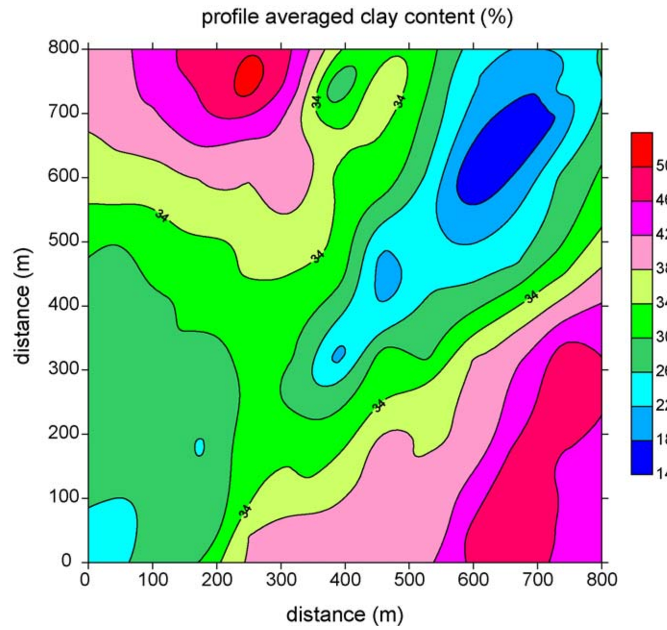

Fig. 2. Spatial distribution maps of soil profile averaged clay content.

2.4. Data collection

Shallow groundwater depth was measured using 38 mm diameter by 3 m deep PVC pipe observation wells having slits in the bottom meter (see Fig. 1 for approximate placement). The soil surface and observation wells were referenced to a common elevation. The depth to water table was measured weekly and shallow groundwater samples were taken every other week. Sodium, calcium, and magnesium concentrations were measured using an inductively coupled plasma spectrophotometer as described by Ayars et al. (1993).

Once each spring (April–June) within 2–3 weeks of crop emergence, and once each fall (August–November) following harvest, we sampled the 1.8 m soil profile. Seventy-five sampling sites were located on an approximately regular grid (see Fig. 1). Soil cores were sectioned into 0.30 m increments, put in plastic bags, weighed, homogenized, and split into two parts: one for chemical analysis, and one for gravimetric water content. The gravimetric water content sample (approximately 400 g) was weighed in the field, oven dried at 105 °C for 48 h in the lab and reweighed. Soil for chemical analyses was stored in the original plastic bags in a refrigerator at 4 °C until a 1:1 soil paste was extracted (U.S. Salinity Lab Staff, 1954). The fall 1996 gravimetric water content samples were used to measure soil particle size distribution. Additional soil samples (locations shown in Fig. 1) were gathered along a line from southwest to northeast to clarify the extent of a relic stream channel. By combining hydrometer and wet sieving methods, a complete particle size distribution was measured (Gee and Bauder, 1986; Skaggs et al., 2001; Shouse et al., 2006). Soil bulk density was estimated at each sampling using the core diameter, depth increment, total soil weight, and the gravimetric water content (Grossman and Reinsch, 2002). Our 1:1 extraction procedure used a water balance to account for all water added to the sample, allowing us to calculate the total mass of all chemical constituents as well as estimate their concentrations at other water contents. ExtractChem (Suarez and Taber, 2007) was used to correct the 1:1 extract concentrations to field water content and MINEQL+ (Schecher and McAvoy, 2003) was used to check for calcite and gypsum supersaturation. We measured the same chemical constituents in soil solution extracts as were measured in the shallow groundwater and drainage waters using the same analytical methods.

Spatial analysis of our data was performed using the Surfer graphics program (Golden Software, Boulder, CO) using ordinary point kriging and a neighborhood of 300 m. All semivariograms were fit to an exponential model (Webster and Oliver, 2001). The method of Blackmore (2000) was used for analyzing the spatial trend and time stability of the apparent SAR. According to this procedure the spatial trend is simply the mean value of a variable calculated at each grid point in the field over all the years, while the temporal stability can be assessed by calculating the coefficient of variation at each grid point over time.

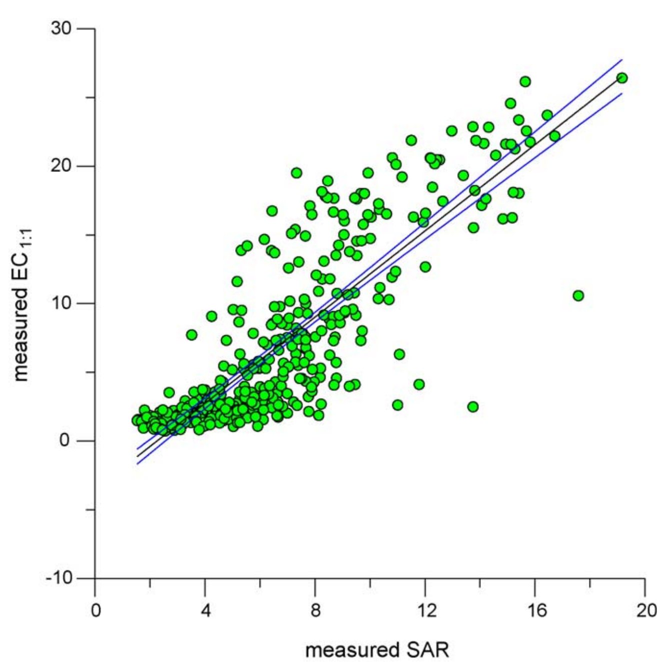

Fig. 3. Relationship between 1:1 soil extract electrical conductivity and SAR (The lines represent the linear trend with the 95% confidence interval on the slope).

3. Results and discussion

Fig. 2 is a map of the profile averaged clay content. A section of the middle of the field, stretching from the southwest corner to the northeast corner (Calflax clay loam), has lower clay content (<24%) that extends to the 1.8-m depth (individual depths not shown). This coarser textured soil material is related to an ancient stream channel that once bisected the field. Spatial patterns for bulk density (shown in Shouse et al., 2006) showed an association with clay content. The bulk density is less in areas with higher clay contents, ranging from 1.22 to 1.26 mg m^{-3} where clay is greater than 40%, and from 1.4 to 1.5 mg m^{-3} where clay is less than 25%. Areas with higher clay contents had much lower saturated hydraulic conductivities and higher soil water retention compared to the lower clay content areas. Regions with higher clay content had lower leaching rates and higher yields (and thus higher evapotranspiration (ET)) (Shouse et al., 2006), and it may be expected that these regions would have higher salinities and higher SAR values compared to sandier sites.

High soil water SARs are often associated with high soil salt concentrations in arid and semi-arid irrigated soil (Nicholaichuk et al., 1988; Nuttall et al., 2003; Ben-Gal and Shani, 2002). Fig. 3 shows a graph of the 1:1 extract SAR vs. soil salinity ($EC_{1:1}$). The SAR is directly related to salinity ($r^2 = 0.71$, $p < 0.05$) for all soil depths measured in May, 1995. Other sampling times show the same correlation with r^2s between 0.7 and 0.8 ($p < 0.05$), and all of the sampling times fit essentially the same regression trend line. The direct correlation between salinity and SAR extended also to the shallow groundwater. These correlations exist because the soils within the field share a common heritage, namely the alluvium derived from sedimentary marine deposits of the Coast Range Mountains on the western side of the San Joaquin Valley (Letey et al., 2002). The correlation between SAR and salinity in soils and groundwater most likely exists throughout Broadview Water District and possibly extends to other fields in the western San Joaquin Valley (Corwin et al., 2003; Nicholaichuk et al., 1988; Nuttall et al., 2003; Ben-Gal and Shani, 2002), especially since the range of these gypsiferous, calcareous soils is extensive (Soil Survey Staff, 2009).

Mean SAR (Fig. 4a) consistently increases with depth at every sampling time. At the shallower soil depths the SAR is between 2 and 6, values that are not considered a sodicity hazard (U.S. Salinity Lab Staff, 1954; Sumner, 1992; Suarez et al., 2006). In the upper soil profile, lower SAR results from adequate leaching with good quality irrigation water, potentially with concomitant dissolution of gypsum and calcite. With depth, and to a lesser extent with time, the SAR increases due to the concentrating effect of crop root water uptake. Calcite is concentrated and the solubility limit is reached; further concentration of the soil solution causes increased precipitation of calcite and the loss of calcium ions. Sodium salts,

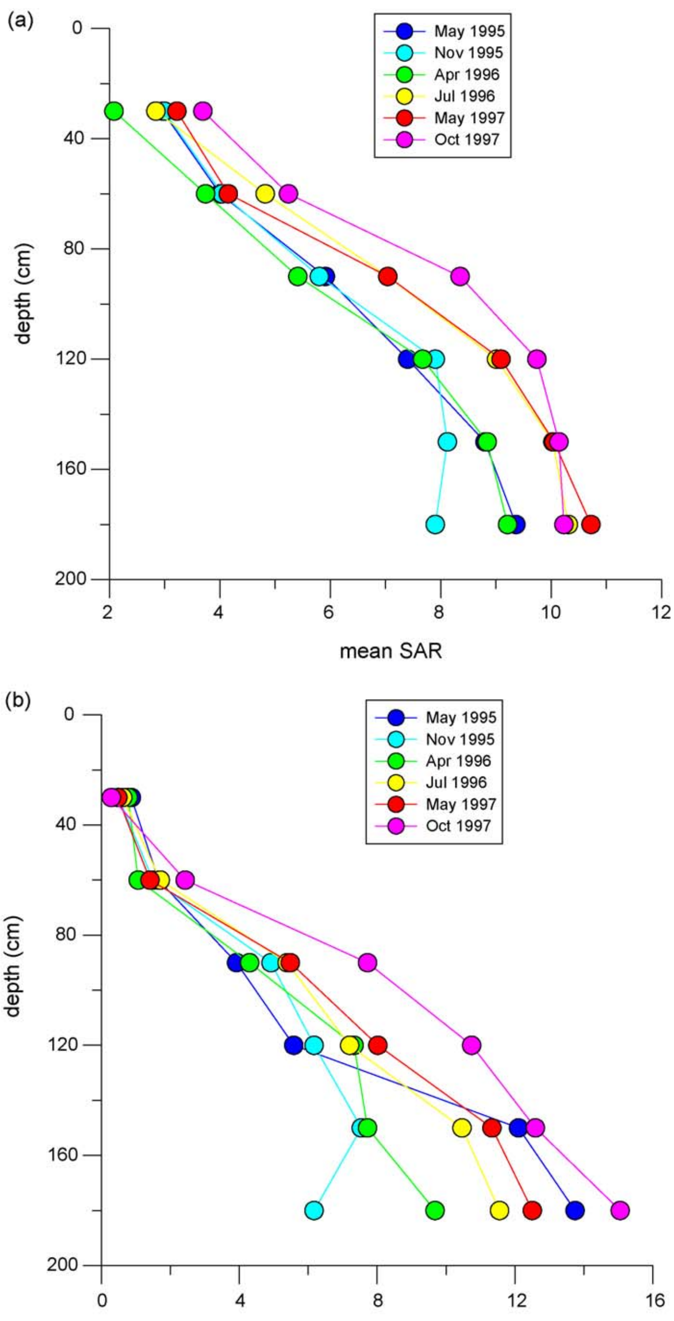

Fig. 4. Depth distribution of the mean and variance of SAR measured at each sampling time.

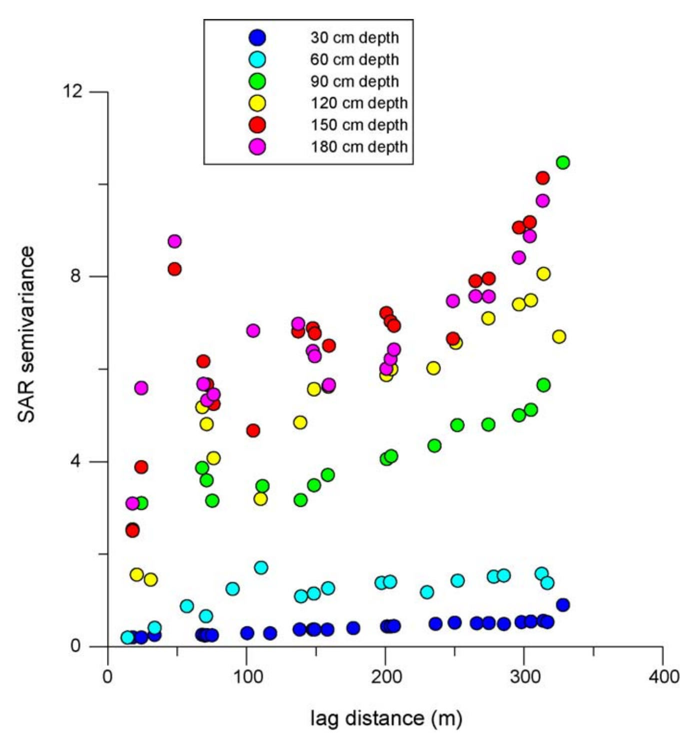

Fig. 5. Semivariograms of SAR for each depth measured in April 1996.

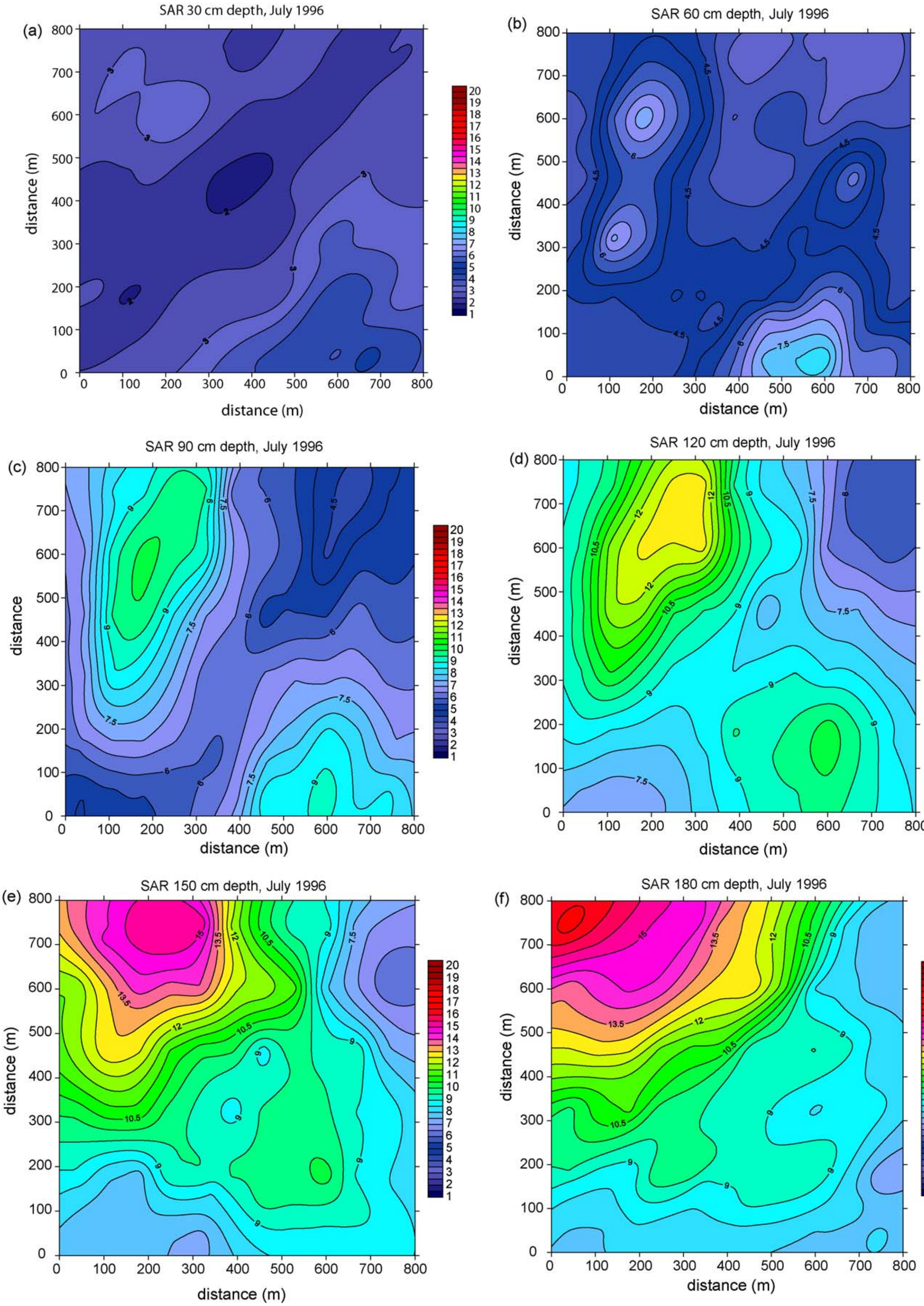

Fig. 6. Spatial distribution of SAR for each depth measured in July 1996.

being more soluble than calcite, remain in solution, causing increased sodium concentration relative to calcium and magnesium, thereby increasing the SAR. SAR also increases with concentration of the soil solution because of the square root term in the denominator. The SAR at the bottom of the soil profile reached a high average value of about 10, still below the high sodicity threshold value of 18 used by the U.S. Salinity Lab Staff (1954) but approaching the value of 13 considered a potential hazard for permeability reduction by NRCS (Soil Survey Staff, 7/7/2009). Calculations with ExtractChem and MINEQL+ predicted that gypsum was under-saturated but calcite was precipitating in the soil solutions at field water content.

Variances for SAR (Fig. 4b) are quite small in the upper reaches of the soil profile indicating considerable field uniformity to a depth of about 0.90 m. This lack of variability is consistent with an irrigation management regime that leaches soluble salts uniformly near the soil surface across the field. With depth, however, SAR variances increase and reach maxima at the bottom of the soil profile near the interface of the shallow groundwater. This observation is due in part to the variability of leaching fractions caused by differences in soil texture and/or infiltration non-uniformity, and by the influence and variability in the upward movement of water from the shallow groundwater table (Shouse et al., 2006).

Variability of salt and boron concentrations in this field was dominated by two distinct soil profiles (Shouse et al., 2006). In the first, the concentration of salt and boron was relatively low and uniform throughout. In the second, the salt and boron concentrations increased with depth to a maximum level and then decreased slightly near the groundwater interface. Soil profile characteristics had similar effects on the SAR. SAR increased with depth throughout the field, but more uniform profiles were located in sandier areas of the field having higher apparent leaching fractions, and also near drain laterals and at the head end of the field (where water infiltrates for a longer period of time). In the more clayey areas and near the tail end of the field, leaching was less and the SAR profiles exhibited higher maxima near the bottom of the soil profile.

The semivariograms of SAR (Fig. 5) show the semivariance increases with distance indicating that near neighbors are similar and that there is structure to the variance. The semivariances follow the measured SAR population variances (Fig. 4b) increasing with depth to the bottom of the profile (1.8 m). This pattern of less field homogeneity with increasing depth was also found in our previous analysis of B variability (Shouse et al., 2006). The semivariogram slopes are steep at short distances, indicating that the transition to dissimilarity of measured SAR values is fairly abrupt (Webster and Oliver, 2001).

Spatial distributions of SAR based on point kriging are contour mapped in Fig. 6. The SAR distributions are more uniform at shallower depths (<0.9 m) than at deeper depths (>0.9 m); the complexity of the spatial distribution increases with depth, and SAR appears to be co-located in the field with the profile averaged clay content (cf. Fig. 2). We previously noted the same patterns for salinity and boron (Shouse et al., 2006). The coincidence of higher SARs with clay is not unexpected because water movement through these areas was slower, and based on visual appearance plant growth was better compared to the sandier regions of the field. Where there is increased plant growth, presumably there is also increased water use, reduced deep drainage, and perhaps greater water use from the shallow groundwater. Previously we showed that the apparent leaching fraction was less than 10% in the high clay areas compared to 50% in the sandy areas (Shouse et al., 2006). With decreased leaching, and the mixed salt chemistry (chloride and sulfate salts) of this field, calcium ions would continue to precipitate as calcite, resulting in higher SAR values. This was corroborated with calculations using ExtractChem and MINEQL+. Increasing salinity values are observed concomitant with the higher SAR values in the lower profile. Data from other sampling times show the same spatial trends and are not shown here.

We have shown that the spatial distribution of SAR in the upper 0.6–0.9 m is relatively uniform due to the irrigation water management practices and the generally good quality of irrigation water (Fig. 6a and b). The spatial heterogeneity of SAR below about 0.9 m increases with depth and to a lesser extent time. To look at the temporal changes in the spatial distribution of SAR, we averaged the SAR for the 0.9–1.8 m depths (Fig. 7), where most of the change is occurring. Fig. 7a shows the spatial distribution of the lower profile averaged SAR for the initial sampling (May 1995). Prior to this sampling no water table management occurred. Fig. 7b shows the spatial distribution of the lower profile averaged SAR for

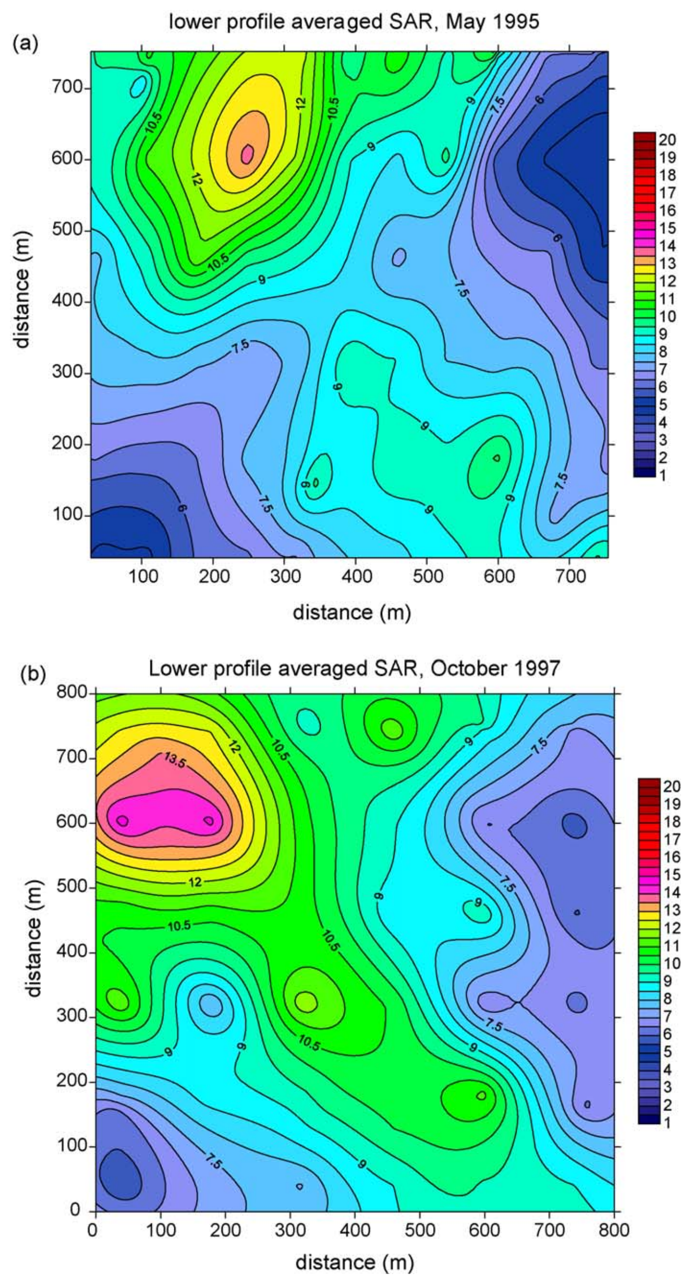

Fig. 7. Spatial distribution of lower profile averaged SAR for May 1995 and October 1997.

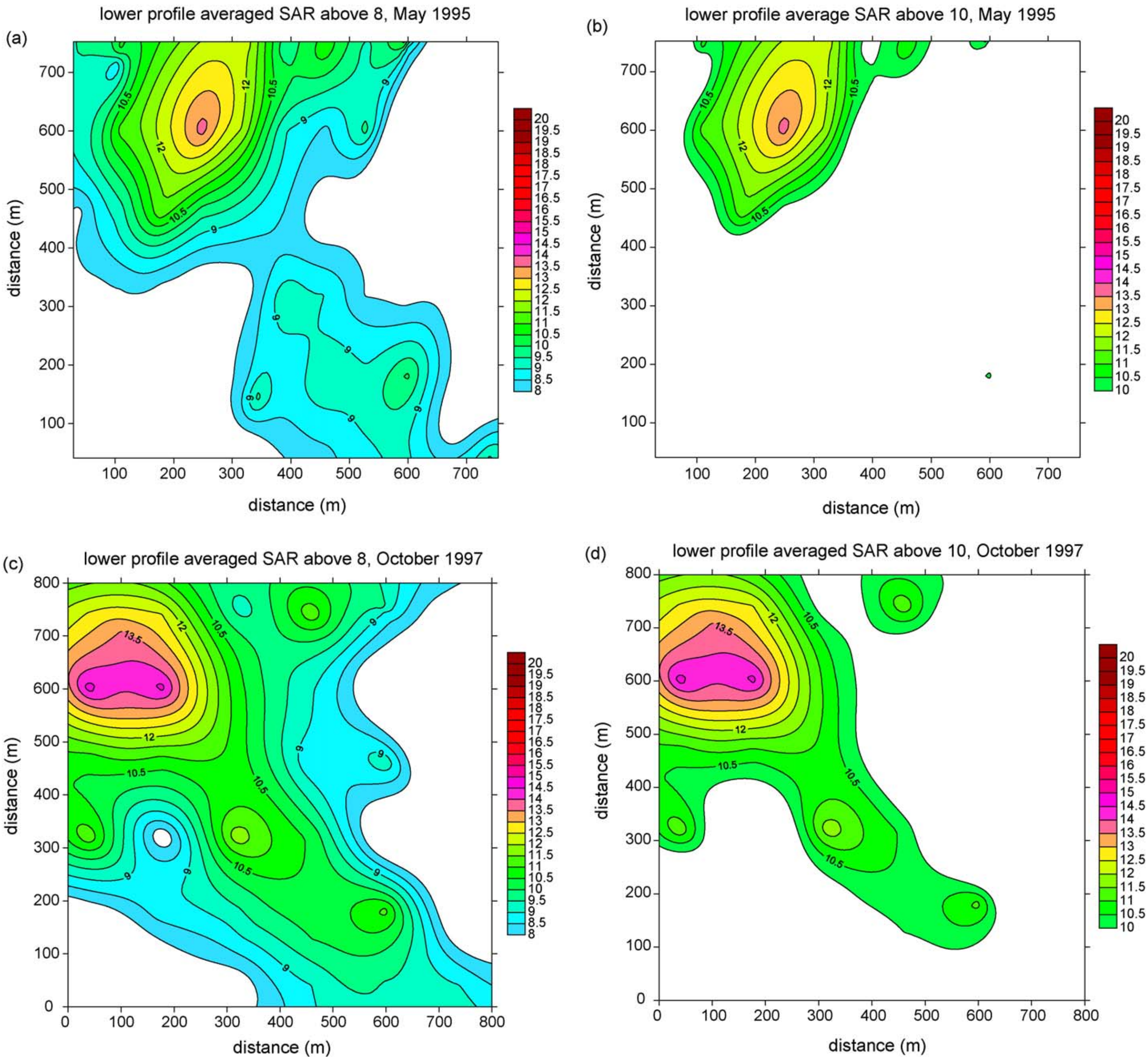

Fig. 8. Spatial distribution of lower profile averaged SAR greater than 8 and greater that 10 for May 1995 and October 1997.

the last sampling (October 1997). These contour maps have all the features of the individual depth maps (see Fig. 6). Areas high in clay have high SAR and high salinity.

To better illustrate the changes that occurred in soil solution SAR between these two times, we delineated for each time areas where depth averaged SAR in the lower profile was above 8 (Fig. 8a and c) and areas where SAR was above 10 (Fig. 8b and d). Comparing Fig. 8a and c, we see that the percentage of the field with lower profile SAR above 8 increased from approximately 60% to approximately 80%, whereas Fig. 8b and d indicate that the area with SAR above 10 increased from about 20% to about 40%. The areas affected are higher in clay. The increase in area affected by sodium corresponds roughly to the area we found to have an apparent leaching fraction lower than 20%, increased solution salinity, and a high time stability (Shouse et al., 2006). Our analysis of the SAR shows a marked increase in the lower soil profile during our 3-year study. The area affected by increased SAR as well as the absolute values of SAR have increased. Since salinities are also increasing, the increased SARs should not be a limiting factor affecting the sustainability of long term use of shallow groundwater for crop production.

4. Summary and conclusions

SAR was measured over 3 years in a field that was managed according to a shallow groundwater management strategy designed to reduce drainage volumes by restricting drain flows and inducing the consumption of shallow groundwater by crops. Our objectives were to characterize the spatial and temporal distributions of SAR, and to assess the potential impact of groundwater management programs on SAR under field conditions.

We found a large section in the middle of the field to have a much coarser soil texture than the rest of the field, and this texture variability proved to be critical to understanding the observed SAR distributions. SAR and clay content were highly correlated in the field: areas with high clay content had high SAR. SAR was relatively low and uniform across the field to a depth of 0.6 m, but both SAR and its field variability increased with depth, along with increases in total salinity.

We were able to discern two distinct soil SAR profile types, one where the SAR was relatively low and uniform with depth and the other where the SAR increased markedly with depth reaching a maximum near the bottom of the profile. The first profile was located in sandier regions and also near drain laterals where the apparent leaching fraction calculated by Shouse et al. (2006) was high and the upward flow of water was assumed to be lowest. The second profile type showing increasing SAR with depth was found in the finer texture soils where there was less leaching and more potential for upward flow. During our 3-year study period, the percentage of the field where the lower profile depth averaged SAR was greater than 8 increased from 60 to 80%, while percentage with SAR greater than 10 increased from 20 to 40%. The increase in SAR was accompanied by an increase in solution salinity indicating that SAR would not be a limiting factor in the design and management of shallow groundwater systems and that the technology should be sustainable into future seasons. Ongoing monitoring of the field would be helpful.

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