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Monitoring management-induced spatio–temporal changes in soil quality through soil sampling directed by apparent electrical conductivity

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Abstract

Characterizing spatial and temporal variability of soil properties at field and landscape scales is tremendously important for a variety of agronomic and environmental concerns including solute transport modeling of non-point source pollutants in the vadose zone, site-specific crop management, and soil quality assessment, to mention a few. Currently, there is a global need to develop tools that evaluate the overall quality of soil to determine the effectiveness and sustainability of farm-management practices. The knowledge now exists for characterizing the spatial variability of soil quality with non-invasive geophysical measurements of apparent soil electrical conductivity (EC_a) using mobile GPS-based systems. The objective of this research was to evaluate EC_a-directed soil sampling as a means for monitoring management-induced spatio–temporal changes in soil quality. Appraisal was made of a specific management practice, the reuse of irrigation drainage water applied to a saline–sodic soil in central California.

A soil quality assessment study was conducted on a 32.4-ha saline–sodic field comprised of 8 rectangular paddocks in California's San Joaquin Valley from August 1999 to April 2002. The study evaluated the spatio–temporal changes that had occurred as a result of irrigation with drainage water over that time period. Using geospatial electromagnetic induction (EMI) measurements of EC_a and a spatial response surface sampling design, 40 sites were selected that reflected the spatial variability of the EC_a measurements. Duplicate samples were taken at eight selected sites (one randomly selected site from each paddock) to study local-scale variability. At each site soil-core samples were taken at 0.3-m intervals to a depth of 1.2 m and analyzed for

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Abbreviations: θ_v, Volumetric water content ($m^3\ m^{-3}$); ρ_b, Bulk density ($Mg\ m^{-3}$); ANOVA, Analysis of variance; CEC, Cation exchange capacity; CV, Coefficient of variation; EC, Electrical conductivity ($dS\ m^{-1}$); $EC_{1:1}$, Electrical conductivity of a 1 to 1 soil to water extract ($dS\ m^{-1}$); EC_{25}, Reference EC at 25 °C ($dS\ m^{-1}$); EC_a, Apparent soil electrical conductivity ($dS\ m^{-1}$); EC_e, Electrical conductivity of the saturation extract ($dS\ m^{-1}$); EMI, Electromagnetic induction; EM_h, Electromagnetic induction measurement in the horizontal coil-mode configuration; EM_v, Electromagnetic induction measurement in the vertical coil-mode configuration; ESP, Exchangeable sodium percentage; GPR, Ground penetrating radar; GPS, Global positioning system; OM, Organic matter; pH_e, pH of the saturation extract; SAR, Sodium adsorption ratio; SJV, San Joaquin Valley; SP, Saturation percentage; SRS, Spatial response surface; TDR, Time domain reflectometry.

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28 physical and chemical properties. Maps created from a geographic information system (GIS) show spatio–temporal changes of four dynamic soil properties (salinity, sodium adsorption ratio, boron, and molybdenum) critical to soil quality, which were strongly and significantly correlated with EC_a.

Data from 1999 indicate the presence of high salinity, which increased with depth, high sodium adsorption ratio (SAR), which also increased with depth, and moderate to high B and Mo, which showed no specific trends with depth. The application of drainage water for 32 months resulted in leaching of B from the top 0.3 of soil, leaching of salinity from the top 0.6 m of soil, and leaching of Na and Mo from the top 1.2 m of soil. The leaching fraction over the time period from 1999–2002 was estimated to be 0.10. The level of salinity in the reused drainage water (i.e., 3–5 dS m^{-1}) allowed infiltration and leaching to occur even though high sodium and high expanding-lattice clay levels posed potential water flow problems. Preliminary spatio–temporal analyses from 1999–2002 indicate at least short-term feasibility of drainage water reuse from the perspective of soil quality when the goal is forage production for grazing livestock. The implications of this research extend beyond the provincial applications of assessing drainage water reuse in central California to the global potential of EC_a-directed soil sampling for evaluating farm-induced management ramifications on soil quality.

Keywords: Apparent soil electrical conductivity; Spatial variability; Landscape-scale processes; Precision agriculture; Salinity; Site-specific management

1. Introduction

Sustainable agriculture is globally recognized as a potentially viable means of meeting the future food demands of an ever growing world population because it seeks to balance agricultural productivity, economic stability, resource utilization and degradation, and environmental impacts. Soil resource management is one aspect of sustainable agriculture that is needed to overcome limitations to productivity while maintaining or enhancing environmental quality. An overall understanding of soil quality will allow management of the soil resource to ensure sustainable food, fiber, and feed production throughout the world. Currently, there is a global need for tools to evaluate the ramifications of soil resource management upon spatio–temporal changes in soil quality to ascertain sustainability of farm-management practices.

Ever since the ground-breaking work of Nielson et al. (1973), which characterized the spatial heterogeneity of soil–water properties for a 150-ha field at the University of California's West Side Field Station in the San Joaquin Valley (SJV), the spatial variability of soil physical and chemical properties has been a topic of major concern to soil scientists. The significance of spatial variability lies in the fact that it is a fundamental component of any landscape-scale soil-related issue including subsurface solute transport, site-specific crop management, and soil quality assessment, to mention a few. Soil quality assessment is based upon a spatio–temporal understanding of the variability of those indicator soil physical, chemical, and biological properties that influence the intended objective function of a soil.

There are a variety of methods for potentially characterizing soil spatio–temporal variability including ground penetrating radar (GPR), aerial photography, multi- and hyper-spectral imagery, time domain reflectometry (TDR), and apparent soil electrical conductivity (EC_a). Of these approaches, the most extensively investigated is EC_a (Corwin and Lesch, 2005a).

There are several reasons why geospatial measurements of EC_a are well-suited for characterizing spatial variability. Geospatial measurements of EC_a are reliable, quick, and easy to take. The mobilization of EC_a measurement equipment is easy and can be accomplished at a reasonable cost. Finally, and most importantly, EC_a is influenced by a variety of soil properties for which the spatial variability of each could be potentially established.

Corwin and Lesch (2005a) provide a compilation of the literature pertaining to the soil physical and chemical properties that are either directly or indirectly measured by EC_a. Because of the pathways of conductance, EC_a is influenced by a complex interaction of soil properties including salinity, saturation percentage (SP), water content, and bulk density (ρ_b). Saturation percentage and ρ_b are both directly influenced by clay content (or texture) and organic matter (OM). Furthermore, the exchange surfaces on clays and OM provide a solid–liquid phase pathway primarily via exchangeable cations; consequently, clay type and content (or texture), cation exchange capacity (CEC), and OM are recognized as additional factors influencing EC_a measurements. Measurements of EC_a must be interpreted with these influencing factors in mind. Another factor influencing EC_a is temperature. Electrolytic conductivity increases at a rate of approximately 1.9% per degree centigrade increase in temperature (Corwin, 2003). Customarily, electrical conductivity (EC) is expressed at a reference temperature of 25 °C, EC_{25}, for purposes of comparison.

Because of the influence of these soil properties on EC_a, the spatial distribution of EC_a within a field provides a potential means of mapping the variability of these and other related soil properties by using geospatial EC_a measurements to direct soil sampling. After the initial, largely observational work of the 1980s, the direction of research involving geophysical measurements of EC_a in soil gradually shifted to mapping within-field variation of EC_a to characterize the spatial distribution of properties that statistically correlated with EC_a. The mapping of within-field variation of EC_a to characterize the spatial distribution of soil properties has its roots in the early salinity mapping work of Halvorson and Rhoades (1976) and Rhoades and Halvorson (1977), who observed the relationship between maps of EC_a and soil seep patterns. The shift in the emphasis of field-related EC_a research from observation to directed-sampling design has gained momentum resulting in the accepted use of geospatial measurements of EC_a as a reliable directed-sampling tool for characterizing spatial variability at field and landscape scales Corwin and Lesch (2003, 2005a,b).

In the past the characterization of soil spatial variability using EC_a-directed soil sampling has focused on four different landscape-scale applications: (i) solute transport modeling in the vadose zone (Corwin et al., 1999), (ii) site-specific crop management (Corwin et al., 2003a; Corwin and Lesch, 2005a), (iii) deep drainage flux (Triantafilis et al., 2003, 2005), and (iv) soil quality assessment (Johnson et al., 2001; Corwin et al., 2003b; Corwin and Lesch, 2005c; Triantafilis and Lesch, 2005). The application of EC_a-directed soil sampling to characterize soil quality has mainly occurred in the Great Plains and arid Southwest of the USA and in Australia. Using EC_a maps to direct soil sampling, Johnson et al. (2001) and Corwin et al. (2003b) spatially characterized the overall soil quality of physical and chemical properties thought to affect yield potential. To characterize soil quality, Johnson et al. (2001) used a stratified soil sampling design with allocation into four geo-referenced EC_a ranges. Correlations were performed between EC_a and the minimum data set of physical, chemical, and biological soil attributes proposed by Doran and Parkin (1996). Their results showed a positive correlation of EC_a with percentage clay, ρ_b, pH, and $EC_{1:1}$ over a soil depth of 0–30 cm, and a negative correlation with soil moisture, total and particulate organic matter, total C and N, microbial biomass C, and microbial biomass N. Corwin et al. (2003b) characterized the soil quality of a saline–sodic soil using a spatial response surface (SRS) sampling design. A positive correlation (significant at the 0.01 level) was found between EC_a and the properties of volumetric water content (θ_v); electrical conductivity of the saturation extract (EC_e); Cl^-, NO_3^-, SO_4^{--}, Na^+, K^+, and Mg^{++} in the saturation extract; SAR (sodium adsorption ratio), exchangeable sodium percentage (ESP); B; Se; Mo. A negative correlation (significant at the 0.01 level) was found for $CaCO_3$, inorganic C, and organic C. Most of these properties are associated with soil quality for arid zone soils. The high positive and negative correlations indicated that the spatial variability of these soil properties was accurately characterized by the SRS sampling design and predictable from the EC_a survey data. However, a number of other soil properties (i.e., ρ_b; percentage clay; pH_e; SP; HCO_3^- and Ca^{+2} in the saturation extract; exchangeable Na^+, K^+, and Mg^{+2}; As; CEC; gypsum; and total N) did not correlate well with EC_a measurements. To spatially characterize these soil properties a design-based sampling scheme using the EC_a measurements may be needed or random or grid soil sampling, which are highly sample and labor intensive, may be required.

The objective of the presented research is to evaluate EC_a-directed soil sampling as a basis for monitoring management-induced spatio–temporal changes in soil quality. Specifically, EC_a-directed soil sampling is used to characterize spatio–temporal changes in soil properties related to arid-zone soil quality at a drainage water reuse site. Characterizing the spatial variability of soil properties with EC_a-directed soil sampling is based on the hypothesis that when EC_a correlates with a soil property or properties, then spatial EC_a information can be used to identify sites that reflect the range and variability of the property or properties.

2. Materials and methods

This study is part of an on-going study of the sustainability of drainage water reuse on forage production for grazing livestock. Drainage water reuse provides an alternative method for the disposal of drainage water in California's central valley. The intended 8-year drainage water reuse study was initiated August 1999. After an initial soil quality assessment in August 1999, an assessment was conducted in April 2002. A third soil quality assessment was conducted in the winter of 2004 and a final soil quality assessment is scheduled for the spring of 2007. The data presented in this paper offers an exploratory analysis of the spatio–temporal trends in soil quality from August 1999 to April 2002.

2.1. Study site description

The study site is a 32.4-ha saline–sodic field located on Westlake Farm, in Kings County on the west side of California's SJV (Fig. 1). The soil at the Westlake Farm site is from the Lethent clay loam series (fine, montmorillonitic, thermic, Typic Natrargid; Arroves and Anderson, 1986). The field has been laser leveled with a slope of zero from north to south and a slope of 0.0009 m m^{-1} from west to east. A forage crop, Bermuda grass (*Cynodon dactylon* (L.) Pers.), was established in spring and summer of 2000. Livestock grazes on the forage crop during the months of active forage production (i.e., April to October).

The field is divided into eight 4-ha rectangular paddocks (or plots), each with dimensions of 75 by 364 m (Fig. 2). The borders of each paddock are defined by earth berms. A tile drain was installed at the center of each paddock. The central drain of each paddock serves an area of roughly 13,650 m^2. Aside from a center drain, each paddock has drains at the paddock's north and south boundaries to ensure that each paddock can be treated as an independent hydrologic unit. Each paddock was irrigated by flood irrigation from west to east guided by corrugated furrows. East-west tile-drainage lines were spaced 37.5 m apart at a depth of 1.2 m using a 7.6-cm diameter perforated plastic pipe with a gravel envelope. Further details of the site preparation and drainage water reuse project experimental design can be found in Kaffka et al. (2002). Instrumentation was installed to monitor drainage water flows and quality on the central drain in Paddocks 2, 3, 6, and 7.

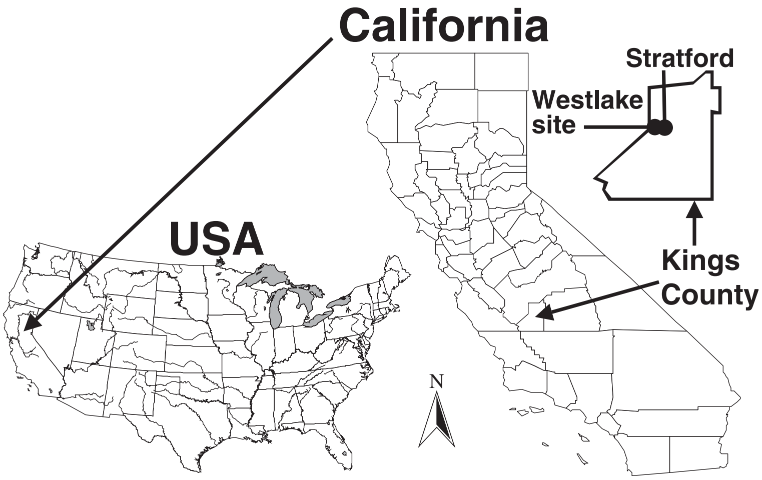

Fig. 1. Site map of the Westlake Farm study site.

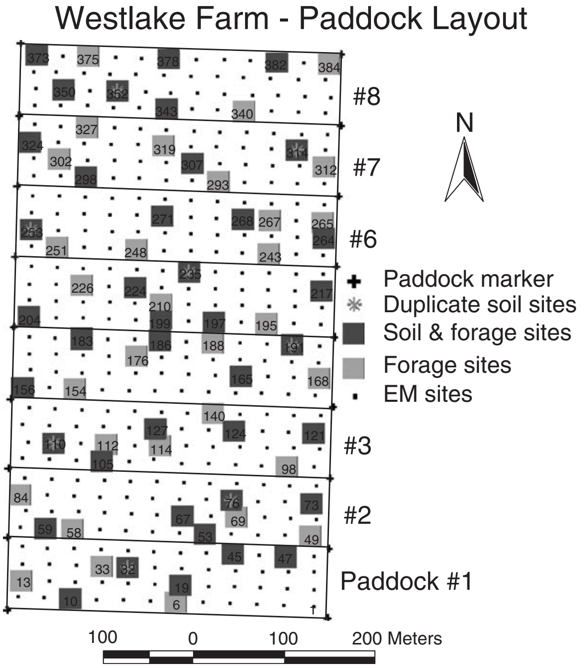

Fig. 2. Paddock layout showing the EMI sites, combined soil and forage sites, forage samples sites, and duplicate soil sites.

2.2. EC_a survey

The initial EC_a survey was conducted from 12–16 Aug. 1999 using mobile electromagnetic induction (EMI) equipment. This survey consisted of a grid of EC_a measurements arranged in a 4 (row) × 12 (position within row) pattern within each of the 8 paddocks for a total of 384 sites across the 32.4-ha study area (Fig. 2). The spacing between the 384 sites was approximately 20 m in the N–S direction and 30 m in the E–W direction. All 384 sites were geo-referenced using a Trimble Pro-XRS GPS receiver with sub-meter precision. At each site, EMI EC_a measurements were taken using a single-dipole EM38 unit[1] (Geonics Ltd., Mississaugua, Ontario, Canada) with the coil configuration oriented in the vertical (EM_v) and in the horizontal (EM_h) position. The horizontal coil configuration concentrates the reading nearer to the soil surface and penetrates to a depth of roughly 1 m, whereas the EMI reading in the vertical configuration penetrates to a depth of 1.5 m and is less sensitive to changes in EC_a near the surface. The moisture condition of the soil at the time of the EMI survey was approximately field capacity. The EC_a measurements taken in 1999 were used to establish the location of 40 sites where soil core samples were taken. A SRS sampling design was used to select the 40 sites from the EC_a data.

A follow-up EC_a survey was conducted from 8–12 Apr. 2002. In the 2002 EC_a survey, a dual-dipole EM38 unit[1] (Geonics Ltd., Mississaugua, Ontario, Canada) mounted on a PVC sled was pulled behind a mobile spray rig. Further details of this mobile EMI equipment can be found in Corwin and Lesch (2005a,b). The dual-dipole EM38 was coupled to a Trimble Pro-XRS GPS and a data logger. The dual-dipole EM38 unit permitted near-continuous, simultaneous EC_a measurements in both the horizontal (EM_h) and vertical (EM_v) dipole configurations collected at a 3-s time interval between pairs of readings. Drift runs were conducted periodically (i.e., roughly every 3 h) throughout the day to adjust for temperature effects on the EC_a measurements. The entire survey consisted of 22,177 sets of EM_h and EM_v measurements of EC_a. With each pair of EM_h and EM_v measurements an associated GPS reading was taken. As was the case in the 1999 EMI survey, the moisture condition of the soil at the time of the EMI survey was approximately field capacity. From the EC_a results taken in 2002, 40 EMI survey locations that were within 1–2 m of the original sampling positions taken in 1999 were identified and used to pinpoint the location of the 40 soil core sample sites.

2.3. Statistical methodology: EC_a-directed soil sampling and analysis of change of spatial data

Currently, two EC_a-directed soil sampling approaches are used: (i) design-based sampling and (ii) model-based sampling. The former consists primarily of the use of unsupervised classification, whereas the latter typically relies on optimized SRS sampling designs (Corwin and Lesch, in press-b). Nathan (1988) and Valliant et al. (2000) discuss the merits of design (probability) and model (prediction) based sampling strategies in detail. Specific model-based sampling approaches, having direct application to agricultural and environmental survey work, are described by

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[1] The citation of particular products or companies is for the convenience of the reader and does not imply any endorsement, guarantee, or preferential treatment by the U.S. Department of Agriculture.

McBratney and Webster (1981), Lesch et al. (1995a,b), van Groenigen et al. (1999), and Lesch (2005).

In the model-based SRS sampling approach (Lesch et al., 1995b; Lesch, 2005), a minimum set of calibration samples are selected based on the observed magnitudes and spatial locations of the EC_a data, with the explicit goal of optimizing the estimation of a regression model (i.e., minimizing the mean square prediction errors produced by the calibration function). The basis for this sampling approach stems directly from traditional response surface sampling methodology (Box and Draper, 1987). Further details concerning the implementation of this sampling approach can be found in Lesch (2005).

The goal of a SRS sampling design is two fold. In addition to selecting site locations that can be used to optimize the fitting of a spatial regression model, the design simultaneously attempts to select a set of locations that are more representative of the total spatial variation for the targeted soil property or properties than can normally be achieved using classical randomization schemes. When samples from these locations are collected repetitively (across two or more time frames), more traditional analysis of variance (ANOVA) modeling techniques can be applied, conditional on the assumption that the selected locations adequately represent the spatial distribution of the targeted soil property. This latter assumption is reasonable provided the targeted soil property is well correlated with the EC_a survey data.

One type of ANOVA model that is well suited for analyzing soil sample data across time frames is the classic two-way mixed linear model (with interaction). This type of model can be effectively used to screen for both static and/or dynamic changes in soil physical and chemical properties over time (Lesch et al., 1998). Let $i=1$, 2 represent the two time frames, $j=1, 2, \ldots, m$ represent the m sample site locations, and assume that replicate samples are available at some locations ($k=1$, 2), then the mixed linear ANOVA model is specified as

$$y_{ijk} = u + t_i + s_j + (ts)_{ij} + e_{ijk} \tag{1}$$

where y represents the soil chemical measurement taken at the jth site and ith time frame, u represents the overall mean estimate, t represents the average shift (or change in the mean) as a function of time, s represents the site (location) effect, ts represents site-by-time interaction effect, and e represents the residual error component. In the mixed linear model, s, ts, and e represent random variance components and typically are assumed to follow independent normal distributions (Montgomery, 1997). When analyzing spatial data, the ts component is sometimes assumed to follow a specific spatial covariance function. However, the distribution assumptions placed on the location effect are irrelevant if one is only interested in assessing change over time, since this effect is inconsequential.

Regardless of the specific distribution assumptions placed on the variance components, the mixed linear model can be used to generate two tests of interest; $t_1-t_2=0$ and $\Phi_{ts}^2=0$, where Φ_{ts}^2 represents the variance component associated with the ts interaction effect. The first test corresponds to a test for no change in the mean soil property level over time, while the second represents a test for no spatially dynamic redistribution in the location-specific soil property levels over time. These two statistical tests can be used to determine which one of the following four scenarios appears to be most likely, given the observed sample data: (i) no change over time, indicated by the corresponding hypothesis $t_1-t_2=0$ and $\Phi_{ts}^2=0$; (ii) static (spatially constant) change over time, indicated by $t_1-t_2\neq0$ and $\Phi_{ts}^2=0$; (iii) dynamic (spatially variable) change over time, but non-significant global shift, indicated by $t_1-t_2=0$ and $\Phi_{ts}^2>0$; and (iv) dynamic (spatially variable) change over time with a significant global shift, indicated by $t_1-t_2\neq0$ and $\Phi_{ts}^2>0$.

2.4. Soil core sampling

Utilizing the grid EC_a data from the 1999 EMI survey and SRS sampling algorithm incorporated into ESAP version 2.01 software (Lesch et al., 2000), 40 soil sample sites were selected that characterized the spatial variability in EC_a both across each paddock and over the entire field. The sample design was generated from EC_a data collected for the entire field, rather than separating the data by paddock. This provided a more reasonable statistical representation of the field-scale spatial variability, but resulted in a non-uniform number of sample sites in some paddocks. Conceptually, 40 sites were chosen to satisfy the following two criteria: (i) to represent about 95% of the observed range in the bivariate EMI survey data and (ii) to be spatially distributed across the 8 paddocks in an approximately uniform manner with about 5 sites within each paddock. Note that the final sampling design employed in this study resulted in 4 to 6 sample sites per paddock (see Fig. 2).

At each of the 40 sites, soil-core samples were taken at 2 points (i.e., two sets of soil cores per site) roughly 5 cm apart. Soil cores were taken at 0.3-m increments to a depth of 1.2 m. One set of soil cores was designated for soil chemical property analysis and the other set for soil physical property analysis. Within each paddock, one site was selected where duplicate soil-core samples were taken at 4, rather than 2, points to establish local-scale variability. Duplicate soil cores were taken at Sites 32, 76, 110, 191, 235, 253, 314, and 352 (Fig. 2). This resulted in 64 additional soil samples. A total of 384 soil samples were taken (160 soil chemical property samples, 160 soil physical property samples, 32 duplicate soil chemical property samples, and 32 duplicate soil physical property samples). Fig. 2 shows the location of all 384 EMI measurement sites, the 40 selected soil-core sites, and the 8 duplicate soil-core sites.

To observe spatio–temporal changes resulting from the application of drainage water, soil core samples were taken during the time period from 19–23 Aug. 1999, which was just after the initial EMI survey, and again 32 months later during the time period from 15–17 Apr. 2002, which was just after the follow-up EMI survey. The 40 locations for both sampling time periods (i.e., Aug. 1999 and Apr. 2002) were within 1–2 m of one another. All soil cores were kept in refrigerated storage prior to their preparation and analysis, which occurred within a few days after their collection.

2.5. Soil physical and chemical analyses

Soil quality has been defined by Doran and Parkin (1996) as “the capacity of a soil to function, within ecosystem and land-use boundaries, to sustain biological productivity, maintain environmental quality, and promote plant and animal health.” The soil cores were analyzed for a range of physical and chemical properties deemed important for the assessment of soil quality of an arid zone soil whose function was the production of forage for livestock. The soil chemical properties included: electrical conductivity of the saturation extract (EC_e); pH_e; anions (HCO_3^-, Cl^-, NO_3^-, SO_4^{--}) and cations (Na^+, K^+, Ca^{++}, Mg^{++}) in the saturation extract; trace elements (B, Se, As, Mo) in the saturation extract; $CaCO_3$; gypsum; cation exchange capacity (CEC); exchangeable Na^+, K^+, Mg^{++} and Ca^{++}; ESP; SAR; total C; and total N. The 192 soil samples designated for analysis of soil physical properties were analyzed for SP, volumetric water content (θ_v), ρ_b, and clay content. Total C and total nitrogen were analyzed, but only for the top two depth increments (0–0.3 and 0.3–0.6 m) for the 1999 samples because of limited analytical resources at that time. The trace elements Se and As were not analyzed for the 2002 samples because their low 1999 levels indicated that they were not a soil quality concern. The properties of ρ_b and clay percentage were also not determined for the 2002 samples because these are assumed to be static properties that are not expected to change over a 32-month period. The soil sample preparation, and chemical and physical methods used for each analysis were from ASA Agronomy Monograph No. 9 Parts 1 and 2 (Klute, 1986; Page et al., 1982), except for the inorganic and organic C, which were analyzed with a UIC Coulometric Carbon Analyzer[1], and total N, which was analyzed with a Leco C-N 2000 Analyzer[1].

2.6. GIS and map preparation

To display and manipulate the spatial data a geographic information system (GIS) was used. All spatial data were entered into the GIS with the commercial GIS software ArcView 3.1 (ESRI, 1992). Maps of soil properties were prepared using ordinary inverse-distance-weighting (IDW) interpolation. This interpolation procedure is generally less accurate than a more rigorous geostatistical kriging (or cokriging) procedure, but still suitable for use in an exploratory spatial assessment when used in conjunction with the mixed linear modeling tests described previously.

3. Results and discussion

Corwin et al. (2003b) provide a detailed discussion of the 1999 soil quality assessment at the Westlake Farm study site. Two findings came out of the initial assessment that are significant to the analysis of the spatio–temporal soil quality trends at this particular study site: (i) EC_e, SAR, Mo, and B were deemed the most important properties for evaluating the study site's soil quality and (ii) the spatial variability of those soil properties that did not correlate with EC_a (i.e., ρ_b; % clay; pH_e; SP; HCO_3^- and Ca^{++} in the saturation extract; exchangeable Na^+, K^+, and Mg^{++}; As; CEC; gypsum; and total N) were not accurately characterized by the EC_a-directed sampling approach.

Assessing soil quality depends upon the intended function of the soil. Sustainability of drainage water reuse at the Westlake Farm site will depend upon spatio–temporal changes to soil properties that either detrimentally impact forage production for grazing livestock or detrimentally impact livestock health. The chemical properties of EC_e, SAR, and B can detrimentally affect either forage production or quality. Increased salinity levels reflected by higher EC_e can reduce forage yields due to toxic ion (e.g., Na toxicity) and osmotic effects. The infiltration of low EC water into the root zone can be adversely affected by high SAR levels in soil reducing the available water for forage production. There is a narrow range over which B is beneficial to plants. Outside that range B will reduce plant yields. Whereas, Mo can detrimentally affect the health of livestock. The digestive system of ruminant livestock is adversely affected by forage containing elevated levels of Mo (i.e., >3–5 mg kg^{-1}). Barshad (1948) found that plants are able to adsorb amounts of Mo harmful to cattle from soils that contain as little as 1.5–5.0 mg kg^{-1} total Mo. These factors established EC_e, SAR, Mo, and B as the most important properties for evaluating the study site's soil quality.

Preliminary spatio–temporal trends in soil quality are presented for the first 32 months of an 8-year project intended to evaluate the sustainability of drainage water reuse as a viable alternative to the disposal of drainage water in California's SJV. The preliminary results are presented to show the spatio–temporal trends in soil quality that have occurred from August 1999 to April 2002.

Table 1
Correlation coefficients between EMI EC_a (both EM_h and EM_v) and soil properties measured over 0–1.2 m for 1999 and 2002

Soil property	EM_h EC_a	EM_v EC_a	EM_h EC_a	EM_v EC_a
	1999		2002	
θ_v (m^3/m^3)	0.62**[a]	0.64**[a]	0.59**	0.63**
ρ_b (Mg/m^3)	−0.35[a]	−0.31[a]	−0.34	−0.30
Clay (%)	0.29[b]	0.25[b]	0.33	0.28
EC_e (dS/m)	0.74**	0.78**	0.89**	0.84**
pH_e	0.01	−0.01	0.40**	0.43**
SP (%)	0.25	0.22	0.22	0.30
Anions in saturation extract (mmolc/L)				
HCO_3^-	0.05	0.06	0.56**	0.59**
Cl^-	0.34*	0.43**	0.77**	0.81**
NO_3^-	0.47**	0.34*	–	–
SO_4^{--}	0.81**	0.82**	0.87**	0.79**
Cations in saturation extract (mmolc/L)				
Na^+	0.76**	0.80**	0.88**	0.83**
K^+	0.69**	0.69**	0.74**	0.71**
Ca^{++}	0.18	0.23	0.07	0.13
Mg^{++}	0.74**	0.65**	0.74**	0.61**
Exchangeable cations (mmolc/kg)				
Na^+	0.19	0.21	0.68**	0.68**
K^+	−0.15	−0.15	−0.13	−0.01
Ca^{++}	−0.34*	−0.36*	0.12	0.15
Mg^{++}	0.14	0.14	0.14	0.12
SAR	0.64**	0.70**	0.84**	0.82**
ESP (%)	0.31*	0.32*	0.28	0.30
B (mg/L)	0.48**	0.42**	0.43**	0.40**
Se (:g/L)	0.56**	0.51**	–	–
As (:g/L)	0.15	0.15	–	–
Mo (:g/L)	0.58**	0.45**	0.45**	0.40**
$CaCO_3$ (g/kg)	−0.44**	−0.37**	−0.28	−0.28
Gypsum (g/kg)	0.19	0.18	0.13	0.10
Total C (g/kg)	−0.52**[,c]	−0.49**[,c]	−0.40**	−0.29
Total N (g/kg)	−0.13[c]	−0.10[c]	−0.21	−0.08

$N=40$.
[a] $N=31$.
[b] $N=41$.
[c] Measured over 0–0.6.
* Significant (test for *r*=0) at $P<0.05$ level.
** Significant (test for *r*=0) at $P<0.01$ level.

3.1. Correlation between EC_a and soil properties

Table 1 provides the 1999 and 2002 correlation coefficients between EMI EC_a (both EM_h and EM_v) and soil properties over the 0–1.2 m depth for 1999 and 2002. In most instances there has been little change between 1999 and 2002 with respect to the correlation between EC_a and the various soil properties. Significant correlation with EC_a exists for the properties of θ_v; EC_e; HCO_3^-, Cl^-, NO_3^-, and SO_4^{--} in the saturation extract; Na^+, K^+, and Mg^{++} in the saturation extract; SAR; B; Se; Mo; and total C. Poorly correlated properties include ρ_b; % clay; pH_e (poorly correlated in 1999 but not in 2002); SP; Ca^{++} in saturation extract; exchangeable Na^+ (poorly correlated in 1999 but not in 2002), K^+, and Mg^{++}; As; gypsum; and total N. Most properties that correlated with EC_a in 1999 tended to remain correlated in 2002. Of particular concern are the properties of EC_e, SAR, Mo, and B. All of these are significantly correlated with EC_a at the ≤ 0.01 level.

There are some soil properties that were significantly correlated with EC_a in 1999, but not in 2002, including exchangeable Ca^{++}, ESP, and $CaCO_3$. Properties that were significantly correlated to EC_a in 2002, but not in 1999, include pH_e, HCO_3^- in the saturation extract, and exchangeable Na^+. These temporal changes are not unexpected since they reflect the complex dynamics of the EC_a measurement. Additionally, these results point out that ground truth soil samples need to be collected during each EC_a survey in order to adequately infer changes in geospatial patterns.

3.2. General temporal trends in soil chemical properties

Tables 2–5 provide the 1999 and 2002 mean and range statistics for the 0–0.3, 0.3–0.6, 0.6–0.9, and

Table 2
Mean and range statistics for 0.0–0.3 m sample depth for 1999 and 2002

Soil property	Mean	Min.	Max.	SD	CV	Skewness	Mean	Min.	Max.	SD	CV	Skewness
	1999						2002					
θ_v (m^3/m^3)	0.30	0.21	0.48	0.06	20.5	0.88	0.29	0.23	0.48	0.04	18.64	0.95
ρ_b (Mg/m^3)	1.29	1.11	1.52	0.10	7.7	0.29	NM	NM	NM	NM	NM	NM
Clay (%)	35.9	22.8	48.3	6.8	19.1	−0.22	NM	NM	NM	NM	NM	NM
EC_e (dS/m)	13.0	5.6	35.7	7.5	57.8	1.81	11.43	4.83	30.60	6.06	53.05	1.74
pH_e	7.6	6.9	8.3	0.3	3.3	−0.04	7.67	6.53	8.18	0.36	4.66	−1.80
SP (%)	58.8	45.9	79.1	7.8	13.2	0.57	60.63	48.38	80.15	9.23	14.28	0.04
Anions in saturation extract (mmolc/L)												
HCO_3^-	5.25	2.78	10.71	1.97	37.6	1.03	2.81	1.51	5.97	0.93	32.92	1.14
Cl^-	21.81	3.29	67.25	15.18	69.6	1.45	18.30	3.14	63.68	14.38	78.60	1.75
NO_3^-	0.70	0.15	8.35	1.36	195.0	4.45	nd	nd	nd	–	–	–
SO_4^{--}	150.00	60.59	535.00	108.36	72.2	2.12	131.67	58.91	383.7	75.76	57.54	2.05
Cations in saturation extract (mmolc/L)												
Na^+	136.78	34.41	490.91	109.88	80.3	1.87	109.54	25.05	368.29	81.38	74.29	1.80
K^+	0.90	0.41	3.65	0.56	62.0	3.11	1.41	0.56	3.11	0.52	36.85	1.21
Ca^{++}	23.92	17.62	28.00	2.09	8.8	−0.59	25.07	21.88	30.50	2.31	9.20	0.62
Mg^{++}	18.60	8.60	100.74	17.56	94.4	3.32	15.93	9.10	54.30	9.24	58.01	3.26
Exchangeable cations (mmolc/kg)												
Na^+	58.8	21.8	172.0	27.8	47.3	1.65	7.06	1.93	19.11	4.15	58.74	1.01
K^+	10.9	3.1	15.5	2.3	21.1	−1.13	1.20	0.69	1.72	0.27	22.15	−0.14
Ca^{++}	85.8	0.0	170.7	43.9	51.2	0.01	59.27	36.22	88.67	16.64	28.07	0.29
Mg^{++}	62.7	31.5	86.8	12.6	20.1	−0.33	7.10	3.97	10.12	1.41	19.92	−0.37
SAR	28.2	8.3	70.2	16.5	58.7	1.08	23.46	5.62	59.50	14.40	61.39	1.22
ESP (%)	28.4	9.5	83.4	14.8	52.0	1.28	35.40	11.23	119.46	21.25	60.03	1.66
B (mg/L)	17.0	1.1	42.5	8.2	48.2	1.21	14.21	2.64	33.23	7.35	51.75	1.00
Se (:g/L)	8.8	0.0	77.0	12.7	144.8	3.76	NM	NM	NM	NM	NM	NM
As (:g/L)	8.2	0.0	54.0	12.4	151.5	1.81	NM	NM	NM	NM	NM	NM
Mo (:g/L)	862.3	442.0	3043.0	532.5	61.8	2.72	632.1	150.0	3291.0	592.1	93.66	2.70
$CaCO_3$ (g/kg)	10.8	0.1	32.3	8.1	75.5	1.00	15.0[a]	0.40[a]	55.7[a]	11.1[a]	74.07[a]	1.57[a]
Gypsum (g/kg)	34.1	11.1	78.0	17.5	51.4	0.68	37.3	2.6	100.4	21.3	57.03	0.94
Total C (g/kg)	7.3	3.2	13.5	1.3	66.0	1.06	0.81	0.30	1.13	0.20	25.13	−0.33
Total N (g/kg)	0.7	0.3	0.9	0.1	22.1	−0.54	0.07	0.04	0.11	0.02	20.96	−0.03

NM = not measured; nd = not detectable. $N=48$ for each sample year.

[a] $N=41$.

Table 3
Mean and range statistics for 0.3–0.6 m sample depth for 1999 and 2002

Soil property	Mean	Min.	Max.	SD	CV	Skewness	Mean	Min.	Max.	SD	CV	Skewness
	1999						2002					
θ_v (m^3/m^3)	0.40	0.29	0.52	0.05	11.8	0.36	0.34	0.27	0.50	0.03	13.03	0.75
ρ_b (Mg/m^3)	1.51	1.31	1.72	0.10	6.5	−0.03	NM	NM	NM	NM	NM	NM
Clay (%)	30.4	21.8	46.9	4.9	16.0	0.82	NM	NM	NM	NM	NM	NM
EC_e (dS/m)	20.2	13.5	34.5	5.3	26.0	1.05	17.46	6.11	34.00	6.55	37.48	0.72
pH_e	7.6	7.0	8.0	0.2	2.6	−0.89	7.87	6.63	8.23	0.33	4.22	−1.96
SP (%)	63.0	45.5	84.1	10.2	16.2	0.01	67.16	50.26	86.90	10.00	14.89	0.24
Anions in saturation extract (mmolc/L)												
HCO_3^-	2.67	0.80	4.92	0.71	26.8	0.40	2.07	1.27	3.50	0.47	22.74	0.82
Cl^-	35.30	12.94	68.98	14.33	40.6	0.50	30.22	3.71	80.89	16.51	54.65	0.72
NO_3^-	0.89	0.15	6.74	1.47	165.2	3.01	nd	nd	nd	–	–	–
SO_4^{--}	239.51	133.21	460.56	76.30	31.9	1.42	205.18	75.99	439.80	85.77	41.80	1.21
Cations in saturation extract (mmolc/L)												
Na^+	237.20	136.00	457.37	77.23	32.6	1.18	190.91	40.73	428.23	90.19	47.24	0.86
K^+	1.03	0.25	2.12	0.41	39.6	0.86	1.29	0.42	2.75	0.49	38.10	0.69
Ca^{++}	22.08	12.45	30.90	2.46	11.1	−0.29	23.73	17.88	29.64	2.22	9.36	0.59
Mg^{++}	20.43	10.00	77.47	12.31	60.3	3.39	20.06	10.10	69.28	12.12	60.42	2.67
Exchangeable cations (mmolc/kg)												
Na^+	79.0	47.1	138.9	19.9	25.2	0.82	11.04	1.85	21.00	4.16	37.69	−0.21
K^+	6.7	3.5	12.0	2.1	31.7	0.76	0.67	0.20	1.27	0.24	36.09	0.11
Ca^{++}	59.6	2.4	263.0	41.9	70.2	2.51	72.80	33.03	130.67	26.41	36.27	0.41
Mg^{++}	49.4	34.5	66.8	9.5	19.2	0.18	6.07	3.58	8.58	1.47	24.30	−0.03
SAR	51.4	30.3	89.5	12.9	25.1	0.87	40.31	9.13	78.87	15.31	37.99	0.08
ESP (%)	41.6	23.7	63.4	9.4	22.7	0.30	80.81	11.26	567.90	84.04	103.99	4.57
B (mg/L)	19.0	13.6	38.1	5.6	29.7	1.90	19.06	6.69	32.35	6.09	31.97	0.77
Se (:g/L)	14.0	0.0	45.0	8.5	60.9	0.79	NM	NM	NM	NM	NM	NM
As (:g/L)	8.8	0.0	58.0	13.2	149.6	1.86	NM	NM	NM	NM	NM	NM
Mo (:g/L)	750.5	180.0	2488.0	430.2	57.3	2.63	576.5	220.0	1783.0	375.8	65.18	1.94
$CaCO_3$ (g/kg)	10.4	0.0	57.6	10.7	102.8	2.23	15.7[a]	0.40[a]	58.8[a]	13.3[a]	84.88[a]	1.58[a]
Gypsum (g/kg)	53.7	11.4	142.7	32.1	59.8	0.94	51.7	6.3	117.5	30.4	58.88	0.59
Total C (g/kg)	4.3	1.9	15.1	1.3	87.3	2.41	0.49	0.19	0.98	0.19	38.70	0.45
Total N (g/kg)	0.5	0.2	3.8	0.5	106.0	6.47	0.04	0.03	0.08	0.01	22.47	0.70

NM = not measured; nd = not detectable. $N=48$ for each sample year.

[a] $N=41$.

0.9–1.2 m depth increments, respectively. The general temporal trend from 1999 to 2002 is a decrease in the mean at nearly all depth increments for EC_e; HCO_3^-, Cl^-, NO_3^-, and SO_4^{--} in the saturation extract; Na^+ in the saturation extract; exchangeable Na^+, K^+, and Mg^{++}; SAR; Mo; total C; and total N. In contrast, pH_e, K^+ and Ca^{++} in the saturation extract, exchangeable Ca^{++}, ESP, and $CaCO_3$ show an increase at nearly all depth increments from 1999 to 2002, whereas Mg^{++} in the saturation extract and gypsum stay nearly the same from 1999 to 2002. Boron decreases at the surface (i.e., 0–0.3 m), but increases in the bottom two depth increments, which reflects retarded transport of B due to adsorption and the subsequent leaching that ensues only near the surface.

Field-scale variation is indicated by the coefficient of variation (CV). In general, there is no substantial change in the values of CVs for each measured soil property from 1999 to 2002. Only exchangeable Ca^{++} consistently shows a change in CVs for all depths from 1999 to 2002 with CVs decreasing. The observed decrease in exchangeable Ca^{++} from 1999 to 2002 is most likely an analytical issue rather than

Table 4
Mean and range statistics for 0.6–0.9 m sample depth for 1999 and 2002

Soil property	Mean	Min.	Max.	SD	CV	Skewness	Mean	Min.	Max.	SD	CV	Skewness
	1999						2002					
θ_v (m^3/m^3)	0.40	0.23	0.51	0.05	12.4	−0.72	0.36	0.20	0.48	0.05	19.31	−0.69
ρ_b (Mg/m^3)	1.52	1.16	1.80	0.12	8.0	−0.42	NM	NM	NM	NM	NM	NM
Clay (%)	26.2	2.5	42.0	6.8	25.9	−0.55	NM	NM	NM	NM	NM	NM
EC_e (dS/m)	22.5	9.7	43.2	6.5	28.7	1.27	22.49	7.94	37.90	6.96	30.96	0.04
pH_e	7.6	7.3	8.1	0.2	2.3	0.32	8.04	7.48	8.34	0.22	2.68	−0.89
SP (%)	59.1	40.0	89.5	11.1	18.8	1.16	64.82	48.11	117.52	14.76	22.78	1.61
Anions in saturation extract (mmolc/L)												
HCO_3^-	2.62	1.10	9.01	1.18	44.8	3.61	2.24	1.23	4.70	0.73	32.47	1.38
Cl^-	47.07	5.48	100.48	21.82	46.3	0.68	47.15	5.28	93.60	24.40	51.74	0.14
NO_3^-	0.65	0.10	4.34	0.81	124.6	2.93	nd	nd	nd	–	–	–
SO_4^{--}	258.84	121.51	570.77	89.44	34.6	1.68	261.55	99.55	473.30	89.68	34.29	0.52
Cations in saturation extract (mmolc/L)												
Na^+	270.23	98.60	603.00	97.80	36.2	1.49	260.61	68.72	498.01	98.06	37.63	0.32
K^+	1.04	0.37	2.50	0.43	41.0	1.22	1.52	0.50	3.08	0.61	40.03	0.36
Ca^{++}	22.12	13.85	28.10	2.24	10.1	−0.79	24.43	20.48	29.38	2.06	8.43	0.33
Mg^{++}	19.07	9.95	46.37	7.80	40.9	1.60	21.95	9.78	55.11	9.70	44.17	1.00
Exchangeable cations (mmolc/kg)												
Na^+	78.8	35.2	132.2	23.8	30.2	0.51	11.68	1.99	31.15	5.32	45.55	1.04
K^+	4.7	1.8	12.4	2.3	48.5	1.35	0.37	0.08	1.29	0.22	58.51	2.09
Ca^{++}	45.5	0.0	173.9	36.8	80.9	1.20	85.69	32.50	160.94	32.61	38.06	0.40
Mg^{++}	43.2	27.2	70.6	11.1	25.7	0.86	4.84	2.38	7.99	1.54	31.79	0.46
SAR	59.0	24.0	107.6	16.6	28.1	0.76	53.35	16.26	91.90	16.00	30.00	−0.11
ESP (%)	47.5	28.9	95.0	13.5	28.4	1.18	81.74	13.33	179.19	36.57	44.74	0.58
B (mg/L)	17.5	9.4	31.3	4.8	27.2	0.87	21.49	11.17	34.19	5.84	27.17	0.51
Se (:g/L)	12.9	0.0	34.0	9.2	71.5	0.67	NM	NM	NM	NM	NM	NM
As (:g/L)	12.9	0.0	116.0	23.4	181.0	2.82	NM	NM	NM	NM	NM	NM
Mo (:g/L)	780.5	183.0	1756.0	338.9	43.4	0.90	661.6	252.0	2372.0	451.5	68.24	2.15
$CaCO_3$ (g/kg)	11.4	0.3	46.7	12.5	110.1	1.46	12.8[a]	0.20[a]	68.3[a]	14.3[a]	112.0[a]	2.46[a]
Gypsum (g/kg)	66.3	5.6	215.3	39.8	60.0	1.30	69.8	11.0	235.9	44.7	63.99	1.43
Total C (g/kg)	NM	NM	NM	NM	NM	NM	0.37	0.17	0.81	0.15	39.76	1.08
Total N (g/kg)	NM	NM	NM	NM	NM	NM	0.04	0.02	0.08	0.01	30.57	2.46

NM = not measured; nd = not detectable. $N=48$ for each sample year.

[a] $N=40$.

a natural phenomenon. The CVs, whether for 1999 and 2002, show a range of spatial variability with some general trends (see Tables 2–5). $CaCO_3$ has high CVs (i.e., CVs > 80) at all depths. Anions and cations in the saturation extract, except for Ca^{++}, exchangeable cations, EC_e, SAR, ESP, B, total C, and total N tend to have low to moderate CVs (i.e., CVs from 20 to 50). Gypsum has moderate CVs. Mo has moderate to high CVs (i.e., CVs from 50 to 80). Saturation percentage has low CVs (i.e., CVs from 10 to 20). The lowest CVs are for pH_e and Ca^{++} in the saturation extract.

A partitioning of the local- and global-scale variability of each soil property was performed by calculating the observed variation both within and between sites using a standard, one-way ANOVA model. The estimated within-site variation component was then divided by the total estimated variation (i.e., the sum of the within and between site variance components) to produce the partitioning estimates shown in Table 6. These results indicate that the greatest local-scale variability occurs at the 0.6–0.9 m depth increment for 2002 (Table 6). The properties showing the greatest local-scale variability are % clay,

Table 5
Mean and range statistics for 0.9–1.2 m sample depth for 1999 and 2002

Soil property	Mean	Min.	Max.	SD	CV	Skewness	Mean	Min.	Max.	SD	CV	Skewness
	1999						2002					
θ_v (m^3/m^3)	0.43	0.31	0.54	0.05	11.4	−0.37	0.41	0.29	0.51	0.06	20.84	−0.35
ρ_b (Mg/m^3)	1.51	1.14	1.75	0.16	10.7	−0.85	NM	NM	NM	NM	NM	NM
Clay (%)	23.3	11.1	36.9	6.3	26.9	0.08	NM	NM	NM	NM	NM	NM
EC_e (dS/m)	25.2	8.0	49.7	7.9	31.5	0.81	24.30	7.84	45.30	8.14	33.51	0.26
pH_e	7.6	7.1	8.0	0.2	2.7	−0.37	8.03	7.20	8.36	0.22	2.69	−1.47
SP (%)	58.7	38.5	93.3	12.9	22.0	0.68	61.29	41.12	122.95	15.28	24.93	1.91
Anions in saturation extract (mmolc/L)												
HCO_3^-	3.06	1.19	7.76	1.49	48.6	1.98	2.30	1.32	3.65	0.58	25.24	0.53
Cl^-	58.66	3.18	152.68	30.15	51.4	0.80	55.55	5.50	108.52	27.14	48.86	−0.17
NO_3^-	0.32	0.12	1.29	0.20	62.8	3.44	nd	nd	nd	–	–	–
SO_4^{--}	291.98	91.03	720.96	114.02	39.0	1.52	280.42	96.04	606.10	106.73	38.06	0.89
Cations in saturation extract (mmolc/L)												
Na^+	311.57	66.10	738.69	126.13	40.5	1.20	286.02	68.57	622.16	116.93	40.88	0.61
K^+	1.06	0.34	2.36	0.47	44.0	0.85	1.55	0.38	3.41	0.70	45.59	0.78
Ca^{++}	22.34	16.68	26.50	2.11	9.5	−0.64	24.98	19.88	29.89	2.41	9.64	0.29
Mg^{++}	22.30	9.33	52.05	10.42	46.7	1.27	22.59	7.84	50.96	9.81	43.43	0.92
Exchangeable cations (mmolc/kg)												
Na^+	87.7	32.4	192.0	32.0	36.5	1.10	12.05	3.07	21.78	4.20	34.81	−0.07
K^+	4.1	1.1	9.8	2.1	51.7	0.88	0.29	0.06	0.79	0.16	55.59	0.89
Ca^{++}	42.5	0.0	154.2	35.4	83.3	0.97	88.03	26.14	158.60	37.02	42.05	0.05
Mg^{++}	46.4	26.6	90.3	14.7	31.7	1.09	4.81	2.35	9.38	1.80	37.52	0.98
SAR	64.9	16.8	120.2	19.5	30.0	0.35	57.46	16.51	103.12	17.96	31.25	−0.03
ESP (%)	51.8	18.2	135.7	20.2	39.0	1.82	86.20	20.59	163.21	30.14	34.97	0.54
B (mg/L)	17.9	6.5	31.8	6.3	35.0	0.45	21.71	7.89	39.01	6.59	30.36	0.48
Se (:g/L)	14.1	0.0	62.0	13.9	98.5	1.80	NM	NM	NM	NM	NM	NM
As (:g/L)	4.4	0.0	30.0	8.1	183.3	2.04	NM	NM	NM	NM	NM	NM
Mo (:g/L)	946.9	330.0	2856.0	450.7	47.6	1.78	720.7	240.0	2991.0	451.5	62.65	3.11
$CaCO_3$ (g/kg)	12.7	0.6	64.8	14.6	115.2	1.79	14.3[a]	0.20[a]	50.3[a]	12.4[a]	86.67[a]	1.27[a]
Gypsum (g/kg)	64.1	0.8	251.4	46.4	72.4	2.06	84.6[a]	8.8[a]	269.5[a]	57.9[a]	68.49[a]	1.46[a]
Total C (g/kg)	NM	NM	NM	NM	NM	NM	0.39	0.14	0.87	0.17	44.34	0.93
Total N (g/kg)	NM	NM	NM	NM	NM	NM	0.03	0.02	0.08	0.01	37.37	2.26

NM = not measured; nd = not detectable. $N = 48$ for each sample year.

[a] $N = 43$.

pH_e, ESP, CEC, and gypsum. The lowest local-scale variability is for SAR. These results are largely consistent with those of 1999.

3.3. Spatio–temporal trends in EC_e, SAR, B, and Mo

The general spatio–temporal trends with depth for EC_e, Cl^-, SAR, B, and Mo were initially examined using mixed linear ANOVA modeling techniques. Table 7 shows the significance levels (p-values) corresponding to the F-tests for no dynamic spatial variation ($\Phi_{ts}^2 = 0$) and no shift in the mean level over time ($t_1 - t_2 = 0$) for each of these soil properties. In each model, the interaction variance component was assumed to follow an ordinary (non-spatial) normal distribution with mean 0 and variance Φ_{ts}^2. The test for a global shift in the mean level over time was based on 1 and 39 degrees of freedom and the spatially dynamic variation test was likewise based on 39 and 16 degrees of freedom, respectively. In all cases the F-tests (and corresponding p-values) are approximate, since only 20% of the sampling locations were replicated (and hence the design is unbalanced).

Table 6
Percent local-scale variation of soil properties by depth for 2002

Soil property	Soil depth increment (m)			
	0–0.3	0.3–0.6	0.6–0.9	0.9–1.2
θ_v (m^3/m^3)	5.3	5.9	3.9	2.1
ρ_b (Mg/m^3)	1.5	1.7	2.7	2.4
Clay (%)	13.8	20.4	13.6	8.6
EC_e (dS/m)	0.8	1.6	5.5	2.8
pH_e	7.1	2.2	14.4	9.1
SP (%)	2.1	2.8	5.8	0.9
SAR	1.0	1.3	3.7	2.1
ESP (%)	5.0	36.8	10.9	15.8
B (mg/L)	9.8	2.6	3.3	2.8
Mo (:g/L)	10.3	0.7	5.1	3.1
CEC (mmolc/kg)	10.8	25.1	9.9	2.0
$CaCO_3$ (g/kg)	6.5	14.1	0.5	6.4
Gypsum (g/kg)	10.9	4.7	11.5	5.8
Total C (g/kg)	4.6	5.7	2.8	6.6
Total N (g/kg)	4.8	5.6	8.2	1.2
Anions in saturation extract (mmolc/L)				
HCO_3^-	9.3	3.7	5.9	5.0
Cl^-	1.6	3.0	5.2	3.4
NO_3^-	–	–	–	–
SO_4^{2-}	0.9	1.1	4.5	2.3
Cations in saturation extract (mmolc/L)				
Na^+	0.9	1.4	4.9	2.5
K^+	3.5	1.5	5.1	1.9
Ca^{2+}	15.1	2.2	5.7	3.7
Mg^{2+}	1.1	2.3	7.8	3.9
Exchangeable cations (mmolc/kg)				
Na^+	2.7	4.0	4.8	4.0
K^+	6.8	5.6	2.8	2.4
Ca^{2+}	2.2	0.9	7.7	3.5
Mg^{2+}	3.8	1.4	4.5	0.5

In the case of EC_e, dynamic spatial variation is clearly present ($p<0.01$) in the 0–0.3 m depth increment, and also detected ($p<0.05$) in the 0.3–0.6 m depth increment. The mean levels are reduced in these two sample depth increments (see Tables 2 and 3). Table 7 shows that no statistically significant changes appear to have occurred below 0.6 m. The Cl^- test results appear to follow these same patterns, although three, rather than two, of the four F-tests are not statistically significant at the 0.05 level. Overall, both the salinity and chloride test results suggest that some preferential leaching of salts has occurred in the near surface (0.0–0.6 m) depth, but negligible leaching has taken place below this depth.

Dynamic spatial variation is clearly present ($p<0.01$) for SAR in the 0–0.3 m depth increment and also appears to be present ($p<0.05$) in the 0.3–0.6 m depth increment. The spatial variation F-tests narrowly miss being significant in the lower two depth increments. Unlike the EC_e and Cl^- levels, the mean SAR levels appear to be significantly reduced across all four sampling depth increments. For B, dynamic spatial variation appears to be clearly present ($p<0.01$) in all but the first sampling depth increment. The mean B level is significantly reduced in the 0–0.3 m depth increment, and significantly elevated in the 0.6–0.9 and 0.9–1.2 m depth increments. Finally, the mean Mo levels appear to be clearly reduced ($p<0.01$) in all but the 0.6–0.9 m depth increment, which narrowly misses being significant at the 0.05 level. However, the degree of dynamic spatial variation in the Mo pattern appears uncertain, as only one of the four dynamic variation tests can be judged to be statistically significant.

The spatio–temporal trends of EC_e, SAR, B, and Mo are the most significant properties to assess in an evaluation of the drainage water reuse impacts on soil quality at the Westlake Farm site because these properties have the greatest influence on soil quality due to their affect on forage productivity and quality and on livestock health. To visually evaluate their spatio–temporal trends, spatial baseline and spatio–temporal difference maps for EC_e, SAR, B, and Mo are shown in Figs. 3–6, respectively.

EC_e and SAR are closely linked and display similar spatial patterns and temporal changes (see Figs. 3 and 4, respectively). The 1999 spatial baseline patterns for EC_e and SAR are displayed in Figs. 3a and 4a, respectively. There are obvious spatial similarities between the two sets of baseline maps. These similarities are the consequence of the fact that Na^+ salts dominate the total salinity, so similarities would be expected. Figs. 3b and 4b show the changes that have occurred in EC_e and SAR, respectively, over the 32-min study period within the 0.0–0.3 m and 0.3–0.6 m depths. The blue areas represent areas of decreases in salinity and SAR, while red areas indicate areas of increases from 1999 to 2002. Comparison of Figs. 3b and 4b shows that spatial patterns are complex, but still reveal general similarities for the patterns of change. The general similarities suggest that leaching patterns are similar. The estimated field average

Table 7
F-test significance levels associated with mixed linear modeling result

Chemical property	Sample depth increment (m)	Pr>*F* for dynamic spatial variation	Pr>*F* for shift in mean level
EC_e	0–0.3	0.0002	0.0415
	0.3–0.6	0.0269	0.0006
	0.6–0.9	0.1610	0.8869
	0.9–1.2	0.3187	0.2700
Cl^-	0–0.3	0.0012	0.1503
	0.3–0.6	0.1414	0.0145
	0.6–0.9	0.5068	0.8244
	0.9–1.2	0.6510	0.2508
SAR	0–0.3	0.0003	0.0135
	0.3–0.6	0.0158	0.0001
	0.6–0.9	0.0670	0.0145
	0.9–1.2	0.0562	0.0006
B	0–0.3	0.8459	0.0022
	0.3–0.6	0.0051	0.8597
	0.6–0.9	0.0025	0.0001
	0.9–1.2	0.0084	0.0002
Mo	0–0.3	0.5153	0.0033
	0.3–0.6	0.0028	0.0022
	0.6–0.9	0.1158	0.0579
	0.9–1.2	0.0642	0.0076

leaching fraction calculated from both irrigation and drainage volumes and from salinity levels in the irrigation and drainage water was approximately 0.1 over the first 32 months of the study. The complex spatial patterns observed in Figs. 3b and 4b indicate leaching is not uniform across the field and preferential leaching patterns are clearly present. Some of the observed leaching patterns within the top 0–0.3 m reflect an association with micro-topographic effects, since areas of greatest decreases in EC_e and SAR tend to be associated with shallow depressions on the laser-leveled field. It is clear from the spatial patterns and from Table 7 that (i) leaching is occurring and (ii) the leaching pathways are complex and far from uniform. Furthermore, Tables 2–5 and 7 indicate that leaching occurs within the top 0.6 m for salinity and extends through the top 1.2 m for sodium.

The 1999 soil quality assessment of the research site by Corwin et al. (2003b) revealed high SAR and low saturated hydraulic conductivity (K_s). This combination of properties could potentially cause infiltration and soil aeration problems if the applied irrigation water has low salinity. However, the observed decrease in EC_e and SAR from 1999 to 2002 indicates that the reused drainage water is sufficiently high in salinity to prevent infiltration problems that could arise due to the dispersion of expanding lattice clays present at the site.

The dominant spatial feature of B distribution for the 1999 assessment is high B (i.e., >10 mg/L) in the southwest corner (Fig. 5a). Boron levels peak at the 0.3–0.6 m depth increment. The spatial extent of the high B in the southwest corner remains roughly the same over the 0–0.6 m depth increment and decreases in extent below 0.6 m. Mean B for the entire field

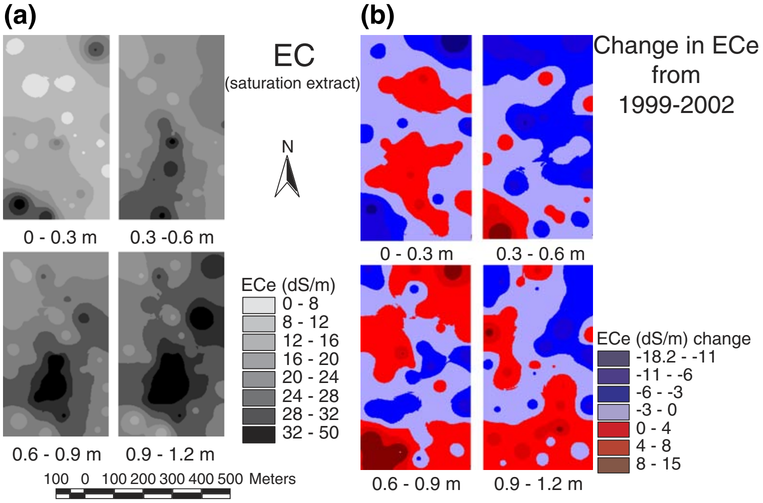
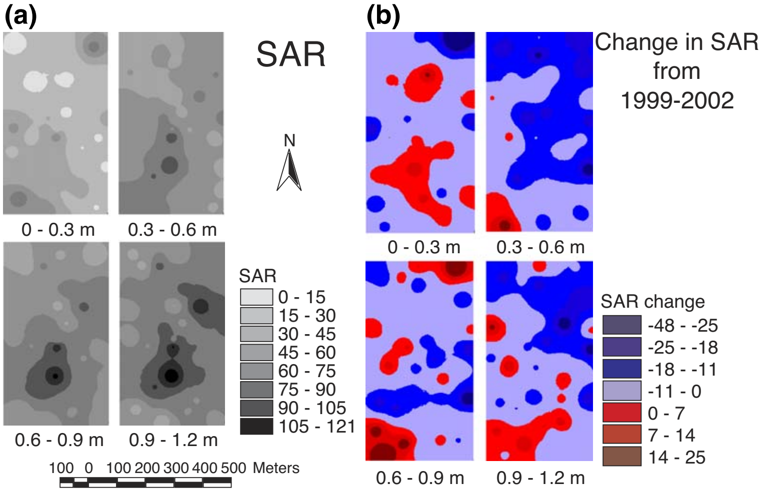


Fig. 3. IDW interpolated maps of (a) EC_e for 1999 at depth increments of 0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m and (b) change in EC_e from 1999 to 2002 at depth increments of 0–0.3 and 0.3–0.6 m.



decreases in the top 0–0.3 m by 16%, remains the same from 0.3–0.6 m, and increases below 0.6 m (Tables 2–5). The changing spatial patterns associated with these leaching and loading effects are shown in Fig. 5b. The spatial patterns of Fig. 5b at the 0–0.3 m depth increment show only gross similarity to the patterns in Figs. 3b and 4b at the same depth increment and few visual similarities below 0.3 m. The lack of similarity reflects the spatially complex nature of the soil physical and chemical properties influencing transport and adsorption of B within the root zone. The general spatio–temporal trend shown in Fig. 5b reflects the results shown in Tables 2–5 and 7, i.e., B is being leaching primarily from the top 0.3 m and to some extent from 0.3–0.6 m into the depths below 0.6 m where it is accumulating. The strong adsorption of B onto clay surfaces retards the movement of B causing it to leach only near the soil surface and to accumulate at the deeper depths (i.e., 0.6–1.2 m).

The dominant spatial features of Mo distribution for the 1999 assessment are high Mo (i.e., >1000 ug/L) in the southwest corner, which increases in spatial extent

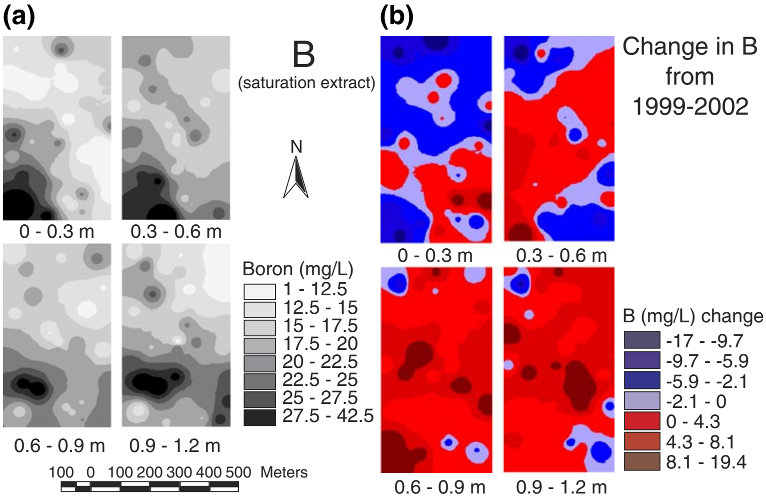


Fig. 5. IDW interpolated maps at depths increments of 0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m for (a) B in 1999 and (b) change in B from 1999 to 2002.

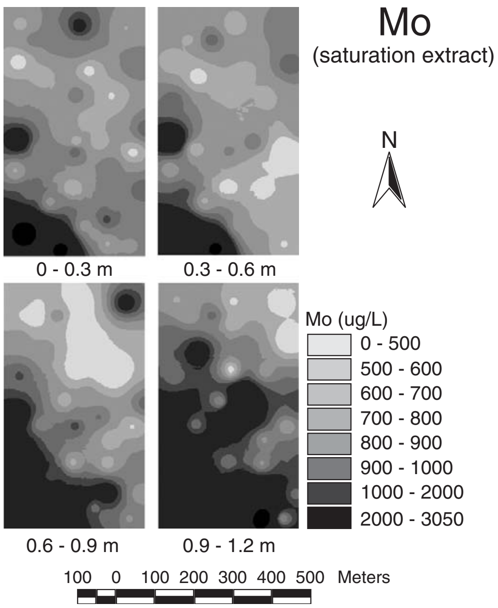


Fig. 6. IDW interpolated maps of Mo for 1999 at depth increments of 0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m.

with depth (Fig. 6), and a profile with the highest level of Mo from 0–0.3 m, decreasing Mo from 0.3–0.6 m, and then Mo increasing over the subsequent depth increments (Tables 2–5). Levels of Mo consistently decrease 15–27% from 1999 to 2002 for each depth increment (Tables 2–5). As shown in Table 7, this reduction appears to be consistently significant (or nearly significant) throughout the soil profile. Since the F-tests for dynamic spatial variation were generally not significant, spatio–temporal trend maps for Mo are not shown. The substantial and almost uniform decrease in Mo for each depth increment from 1999 to 2002 indicates that Mo is highly mobile in the saline–sodic soil at the study site.

Even though Mo is a mobile trace element, the fact that it is more readily leached than salinity is unexpected. There are three possible scenarios that might help explain the unexpectedly high observed decrease in field average Mo levels at each depth. The first is that pH and/or ionic strength effects are reducing the adsorptive affinity and capacity of the soil, thus causing Mo to be readily leached from the soil profile. The second is the formation of soluble Mo complexes that enhance the movement of Mo through the soil profile. Both of these scenarios would result in the removal of Mo by leaching, which reduces its threat to grazing livestock because the possibility of elevated levels of Mo in the forage is lowered. However, Mo in a closed drainage water reuse system would be returned to the system; consequently, Mo must be managed to isolate it in the lower soil depths with minimal leaching into tile drains to minimize its reintroduction. A third less likely scenario is that Mo reduction is related to plant uptake of Mo through the entire root zone (or at least below 0.6 m where little leaching is occurring). If this latter scenario is occurring, then a greater threat to grazing livestock might very well exist. Combinations of these scenarios may also be possible. Further investigation into this issue is ongoing through the periodic collection and analysis of forage samples.

4. Summary and conclusions

An assessment of spatio–temporal changes in soil properties related to soil quality was conducted to provide a preliminary evaluation of the sustainability of drainage water reuse on the west side of the SJV in an effort to ascertain its potential as an alternative for drainage water disposal. The Westlake Farm study site is representative of land that has been removed from productivity due to its poor soil quality (i.e., high salinity, high sodicity, and elevated Mo and B levels), but has potential value if properly managed. Proper management of a saline–sodic soil with elevated B levels requires a cropping strategy that uses salt and B tolerant crops and irrigation management that assures adequate infiltration and leaching to prevent additional buildup of salinity, sodicity, and trace elements (i.e., B and Mo) that will impair crop productivity and quality. The current study provides a unique opportunity to evaluate an alternative means of drainage water disposal, which is an issue of major concern in California's central valley.

Soil quality is a concept that has grown out of a concern for the future sustainability of agriculture (Warkentin, 1995). Sustainability implies maintenance or improvement over time. Evaluation of spatio–temporal changes in soil quality is one aspect of ascertaining the sustainability of drainage water reuse in the SJV. Preliminary results from this drainage water reuse study indicate general spatio–temporal soil quality trends that support the use of drainage water for production of a salt-tolerant forage crop. These trends include reduction in SAR and leaching of salinity and B from the near surface, while still maintaining infiltration. From 1999 to 2002 soil quality has generally improved due to the removal of salts, sodium, B, and Mo from the upper portion of the root zone. Results indicate that leaching of salinity occurred over the top 0.6 m of soil. Leaching of sodium reduced SAR, which will reduce dispersion and increase infiltration. The high adsorption affinity of B for clay surfaces has retarded its movement through the soil profile so that B leaching is only occurring in the top 0.3 m. The fairly uniform decrease in Mo with depth down to 1.2 m suggests that Mo is highly mobile in the saline–sodic soil system. Whether the reduction in Mo throughout the top 1.2 m is due entirely to leaching is unclear and remains to be established. However, the reduction in sodium from the top 1.2 m also suggests that more leaching may be occurring than is suggested by the reduction in salinity in the top 0.6 m.

It is cautioned that these are preliminary results indicating short-term trends and should not be taken as a confirmation of drainage water reuse in the SJV as a viable means of drainage water disposal. Furthermore, soil quality assessment is only one aspect of the overall drainage water reuse system. Sustainability of drainage water reuse on forage for grazing livestock can only be ascertained after a long-term (i.e., 5–10 years), holistic examination that looks not only at spatio–temporal changes in soil quality, but impacts on crop and livestock productivity and quality, environmental impacts, and economic sustainability.

Nevertheless, from a soil quality perspective several factors point to a high likelihood of sustainability of drainage water reuse on forage crops on the west side of the SJV. First, even though the saline–sodic soil at the study site has a low saturated hydraulic conductivity, the reused drainage water is sufficiently saline (i.e., 3–5 dS m^{-1}) to compensate for the high exchangeable Na^+ levels that may otherwise cause infiltration problems. This has resulted in leaching reflected by lower EC_e, SAR, B, and Mo levels within the top 0.3 m and deeper. Second, the soil at the study site contains expanding lattice clays that cause cracks to develop at the surface when the soil dries. These cracks provide infiltration pathways for water flow. Third, the root system of Bermuda grass helps to maintain water-flow pathways and enhances soil aggregation. Fourth, the drainage system provides an outlet to prevent the formation of a shallow water table that would limit the systems longevity. All of these factors aid in getting water flow through a soil that is prone to impermeability.

For landscape- and field-scale characterization of soil quality the use of random and regular interval sampling requires numerous samples beyond what is generally practical. To study spatio–temporal changes in soil quality the resource demands are even higher due to the need to document changes over time with repeated sampling at regular intervals. The spatial patterns obtained from geospatial EC_a measurements are an ideal means for directing soil sampling using either model- or design-based sampling designs. Geospatial measurements of EC_a provide one of the most cost-effective means of characterizing spatial variability, in particular spatio–temporal characterization of soil quality.

To add to the base of knowledge, new data and statistical analyses have been presented that demonstrate the flexibility and utility of EC_a-directed soil sampling as a basis for assessing management-induced spatio–temporal changes in soil quality. While only one type of management applied at one location was considered, the implication extends beyond the localized, though significant, finding that EC_a can be used to monitor drainage water reuse in a saline–sodic soil system. More importantly, when EC_a is correlated with soil properties associated with soil quality (and/or productivity), EC_a-directed soil sampling is an effective tool to broadly evaluate the spatio–temporal impact of management on soil resources. Assessment and interpretation guidelines are currently available to document the effects of current and alternative soil and crop management strategies on soil resources (Corwin and Lesch, 2003, 2005b; Corwin et al., 2003b).

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