Short-Term Sustainability of Drainage Water Reuse: Spatio-Temporal Impacts on Soil Chemical Properties

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Greater urban demand for finite water resources, increased frequency of drought resulting from erratic weather, and increased pressure to reduce drainage water volumes have intensified the need to reuse drainage water. A study was initiated in 1999 on a 32.4-ha saline-sodic field (Lethent clay loam series; fine, montmorillonitic, thermic, Typic Natrargid) located on the west side of California's San Joaquin Valley (WSJV) with the objective of evaluating the sustainability of drainage water reuse with respect to impact on soil quality. An evaluation after 5 yr of irrigation with drainage water is presented. Geo-referenced measurements of apparent soil electrical conductivity (EC_a) were used to direct soil sampling at 40 sites to characterize the spatial variability of soil properties (i.e., salinity, Se, Na, B, and Mo) crucial to the soil's intended use of growing Bermuda grass (*Cynodon dactylon* (l.) Pers.) for livestock consumption. Soil samples were taken at 0.3-m increments to a depth of 1.2 m at each site in August 1999, April 2002, and November 2004. Drainage water varying in salinity (0.8–16.2 dS m^{-1}), SAR (5.4–52.4), Mo (80–400 μg L^{-1}), and Se (<1–700 μg L^{-1}) was applied to the field since July 2000. An analysis of the general temporal trend shows that overall soil quality has improved due to leaching of B from the top 0.6 m of soil; salinity and Na from the top 1.2 m, but primarily from 0 to 0.6 m; and Mo from the top 1.2 m. Short-term sustainability of drainage water reuse is supported by the results.

Irrigation and drainage go hand in hand. This close association is evident in the concerns facing the highly productive irrigated agriculture of California. Irrigated agriculture in California's western San Joaquin Valley (WSJV) faces two primary concerns: maintaining sufficient irrigation water supplies and disposing of drainage water. Urban pressures for more water and an increased occurrence of drought due to erratic weather patterns have increased the public's scrutiny of irrigated agriculture's water demands, while rising water tables beneath agricultural lands in the WSJV and the lack of a drainage water disposal solution threaten crop productivity to the point where once productive lands are now fallow. Such concerns are not confined to California, but can be found throughout the world in areas like the Middle East and countries like China, Mexico, Pakistan, India, and other locations where irrigated agriculture occurs on marginal arid and semiarid soils (Hillel and Vlek, 2005).

It is well documented that drainage waters in the WSJV often contain high concentrations of salinity and trace elements (Deverel et al., 1984; Deverel and Fujii, 1988; Deverel and Millard, 1988; Fujii and Deverel, 1989; Fujii and Swain, 1995). Furthermore, these drainage waters can deleteriously affect plants, grazing livestock, and wildlife because of the presence of salinity and toxic trace elements such as As, B, Mo, and Se (Letey et al., 1986; Shannon, 1997). Currently, no comprehensive plan for drainage water disposal exists for the WSJV due to the closing of Kesterson Reservoir to protect waterfowl from the adverse effects of Se and the political and economic decision not to build the infrastructure (e.g., drain to the Pacific Ocean) needed to dispose of drainage waters. Disposal in surface waters is not a viable alternative because restrictive limits on the salinity of drainage waters disposed in surface waters have been imposed. Without a means of disposing of drainage water, current WSJV farm practices are not sustainable due to the buildup of salinity in the soil profile.

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Abbreviations: EC_a, apparent soil electrical conductivity; EC_e, electrical conductivity of the saturation extract (dS m^{-1}); EC_{dw}, electrical conductance of the applied drainage water (dS m^{-1}); EM, electromagnetic induction; EM_h, electromagnetic induction measured in the horizontal soil configuration; EM_v, electromagnetic induction measured in the vertical coil configuration; SAR, sodium adsorption ratio; WSJV, western San Joaquin Valley.

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The most common way to handle drainage water currently is disposal to ground water beneath irrigated lands or to permanent evaporation ponds. The soil quality of 340,000 ha of land in the WSJV is adversely affected by the presence of shallow or perched water within 0 to 1.5 m of the soil surface. These lands would benefit from the installation of subsurface tile drainage, and ground water degradation would be reduced. Existing solar evaporation ponds serve artificial drainage systems for only about 18,000 ha (Letey and Oster, 1993). Based on the experience of pond operators, 1 ha of pond serves about 10 ha of artificially drained land. If all lands with shallow water tables were drained, approximately 34,000 ha of land would be needed for evaporation ponds, an amount almost twenty times greater than that currently available. Even though less productive lands are generally used for evaporation ponds, 34,000 ha removes substantial land from productivity.

The reuse of drainage water on marginal lands is an alternative approach that can provide an additional source of irrigation water and reduce the volume of drainage water thereby reducing demand on water resources and need for evaporation ponds while bringing less- or non-productive lands back into productivity. Reusing drainage water would reduce its volume and the amount of land needed for its disposal by up to an order of magnitude (Oster, 1997), lowering the cost of disposal and reducing the exposure of wildlife to potentially toxic waters. However, the recycling of drainage water containing salts and trace elements would re-introduce potentially harmful chemical constituents that can degrade soil quality (e.g., salts and trace elements).

Increased salinity levels can reduce forage yields due to toxic ion (e.g., Na toxicity) and osmotic effects, while increased trace element levels (i.e., Se, As, Mo, or B) can threaten plants and livestock. Selenium is toxic to shore birds and migratory waterfowl when it concentrates in the food chain of evaporation ponds, some of which are productive artificial wetlands (Skorupa, 1998). Boron can reduce plant yields (Nable et al., 1997). Molybdenum can cause harmful effects to ruminant animals (Barshad, 1948). Barshad (1948) found that plants are able to absorb amounts of Mo harmful to cattle from soils that contain as little as 1.5 to 5.0 mg kg^{-1} total Mo.

Maintenance of soil physical properties is a further concern when reusing drainage water. As the level of sodicity increases, greater levels of salinity are required to prevent deterioration of water infiltration and redistribution, and aeration (Shainberg and Letey, 1984; Lima et al., 1990; Criscimanno et al., 1995; Oster and Jayawardane, 1998). Irrigation with saline-sodic drainage water ($3 < \text{EC} < 10$ dS m^{-1}, where EC = electrical conductivity; $5 < \text{SAR} < 35$, where SAR = sodium adsorption ratio) will result in soil salinities ranging from 3 to 30 dS m^{-1} and SARs ranging from 5 to 60 for a leaching fraction (LF) of about 30%. However, resulting levels of salinity should compensate for damage from increased levels of exchangeable Na, which otherwise would impair hydraulic conductivity and reduce soil aeration (Oster and Jayawardane, 1998). Infiltration rates can be maintained with gypsum applied to the soil surface before rainfall begins or before application of non-saline irrigation water (Kazman et al., 1983; Keren et al., 1990; Oster et al., 1999). Consequently, careful management of reused drainage water is of paramount importance.

There are several ways to reuse drainage water. Methods for reusing drainage water were developed and evaluated by Rhoades (1989), Bradford and Letey (1993), Grattan and Rhoades (1990), Cervinka (1994), and others. Posnikoff and Knapp (1996) provided a favorable economic assessment of the potential for reuse. Sequential reuse is the application of drainage water from one or more fields to irrigate salt-tolerant crops on another. Less tolerant crops might require cyclic (re)use of good quality water to establish the crop and drainage water reuse for part of or all of the remainder of the growing season. Blending drainage water with good quality irrigation water is another strategy and has been used by farmers in the Broadview Irrigation District (Wichelns et al., 1990). The simplest method to understand and manage is using drainage water down gradient from the fields where it is collected to establish and irrigate salt-tolerant perennial crops such as Bermuda grass.

Numerous papers have dealt with drainage water reuse (Westcot, 1988; Rhoades, 1989; Grattan and Rhoades, 1990; Ayars et al., 1993; Tanji and Karajeh, 1993; Willardson et al., 1997; Goyal et al., 1999; Mitchell et al., 2000; Grattan and Oster, 2002; Oster and Grattan, 2002; Grattan et al., 2004; Grieve et al., 2004). However, few long-term (i.e., more than 5 yr) evaluations of drainage water reuse have been conducted. A 9-yr evaluation of drainage water reuse on the yield of a cotton-safflower rotation by Goyal et al. (1999) found that detrimental effects on cotton lint yield did not occur until the fifth year by waters of salinity greater than 3000 mg L^{-1} (i.e., about 5 dS m^{-1}) as long as leaching occurred with pre-plant irrigations with low salinity water. However, no study has made a detailed evaluation of the short- (5 yr) and long-term (10 yr or more) spatio-temporal impacts of drainage water reuse on soil chemical properties.

High quality forages for dairy cattle, beef cattle, and sheep are in short supply in the Central Valley of California. Alfalfa acreage (the principal dairy forage) has been restricted due to water costs, urbanization, and competitive perennial crops. However, drainage water could be an abundant water resource. If forage of sufficient quality can be produced, the growing need for forage in the WSJV could be met by reusing drainage water on low-productivity, poor quality soils.

We hypothesize that, if managed appropriately, the quality and productivity of forage grown on previously low-productivity, saline-sodic soils of the WSJV can be maintained at sufficiently high levels to be sustainable when irrigated with drainage water. It is the objective of this paper to evaluate the sustainability of drainage water reuse on a saline-sodic soil in the WSJV from the perspective of the impact on soil chemical properties crucial to the soil's intended use of producing Bermuda grass for forage by livestock.

Methods and Materials

A 10-yr drainage water reuse study was initiated in August 1999. After an initial soil quality assessment in August 1999, a quarter-term assessment was conducted in April 2002, and a mid-term (5-yr) assessment was conducted in November 2004. A final

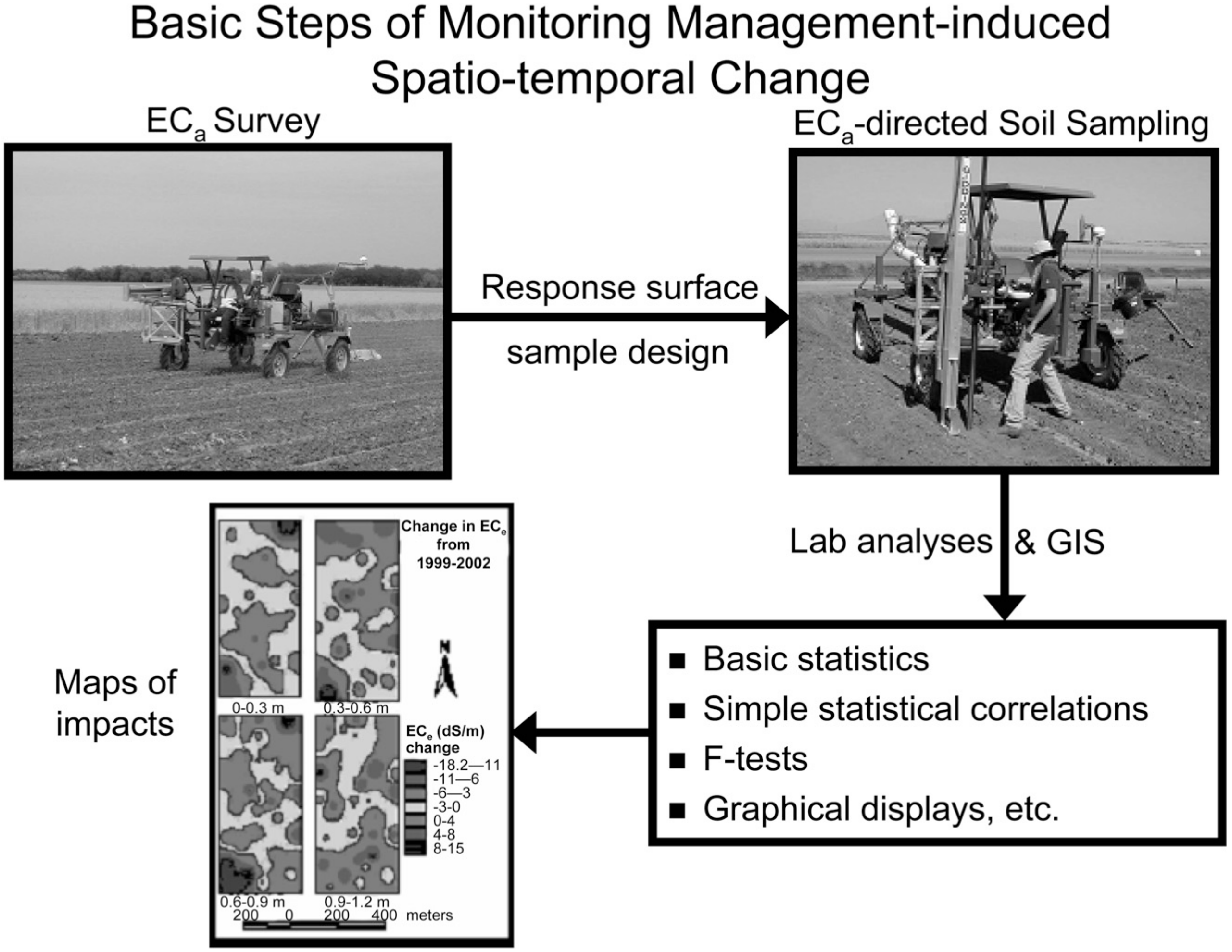

Fig. 1. Flow diagram showing the basic steps involved in characterizing spatial variability with apparent soil electrical conductivity (EC$_a$) directed soil sampling for monitoring management-induced spatio-temporal change.

evaluation is planned for spring or summer of 2009. The data presented in this paper offers an exploratory analysis of the spatio-temporal trends in soil quality that have developed from August 1999 to November 2004. This analysis will serve as a guide from which future detrimental or beneficial spatio-temporal changes can be evaluated by specifying trends of greatest potential concern.

Monitoring the spatio-temporal impacts of drainage water reuse on soil chemical properties was conducted using the guidelines and protocols developed by Corwin and Lesch (2003, 2005a, 2005b) and methodology developed by Corwin et al. (2006). The protocols by Corwin and Lesch (2005a) rely on geospatial measurements of apparent soil electrical conductivity (EC$_a$) to characterize soil spatial variability using EC$_a$–directed soil sampling. The EC$_a$–directed soil sampling approach for characterizing soil spatial variability consists of seven basic steps: (i) site description and EC$_a$ survey design, (ii) EC$_a$ data collection with GPS-based EC$_a$ equipment, (iii) soil sampling design directed by geospatial EC$_a$ data, (iv) soil core sampling at specified sites, (v) laboratory analysis of soil physical and chemical properties pertinent to the objectives, (vi) spatial statistical analysis to determine properties influencing EC$_a$, and (vii) geographic information system (GIS) database development and graphic display. Corwin et al. (2006) used these protocols to develop the methodology for monitoring management-induced spatio-temporal changes depicted in the flow diagram of Fig. 1.

Study Site Description

The study site is a 32.4-ha saline-sodic field (latitude 36° 11’ 24.827” N, longitude 119° 52’ 45.455” W) located on Westlake Farms, which resides in Kings County on the WSJV (Fig. 2). The field consists of 8 paddocks (75 m × 364 m), which enable livestock to be isolated away from paddocks that are being irrigated or are too moist to carry livestock traffic. The soil at the Westlake Farm site is part of the Lethent clay loam series (fine, montmorillonitic, thermic, Typic Natrargid; USDA, 1986). Details of the site preparation can be found in Kaffka et al. (2002) and Corwin et al. (2003).

Apparent Soil Electrical Conductivity (EC$_a$) Survey

Apparent soil electrical conductivity surveys were conducted roughly 30 mo apart from 1999 to 2004. The initial EC$_a$ survey was conducted from 12–16 Aug. 1999. This survey consisted of a grid of EC$_a$ measurements arranged in a 4 (row) × 12 (position within row) pattern within each of the eight paddocks for a total of 384 sites across the 32.4-ha study area (Fig. 3a). All 384

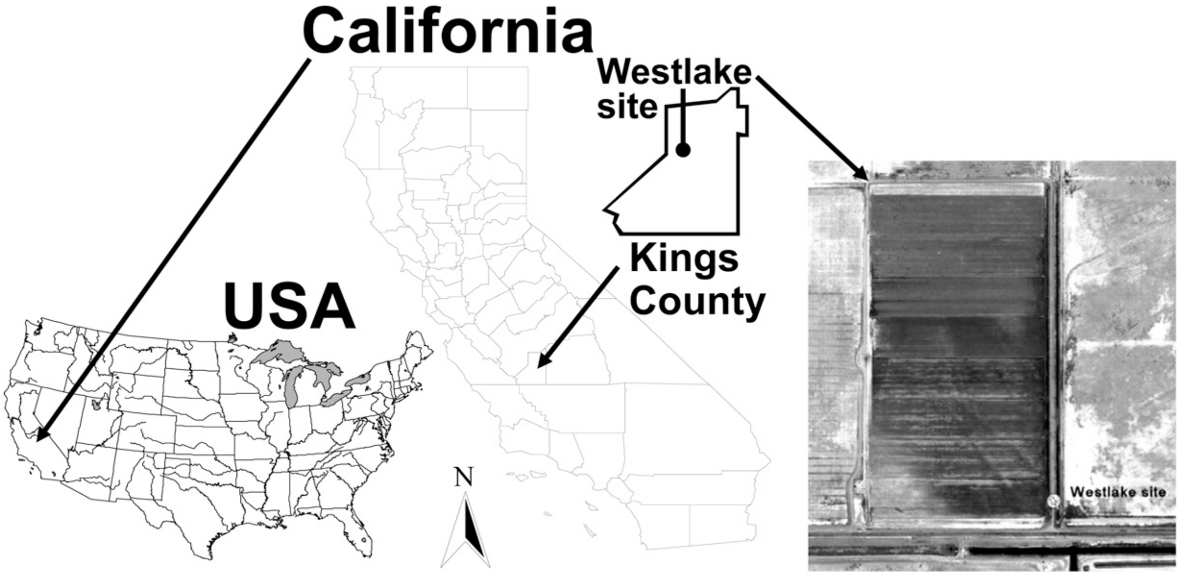

Fig. 2. Map showing location of Westlake Farm 32.4-ha study site on the west side of California's San Joaquin Valley.

sites were geo-referenced using a Trimble Pro-XRS GPS system with sub-meter precision. At each site, EM EC_a measurements were taken 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 0.75 to 1 m, whereas the EM reading in the vertical configuration penetrates to a depth of 1.2 to 1.5 m and concentrates the reading less at the surface. Before 2002, our EM equipment did not permit the continuous measurement of EC_a. Rather, only single measurements from one point to the next could be taken; consequently, mobile continuous-measurement electrical resistivity (ER) equipment was used to get a more detailed EC_a map by conducting an intensive survey with EC_a measurements taken at 7288 locations (Fig. 3b). Electrical resistivity provides a single EC_a measurement at each location with a depth of penetration dependent on the spacing between the electrodes and the electrode configuration (e.g., Wenner or Schlumberger). A Wenner array electrode configura-

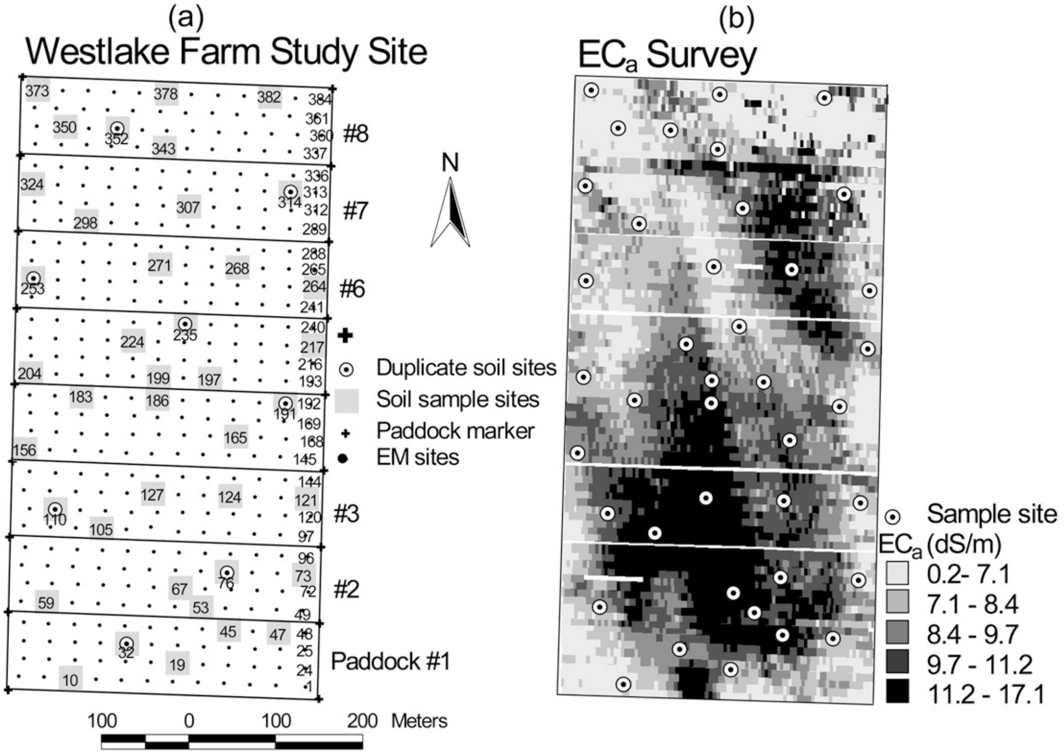

Fig. 3. Maps showing (a) paddock layout, electromagnetic induction (EM) measurement locations, and 40 soil sampling sites with duplicate sample sites and (b) electrical resistivity apparent soil electrical conductivity (EC_a) survey with soil sample sites shown with a bull's eye.

tion was used with a spacing of 1.2 m, which resulted in a depth of penetration of 1.2 m. Due to similar depths of penetration, the spatial patterns of the ER EC_a measurements are most comparable to those of the EM_v EC_a measurements.

The second EC_a survey was conducted from 8–12 Apr. 2002 using mobile EM equipment. In the 2002 survey, a dual-dipole EM38 unit (Geonics Ltd., Mississaugua, Ontario, Canada) mounted on a PVC sled was pulled behind a mobile spray rig with adequate clearance to traverse fields containing a crop cover. Further details of this mobile EM equipment can be found in Corwin and Lesch (2005a, 2005b). The dual-dipole EM38 was coupled to a Trimble Pro-XRS GPS and a data logger. The dual-dipole EM38 unit permitted continuous, simultaneous EC_a measurements in both the horizontal (EM_h) and vertical (EM_v) dipole configurations to be collected at a 3-s time interval between readings. The entire survey consisted of EC_a measurements (i.e., both EM_h and EM_v) taken at 22,177 locations within the field. An associated GPS reading was taken with each set of EM_h and EM_v measurements. The third EC_a survey was conducted from 14–15 Dec. 2004. This survey followed the same routine as used in the April 2002 survey.

Variations, if any, in the patterns of the geospatial EC_a measurements were used to indicate spatial changes in dynamic properties, such as salinity, which influence the EC_a measurement.

Statistical Sampling Methodology

Soil core sample sites were selected using geo-referenced EC_a survey data as a surrogate for the spatial variation of soil properties known to influence EC_a including soil texture, water content, salinity, organic matter, cation exchange capacity, bulk density, and organic matter. The sampling approach is discussed in detail in Lesch et al. (1995) and Lesch (2005). In this model-based sampling approach, 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) and minimizing the clustering of sample sites. This sampling approach stems directly from traditional response surface sampling design (RSSD) methodology (Box and Draper, 1987). A RSSD algorithm is the backbone of the ESAP software developed by Lesch et al. (2000), which is specifically designed for use with ground-based EM signal readings. For a given study site, soil chemical and physical properties measured at the sample locations specified by the RSSD that correlate with the associated EC_a will be spatially characterized. Those properties that do not correlate with EC_a will require some other sampling approach such as random, grid, or stratified random sampling to be spatially characterized.

Utilizing the grid EC_a data from the 1999 EM survey and ESAP 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 (Paddocks 5, 6, 7, and 8). Conceptually, 40 sites were chosen to satisfy the following two criteria: (i) to represent about 95% of the observed range in the bivariate EM survey data and (ii) to be spatially distributed across the eight paddocks in an approximately uniform manner with about five sites within each paddock. Note that the final sampling design employed in this study resulted in four to six sample sites per paddock (see Fig. 3).

Soil Core Sampling

At each of the 40 sites, soil-core samples were taken at two 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 four, rather than two, points to establish local-scale variability. Duplicate soil cores were taken at Sites 32, 76, 110, 191, 235, 253, 314, and 352 (Fig. 3a). 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). Figure 3a shows the location of all 384 EM measurement sites, the 40 selected soil-core sites, and the eight duplicate soil-core sites.

To observe temporal changes resulting from the application of drainage water, soil core samples were taken at the same 40 selected sample site locations at roughly 30-mo intervals: 19–23 Aug. 1999, 15–17 Apr. 2002, and 30 Nov.-3 Dec. 2004. All soil cores were kept in refrigerated storage before being air-dried and sieved (2-mm sieve), which occurred within a few days after their collection.

Soil Physical and Chemical Analyses

The soil cores were analyzed for a range of physical and chemical properties considered important for the assessment of soil quality of an arid zone soil when the goal was the production of forage. 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^{2+}, Mg^{2+}) 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^{2+}, and Ca^{2+}; exchangeable sodium percentage (ESP); SAR; total C; and total N. The 192 soil samples designated for analysis of soil physical properties were analyzed for saturation percentage (SP), volumetric water content (θ_v), bulk density (ρ_b), and clay content. Total C and N were analyzed for 1999 and 2004 samples, but not for the bottom two depth increments (0.6–0.9 and 0.9–1.2 m) of 1999 because of limited analytical resources at that time. The properties of clay % were also not determined for the 2002 and 2004 samples because these are static properties that are not expected to change over the study time 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; respectively), except for total C and N, which were analyzed with a Leco C-N 2000 Analyzer (Leco Corp., St. Joseph, MI).

GIS and Map Preparation

A geographic information system (GIS) was used to display and manipulate the spatial data. All spatial data were entered into the GIS with the commercial GIS software ArcView 3.3. Interpolated maps of the soil chemical properties most significantly influencing soil quality (i.e., EC_e, SAR, B, Mo, Se, and pH_e) were prepared using ordinary inverse-distance-weighting (IDW) interpolation. A comparison of IDW interpolation to kriging for all three sampling times using jackknifing showed a general improvement in prediction of the interpolated values using IDW; consequently, all interpolations were done using IDW.

Irrigation with Drainage Water

A forage crop of Bermudagrass (*Cynodon dactylon* (L.) Pers.) was established in the late spring and early summer of 2000. Once the Bermudagrass was established, the reuse of drainage water as the primary source of irrigation water began. Drainage water was first applied in July of 2000. Drainage water from an on-farm holding pond was applied to the field site usually from July to September for each of the 5 yr (i.e., 2000–2004). The eight paddocks allowed the livestock that fed on the Bermudagrass to be rotated from one paddock to the next to prevent the livestock from compacting soil in those paddocks that were being irrigated with drainage water. There were sufficient resources to continuously monitor only four of the eight paddocks. However, all eight paddocks received drainage water from the holding pond. The four paddocks were continuously monitored for drainage water volume applied and electrical conductance of the applied drainage water (EC_{dw}). The four monitored paddocks included Paddocks 2, 3, 6, and 7.

Table 1 shows dates of irrigation with drainage water, volumes of drainage water applied, and EC_{dw} for each of the four monitored paddocks over the 5 yr of the study. The electrical conductance of the applied drainage water ranged from 0.6 to 16.2 dS m^{-1}. The volume-weighted average EC_{dw}s applied to Paddocks 2, 3, 6, and 7 for all 5 yr were 4.08, 3.98, 4.27, and 4.15 dS m^{-1}, respectively. The overall volume-weighted average EC_{dw} applied was 4.13 dS m^{-1}. The reused drainage water varied considerably in electrical conductance due to (i) continuous fluctuations in drainage volumes and salinities from the lands draining into the holding pond as irrigation and cropping strategies changed, (ii) rotations in and out of productivity of lands widely ranging in their levels of soil salinity, and (iii) the fact that at times the holding pond had insufficient drainage water volume to meet demand; consequently, high quality (i.e., low salinity <1 dS m^{-1}) canal water was occasionally used for irrigation.

Table 2 shows a detailed analysis of the chemical composition of the reused drainage water. The detailed chemical analysis was conducted on water samples taken from the irrigation delivery system once each year during the months of July-August from 2000–2004. Available resources only allowed for the detailed analysis of a single water sample each year. The range in the chemical composition for those properties potentially influencing water quality varied with EC ranging from 1.80 to 16.26 dS m^{-1}, SAR from 5.4 to 52.4, B from 0.4 to 15.1 mg L^{-1}, Se from <0.01 to 0.70 mg L^{-1}, and Mo from 0.08 to 0.40 mg L^{-1}.

The drainage water composition tended to reflect the properties of the soil, irrigation management, and crop history of those lands in production that drained into the holding pond. Drainage water in the holding pond tended to be highest in salinity and in trace elements (B, Se, and Mo) during 2001 (see Tables 1 and 2) due to irrigation and subsequent drainage of poor quality lands. The drainage water in the holding pond during the years of 2000 and 2002–2004 was less saline and contained lower trace element concentrations because the holding pond received drainage water from lands that were of higher quality and more heavily leached.

Rainfall from 2000–2004 was not a significant input factor since this is an arid climate where most years were below average rainfall due to ongoing drought conditions in California's WSJV. The little rainfall that occurred did not penetrate due to high Na levels in the soil, which caused the rainfall to pond on the soil surface and subsequently evaporate.

Results and Discussion

A detailed discussion of the initial conditions at the Westlake Farm study site is provided by Corwin et al. (2003). The preliminary soil quality assessment of the study site from the 1999 EC_a survey and soil sampling revealed that the major properties of concern due to their potential direct or indirect impact on forage yield and quality included EC_e, SAR, B, and Mo, with potential concern for Se (Corwin et al., 2003). The discussion of results will primarily focus on these five soil chemical properties following a brief discussion of general temporal trends in soil properties that occurred from 1999 to 2004.

General Temporal Trends

Table 3 shows the correlations between EC_a measurements taken with EM in 1999 and soil physical and chemical properties for the three soil sampling times of 1999, 2002, and 2004. Correlations that are significant ($P \leq 0.05$) indicate those soil properties that were spatially characterized with the RSSD approach using the 1999 EM EC_a survey; as a result, the field means and spatial distributions of these significantly correlated properties can be considered reliable. Soil properties that are not significantly correlated with EC_a were not spatially characterized with the RSSD approach, which casts doubt on the accuracy of their field means. Rather, grid, random, stratified random, or some other sampling approach is needed to represent the within-field spatial variation of these properties. For all three sample times, salinity (i.e., EC_e), SAR, and B are significantly correlated with EC_a, as are θ_v, Cl^-, SO_4^-, Na^+, and Mg^{2+}. For 1999, the following properties are significantly correlated ($P \leq 0.05$) with EC_a for both EM_h and EM_v: θ_v; EC_e; the anions Cl^-, NO_3^-, and SO_4^-; the cations Na^+, K^+, and Mg^{2+}; the exchangeable cation Ca^{2+}; SAR; ESP; the trace elements B, Se, and Mo; $CaCO_3$; and total C. For 2002, the properties significantly correlated ($P \leq 0.05$) with 1999 EM EC_a for both EM_h and EM_v include θ_v; EC_e; pH_e; the anions Cl^-, HCO_3^-, and SO_4^-; the cations Na^+, K^+, and Mg^{2+}; exchangeable Na^+; SAR; and the trace elements B and Mo. For 2004, the properties correlated to 1999 EM EC_a for both EM_h and EM_v include θ_v; EC_e; pH_e; the anions Cl^-, NO_3^-, and SO_4^-; exchangeable Na^+ and Mg^{2+}; SAR; and B.

Table 1. Drainage water (DW) reuse volumes and electrical conductance (EC_{dw}) as applied to each of the four monitored paddocks from 2000–2004. NM = not measured.

	Paddock 2			Paddock 3			Paddock 6			Paddock 7		
Year	Date	DW	EC_{dw}	Date	DW	EC_{dw}	Date	DW	EC_{dw}	Date	DW	EC_{dw}
		m^3	dS m^{-1}		m^3	dS m^{-1}		m^3	dS m^{-1}		m^3	dS m^{-1}
2000	14–18 July	10,856	NM	14–18 July	10,856	NM	25 July	13,200	3.3	25 July	13,200	3.3
	4 Aug.	8019	3.0	4 Aug.	8019	3.0	5 Aug.	8019	3.0	5 Aug.	8019	3.0
	8–16 Sept.	7895	1.9	15 Sept.	7895	1.9	22 Aug.–16 Sept.	20,972	4.1	25–30 Aug.	20,972	3.6
2001	5 June	5929	8.7	5 June	5929	8.7	6 June	5929	8.7	7 June	5929	8.7
	18 July	4313	14.4	18 July	4313	14.4	18J uly	4140	14.4	19 July	4833	14.4
	2 Aug.	2697	11.5	2 Aug.	2697	11.5	3 Aug.	3439	11.5	2 Aug.	3606	11.5
	24 Aug.	3876	16.2	24 Aug.	3876	16.2	22 Aug.	4958	16.2	21 Aug.	4958	16.2
	14 Sept.	3543	NM	14 Sept.	3543	NM	12 Sept.	3474	NM	12 Sept.	3280	NM
	28 Sept.	2101	NM	28 Sept.	2101	NM	26 Sept.	2635	NM	26 Sept.	2614	NM
2002	7 June	3744	5.3	12 Mar.	2580	NM	Mar. 14	2580	NM	Mar. 14	3744	NM
	27 June	4244	5.6	25 Apr.	2829	4.0	11 June	3744	3.3	9–13 June	4743	3.3
	1 Aug.	4577	5.4	8 Aug.	3328	4.3	24–28 June	7905	7.0	1 Aug.	4327	3.7
	19 Aug.	5325	1.4	19 Aug.	5325	1.4	13 Aug.	4660	4.6	27 Aug.	4161	NM
	14 Sept.	5242	2.6	15 Sept.	2580	2.4	19 Aug.	5325	1.4	6 Oct.	5825	2.1
	4 Oct.	3162	2.0	25 Oct.	4077	1.9	15 Sept.	2580	2.4	18 Dec.	5076	NM
							18 Dec.	3328	NM			
2003	12 Apr.	2871	3.5	12 Apr.	3104	3.3	13 Apr.	3628	3.0	13 Apr.	4926	3.1
	23 May	3062	10.1	28 May	3578	3.2	24 May	5142	2.3	29 May	5142	4.0
	21–22 June	3420	1.4	22 June	3179	1.7	24–25 June	4011	1.3	3–4 July	5858	0.9
	26 July	4934	5.7	3 July	2588	0.9	13–14 July	4086	6.8	14 July	2804	4.0
	14 Aug.	3495	0.8	27 and 29 July	4411	5.1	30–31 July	4477	2.3	1 Aug.	4909	0.6
	3 Oct.	5750	1.7	14 Aug.	3029	0.7	16–17 Aug.	5392	0.9	6 Sept.	4185	2.0
				4 Sept.	3803	2.2	26 Aug.	3087	2.4	6 Oct.	4618	1.6
				4 Oct.	6540	1.7	6 Sept.	3695	1.9			
							5 Oct.	5192	1.7			
2004	30–31 Mar.	2162	10.6	30–31 Mar.	2162	10.6	30–31 Mar.	2162	10.6	30–31 Mar.	2162	10.6
	6–8 July	4249	2.1	6–8 July	4249	2.1	6–8 July	4249	2.1	6–8 July	4249	2.1
	25–28 July	4965	2.3	25–28 July	4965	2.3	25–28 July	4965	2.3	25–28 July	4965	2.3
	28–29 July	1745	2.1	28–29 July	1745	2.1	28–29 July	1745	2.1	28–29 July	1745	2.1
	13–18 Aug.	6322	2.0	13–18 Aug.	6322	2.0	13–18 Aug.	6322	2.0	13–18 Aug.	6322	2.0
	29–30 Aug.	7987	1.9	29–30 Aug.	7987	1.9	29–30 Aug.	7987	1.9	29–30 Aug.	7987	1.9
	1–2 Sept.	1869	1.8	1–2 Sept.	1869	1.8	1–2 Sept.	1869	1.8	1–2 Sept.	1869	1.8
	1–2 Sept.	2248	1.8	1–2 Sept.	2248	1.8	1–2 Sept.	2248	1.8	1–2 Sept.	2248	1.8

These correlations indicate that the RSSD sampling design did not accurately characterize the spatial distribution of ρ_b, clay percentage, HCO_3^-, Ca^{2+}, the exchangeable cations K^+ and Mg^{2+}, and As. Furthermore, the reuse of drainage water by flood irrigation caused

Table 2. Chemical composition of reused drainage water applied to Westlake Farm field site. Water samples were taken each year during the summer from 2000–2004.

	Date of reused drainage water sample				
Chemical parameter	22 Aug. 2000	24 Aug. 2001	13 Aug. 2002	26 July 2003	29 Aug. 2004
EC (dS m^{-1})†	4.33	16.26	4.65	5.08	1.80
pH	7.4	7.9	7.4	7.4	7.7
Na^+ (meq L^{-1})	23.7	219.3	28.0	37.4	11.5
K^+ (meq L^{-1})	0.5	0.5	0.5	1.3	0.2
Ca^{++} (meq L^{-1})	25.3	21.9	24.3	25.6	6.3
Mg^{++} (meq L^{-1})	13.5	13.1	14.7	10.6	1.2
SO_4^- (meq L^{-1})	54.6	204.3	59.1	65.5	12.1
Cl^- (meq L^{-1})	6.4	34.5	6.2	4.9	5.5
HCO_3^- (meq L^{-1})	1.2	1.5	1.2	1.3	1.3
NO_3^- (meq L^{-1})	<0.1	0.6	<0.1	<0.1	0.7
PO_4^{--} (meq L^{-1})	<0.1	<0.1	<0.1	<0.1	<0.1
SAR	5.4	52.4	6.3	8.8	5.9
B (mg L^{-1})	1.3	15.1	1.6	2.0	0.4
Se (µg L^{-1})	60	700	20	30	<1
As (µg L^{-1})	<1	<1	<1	<1	<1
Mo (µg L^{-1})	120	400	120	160	80

† EC = electrical conductance; SAR = sodium adsorption ratio.

a redistribution of the spatial patterns of several soil properties (i.e., pH_e, Se, and Mo), which departed from the spatial EC_a patterns observed in 1999, where pH_e was not significantly correlated in 1999 but was in 2004, and Se and Mo were significantly correlated in 1999 but not in 2004.

Tables 4 through 7 list the basic statistics of all measured soil properties for 1999 and 2004 at the depth increments of 0 to 0.3, 0.3 to 0.6, 0.6 to 0.9, and 0.9 to 1.2 m, respectively. These statistics show the extent of change in soil properties that occurred over the 5-yr period. The general temporal trend from 1999 to 2004 is a decrease in the mean at nearly all depth increments for EC_e, most anions and cations, SAR, B, As, Mo, and total C and N. The general downward trend of many dynamic soil properties, particularly EC_e, SAR, B, As, and Mo, reflects the effect of leaching that has occurred through the soil profile from the application of 2 to 10 dS m^{-1} drainage water to the saline-sodic field. From 1999 to 2004 Se is the only chemical property to show an increase at all depth increments and pH_e remains stable at 7.6 to 7.7 in the shallow depths (i.e., 0–0.6 m), but increases from 7.6 to 7.8 at 0.6 to 0.9 m and from 7.6 to 7.9 at 0.9 to 1.2 m. The increase in Se is presumably from low levels of Se present in the reused drainage water.

The coefficient of variation (CV) indicates the field-scale variation. There are few substantial changes in CVs from 1999 to 2004. Only Se and As show significant changes in CVs from 1999 to 2004 with two- to four-fold increases at most depths. On a relative basis CVs for θ_v, pH_e, ρ_b, SP, and Ca^{2+} in the saturation extract are low (i.e., CV < 27) and CVs for NO_3^- in the saturation extract, Se, As, and $CaCO_3$ are high (i.e., CV > 70).

A one-way ANOVA model was used to partition the local- and global-scale variability of each soil property by calculating the observed variation within and between sites. Table 8 shows the estimated local-scale variation of soil properties by depth for 2004. The results in Table 8 indicate that the greatest local-scale variation for most properties occurs at the 0.3- to 0.6-m depth increment for 2004. The properties having the greatest local-scale variability for the entire soil profile (i.e., 0–1.2 m) include ρ_b and clay percentage. The high local-scale variability for density and clay content are likely related to sampling at arbitrarily specified depths rather than diagnostic horizons. The lowest local-scale variability for the entire soil profile is for EC_e and SAR.

Spatio-Temporal Trends of EC_e, SAR, B, and Mo

The field means by depth for EC_e, SAR, B, and Mo are shown in Fig. 4 for all three sampling times. Salinity, as measured by EC_e, decreases at all depths for each of the three sampling times (Fig. 4a), resulting in an 11% decrease in salinity from 1999 to 2004 in the top 1.2 m soil profile on a mass basis. A comparable 11% decrease occurs for SAR from 1999 to 2004 with the sharpest decrease from 1999 to 2002 (Fig. 4b). Boron decreases 21% from 1999 to 2004 (Fig. 4c) and Mo decreases by 56% from 1999 to 2004 for the 1.2-m soil profile (Fig. 4d).

Adsorption of B and Mo is strongly dependent on pH (Goldberg et al., 2002, 2005). The maximum adsorption for B occurs around pH 9 (Goldberg et al., 2005). The maximum adsorption for Mo occurs in the pH range 2 to 4 with rapid decrease in adsorption with increasing pH (Goldberg et al., 2002). For the pHs of the soil in this study, B is more strongly adsorbed and is less mobile than Mo (Corwin et al., 1999). However, B does not adsorb to a great extent. From 1999 to 2002 B is leached from the top 0.3 m and accumulates deeper in the soil profile (i.e., 0.6–1.2 m). The strong adsorption of B to clay surfaces is responsible for its retarded movement and accumulation in the lower profile in contrast to salinity and Na. The greatest overall decrease of B occurs from 2002 to 2004. Molybdenum is very mobile. Each successive sampling shows a

Table 3. Correlation coefficients between electromagnetic induction EC_a (both EM_h and EM_v) and soil properties measured over 0 to 1.2 m for 1999, 2002, and 2004. *N* = 40.

	1999‡		2002		2004	
Soil property	**EM_h EC_a†**	**EM_v EC_a**	**EM_h EC_a**	**EM_v EC_a**	**EM_h EC_a**	**EM_v EC_a**
θ_v (m^3/m^3)	0.62**§	0.64**§	0.59**	0.63**	0.55**	0.54**
ρ_b (Mg/m^3)	−0.35§	−0.31§	−0.34	−0.30	0.08	0.07
Clay (%)	0.29¶	0.25¶	0.33	0.28	0.30	0.29
EC_e (dS/m)	0.74**	0.78**	0.89**	0.84**	0.90**	0.90**
pH_e	0.01	-0.01	0.40**	0.43**	0.55**	0.58**
SP (%)	0.25	0.22	0.22	0.30	0.22	0.19
Anions in saturation extract ($mmol_c$/L)						
HCO_3^-	0.05	0.06	0.56**	0.59**	−0.04	−0.01
Cl^-	0.34*	0.43**	0.77**	0.81**	0.80**	0.82**
NO_3^-	0.47**	0.34*	NM	NM	0.46**	0.53**
SO_4^-	0.81**	0.82**	0.87**	0.79**	0.88**	0.87**
Cations in saturation extract ($mmol_c$/L)						
Na^+	0.76**	0.80**	0.88**	0.83**	0.87**	0.88**
K^+	0.69**	0.69**	0.74**	0.71**	−0.37*	−0.30
Ca^{++}	0.18	0.23	0.07	0.13	−0.35*	−0.27
Mg^{++}	0.74**	0.65**	0.74**	0.61	0.71**	0.66**
Exchangeable cations ($mmol_c$/kg)						
Na^+	0.19	0.21	0.68**	0.68**	NM	NM
K^+	−0.15	−0.15	−0.13	−0.01	NM	NM
Ca^{++}	−0.34*	−0.36*	0.12	0.15	NM	NM
Mg^{++}	0.14	0.14	0.14	0.12	NM	NM
SAR	0.64**	0.70**	0.84**	0.82**	0.87**	0.88**
ESP (%)	0.31*	0.32*	0.28	0.30	NM	NM
B (mg/L)	0.48**	0.42**	0.43**	0.40**	0.68**	0.59**
Se (µg/L)	0.56**	0.51**	NM	NM	−0.14	−0.15
As (µg/L)	0.15	0.15	NM	NM	−0.15	−0.19
Mo (µg/L)	0.58**	0.45**	0.45**	0.40**	−0.13	−0.27
$CaCO_3$ (g/kg)	−0.44**	−0.37**	−0.28	−0.28	−0.31	−0.33
Gypsum (g/kg)	0.19	0.18	0.13	−0.10	0.15	−0.11
Total C (g/kg)	−0.52**#	−0.49**#	−0.40**	−0.29	−0.42	−0.30
Total N (g/kg)	−0.13#	−0.10#	−0.21	−0.08	0.22	0.29

* Significant (test for $|r| = 0$) at $P \leq 0.05$ level.

** Significant (test for $|r| = 0$) at $P \leq 0.01$ level.

† EC_a = apparent soil electrical conductivity, EM_h = electromagnetic induction measured in the horizontal soil configuration, EM_v = electromagnetic induction measured in the vertical coil configuration, θ_v = volumetric water content, ρ_b = bulk density, EC_e = electrical conductivity of saturation extract, sp. = saturation percentage, SAR = sodium adsorption ratio, ESP = exchangeable sodium percentage, NM = not measured.

‡ Taken from Corwin et al. (2003).

§ *N* = 31.

¶ *N* = 41.

\# Measured over 0 to 0.6 m.

Table 4. Mean and range statistics for 0.0–0.3 m sample depth for 1999 and 2004. *N* = 48 for each sample year.

	1999‡						2004					
Soil property	Mean	Min.	Max.	SD	CV	Skewness	Mean	Min.	Max.	SD	CV	Skewness
θ_v (m^3/m^3)†	0.30	0.21	0.48	0.06	20.5	0.88	0.32	0.22	0.39	0.03	9.1	−0.11
ρ_b (Mg/m^3)	1.29	1.11	1.52	0.10	7.7	0.29	1.03	0.80	1.31	0.11	10.6	0.34
Clay (%)	35.9	22.8	48.3	6.8	19.1	−0.22	35.1	21.5	48.8	7.1	19.9	−0.19
EC_e (dS/m)	13.0	5.6	35.7	7.5	57.8	1.81	11.17	4.48	28.50	5.58	50.0	1.16
pH_e	7.6	6.9	8.3	0.3	3.3	−0.04	7.68	6.73	8.85	0.43	5.4	−0.03
SP (%)	58.8	45.9	79.1	7.8	13.2	0.57	65.88	53.12	82.13	7.53	11.4	0.38
Anions in saturation extract ($mmol_c/L$)												
HCO_3^-	5.25	2.78	10.71	1.97	37.6	1.03	4.17	1.90	10.31	1.59	38.2	1.52
Cl^-	21.81	3.29	67.25	15.18	69.6	1.45	21.55	5.13	62.35	13.74	63.8	1.19
NO_3^-	0.70	0.15	8.35	1.36	195.0	4.45	0.22	0.0	2.79	0.64	291.7	3.46
SO_4^-	150.00	60.59	535.00	108.36	72.2	2.12	122.31	78.27	341.2	59.30	48.5	1.59
Cations in saturation extract ($mmol_c/L$)												
Na^+	136.78	34.41	490.91	109.88	80.3	1.87	106.98	25.74	366.45	70.81	66.2	1.40
K^+	0.90	0.41	3.65	0.56	62.0	3.11	1.48	0.72	3.51	0.57	38.2	1.59
Ca^{++}	23.92	17.62	28.00	2.09	8.8	−0.59	23.96	18.75	28.47	2.39	10.0	−0.11
Mg^{++}	18.60	8.60	100.74	17.56	94.4	3.32	19.30	13.05	37.86	5.39	27.9	2.34
Exchangeable cations ($mmol_c/kg$)												
Na^+	58.8	21.8	172.0	27.8	47.3	1.65	39.8	10.4	70.2	23.3	48.8	0.31
K^+	10.9	3.1	15.5	2.3	21.1	−1.13	13.1	4.1	17.2	1.7	20.2	−0.91
Ca^{++}	85.8	0.0	170.7	43.9	51.2	0.01	84.2	1.1	167.2	28.4	47.2	0.05
Mg^{++}	62.7	31.5	86.8	12.6	20.1	−0.33	65.1	34.7	95.4	12.2	23.3	0.02
SAR	28.2	8.3	70.2	16.5	58.7	1.08	22.8	5.5	65.9	14.1	62.0	1.03
ESP (%)	28.4	9.5	83.4	14.8	52.0	1.28	20.8	5.5	44.7	8.3	50.1	1.01
B (mg/L)	17.0	1.1	42.5	8.2	48.2	1.21	9.12	1.83	28.87	5.92	65.0	1.37
Se (µg/L)	8.8	0.0	77.0	12.7	144.8	3.76	57.0	0.0	399.0	89.0	156.9	2.15
As (µg/L)	8.2	0.0	54.0	12.4	151.5	1.81	1.00	0.0	37.0	5.0	601.9	6.68
Mo (µg/L)	862.3	442.0	3043.0	532.5	61.8	2.72	555.0	0.0	2484.0	510.0	92.7	2.32
$CaCO_3$ (g/kg)	10.8	0.1	32.3	8.1	75.5	1.00	10.1	0.44	55.1	10.9	80.1	1.60
Gypsum (g/kg)	34.1	11.1	78.0	17.5	51.4	0.68	33.3	4.4	89.4	20.2	55.5	0.69
Total C (g/kg)	0.7	0.32	1.35	0.13	66.0	1.06	0.88	0.32	1.20	0.22	26.11	−0.30
Total N (g/kg)	0.7	0.3	0.9	0.1	22.1	−0.54	0.10	0.06	0.13	0.03	24.2	−0.13

† θ_v = volumetric water content, ρ_b = bulk density, EC_e = electrical conductivity of saturation extract, sp. = saturation percentage, SAR = sodium adsorption ratio, ESP = exchangeable sodium percentage, NM = not measured.

‡ Taken from Corwin et al. (2003).

decrease with greater leaching of Mo occurring from the lower portion of the profile (i.e., 0.3–1.2 m). The slower removal of Mo from the top 0.3 m suggests bypass or greater retention due to adsorption. The change in field means from 1999 to 2004 for EC_e and Cl^- at the top two depth increments (see Tables 4 and 5) suggests that bypass may be occurring as indicated by the greater removal of salinity from the 0.3- to 0.6 m depth increment than from the 0- to 0.3-m depth increment. However, bypass is not substantiated since similar high concentrations near the surface would have remained over time for other solutes (e.g., Na^+, B) and similar concentration profiles would have developed if bypass had occurred. Furthermore, factors influencing increased Mo adsorption (e.g., pH or ionic strength) near the surface were not present to create the concentration distribution profile that occurred for Mo from 1999 to 2004. The recycling of Mo by plant roots may be a possible explanation. Since about 2002, an invasion of sweet clover (*Melilotus*), which contained high concentrations of Mo in tissue samples, may have recycled Mo. However, this remains to be confirmed as a plausible mechanism through more extensive plant tissue analysis, which is part of a complementary study involving drainage water reuse impacts on forage quality and quantity.

Figure 4 is particularly useful for understanding the trends that occurred over time for the field as a whole, but does not provide any insight into the spatial changes that occurred. Spatio-temporal changes in distribution for EC_e, SAR, B, and Mo are shown in Fig. 5a,b,c through 8a,b,c, respectively. Each figure contains three sets of four maps corresponding to the three sampling times (i.e., 1999, 2002, and 2004) and four depth increments (i.e., 0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m). Each map provides an areal distribution of the soil property at a specified depth increment and time. Each set of four maps consisting of the four depth increments at a specified time provides a snapshot in time of the property's three-dimensional distribution. The last set of maps in each figure (i.e., Fig. 5d, 6d, 7d, and 8d) indicates the net change in a soil property from 1999 to 2004, with light areas indicating a net loss and dark areas indicating a net gain. Figures 5 through 8 provide a visual means of evaluating the spatial baselines, spatio-temporal trends, and spatial net changes for EC_e, SAR, B, and Mo.

To establish the significance of changes by depth for dynamic spatial variation and for shift in the field mean over time, mixed linear ANOVA modeling techniques were used. Details of the ANOVA modeling approach are described in Corwin et al. (2006). Table 9 shows the significance levels (*p* values) by

Table 5. Mean and range statistics for 0.3–0.6 m sample depth for 1999 and 2004. *N* = 48 for each sample year.

	1999‡						2004					
Soil property	Mean	Min.	Max.	SD	CV	Skewness	Mean	Min.	Max.	SD	CV	Skewness
θ_v (m^3/m^3)†	0.40	0.29	0.52	0.05	11.8	0.36	0.34	0.29	0.41	0.03	7.3	0.44
ρ_b (Mg/m^3)	1.51	1.31	1.72	0.10	6.5	−0.03	1.12	0.93	1.30	0.08	7.0	−0.07
Clay (%)	30.4	21.8	46.9	4.9	16.0	0.82	29.4	21.5	46.1	4.8	15.5	0.88
EC_e (dS/m)	20.2	13.5	34.5	5.3	26.0	1.05	16.53	4.38	35.5	7.73	46.7	0.55
pH_e	7.6	7.0	8.0	0.2	2.6	−0.89	7.60	6.70	8.30	0.41	5.4	−0.44
SP (%)	63.0	45.5	84.1	10.2	16.2	0.01	65.09	44.32	98.89	11.95	18.4	0.50
Anions in saturation extract ($mmol_c$/L)												
HCO_3^-	2.67	0.80	4.92	0.71	26.8	0.40	1.90	1.05	3.53	0.48	25.1	0.82
Cl^-	35.30	12.94	68.98	14.33	40.6	0.50	31.59	4.92	106.04	20.12	63.7	1.20
NO_3^-	0.89	0.15	6.74	1.47	165.2	3.01	0.26	0.0	2.95	0.57	221.5	3.60
SO_4^-	239.51	133.21	460.56	76.30	31.9	1.42	193.00	57.10	428.8	95.31	49.4	0.92
Cations in saturation extract ($mmol_c$/L)												
Na^+	237.20	136.00	457.37	77.23	32.6	1.18	192.37	23.38	510.10	112.44	58.5	0.80
K^+	1.03	0.25	2.12	0.41	39.6	0.86	0.55	0.12	0.32	0.24	43.4	0.79
Ca^{++}	22.08	12.45	30.90	2.46	11.1	−0.29	21.99	14.69	26.43	2.04	9.3	−0.63
Mg^{++}	20.43	10.00	77.47	12.31	60.3	3.39	20.04	8.62	62.19	10.56	52.7	2.81
Exchangeable cations ($mmol_c$/kg)												
Na^+	79.0	47.1	138.9	19.9	25.2	0.82	62.7	23.6	141.2	25.3	27.7	0.96
K^+	6.7	3.5	12.0	2.1	31.7	0.76	4.4	2.4	13.3	2.4	34.1	1.55
Ca^{++}	59.6	2.4	263.0	41.9	70.2	2.51	60.7	5.6	234.8	38.8	40.1	1.98
Mg^{++}	49.4	34.5	66.8	9.5	19.2	0.18	54.4	30.5	70.2	11.3	25.5	0.36
SAR	51.4	30.3	89.5	12.9	25.1	0.87	41.3	5.2	100.2	21.3	51.4	0.41
ESP (%)	41.6	23.7	63.4	9.4	22.7	0.30	32.6	10.9	52.0	9.3	17.6	−0.04
B (mg/L)	19.0	13.6	38.1	5.6	29.7	1.90	14.15	1.39	33.73	7.50	53.0	0.78
Se (µg/L)	14.0	0.0	45.0	8.5	60.9	0.79	77.0	0.0	529.0	140.0	181.0	2.06
As (µg/L)	8.8	0.0	58.0	13.2	149.6	1.86	3.0	0.0	155.0	22.0	659.0	6.90
Mo (µg/L)	750.5	180.0	2488.0	430.2	57.3	2.63	360.0	0.0	1589.0	343.0	95.4	1.57
$CaCO_3$ (g/kg)	10.4	0.0	57.6	10.7	102.8	2.23	10.7	0.41	52.2	12.2	86.6	1.49
Gypsum (g/kg)	53.7	11.4	142.7	32.1	59.8	0.94	52.1	5.9	112.9	28.8	59.1	0.71
Total C (g/kg)	0.43	0.19	1.51	0.13	87.3	2.41	0.44	0.17	0.92	0.18	36.6	0.44
Total N (g/kg)	0.5	0.2	3.8	0.5	106.0	6.47	0.8	0.04	0.14	0.02	25.5	0.80

† θ_v = volumetric water content, ρ_b = bulk density, EC_e = electrical conductivity of saturation extract, sp. = saturation percentage, SAR = sodium adsorption ratio, ESP = exchangeable sodium percentage, NM = not measured.

‡ Taken from Corwin et al. (2003).

depth increment 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 EC_e, Cl^-, SAR, B, Mo, and pH_e.

General similarities are evident in the baseline spatial patterns of EC_e, SAR, B, and Mo (see Fig. 5a, 6a, 7a, and 8a, respectively), suggesting that these patterns are the consequence of a common process, most likely water flow through the root zone (i.e., 0–1.2 m). Profiles for each of these properties show increases with depth, which are typical of leaching in the presence of a crop. Furthermore, a general spatial pattern of higher levels in the south than the north, with a noticeable trend from southwest to northeast, suggests that high levels of Na in the southern half of the field, particularly in the southwest at the surface (0–0.3 m), dispersed the soil, which restricted infiltration and subsequent leaching, and resulted in the trend of high to low levels of EC_e, SAR, B, and Mo from southwest to northeast. Even though the spatial patterns for each property are complex, the general southwest-northeast patterns have held from 1999–2004, with changes occurring primarily in magnitude as a result of the leaching process.

Spatio-temporal changes of EC_e and SAR from 1999 to 2004 (see Fig. 5 and 6) are very similar, which is the consequence of the dominance of Na^+ salts to the total salinity. The changes that occur in EC_e and SAR from 1999 to 2004 show a general decrease in EC_e, particularly in the top 0.6 m. This is substantiated by the F-test significance levels of EC_e and SAR in Table 9, which show that the shift in mean level is significant for the top 0.6 m and may even extend as far as 1.2 m at various locations in the field. Even though the general spatial patterns appear stable from 1999 to 2004 in Fig. 5 a,b,c and 6 a,b,c, there are definite spatial changes occurring, which is substantiated by the dynamic spatial variation results of Table 9. Specifically, Fig. 5d and 6d indicate that leaching of salts and Na is occurring to a greater extent in the northern half of the field, with removal at all depths, while in the southern half and especially in the southwest corner there is an accumulation of salts and Na in the lower depths (i.e., 0.6–1.2 m). This pattern is most readily explained by the dispersion effect due to high SAR in the south, which creates infiltration and permeability problems. Even though the decrease in EC_e and SAR is not as dramatic in the south as the north, it is readily apparent that salts and Na are being leached throughout the field to a significant extent and the saline-sodic soil is being reclaimed.

The spatial baselines, spatio-temporal trends, and spatial net changes for B and Mo show some strong similarities, differing

Table 6. Mean and range statistics for 0.6–0.9 m sample depth for 1999 and 2004. *N* = 48 for each sample year.

	1999‡						2004					
Soil property	**Mean**	**Min.**	**Max.**	**SD**	**CV**	**Skewness**	**Mean**	**Min.**	**Max.**	**SD**	**CV**	**Skewness**
θ_v (m^3/m^3)†	0.40	0.23	0.51	0.05	12.4	−0.72	0.36	0.27	0.41	0.03	8.7	−0.84
ρ_b (Mg/m^3)	1.52	1.16	1.80	0.12	8.0	−0.42	1.11	0.88	1.29	0.15	13.4	−2.18
Clay (%)	26.2	2.5	42.0	6.8	25.9	−0.55	26.9	4.0	43.1	7.2	24.9	−0.51
EC_e (dS/m)	22.5	9.7	43.2	6.5	28.7	1.27	21.53	4.65	48.3	10.02	46.6	0.49
pH_e	7.6	7.3	8.1	0.2	2.3	0.32	7.79	7.12	8.40	0.33	4.2	−0.26
SP (%)	59.1	40.0	89.5	11.1	18.8	1.16	62.19	40.85	95.77	12.81	20.6	0.69
Anions in saturation extract (mmol$_c$/L)												
HCO_3^-	2.62	1.10	9.01	1.18	44.8	3.61	1.73	1.07	2.49	0.37	21.2	0.35
Cl^-	47.07	5.48	100.48	21.82	46.3	0.68	46.55	6.19	154.01	32.28	69.4	1.11
NO_3^-	0.65	0.10	4.34	0.81	124.6	2.93	0.38	0.0	4.91	0.84	220.2	3.91
SO_4^-	258.84	121.51	570.77	89.44	34.6	1.68	262.27	59.12	648.10	130.87	49.9	0.83
Cations in saturation extract (mmol$_c$/L)												
Na^+	270.23	98.60	603.00	97.80	36.2	1.49	276.05	27.96	744.2	155.21	56.2	0.82
K^+	1.04	0.37	2.50	0.43	41.0	1.22	0.49	0.0	1.18	0.26	52.8	0.84
Ca^{++}	22.12	13.85	28.10	2.24	10.1	−0.79	22.12	19.80	25.43	1.53	6.9	0.54
Mg^{++}	19.07	9.95	46.37	7.80	40.9	1.60	22.60	8.97	53.57	11.32	50.1	1.22
Exchangeable cations (mmol$_c$/kg)												
Na^+	78.8	35.2	132.2	23.8	30.2	0.51	80.9	24.3	156.0	27.6	41.3	0.78
K^+	4.7	1.8	12.4	2.3	48.5	1.35	3.4	1.8	9.9	2.1	44.1	1.53
Ca^{++}	45.5	0.0	173.9	36.8	80.9	1.20	44.7	9.3	89.6	26.4	60.1	0.43
Mg^{++}	43.2	27.2	70.6	11.1	25.7	0.86	48.1	24.8	73.7	12.3	27.9	0.03
SAR	59.0	24.0	107.6	16.6	28.1	0.76	56.9	6.3	121.0	26.5	46.5	0.09
ESP (%)	47.5	28.9	95.0	13.5	28.4	1.18	46.6	25.4	91.9	12.1	24.7	1.12
B (mg/L)	17.5	9.4	31.3	4.8	27.2	0.87	16.71	1.64	37.20	7.28	43.6	0.39
Se (µg/L)	12.9	0.0	34.0	9.2	71.5	0.67	70.0	0.0	704.0	141.0	203.3	2.67
As (µg/L)	12.9	0.0	116.0	23.4	181.0	2.82	5.0	0.0	88.0	18.0	396.0	3.89
Mo (µg/L)	780.5	183.0	1756.0	338.9	43.4	0.90	294.0	0.0	1690.0	350.0	119.3	1.74
$CaCO_3$ (g/kg)	11.4	0.3	46.7	12.5	110.1	1.46	11.0	0.21	38.8	9.9	98.1	1.33
Gypsum (g/kg)	66.3	5.6	215.3	39.8	60.0	1.30	69.0	10.9	223.1	41.2	59.9	1.41
Total C (g/kg)	NM	NM	NM	NM	NM	NM	0.36	0.13	0.77	0.12	38.8	0.89
Total N (g/kg)	NM	NM	NM	NM	NM	NM	0.04	0.03	0.08	0.01	39.9	2.25

† θ_v = volumetric water content, ρ_b = bulk density, EC_e = electrical conductivity of saturation extract, sp. = saturation percentage, SAR = sodium adsorption ratio, ESP = exchangeable sodium percentage, NM = not measured.

‡ Taken from Corwin et al. (2003).

only in the extent of the removal of B through the soil profile because of its lower mobility resulting from adsorption to clay surfaces. Figures 7a and 8a show the spatial baselines for B and Mo, respectively. The baselines increase in concentration with depth and show a general spatial trend of high to low concentrations from the southwest corner to the northeast corner. Spatio-temporal trends from 1999 to 2004 show that leaching has resulted in B and Mo concentration decreases at all depth increments, with higher levels of B and Mo still present in the south than in the north (Fig. 7a,b,c and 8a,b,c). Molybdenum is observably more mobile than B; consequently, by 2004 a greater proportion of Mo has been leached from all depth increments. Even though the field means at every depth were lower for both B and Mo (Tables 4–7), the F-test significance levels for shift in field means from 1999 to 2004 shows that there was a significant ($P \leq 0.05$) decrease in B concentration only in the top 0.6 m, whereas a significant decrease in Mo concentration occurred for the entire 1.2 m soil profile (Table 9). However, from 2002 to 2004 Mo reappears at the shallow depths (i.e., 0–0.3 and 0.3–0.6 m), but only at the north end of the field (see Fig. 8c). The net change from 1999 to 2004 shows substantial leaching of B and Mo at all depths, with the greatest leaching of B occurring in the north at all depths and the greatest leaching of Mo in the south at the deeper depths (Fig. 7c and 8c). The leaching of greater amounts of Mo at the deeper depths is simply due to the higher concentration of Mo at the lower depths in the south. Basically, Mo is being leached from the entire 1.2-m soil profile throughout the field, except for its reappearance at the shallow depths in the north.

Dynamic spatial variation, which indicates change in spatial variation across the field from 1999 to 2004, is present (i.e., $P < 0.05$) for EC_e at all depths; Cl^- at depths 0 to 0.3, 0.3 to 0.6, and 0.6 to 0.9 m; SAR at all depths; B at all depths; Mo at depths 0.3 to 0.6, 0.6 to 0.9, and 0.9 to 1.2 m; and pH_e at depths 0.3 to 0.6, 0.6 to 0.9, and 0.9 to 1.2 m (Table 9). The extent of dynamic spatial variation observed for the properties considered to influence soil quality at the study site (i.e., EC_e, SAR, B, and Mo) suggests that future monitoring efforts to characterize spatial changes in patterns of these properties should also consider new site selections based on new EC_a survey data in addition to using the 40 original sample sites, which characterize the temporal change.

Need for Trace Element Monitoring

Even though the general trend has been reclamation of the saline-sodic field through the leaching of salts and trace elements

Table 7. Mean and range statistics for 0.9–1.2 m sample depth for 1999 and 2004. *N* = 48 for each sample year.

	1999‡						2004					
Soil property	**Mean**	**Min.**	**Max.**	**SD**	**CV**	**Skewness**	**Mean**	**Min.**	**Max.**	**SD**	**CV**	**Skewness**
θ_v (m^3/m^3)†	0.43	0.31	0.54	0.05	11.4	−0.37	0.36	0.27	0.44	0.04	10.3	−0.55
ρ_b (Mg/m^3)	1.51	1.14	1.75	0.16	10.7	−0.85	1.02	0.81	1.34	0.19	19.0	−0.40
Clay (%)	23.3	11.1	36.9	6.3	26.9	0.08	23.2	10.9	35.9	6.5	25.4	0.10
EC_e (dS/m)	25.2	8.0	49.7	7.9	31.5	0.81	22.51	4.33	47.40	9.80	43.5	0.24
pH_e	7.6	7.1	8.0	0.2	2.7	−0.37	7.92	7.04	8.44	0.31	3.9	−0.79
SP (%)	58.7	38.5	93.3	12.9	22.0	0.68	64.22	38.42	134.05	17.46	27.2	1.46
Anions in saturation extract ($mmol_c/L$)												
HCO_3^-	3.06	1.19	7.76	1.49	48.6	1.98	1.72	1.05	2.60	0.44	25.3	0.26
Cl^-	58.66	3.18	152.68	30.15	51.4	0.80	51.69	6.40	129.68	32.04	62.0	0.40
NO_3^-	0.32	0.12	1.29	0.20	62.8	3.44	0.51	0.0	3.13	0.97	191.5	2.03
SO_4^-	291.98	91.03	720.96	114.02	39.0	1.52	268.48	45.73	637.10	127.63	47.5	0.70
Cations in saturation extract ($mmol_c/L$)												
Na^+	311.57	66.10	738.69	126.13	40.5	1.20	290.64	23.73	739.50	155.12	53.4	0.67
K^+	1.06	0.34	2.36	0.47	44.0	0.85	0.43	0.0	1.20	0.25	58.1	0.94
Ca^{++}	22.34	16.68	26.50	2.11	9.5	−0.64	21.49	3.77	25.94	3.04	14.1	−4.21
Mg^{++}	22.30	9.33	52.05	10.42	46.7	1.27	21.73	4.03	48.68	9.88	45.5	0.94
Exchangeable cations ($mmol_c/kg$)												
Na^+	87.7	32.4	192.0	32.0	36.5	1.10	79.6	16.3	188.8	34.6	37.7	1.02
K^+	4.1	1.1	9.8	2.1	51.7	0.88	2.9	0.4	7.6	1.3	49.3	0.91
Ca^{++}	42.5	0.0	154.2	35.4	83.3	0.97	43.2	0.6	144.3	27.1	73.1	0.92
Mg^{++}	46.4	26.6	90.3	14.7	31.7	1.09	45.4	21.6	84.9	14.0	28.9	0.91
SAR	64.9	16.8	120.2	19.5	30.0	0.35	60.9	5.4	127.8	26.5	43.6	0.05
ESP (%)	51.8	18.2	135.7	20.2	39.0	1.82	46.8	14.9	101.3	16.7	34.2	1.02
B (mg/L)	17.9	6.5	31.8	6.3	35.0	0.45	16.45	1.33	34.48	7.32	44.5	0.15
Se (µg/L)	14.1	0.0	62.0	13.9	98.5	1.80	81.0	0.0	677.0	161.0	199.8	2.62
As (µg/L)	4.4	0.0	30.0	8.1	183.3	2.04	3.0	0.0	130.0	19.0	692.8	6.93
Mo (µg/L)	946.9	330.0	2856.0	450.7	47.6	1.78	278.0	0.0	1159.0	272.0	98.0	0.90
$CaCO_3$ (g/kg)	12.7	0.6	64.8	14.6	115.2	1.79	13.0	0.18	40.4	11.9	88.6	1.31
Gypsum (g/kg)	64.1	0.8	251.4	46.4	72.4	2.06	83.6	8.7	256.1	55.9	67.7	1.42
Total C (g/kg)	NM	NM	NM	NM	NM	NM	0.33	0.11	0.78	0.15	43.3	0.89
Total N (g/kg)	NM	NM	NM	NM	NM	NM	0.04	0.02	0.08	0.01	36.6	2.01

† θ_v = volumetric water content, ρ_b = bulk density, EC_e = electrical conductivity of saturation extract, sp. = saturation percentage, SAR = sodium adsorption ratio, ESP = exchangeable sodium percentage, NM = not measured.

‡ Taken from Corwin et al. (2003).

and the short-term prognosis of drainage water reuse is favorable, there is an associated caveat, which could have long-term ramifications for the viability and sustainability of drainage water reuse on California's WSJV. Aside from the apparent reappearance of Mo at the shallow depths in the north end of the field, Se has started to accumulate in the field's northern end (see Tables 4–7 and Fig. 9a). In 1999 no sample was found to have a Se concentration above 100 μg L^{-1} and field means were only 8, 14, 13, and 14 μg L^{-1} for the depth increments of 0 to 0.3, 0.3 to 0.6, 0.6 to 0.9, and 0.9 to 1.2 m, respectively. However, by 2004 Se concentrations as high as 704 μg L^{-1} were found and field means increased from five- to sevenfold to 57, 77, 70, 81 μg L^{-1} for the depth increments of 0 to 0.3, 0.3 to 0.6, 0.6 to 0.9, and 0.9 to 1.2 m, respectively. Even though these levels of Se are of no immediate concern, the unexpected increase of Se over the past 5 yr points to the need for careful and thorough observation to establish if a long-term threat may exist. Because measured Mo and Se concentrations in 2004 did not correlate with spatially associated EC_a measurements taken with EM in 1999, their spatial distributions and field means are unreliable. However, even though the accuracy of the means and spatial distributions of Mo and Se for 2004 are in doubt, there is no question that both are present at the north end of the field and their appearance is a potential concern worthy of future close observation.

Presumably, the increase in Mo and Se is a consequence of its presence in the drainage water applied to the field because chemical factors (e.g., pH and ionic strength) do not provide a reasonable explanation. Molybdenum becomes less correlated with EC_a over time as shown in Table 3, which points to an external input of Mo or more specifically the applied drainage water as the source. The anomalous behavior of the restriction of Mo and Se primarily to the north end of the field in 2004 with little Mo and Se present at the southern end of the field is difficult to rationalize (Fig. 9a). In 1999, Se was spatially distributed similar to EC_e, SAR, B, and Mo, with higher levels in the south and lower in the north. The distribution of Se from 1999 to 2004 has unexpectedly reversed. Similarly, the reappearance of Mo in shallow depths (i.e., 0–0.3 and 0.3–0.6 m) at the north end of the field is unexpected and inexplicable.

In general, the levels of salinity, sodium, B, Se, and Mo in the reused drainage water decline from 2001 to 2004 (Table 1 and 2). The levels of salinity in the drainage water did abruptly change from time to time (see Table 1) and possibly Mo and Se changed as well. Drainage water with higher Mo and Se could have been ap-

Table 8. Percent local-scale variation of soil properties by depth for 2004.

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)†	4.6	5.9	10.7	5.4
ρ_b (Mg/m^3)	6.2	18.2	6.4	6.6
Clay (%)	13.8	20.4	13.6	8.6
EC_e (dS/m)	0.7	1.1	1.4	1.0
pH_e	8.4	3.4	5.3	1.0
SP (%)	3.1	7.2	3.1	0.8
SAR	0.6	0.9	1.7	0.7
B (mg/L)	0.7	1.7	2.2	2.4
Mo (µg/L)	10.2	1.5	3.2	5.0
$CaCO_3$ (g/kg)	3.5	11.2	0.6	5.9
Gypsum (g/kg)	11.1	4.2	10.0	5.7
Total C (g/kg)	2.8	3.6	2.2	4.6
Total N (g/kg)	2.6	3.7	6.0	1.0
Anions in saturation extract (mmol$_c$/L)				
HCO_3^-	7.2	15.3	5.2	3.5
Cl^-	2.6	4.7	4.5	1.1
NO_3^-	0.7	20.0	6.2	0.3
SO_4^{2-}	1.8	1.8	1.6	1.5
Cations in saturation extract (mmol$_c$/L)				
Na^+	1.0	1.5	1.8	1.0
K^+	2.6	4.7	4.5	1.1
Ca^{2+}	16.3	6.2	7.3	2.3
Mg^{2+}	7.1	5.9	2.7	3.2

† θ_v = volumetric water content, ρ_b = bulk density, EC_e = electrical conductivity of saturation extract, sp. = saturation percentage, SAR = sodium adsorption ratio.

plied to the northern end in 2004, but there is no data to directly confirm or disprove this supposition. Even though the appearance of Se was not anticipated to be a serious consideration early in the experiment due the low levels of Se initially found in the reused drainage water sample taken on 22 Aug. 2000 (see Table 2), it has evolved into an area of potential concern, which needs to be monitored in the future to confirm this trend.

The chemical composition of the reused drainage water samples in Table 2 and the drainage water reuse volumes and EC_{dw} in Table 1 may provide a clue to the increase in soil Mo and Se in 2004. Because a detailed chemical analysis of the reused drainage water was only conducted once each year, the water sample analysis on 29 Aug. 2004 in Table 2 does not directly indicate levels of Se and Mo that are sufficiently high to cause an increase in soil Se and Mo. For the most part, the 29 Aug. 2004 sample is representative of the drainage water applied for all of 2004 except for 30 and 31 Mar. 2004 (see Table 1) where the salinity was high (i.e., EC_{dw} = 10.6 dS m^{-1}). As indicated in Table 2, high salinity drainage water is associated with higher levels of Se (i.e., 700 µg L^{-1}) and Mo (i.e., 400 µg L^{-1}). It is conceivable that the high salinity drainage water that was applied on 30 and 31 Mar. 2004 is the source of the increased soil Se and Mo found in 2004, but why the Se and Mo is largely confined to the northern end of the field in 2004 is still not apparent.

Because this study is a real-world assessment of drainage water reuse impacts where the farmer has complete control over the sources of drainage water applied to the field, there is a definite possibility that the appearance and reappearance of Se and Mo, respectively, is due to their presence in drainage water applied to the field. The farmer was opportunistic and applied whatever source of drainage water was available; consequently, the compositions of the drainage water changed over time, as reflected by the wide range of salinities (i.e., 0.8–16.2 dS m^{-1}). There are several causes for the wide variation in chemical composition shown in annual spot checks of the reused drainage water samples. First and foremost, the spatial extent of land that was irrigated and the extent of the irrigation resulting in drainage were in constant flux since lands were rotated in and out of production. Furthermore, crop and irrigation management strategies were constantly changing from one year to the next reflecting the farmer's decisions based on commodity market pressures and drought conditions. At

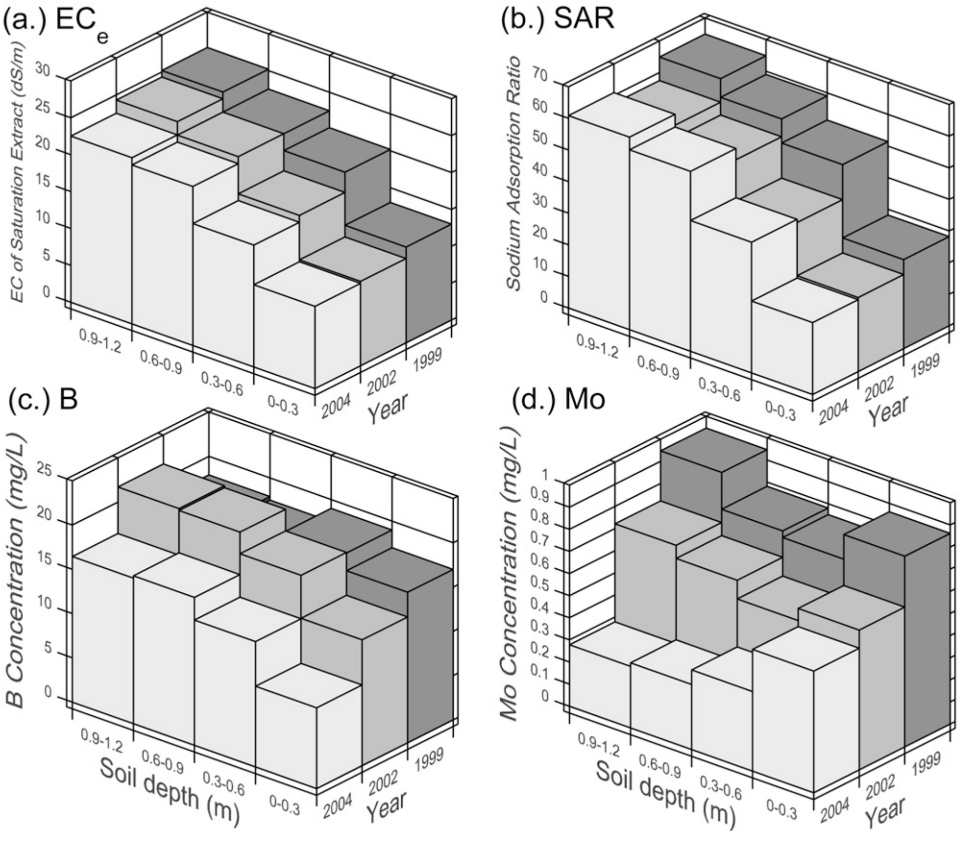

Fig. 4. Bar charts showing the field means by depth increment (0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m) over the sample times of 199, 2002, and 2004 for (a) salinity (electrical conductivity of the saturation extract, EC_e), (b) sodium adsorption ratio (SAR), (c) boron (B), and (d) molybdenum (Mo).

times drainage water volumes were so low in the holding pond that high quality canal irrigation water was used to prevent the loss of the forage. Unfortunately, resources were not available to monitor continuously the chemical compositions of water inputs, only sporadic checks of the reused drainage water composition were possible. These checks cannot confirm reused drainage water as the source of Mo and Se that has appeared in the field in 2004.

Another property that needs close attention in the future is pH_e. Table 9 shows a shift in the field mean pH_e level from 1999 to 2004 for the bottom two depth increments (i.e., 0.6–0.9 and 0.9–1.2 m), but this masks interesting spatial changes. The changes that have occurred with pH_e are spatially divided into north and south effects (Fig. 9b). In the northern half of the field the pH_e has declined from 1999 to 2004 for the top two depth increments (i.e., 0–0.3 and 0.3–0.6 m) and has increased for the bottom two depth increments (i.e., 0.6–0.9 and 0.9–1.2 m). In the southern half of the field the pH_e has increased from 1999 to 2004 throughout the 1.2-m soil profile. The rise in pH_e in the southern half of the field is due to the fact that the southern half is highly saline-sodic while the northern half is primarily saline. Unlike saline soils, leaching will raise the pH of saline-sodic soils because once the neutral soluble salts are removed the exchangeable Na hydrolyzes and increases the OH^- concentration of the soil solution, raising the soil pH. Ostensibly, the 0.8 to 16.2 dS m^{-1} drainage water (averaging 4.13 dS m^{-1}) used for irrigation was not sufficiently high in salts to prevent the hydrolysis of Na in portions of the south end of the field. The future use of gypsum may be needed in these areas to assure that Na ions do not disperse the mineral colloids, creating impervious areas.

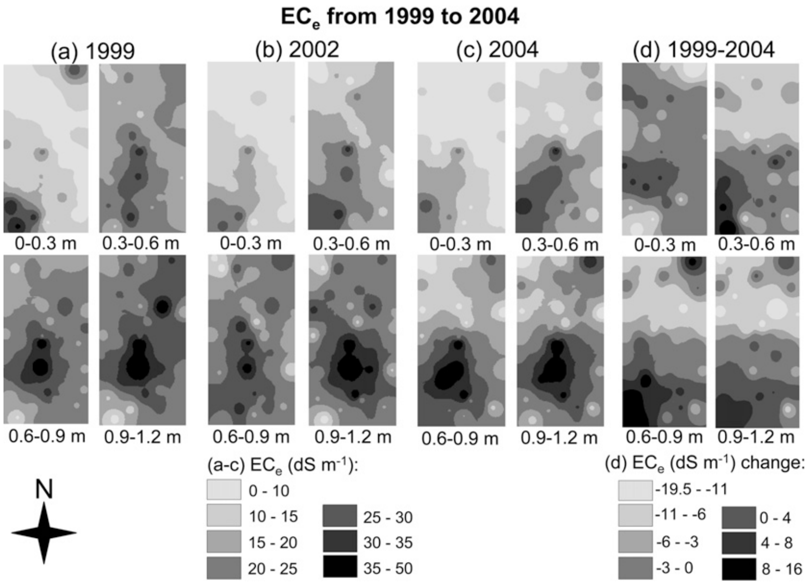

Fig. 5. Maps showing the change in spatial patterns of salinity (electrical conductivity of the saturation extract, EC_e) by depth increment (0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m) for the sampling times of (a) 1999, (b) 2002, and (c) 2004. (d) Maps showing spatial patterns of the net change of EC_e from 1999 to 2004 by depth.

Summary and Conclusions

From 1999 to 2004, the application of 0.8 to 16.2 dS m^{-1} drainage water on saline-sodic soil resulted in removal of salinity and Na from the top 1.2 m of the soil profile, especially from the top 0 to 0.6 m; removal of B from the top 0.6 m; and removal of Mo from the top 1.2 m. From the perspective of impacts on soil chemical properties, drainage water reuse on saline-sodic soil demonstrates short-term sustainability with few detrimental impacts. However, continued monitoring is needed for at least 5 additional years with close attention given to the slow accumulation of Se, particularly at the north end of the field; to the reappearance of Mo in the north; and to a trend of increasing pH, particularly at lower depths.

The key to successful drainage water reuse on agricultural land is maintaining or improving soil quality. Soil quality as defined by Doran and Parkin (1994) is "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." An evaluation of the short-term sustainability of drainage water reuse on marginally productive soil in California's WSJV has shown that drainage water reuse can be suitably managed (i) to improve the soil quality of a saline-sodic soil by leaching salinity, Na, and detrimental trace elements (B and Mo) thereby reclaiming the soil and returning it to productivity; (ii) to transform drainage water from an environmental burden into

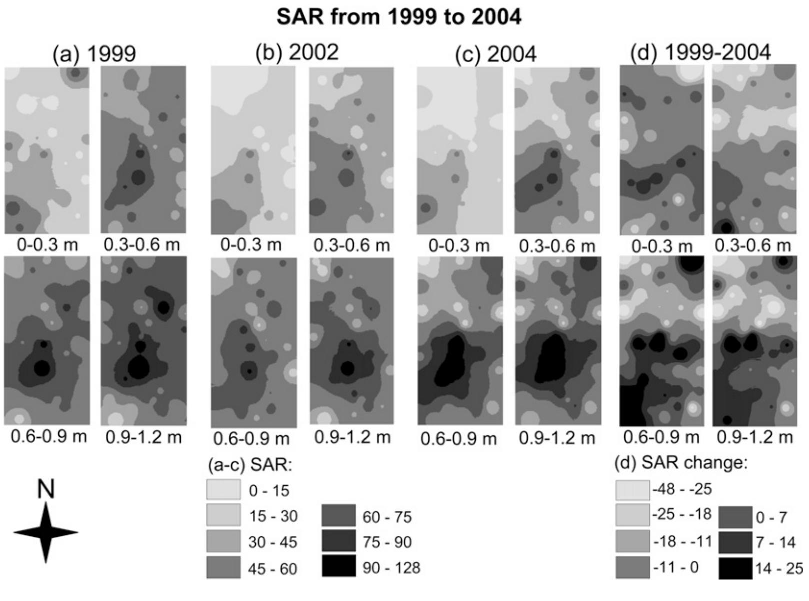

Fig. 6. Maps showing the change in spatial patterns of sodium adsorption ratio (SAR) by depth increment (0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m) for the sampling times of (a) 1999, (b) 2002, and (c) 2004. (d) Maps showing spatial patterns of the net change of SAR from 1999 to 2004 by depth.

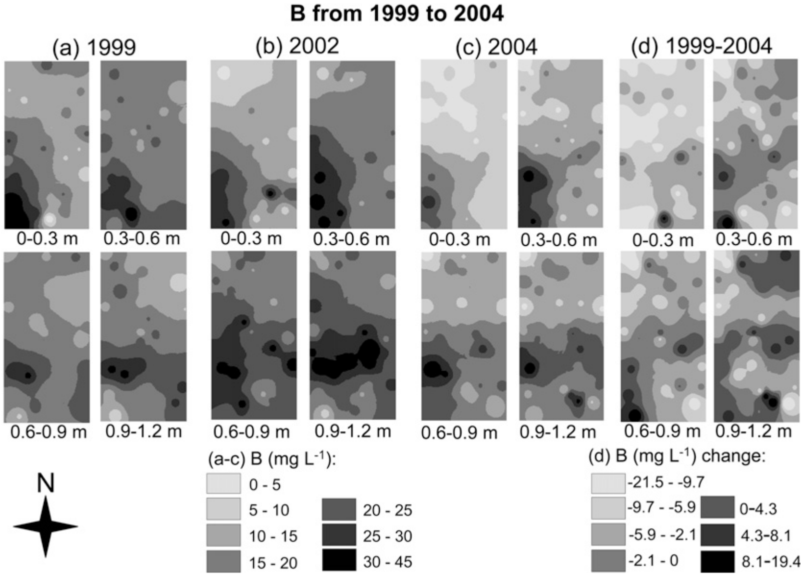

Fig. 7. Maps showing the change in spatial patterns of boron (B) by depth increment (0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m) for the sampling times of (a) 1999, (b) 2002, and (c) 2004. (d) Maps showing spatial patterns of the net change of B from 1999 to 2004 by depth.

a water resource that produces forage to support livestock; (iii) to reduce the degradation of ground water by encouraging the installation of drainage systems to collect drainage water for reuse; and (iv) to reduce dramatically the volume of drainage water disposed in evaporation ponds. Of course, the successful management of the reuse of any degraded water rests on an understanding of the chemical composition of the degraded water and of the soil to which it is applied.

Based on a 5-yr evaluation, drainage water reuse shows considerable potential as an alternative to the disposal of drainage water in California's central valley. Not only can drainage volumes be reduced, mitigating the need for non-productive evaporation ponds, but an alternative water resource becomes available that can be used to economic advantage by reclaiming marginally productive saline-sodic soils and bringing them back into agricultural production. The reuse of drainage water to reclaim saline-sodic soil is noteworthy because higher quality irrigation water could not successfully reclaim these soils, unless gypsum is added to maintain infiltration, due to infiltration problems that occur from the application of waters with low salinity levels onto soils with high exchangeable Na$^+$ levels.

There is a high likelihood of continued sustainability for drainage water reuse at the Westlake Farm study site due to existing conditions, which should be kept in mind when considering other sites in the WSJV or elsewhere for drainage water reuse. First, as previously mentioned drainage water reuse is most effective on sodic soils because there is sufficient salinity in drainage water to minimize infiltration problems that would occur for low salinity waters due to the high exchangeable Na$^+$ levels that could disperse soils posing an infiltration and permeability problem. Second, the high expanding-lattice clay content near the surface produces cracks when the surface is allowed to dry, which provide water flow pathways until the soil is sufficiently wet for the expanding clays to close the cracks. Third, Bermudagrass, aside from being salt tolerant, helps to produce water flow pathways by enhancing soil aggregation and providing flow paths along its root system. Finally, the drainage system provides an outlet to prevent the formation of a shallow water table, which would quickly shorten the longevity of the system. Each of these factors helps in managing the reuse of drainage water by providing conditions that will permit the leaching of salts.

Even though the extrapolation of short-term results points to a positive prognosis, the long-term sustainability of drainage water reuse in California's Central Valley still needs to be evaluated because of the potential for the slow accumulation of detrimental trace elements. Future monitoring efforts must focus attention on Mo, Se, and pH, with greatest concern given to Mo and Se. Even though current levels of Se and Mo are well below threshold levels of concern, their continued accumulation could indicate the need to reevaluate the reuse of drainage water. Because of the complexity of soil-water-plant interactions on marginal soils, trends are difficult to model. Only after monitoring for an additional 5 to 10 yr will sufficient data be collected to make a final evaluation of the long-term sustainability of drainage water reuse on California's WSJV.

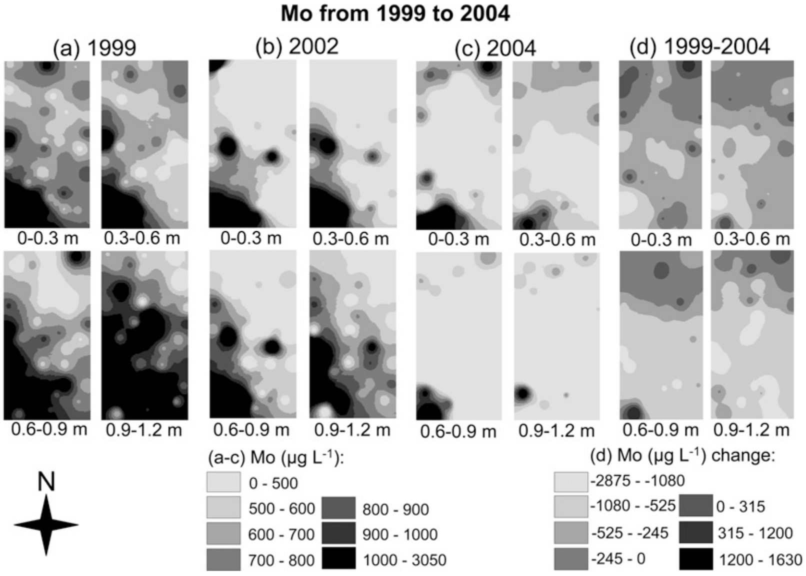

Fig. 8. Maps showing the change in spatial patterns of molybdenum (Mo) by depth increment (0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m) for the sampling times of (a) 1999, (b) 2002, and (c) 2004. (d) Maps showing spatial patterns of the net change of Mo from 1999 to 2004 by depth.

Table 9. F-test significance levels associated with mixed linear modeling result for 2004 vs. 1999 data.

Chemical property	Sample depth increment (m)	Pr > F for dynamic spatial variation	Pr > F for shift in mean level
EC_e†	0–0.3	0.0001	0.0504
	0.3–0.6	0.0037	0.0002
	0.6–0.9	0.0024	0.4529
	0.9–1.2	0.0070	0.0329
Cl^-	0–0.3	0.0016	0.9850
	0.3–0.6	0.0036	0.2468
	0.6–0.9	0.0131	0.8625
	0.9–1.2	0.0703	0.1005
SAR	0–0.3	0.0001	0.0130
	0.3–0.6	0.0009	0.0001
	0.6–0.9	0.0031	0.6793
	0.9–1.2	0.0006	0.1914
B	0–0.3	0.0320	0.0001
	0.3–0.6	0.0005	0.0001
	0.6–0.9	0.0006	0.2346
	0.9–1.2	0.0015	0.1421
Mo	0–0.3	0.3472	0.0001
	0.3–0.6	0.0006	0.0001
	0.6–0.9	0.0031	0.0001
	0.9–1.2	0.0247	0.0001
pH_e	0–0.3	0.5576	0.0825
	0.3–0.6	0.0040	0.8068
	0.6–0.9	0.0168	0.0025
	0.9–1.2	0.0030	0.0001

† EC_e = electrical conductivity of saturation extract, SAR = sodium adsorption ratio.

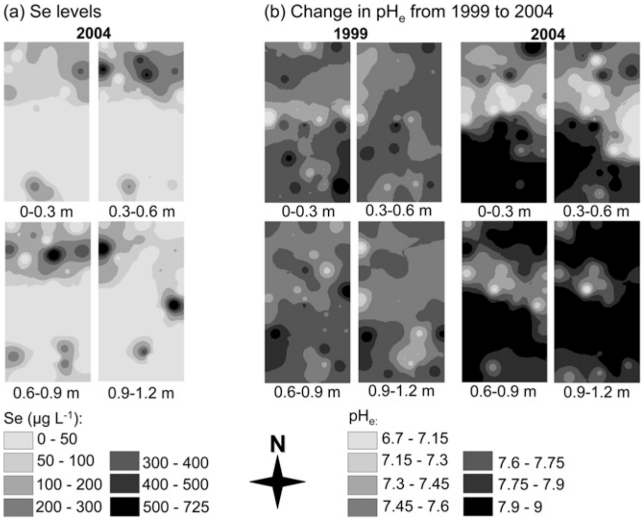

Fig. 9. Maps showing (a) the distribution of selenium (Se) by depth increment (0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m) for 2004 and (b) the change in pH_e (pH of the saturation extract) by depth (0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m) from 1999 to 2004.

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