



Assessment and field-scale mapping of soil quality properties of a saline-sodic soil

D.L. Corwin^{a,*}, S.R. Kaffka^b, J.W. Hopmans^c, Y. Mori^d,
J.W. van Groenigen^e, C. van Kessel^b,
S.M. Lesch^a, J.D. Oster^f

^aUSDA-ARS, George E. Brown, Jr. Salinity Laboratory, 450 West Big Springs Road,
Riverside, CA 92507-4617, USA

^bDepartment of Agronomy and Range Science, University of California, Davis, CA, USA

^cDepartment of Land, Air, and Water Resources, University of California, Davis, CA, USA

^dLife and Environmental Sciences, Shimane University, Japan

^eDepartment of Environmental Science, University of California, Riverside, CA, USA

^fDepartment of Water and Environment, Alterra Wageningen, The Netherlands

Abstract

Salt-affected soils could produce useful forages when irrigated with saline drainage water. To assess the productive potential and sustainability of using drainage water for forage production, a saline-sodic site (32.4 ha) in California's San Joaquin Valley was characterized for soil quality. The objectives were (1) to spatially characterize initial soil physicochemical properties relevant to maintaining soil quality on an arid zone soil and (2) to characterize soil quality relationships and spatial variability.

An initial mobile electromagnetic (EM) induction survey was conducted in 1999, with bulk soil electrical conductivity (EC_a) readings taken at 384 geo-referenced locations, followed by an intensive mobile fixed-array survey with a total of 7288 geo-referenced EC_a readings. Using the EM data and a spatial statistics program (ESAP v2.0), 40 sites were selected that reflected the spatial heterogeneity of the EC_a measurements for the study area. At these sites, soil-core samples were taken at 0.3-m intervals to a depth of 1.2 m. Duplicate samples were taken at eight sites to study the local-scale variability of soil properties. Soil-core samples were analyzed for a variety of physical and chemical properties related to the soil quality of arid zone soils.

Soils were found to be highly spatially heterogeneous. For composite soil-core samples taken to a depth of 1.2 m, EC_e (electrical conductivity of the saturation extract) varied from 12.8 to 36.6 dS m^{-1} , SAR from 28.8 to 88.8, and clay content from 2.5% to 48.3%. B and Mo concentrations varied from 11.5 to 32.2 $mg\ l^{-1}$ and 476.8 to 1959.6 $\mu g\ l^{-1}$, respectively. $CaCO_3$, NO_3^- in the saturation extract, exchangeable Ca^{2+} , Se, and As consistently had the highest coefficients of variation (CV)

* Corresponding author. Tel.: +1-909-369-4819; fax: +1-909-342-4962.

E-mail address: dcorwin@ussl.ars.usda.gov (D.L. Corwin).

while pH_e , ρ_b , and Ca^{2+} in the saturation extract consistently had the lowest CVs at all depths. A one-way analysis of variance (ANOVA) was used to spatially partition the local- and global-scale variability. Local-scale variability was greatest for pH_e . Laboratory measurements of saturated hydraulic conductivity (K_s) were very low (0.0000846–0.0456 cm h^{-1}), whereas field measurements were considerably higher (0.49–1.79 cm h^{-1}). Based on the Cl^- data, the leaching fraction (LF) for the entire study area was estimated to be 17%.

Soil quality was reflected in yield and chemical analysis of forage. Forage Mo contents determined from newly established Bermuda grass varied from 1 to 5 mg kg^{-1} on a dry matter basis, and Cu/Mo ratios averaged 3.3, while forage yield in the establishment year declined with EC_e , and failed to grow above EC_e levels of approximately 22 dS m^{-1} . The initial soil quality assessment of the research site indicated that the sustainability of drainage water reuse at this location would depend upon maintaining a sufficient LF with careful consideration and management of salinity, boron, molybdenum, and sodium levels.

© 2003 Elsevier Science B.V. All rights reserved.

Keywords: Electrical conductivity; EC_a ; Drainage water reuse; EM; Salinity; San Joaquin Valley

1. Introduction

The Central Valley of California, consisting of the Sacramento and San Joaquin Valleys, is one of the most agriculturally productive areas of the world and is of broad agro-ecological significance. Due to the lack of an outlet for drainage water in the western San Joaquin Valley (WSJV), continued agricultural productivity is threatened. Because restrictions have been imposed on the discharge of drainage water to surface water supplies, the most common way to handle drainage water is disposal to groundwater beneath irrigated lands or to permanent evaporation ponds.

Because of restrictions on discharge of saline drainage to surface water, current farm practices for handling drainage are not sustainable. The soil quality of 300,000 ha of land in the WSJV is adversely affected by the presence of shallow or perched water (0.3–1.5 m). Without a means of disposing of drainage water, increasing amounts of farmland in the WSJV will become salt impaired, and will be removed from agricultural productivity. If all lands with shallow water tables were drained, approximately 30,000 ha of land would be needed for evaporation ponds, an amount almost 20 times greater than that currently available. Regulations limit the expansion or development of new evaporation ponds. Besides possible inadvertent effects on wildlife, the dedication of 10% of all affected land to salt ponds would be too expensive for most farm operations. This makes broad-scale reliance upon evaporation ponds impractical, while the continued disposal of drainage water to groundwater is untenable.

An alternative means of reducing drainage water volumes is their reuse on agricultural lands. Methods for reusing drainage water were developed and evaluated by Rhoades (1989), Grattan and Rhoades (1990), Bradford and Letey (1993), Cervinka (1994), and others. Posnikoff and Knapp (1996) provided a favorable economic assessment of the potential for drainage water reuse. However, agricultural drainage water in the WSJV is sufficiently saline to be detrimental to most crops and often contains toxic trace elements

such as Se that can harm wildlife (Deverel et al., 1984; Deverel and Fujii, 1988; Deverel and Millard, 1988; Fujii and Deverel, 1989; Fujii and Swain, 1995).

Using drainage water to produce forages that are salt-tolerant would reduce the volume of drainage water and the amount of land needed for its disposal by up to an order of magnitude (Oster, 1997), thereby lowering the cost of disposal and limiting the exposure of wildlife to potentially toxic waters. High-quality forages for dairy cattle, beef cattle, and sheep are in short supply in the Central Valley of California. Salt-tolerant forages may help provide sufficient supplies. Grasses like Bermuda grass will use more water than most tree species in the same locations because of greater salt tolerance, and have an obvious use for livestock.

If systems of high-quality forage production that use drainage waters are coupled to livestock enterprises in creative ways, drainage water would become an asset rather than a problem. The use of forage–livestock systems, together with careful management of a small number of drainage ponds, will address the most serious problem affecting the long-term sustainability of farming in the WSJV, and provide additional benefits to areas throughout the world affected by similar problems.

Soil quality has been defined by Doran and Parkin (1994) 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.” Maintaining soil quality is essential to the success of forage–livestock systems for drainage water reduction. Accumulation of trace elements in soils and crops to toxic levels will result in failure of the system and the creation of a new pollution problem. Deterioration of soil physical structure from salt and sodium added with irrigation water could restrict crop establishment and growth.

Drainage waters in the WSJV often contain high concentrations of salinity and trace elements such as arsenic, boron, molybdenum, and selenium (Deverel et al., 1984; Deverel and Fujii, 1988; Deverel and Millard, 1988; Fujii and Deverel, 1989; Fujii and Swain, 1995). The recycling of drainage water containing leached salts and trace elements would reintroduce potentially harmful chemical constituents that can degrade soil quality. Increased salinity levels can reduce forage yields due to toxic ion (e.g., Na toxicity) and osmotic effects. Increased levels of trace elements such as B, Se, As, or Mo can threaten plants and livestock. B reduces plant yields, while Mo, Se, and As can harm ruminant animals. Barshad (1948) found that plants are able to absorb amounts of Mo harmful to cattle from soils that contain as little as 1.5–5.0 mg kg⁻¹ total Mo.

Maintenance of soil physical properties is a concern when reusing drainage water. As the level of sodicity increases, greater levels of salinity in irrigation water 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 typically found in the WSJV ($6 < EC < 20$ dS m⁻¹; $5 < SAR < 35$) will result in soil salinities ranging from 6 to 60 dS m⁻¹ and SARs ranging from 5 to 60 if the leaching fraction (LF) is approximately 30%. As the LF declines, these levels should increase. Expected levels of salinity in the irrigation water should compensate for increased levels of exchangeable Na, which otherwise could impair hydraulic conductivity and reduce soil aeration (Oster and Jayawardane, 1998).

Much of the research regarding soil quality has been conducted in the Midwestern and Eastern USA, primarily on intensively managed cropland systems (Wienhold et al., 2001). Except for the work of Deveral and Fujii (Deverel et al., 1984; Deverel and Fujii, 1988; Deverel and Millard, 1988; Fujii and Deverel, 1989; Fujii and Swain, 1995), little soil quality work has been conducted on cropped soils of the arid southwestern USA. This paper reports the results of the initial assessment of soil and forage quality for a 32.4-ha research site in the WSJV, which is being used to study the sustainability of drainage water reuse on forage crops as a means of reducing drainage volumes in the SJV. The objectives are (1) to spatially characterize initial soil physicochemical properties relevant to maintaining soil quality on an arid zone soil and (2) to characterize soil quality relationships and spatial variability.

2. Materials and methods

2.1. Study-site description

The study site is located on Westlake Farms, which resides in Kings County on the west side of California's San Joaquin Valley (Fig. 1). The soil at the Westlake Farm site is part of the Lethent clay loam series (fine, montmorillonitic, thermic, Typic Natrargid, USDA, 1986). The site is a 32.4-ha field that has been prepared as a drainage water reuse study area. The field was laser leveled with a slope of zero from north to south and a slope of 0.0009 m m^{-1} from west to east. Drainage lines were spaced at 37.5 m at a depth of 1.2 m using 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).

Eight 4-ha rectangular paddocks (or plots) with dimensions of $75 \times 364 \text{ m}$ were laid out with their borders defined by earth berms (Fig. 2). Paddock 1 is in the southernmost part of the field, with paddock 8 in the north. A tile drain was installed at the center of each paddock. The central drain of each paddock serves an area of roughly $13,650 \text{ m}^2$. 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 (see Fig. 2). Instrumentation was installed to monitor drainage water flows and quality on the central drain in paddocks 2, 3, 6, and 7. Bermuda grass (*Cynodon dactylon* (L.) Pers.) was established in spring and summer of 2000.

2.2. Soil-quality assessment

Patterns of soil quality result from the operation and interaction of soil-forming processes over a continuum of spatial and temporal scales (Trangmar et al., 1985). At the field and regional scale, and for the sake of future modeling efforts, spatial variability must be considered in assessing soil quality (Halvorson et al., 1997). Spatial variability influences the size and number of soil samples needed to characterize the properties that constitute the soil quality of an area of interest. Because the analysis of soil samples is costly and labor-intensive, it is impractical to arbitrarily establish an intense soil-

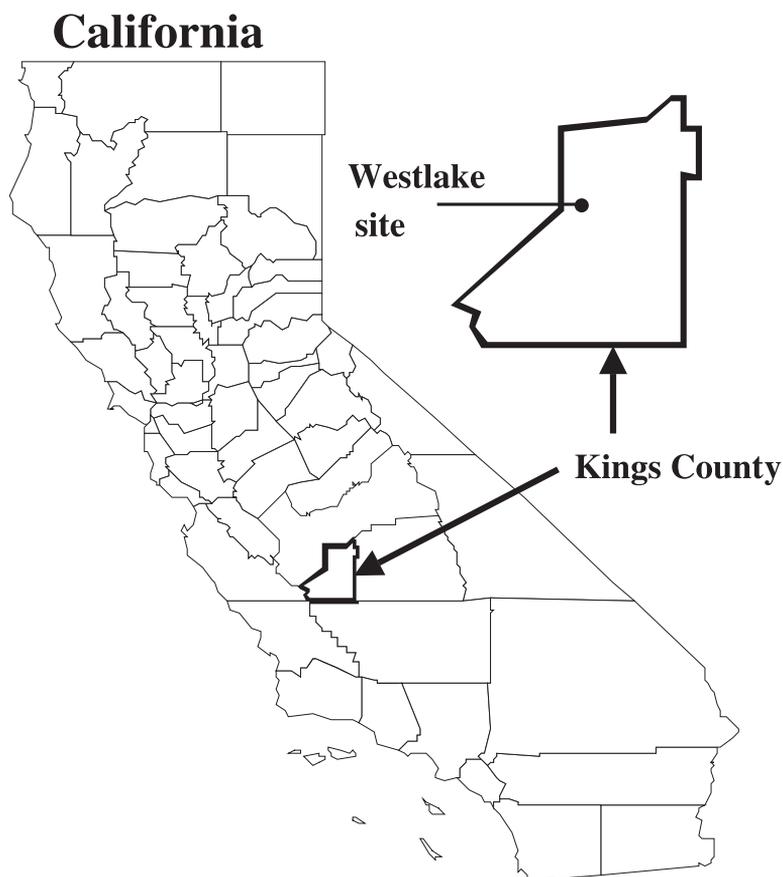


Fig. 1. Geographic location of the Westlake Farm study site.

sampling grid to try to characterize soil quality. Rather, a rapid and inexpensive means of ascertaining the spatial distribution of soil properties associated with soil quality is more desirable. In arid soils, the properties of salinity, texture, and water content are among the most important properties related to soil quality since the amount of soil organic matter is generally extremely small. The measurement of bulk soil electrical conductivity (EC_a) is a rapid, easy, and inexpensive means of obtaining large amounts of spatially referenced information about soil properties either directly or indirectly related to EC_a . Because EC_a is influenced by salinity, water content, bulk density, and texture, which are all important properties related to the quality of arid zone soils, an EC_a survey provides potentially invaluable information for spatially characterizing the quality of an arid zone soil.

The ability to monitor spatial distributions of soil salinity has been demonstrated by Rhoades et al. (1999) at the George E. Brown, Jr. Salinity Laboratory. They developed assessment methodologies for measuring EC_a from geophysical techniques utilizing electromagnetic induction (EM) and fixed-array electrical resistivity. Spatial distributions

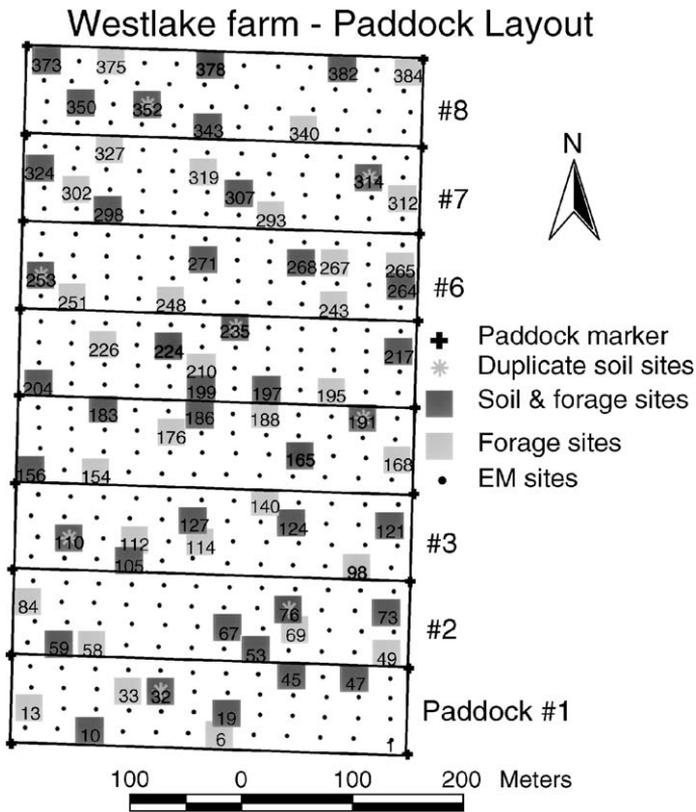


Fig. 2. Paddock layout showing the EM sites, forage sample sites, combined soil and forage sample sites, and duplicate soil sample sites.

of EC_a were measured noninvasively with mobile EM equipment (Carter et al., 1993; Rhoades, 1993) and were measured invasively with a mobilized, tractor-mounted, “fixed-array”, four-electrode unit (Carter et al., 1993; Rhoades, 1992, 1993). The aforementioned mobile EC_a equipment for conducting an EC_a survey were used to assess the soil quality of the Westlake Farm site.

To characterize the spatial variability of properties related to soil quality at both the local and field scale, a survey of EC_a , which followed the guidelines set by Corwin and Lesch (2003), was utilized as spatial information to devise a soil-sampling scheme. The soil-quality assessment to initially characterize the research site was conducted in August of 1999. Mapping of EC_a and soil sampling began 12 August 1999, and occurred over a 2-week period of time. All measurements were geo-referenced with a global positioning system (GPS). The soil-quality assessment consisted of five phases: (1) initial cursory EM EC_a survey, (2) soil sample design, (3) intensive fixed-array EC_a survey, (4) soil chemical and physical analyses, and (5) GIS and map preparation of geo-referenced soil-quality data.

2.2.1. *Cursory EM survey*

The initial survey consisted of a grid of EC_a measurements taken with mobile EM equipment using the Geonics EM-38 electrical conductivity meter.¹ The grid was arranged in a 4×12 pattern within each of the eight paddocks for a total of 384 sites across the 32.4 ha (80-acre) study area (Fig. 2). 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 results in an EM measurement that concentrates the reading nearer to the soil surface and penetrates to a depth of roughly 1 m, whereas the EM reading in the vertical configuration penetrates to a depth of 1.5 m and concentrates the reading less at the surface. From the EM measurements the geometric mean and profile ratio were calculated for each site. The geometric mean was defined as the square root of the product of EM_v and EM_h . The profile ratio was defined as EM_h/EM_v . The profile ratio provided an indication of the shape of the EC_a profile (regular, inverted or uniform). Profile ratios equal to 1 indicate a uniform profile, profile ratios < 1 indicate an increasing profile with depth, and profile ratios > 1 indicate an inverted profile (i.e., conductivity decreases with depth). In essence, the profile ratio is analogous to the leaching fraction, while the geometric mean approximates the cumulative level of salinity in the root zone. All 384 sites were geo-referenced.

2.2.2. *Soil-sample design*

Utilizing the data from the cursory EM survey and response surface sample design software (ESAPv2.0) developed by Lesch et al. (1995, 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 nonuniform number of sample sites per paddock.

Conceptually, 40 sites were chosen to satisfy the following three criteria: (i) to represent about 95% of the observed range in the geometric mean EM data, (ii) to represent about 95% of the observed range in the EM profile ratio data, and (iii) 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. 2). More specific details on how a spatial response surface sample design can be used to simultaneously achieve these criteria can be found in Lesch et al. (1995).

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,

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.

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 all 384 EM measurement sites, the 40 selected soil-core sites, and the 8 duplicate soil-core sites.

2.2.3. Intensive fixed-array survey

After the EM survey and the soil samples were taken, an intensive EC_a survey was conducted with mobile fixed-array equipment. The justification for conducting a cursory and intensive EC_a survey, each with a different type of EC_a measurement equipment, is based on the type of information that each provided. The mobile EM equipment permitted only a point-to-point measurement of EC_a ; but at each point, EM_v and EM_h measurements of EC_a were taken. These provided a measure of cumulative electrical conductivity and a profile ratio that reflected chemical and hydraulic properties, respectively. As previously described, both pieces of information were valuable for sample-site selection. In contrast, the mobile fixed-array equipment provided a continuous measure of EC_a , thereby providing a detailed map of EC_a .

The fixed-array equipment was set to determine EC_a to a depth comparable to the root zone (i.e., 1.2 m). A total of 7288 fixed-array EC_a measurements were taken across the 32.4-ha study area. Eighty east–west transects were made with approximately 90 EC_a measurements taken in each transect. Transects were spaced roughly 8-m apart and there was roughly 4 m between each EC_a measurement within a traverse.

The intensive EC_a survey provided initial detailed spatial data that will be used to assess the temporal and spatial change in salinity patterns as drainage water is added to the soil over the duration of the project. With respect to initial soil quality, the intensive survey provides a detailed map of the spatial distribution of the soil properties correlated with the EC_a measurement.

2.2.4. Soil chemical and physical analyses

A number of physicochemical properties related to soil quality were determined from the 192 soil samples designated for analysis of soil chemical properties: 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+} ; ESP; and SAR. 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. Inorganic C, organic C, and total nitrogen were analyzed, but only for the top two depth increments (0–0.3 and 0.3–0.6 m) because of limited analytical resources. The chemical and physical methods used for each analysis were from ASA Agronomy Monograph No. 9 (Parts 1 and 2, 1986), except for the inorganic and organic C, which were analyzed with a UIC Coulometric Carbon Analyzer¹, and total N, which was analyzed with a Leco C–N 2000 Analyzer¹.

Undisturbed soil cores were collected at five sites in paddock 1 (sites 10, 19, 32, 45, and 47; Fig. 2), at the 0.25- and 0.5-m soil depths. PVC liners (3.58 cm inside diameter) were pushed into the soil to the desired soil depths. The samples were capped and

refrigerated for storage. These undisturbed soil samples were used for saturated hydraulic conductivity (K_s) analysis and for the analysis of associated physical (texture and ρ_b) and chemical properties (EC_e and SAR) that influence K_s .

Saturated hydraulic conductivity was measured by the constant head and falling head methods (Klute and Dirksen, 1986) using 0.01 M $CaCl_2$ solution. For those undisturbed cores with K_s values smaller than 0.01 cm h^{-1} , the falling head method was inaccurate because of the near-zero outflow volumes. Instead, the falling head method was modified by increasing the hydraulic gradient across the soil sample. The adaptation involved increasing the total water pressure at the inflow end of the soil sample using a positive air pressure source. With the 'pressurized falling head' method, the total head was varied between 200 and 600 cm, and outflow rates were measured accordingly. At the conclusion of the K_s experiment, samples were oven-dried from which ρ_b values were determined. Soil solution extracts of saturation pastes were analyzed for EC_e ($dS \text{ m}^{-1}$) and SARs were computed. Soil particle size analysis was determined by the hydrometer method (Gee and Bauder, 1986). For comparison purposes, in situ K_s was determined between the depths of 0.9 and 1.5 m at three sites in paddocks 6 and 7 (sites 264, 271, and 312) with the method of Youngs (2001).

2.2.5. GIS and map preparation

All spatial data were entered into a geographic information system (GIS) using the commercial GIS software ArcView 3.1. Maps of the soil physicochemical properties were prepared by interpolating the measurements at the 40 sample sites using inverse-distance-weighting (IDW) interpolation. Previous studies comparing interpolation methods for mapping soil properties have found kriging better (Laslett et al., 1987; Warrick et al., 1988; Leenaers et al., 1990; Kravchenko and Bullock, 1999), while others have shown inverse distance weighting (IDW) to be superior (Weber and Englund, 1992; Wollenhaupt et al., 1994; Gotway et al., 1996). At this particular study site, IDW was selected as the preferred method of interpolation because it was more accurate than kriging based on the use of the mean squared error as the main criterion when comparing measured to predicted values for the majority of physicochemical properties. Maps of mobile EM and fixed-array EC_a measurements were also prepared.

2.3. Bermuda grass establishment and sampling

Bermuda grass (*C. dactylon* (L.) Pers.) was planted at the site in spring (paddocks 1 to 4) and summer (paddocks 5 to 8) of 2000. Cultivar (cv.) *Giant* was planted in paddocks 1 to 4 to allow both grazing and hay making, while cv. *Common* was planted in paddocks 5 to 8, for grazing purposes only. Systematic sampling of paddocks 1 to 4 began in late September of 2000, and of all eight paddocks in June 2001. Forage was sampled at the same locations initially selected for soil sampling and at 36 additional locations (Fig. 2). As before, the 36 locations were selected using the ESAP software and EM EC_a data. Plant material was collected from two $0.3 \times 1\text{-m}$ grids at each location, placed opposite each other approximately 1 m from the soil sample point. Forage sample sites were located using a Trimble GPS¹ with submeter accuracy. At each sampling, forage was collected at a new compass direction to avoid resampling the same site. Sampling in this manner provides an estimate of standing biomass. While sampling, forage height before and after

harvest was measured. Samples were dried, usually starting late the same day of collection, in a forced air dryer at 35 °C. After drying, forages were ground through a Wiley mill with a 1-mm screen and subsampled for quality analysis. Forages were analyzed for total N, crude protein (CP), ash, acid detergent fiber (ADF), neutral detergent fiber (NDF), total P, K, S, Ca, Mg, Na, B, Zn, Mn, Fe, Cu, Mo, Se, and Cl. Samples for P, K, S, Ca, Mg, Na, B, Zn, Mn, Fe, Cu, and Mo were digested with nitric acid and hydrogen peroxide with microwave heating. The resulting digest was analyzed by inductively coupled plasma atomic emission spectrometer (ICP-AES) for P, K, S, Ca, and B, and by atomic absorption spectrometry for Na, Zn, Cu, Mn, and Fe (Sah and Miller, 1992). Selenium was analyzed by vapor generation inductively coupled plasma emission spectrometer (VG-ICP) following wet oxidation with nitric, perchloric, and sulfuric acids, and reduction with hydrochloric acid (Tracy and Moller, 1990). Chloride was determined using electrometric titration of the extract with silver using an automatic chloride titrator following extraction with 2% acetic acid (Johnson and Ulrich, 1959; Rhoades, 1982). Crude protein was determined based on total nitrogen content from analysis by combustion in an induction furnace and a thermal conductivity detection (Sweeney, 1989). Acid digestible fiber and neutral digestible fiber were determined using the filter bag method on an automated fiber system (Komarek et al., 1993, 1994). Ash was determined by gravimetric determination following high temperature combustion (AOAC method 942.05; Cunniff, 1995).

3. Results and discussion

3.1. Soil salinity, other physicochemical properties, and salinity mapping

EC_a measurements provide a rapid and easy means of establishing the three-dimensional distribution of soil salinity. Fig. 3 shows IDW interpolated maps of EM_h and EM_v measurements taken at the 384 sites. Statistical correlations performed between EM_v EC_a and EC_e and between EM_h EC_a and EC_e for various depth increments (i.e., 0–0.3, 0.3–0.6, 0.6–0.9, 0.9–1.2, and 0–1.2 m) show that the highest correlation occurs for the composite depth increment 0–1.2 m. The correlation coefficients between EM_v EC_a and EC_e, and between EM_h EC_a and EC_e for the composite 0–1.2 m depth increment are $r=0.81$ and $r=0.74$, respectively. Thus, the map of EM_v EC_a in Fig. 3 most closely reflects the spatial distribution of root zone salinity.

The EM EC_a data were valuable for sample design because it provided a quick and easy means of getting the cumulative EC_a through the root zone from the geometric mean. The EM EC_a data also provided a means of characterizing the shape of the EC_a profile from the profile ratio (i.e., EM_h/EM_v), thereby potentially indicating the degree of leaching that has occurred. Because both EM_v and EM_h EC_a were highly correlated with EC_e, the profile ratio reveals general leaching patterns of soluble salts (Fig. 4). Areas with a profile ratio >1 indicate that the past net flow of water and salts in these areas has been upward, whereas profile ratios <1 have net downward flows. Fig. 4 shows that all 384 locations had profile ratios <1.

The past leaching of water and salts at field scale is further substantiated by the increase in Cl⁻ concentration with depth (Tables 1–4). By using the average Cl⁻ concentration for

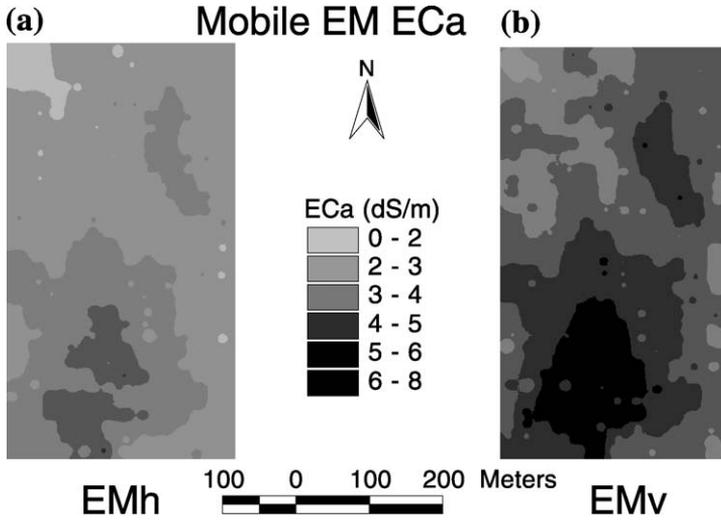


Fig. 3. IDW interpolated maps of (a) EM_h EC_a (dS m⁻¹) and (b) EM_v EC_a (dS m⁻¹) measurements taken at 384 sites.

the 0.9–1.2 m zone (Table 4) in the soil profile (i.e., the depth below the root zone, 58.7 mmolc l⁻¹) and an estimated Cl⁻ concentration in irrigation water used both currently and historically of 10 mmolc l⁻¹ (Kings River water), an average leaching fraction of approximately 17% is estimated.

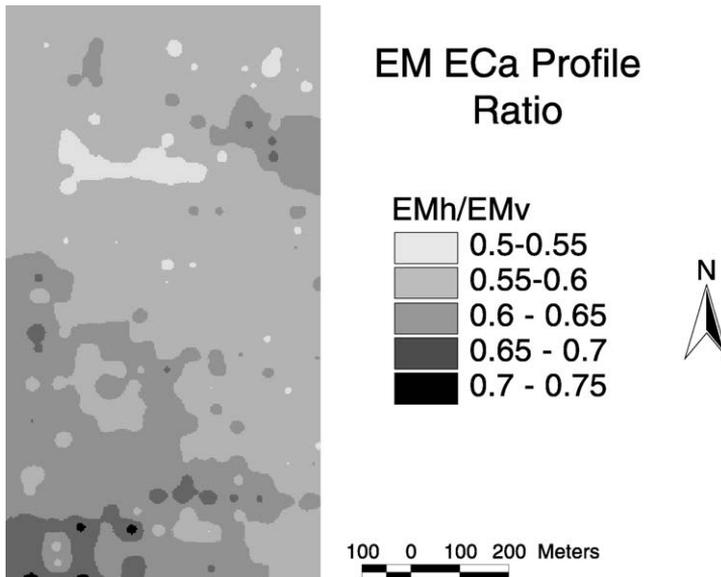


Fig. 4. IDW interpolated map of the EM EC_a profile ratio (i.e., EM_h/EM_v) at 384 sites.

Table 1
Mean and range statistics for 0.0–0.3 m sample depth

Soil property	Mean	Minimum	Maximum	Range	S.D.	S.E.	CV	Skewness	Pr < W*
θ_v (m ³ /m ³)	0.30	0.21	0.48	0.27	0.06	0.01	20.5	0.88	0.04
ρ_b (Mg/m ³)	1.29	1.11	1.52	0.41	0.10	0.02	7.7	0.29	0.62
Clay (%)	35.9	22.8	48.3	25.5	6.8	1.1	19.1	-0.22	0.49
EC _c (dS/m)	13.0	5.6	35.7	30.1	7.5	1.1	57.8	1.81	0.00
pH _c	7.6	6.9	8.3	1.4	0.3	0.04	3.3	-0.04	0.09
SP (%)	58.8	45.9	79.1	33.2	7.8	1.1	13.2	0.57	0.09
Anions in saturation extract (mmolc/l)									
HCO ₃ ⁻	5.25	2.78	10.71	7.93	1.97	0.29	37.6	1.03	0.00
Cl ⁻	21.81	3.29	67.25	63.96	15.18	2.19	69.6	1.45	0.00
NO ₃ ⁻	0.70	0.15	8.35	8.20	1.36	0.20	195.0	4.45	0.00
SO ₄ ⁻	150.00	60.59	535.00	474.41	108.36	15.64	72.2	2.12	0.00
Cations in saturation extract (mmolc/l)									
Na ⁺	136.78	34.41	490.91	456.50	109.88	15.86	80.3	1.87	0.00
K ⁺	0.90	0.41	3.65	3.24	0.56	0.08	62.0	3.11	0.00
Ca ²⁺	23.92	17.62	28.00	10.38	2.09	0.30	8.8	-0.59	0.10
Mg ²⁺	18.60	8.60	100.74	92.14	17.56	2.54	94.4	3.32	0.00
Exchangeable cations (mmolc/kg)									
Na ⁺	58.8	21.8	172.0	150.2	27.8	4.0	47.3	1.65	0.00
K ⁺	10.9	3.1	15.5	12.4	2.3	0.3	21.1	-1.13	0.01
Ca ²⁺	85.8	0.0	170.7	170.7	43.9	6.3	51.2	0.01	0.37
Mg ²⁺	62.7	31.5	86.8	55.3	12.6	1.8	20.1	-0.33	0.20
SAR	28.2	8.3	70.2	61.9	16.5	2.4	58.7	1.08	0.00
ESP (%)	28.4	9.5	83.4	73.9	14.8	2.1	52.0	1.28	0.00
B (mg/l)	17.0	1.1	42.5	41.4	8.2	1.2	48.2	1.21	0.00
Se (μg/l)	8.8	0.0	77.0	77.0	12.7	1.83	144.8	3.76	0.00
As (μg/l)	8.2	0.0	54.0	54.0	12.4	1.79	151.5	1.81	0.00
Mo (μg/l)	862.3	442.0	3043.0	2601.0	532.5	76.9	61.8	2.72	0.00
CEC (mmolc/kg)	216.7	126.8	287.4	160.6	38.7	5.6	17.8	-0.09	0.33
CaCO ₃ (g/kg)	10.8	0.1	32.3	32.2	8.1	1.2	75.5	1.00	0.00
Gypsum (g/kg)	34.1	11.1	78.0	66.9	17.5	2.5	51.4	0.68	0.00
Inorganic C (g/kg)	1.1	0.0	4.6	4.6	1.2	0.2	108.0	1.06	0.00
Organic C (g/kg)	6.2	3.2	8.9	5.7	1.5	0.2	24.8	-0.18	0.05
N (g/kg)	0.7	0.3	0.9	0.6	0.1	0.0	22.1	-0.54	0.17

N=48.

*Distribution statistics: Shapiro–Wilk probability level for test of normality; values <0.05 indicate non-normality.

Fig. 5 shows a map of intensive EC_a measurements taken with the mobile fixed-array equipment, which estimates cumulative salinity distributions to a depth of 1.2 m. More specifically, the intensive survey provides a detailed map that depicts the spatial extent and magnitude of salt accumulation through the root zone (i.e., top 1.2 m). The map also explicitly shows the degree of spatial variability of salinity.

As expected, there is a general EC_a pattern similarity between Figs. 3b and 5. Nonetheless, there are differences. The spatial differences are primarily due to interpolation, while differences in magnitude are due to instrumentation. Fig. 5 is a map of 7288 EC_a measurements, whereas Fig. 3b is an interpolated map of 384 EC_a measurements. Spatial differences due to interpolation are unavoidable. Differences in absolute EC_a

Table 2
Mean and range statistics for 0.3–0.6 m sample depth

Soil property	Mean	Minimum	Maximum	Range	S.D.	S.E.	CV	Skewness	Pr < W*
θ_v (m^3/m^3)	0.40	0.29	0.52	0.23	0.05	0.01	11.8	0.36	0.85
ρ_b (Mg/m^3)	1.51	1.31	1.72	0.41	0.10	0.02	6.5	-0.03	0.65
Clay (%)	30.4	21.8	46.9	25.1	4.9	0.8	16.0	0.82	0.05
EC _e (dS/m)	20.2	13.5	34.5	21.0	5.3	0.8	26.0	1.05	0.00
pH _e	7.6	7.0	8.0	1.0	0.2	0.0	2.6	-0.89	0.02
SP (%)	63.0	45.5	84.1	38.6	10.2	1.5	16.2	0.01	0.16
Anions in saturation extract (mmolc/l)									
HCO ₃ ⁻	2.67	0.80	4.92	4.12	0.71	0.10	26.8	0.40	0.19
Cl ⁻	35.30	12.94	68.98	56.04	14.33	2.07	40.6	0.50	0.06
NO ₃ ⁻	0.89	0.15	6.74	6.59	1.47	0.21	165.2	3.01	0.00
SO ₄ ⁻	239.51	133.21	460.56	327.35	76.30	11.01	31.9	1.42	0.00
Cations in saturation extract (mmolc/l)									
Na ⁺	237.20	136.00	457.37	321.37	77.23	11.15	32.6	1.18	0.00
K ⁺	1.03	0.25	2.12	1.87	0.41	0.06	39.6	0.86	0.02
Ca ²⁺	22.08	12.45	30.90	18.45	2.46	0.36	11.1	-0.29	0.00
Mg ²⁺	20.43	10.00	77.47	67.47	12.31	1.78	60.3	3.39	0.00
Exchangeable cations (mmolc/kg)									
Na ⁺	79.0	47.1	138.9	91.8	19.9	2.9	25.2	0.82	0.03
K ⁺	6.7	3.5	12.0	8.5	2.1	0.3	31.7	0.76	0.02
Ca ²⁺	59.6	2.4	263.0	260.6	41.9	6.0	70.2	2.51	0.00
Mg ²⁺	49.4	34.5	66.8	32.3	9.5	1.4	19.2	0.18	0.06
SAR	51.4	30.3	89.5	59.2	12.9	1.9	25.1	0.87	0.02
ESP (%)	41.6	23.7	63.4	39.7	9.4	1.4	22.7	0.30	0.28
B (mg/l)	19.0	13.6	38.1	24.5	5.6	0.8	29.7	1.90	0.00
Se (μg/l)	14.0	0.0	45.0	45.0	8.5	1.2	60.9	0.79	0.01
As (μg/l)	8.8	0.0	58.0	58.0	13.2	1.9	149.6	1.86	0.00
Mo (μg/l)	750.5	180.0	2488.0	2308.0	430.2	62.1	57.3	2.63	0.00
CEC (mmolc/kg)	194.8	118.5	396.3	277.8	51.1	7.4	26.2	1.56	0.00
CaCO ₃ (g/kg)	10.4	0.0	57.6	57.6	10.7	1.5	102.8	2.23	0.00
Gypsum (g/kg)	53.7	11.4	142.7	131.3	32.1	4.6	59.8	0.94	0.00
Inorganic C (g/kg)	1.1	0.0	8.7	8.7	1.6	0.2	144.5	2.81	0.00
Organic C (g/kg)	3.2	1.9	6.4	4.5	1.0	0.1	30.1	1.06	0.01
N (g/kg)	0.5	0.2	3.8	3.6	0.5	0.1	106.0	6.47	0.00

N=48.

*Distribution statistics: Shapiro–Wilk probability level for test of normality; values <0.05 indicate non-normality.

values are because EM EC_a, as measured with the Geonics EM-38, is influenced by a sensitivity function that is most sensitive to the EC_a of the soil at a depth of about 0.4 m (McNeill, 1980), whereas the measurement of EC_a with the fixed-array equipment is uniformly sensitive through the profile (0–1.2 m). This discrepancy does not permit a one-to-one comparison between EM EC_a and the fixed-array EC_a. The comparison can only be on a relative, not absolute, basis.

The mobile EM and fixed-array surveys (see Figs. 3b and 5) indicate a large area of very high EC_a measurements in the southwestern portion of the study area that extends through a substantial portion of paddocks 2, 3, and 4. In addition, a smaller area of high EC_a exists in the northeast primarily spreading through paddocks 6 and 7. Figs. 4 and 6a

Table 3
Mean and range statistics for 0.6–0.9 m sample depth

Soil property	Mean	Minimum	Maximum	Range	S.D.	S.E.	CV	Skewness	Pr < W*
θ_v (m^3/m^3)	0.40	0.23	0.51	0.28	0.05	0.01	12.4	-0.72	0.07
ρ_b (Mg/m^3)	1.52	1.16	1.80	0.64	0.12	0.02	8.0	-0.42	0.60
Clay (%)	26.2	2.5	42.0	39.5	6.8	1.1	25.9	-0.55	0.01
EC _c (dS/m)	22.5	9.7	43.2	33.5	6.5	0.9	28.7	1.27	0.00
pH _c	7.6	7.3	8.1	0.8	0.2	0.0	2.3	0.32	0.53
SP (%)	59.1	40.0	89.5	49.5	11.1	1.6	18.8	1.16	0.00
Anions in saturation extract (mmolc/l)									
HCO ₃ ⁻	2.62	1.10	9.01	7.91	1.18	0.17	44.8	3.61	0.00
Cl ⁻	47.07	5.48	100.48	95.00	21.82	3.15	46.3	0.68	0.06
NO ₃ ⁻	0.65	0.10	4.34	4.24	0.81	0.12	124.6	2.93	0.00
SO ₄ ⁻	258.84	121.51	570.77	449.26	89.44	12.91	34.6	1.68	0.00
Cations in saturation extract (mmolc/l)									
Na ⁺	270.23	98.60	603.00	504.40	97.80	14.12	36.2	1.49	0.00
K ⁺	1.04	0.37	2.50	2.13	0.43	0.06	41.0	1.22	0.00
Ca ²⁺	22.12	13.85	28.10	14.25	2.24	0.32	10.1	-0.79	0.00
Mg ²⁺	19.07	9.95	46.37	36.42	7.80	1.13	40.9	1.60	0.00
Exchangeable cations (mmolc/kg)									
Na ⁺	78.8	35.2	132.2	97.0	23.8	3.4	30.2	0.51	0.17
K ⁺	4.7	1.8	12.4	10.6	2.3	0.3	48.5	1.35	0.00
Ca ²⁺	45.5	0.0	173.9	173.9	36.8	5.3	80.9	1.20	0.00
Mg ²⁺	43.2	27.2	70.6	43.4	11.1	1.6	25.7	0.86	0.00
SAR	59.0	24.0	107.6	83.6	16.6	2.4	28.1	0.76	0.12
ESP (%)	47.5	28.9	95.0	66.1	13.5	1.9	28.4	1.18	0.00
B (mg/l)	17.5	9.4	31.3	21.9	4.8	0.7	27.2	0.87	0.02
Se (μ g/l)	12.9	0.0	34.0	34.0	9.2	1.3	71.5	0.67	0.02
As (μ g/l)	12.9	0.0	116.0	116.0	23.4	3.4	181.0	2.82	0.00
Mo (μ g/l)	780.5	183.0	1756.0	1573.0	338.9	48.9	43.4	0.90	0.02
CEC (mmolc/kg)	170.2	99.6	338.5	238.9	48.5	7.00	28.5	1.47	0.00
CaCO ₃ (g/kg)	11.4	0.3	46.7	46.4	12.5	1.8	110.1	1.46	0.00
Gypsum (g/kg)	66.3	5.6	215.3	209.7	39.8	5.7	60.0	1.30	0.00

$N=48$.

*Distribution statistics: Shapiro–Wilk probability level for test of normality; values <0.05 indicate non-normality.

indicate that soil salinity increases with depth throughout the study area. In contrast to the EM profile ratio data, the ground-truth chemical analysis data show that the salinity profile is inverted in two small areas near the southwest and northeast corners as indicated by the high EC_c measurements at 0–0.3 m overlying lower EC_c measurements at 0.3–0.6, 0.6–0.9, and 0.9–1.2 m (Fig. 6a). Specifically, this occurred at only four soil sample locations #10, #32, #59, and #382 (see Fig. 2). Nevertheless, the profile ratio from the noninvasive EM EC_a measurements is a reasonable means of characterizing the general shape of the profile.

Tables 1–4 are compilations by depth (0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m, respectively) of the statistical data characterizing significant physicochemical properties of soil quality for the entire study area. The mean, minimum, and maximum values show the study area to be a severely saline-sodic soil that is heavy textured at the surface (0–0.3 m) and has low to very high levels of B and moderate to high levels of Mo. Typically, water

Table 4
Mean and range statistics for 0.9–1.2 m sample depth

Soil property	Mean	Minimum	Maximum	Range	S.D.	S.E.	CV	Skewness	Pr < W*
θ_v (m^3/m^3)	0.43	0.31	0.54	0.23	0.05	0.01	11.4	-0.37	0.95
ρ_b (Mg/m^3)	1.51	1.14	1.75	0.61	0.16	0.03	10.7	-0.85	0.03
Clay (%)	23.3	11.1	36.9	25.8	6.3	1.1	26.9	0.08	0.26
EC _e (dS/m)	25.2	8.0	49.7	41.7	7.9	1.1	31.5	0.81	0.02
pH _e	7.6	7.1	8.0	0.9	0.2	0.0	2.7	-0.37	0.26
SP (%)	58.7	38.5	93.3	54.8	12.9	1.9	22.0	0.68	0.08
Anions in saturation extract (mmol/l)									
HCO ₃ ⁻	3.06	1.19	7.76	6.57	1.49	0.22	48.6	1.98	0.00
Cl ⁻	58.66	3.18	152.68	149.50	30.15	4.35	51.4	0.80	0.02
NO ₃ ⁻	0.32	0.12	1.29	1.16	0.20	0.03	62.8	3.44	0.00
SO ₄ ⁻	291.98	91.03	720.96	629.93	114.02	16.46	39.0	1.52	0.00
Cations in saturation extract (mmol/l)									
Na ⁺	311.57	66.10	738.69	672.59	126.13	18.21	40.5	1.20	0.00
K ⁺	1.06	0.34	2.36	2.02	0.47	0.07	44.0	0.85	0.02
Ca ²⁺	22.34	16.68	26.50	9.82	2.11	0.31	9.5	-0.64	0.13
Mg ²⁺	22.30	9.33	52.05	42.72	10.42	1.50	46.7	1.27	0.00
Exchangeable cations (mmol/kg)									
Na ⁺	87.7	32.4	192.0	159.6	32.0	4.6	36.5	1.10	0.01
K ⁺	4.1	1.1	9.8	8.7	2.1	0.3	51.7	0.88	0.01
Ca ²⁺	42.5	0.0	154.2	154.2	35.4	5.1	83.3	0.97	0.00
Mg ²⁺	46.4	26.6	90.3	63.7	14.7	2.1	31.7	1.09	0.00
SAR	64.9	16.8	120.2	103.4	19.5	2.8	30.0	0.35	0.41
ESP (%)	51.8	18.2	135.7	117.5	20.2	2.9	39.0	1.82	0.00
B (mg/l)	17.9	6.5	31.8	25.3	6.3	0.9	35.0	0.45	0.27
Se (μ g/l)	14.1	0.0	62.0	62.0	13.9	2.0	98.5	1.80	0.00
As (μ g/l)	4.4	0.0	30.0	30.0	8.1	1.2	183.3	2.04	0.00
Mo (μ g/l)	946.9	330.0	2856.0	2526.0	450.7	65.0	47.6	1.78	0.00
CEC (mmol/kg)	175.3	94.6	307.5	212.9	47.1	6.8	26.9	0.81	0.03
CaCO ₃ (g/kg)	12.7	0.6	64.8	64.2	14.6	2.1	115.2	1.79	0.00
Gypsum (g/kg)	64.1	0.8	251.4	250.6	46.4	6.7	72.4	2.06	0.00

N = 48.

* Distribution statistics: Shapiro–Wilk probability level for test of normality; values < 0.05 indicate non-normality.

content increases with depth (Tables 1–4). Bulk density is the lowest in the top depth increment (0–0.3 m) with an average value of 1.29 Mg m^{-3} , and is stable at 1.51 Mg m^{-3} for the remaining depths. pH_e typically averages around 7.5–7.6 for all depths and usually falls within the range of 7–8. Saturation percentage ranges from less than 40% to over 90% with the greatest range occurring at the bottom depth (0.9–1.2 m). The levels of Se and As are low with Se never exceeding $77 \mu\text{g l}^{-1}$ and As never exceeding $116 \mu\text{g l}^{-1}$ at any depth increment (Tables 1–4).

The general spatial patterns of Cl⁻ (Fig. 6b) and SAR (Fig. 6c) tend to be associated with EC_e (Fig. 6a) and, like EC_e, tend to increase with depth (see Tables 1–4). To a lesser extent, the spatial patterns of ESP (Fig. 6d) follow those of EC_e. The association between salinity (EC_e) and the soil chemical properties of Cl⁻, SAR, and ESP is reflected by correlation coefficients determined for EC_e and Cl⁻ ($r=0.76$), EC_e and SAR ($r=0.95$), and EC_e and ESP ($r=0.54$) using values for composite soil cores over the depth of 0–1.2

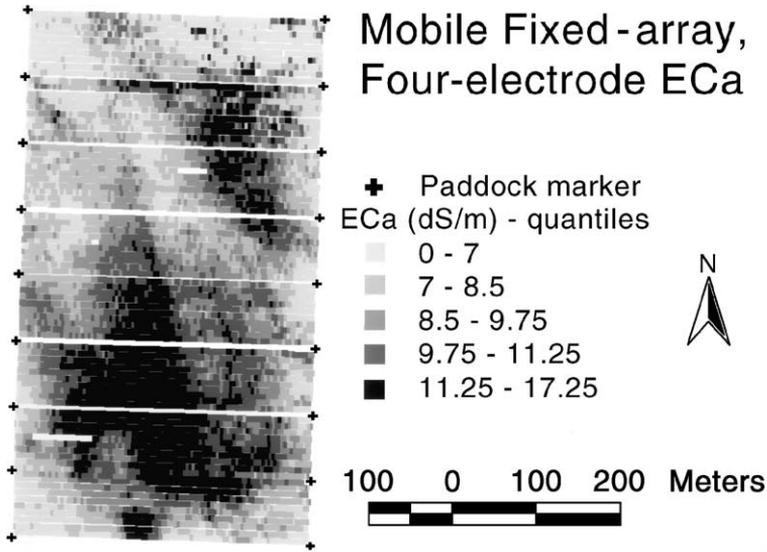


Fig. 5. A detailed map of 7288 EC_a measurements taken to a depth of 1.2 m with mobile fixed-array equipment.

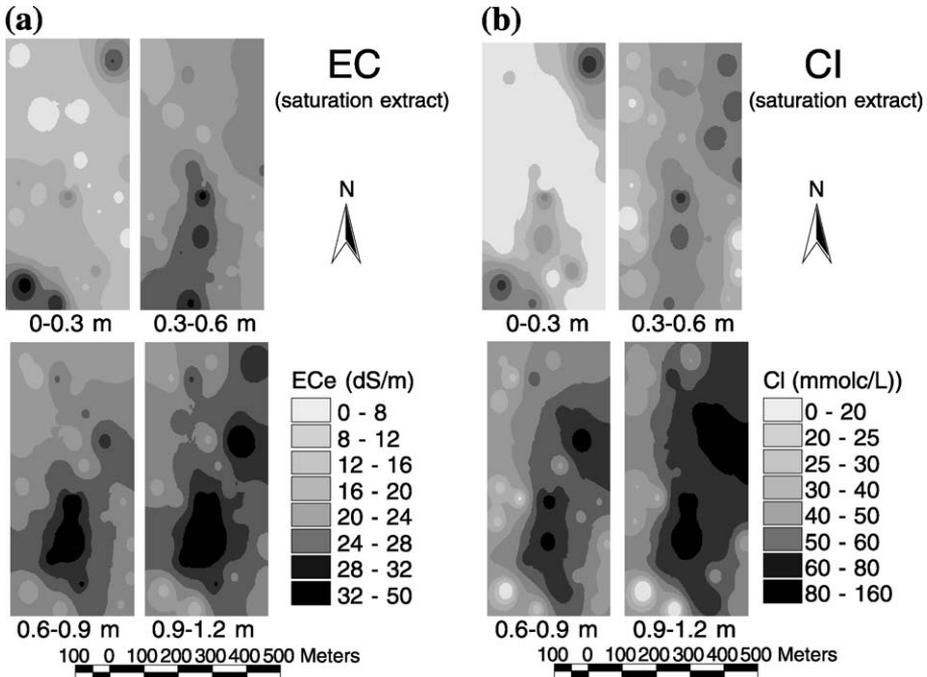


Fig. 6. IDW interpolated maps of measurements taken at 384 sites for the depths of 0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m for (a) EC_c ($dS\ m^{-1}$), (b) Cl^- ($mmolc\ l^{-1}$), (c) SAR, and (d) ESP.

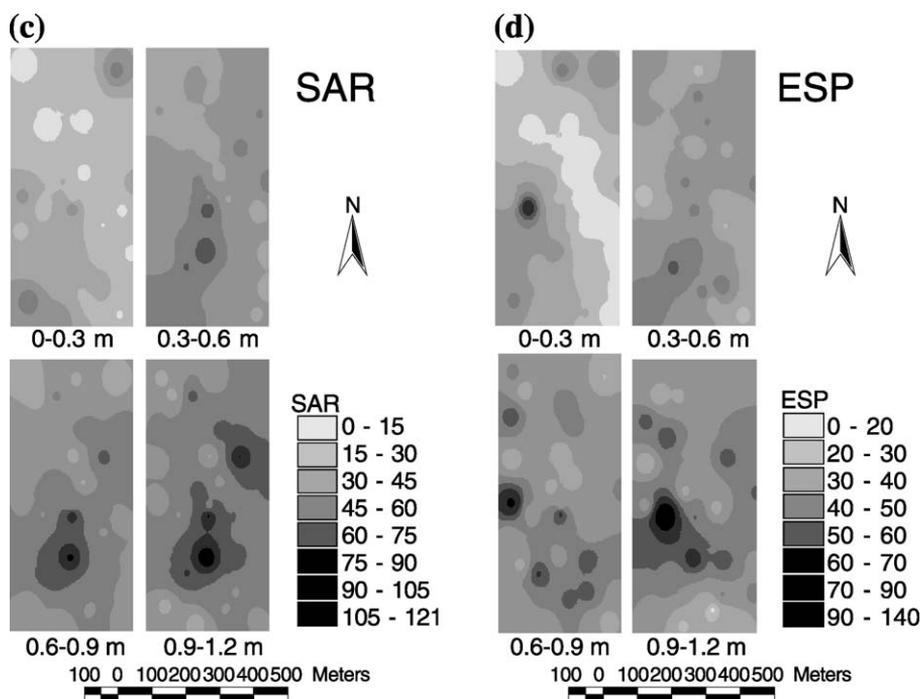


Fig. 6 (continued).

m. CEC correlates well with SP ($r=0.60$), indicating the influence that clay content has on the properties of CEC and SP.

The general spatial patterns of Mo and B (Fig. 7a and b, respectively) tend to coincide, particularly in the top two depths (0–0.3 m and 0.3–0.6 m). The correlation coefficient between Mo and B using values for composite soil cores over the depth of 0–1.2 m is $r=0.59$. The high levels of B in the shallow depths (0–0.3 and 0.3–0.6 m) of the southwest corner (Fig. 7b) pose a potential threat to crop growth. High levels of Mo in the southwest corner at all depths (Fig. 7a) define areas where Mo uptake by plants can pose a threat to ruminant animals. In the top depth increment (i.e., 0–0.3 m), Mo and B are positively correlated with salinity:

$$B = 6.0 + 0.84EC_e \quad (r^2 = 0.60) \quad (1a)$$

$$Mo = 88.2 + 59.9EC_e \quad (r^2 = 0.72) \quad (1b)$$

where B, Mo, and EC_e are the boron concentration (mg l^{-1}), Mo concentration ($\mu\text{g l}^{-1}$), and electrical conductivity (dS m^{-1}) of the saturation extract, respectively. This suggests that locations with high salinity may also be locations with forages high in these trace elements. Except for a high mean concentration at 0–0.3 m, Mo tends to increase with depth similar to soil salinity, i.e., EC_e (see Tables 1–4). The high levels of Mo and B,

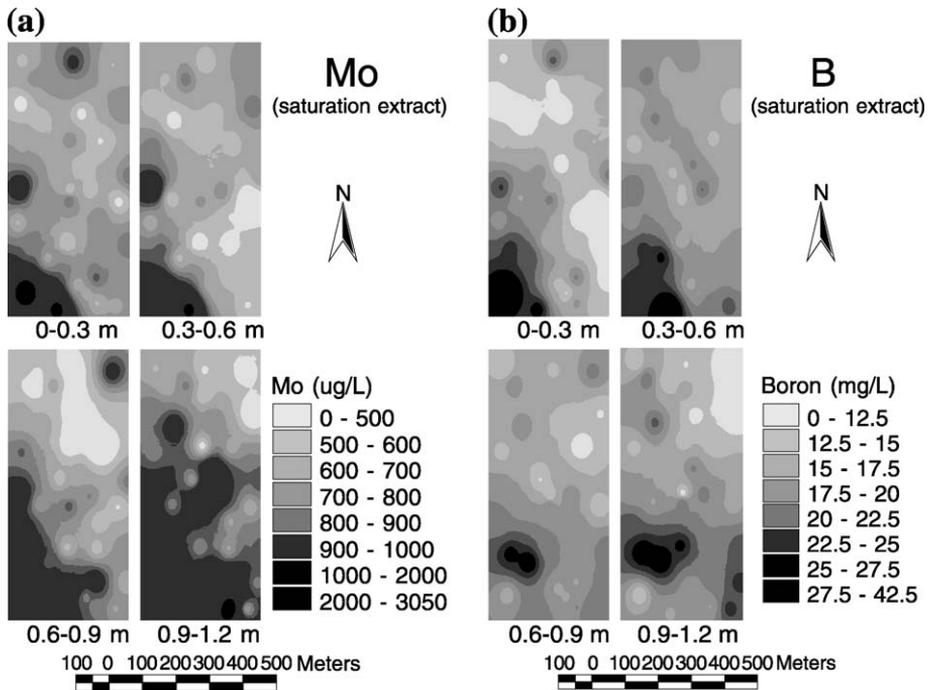


Fig. 7. IDW interpolated maps of measurements taken at 384 sites for the depths of 0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m for (a) Mo ($\mu\text{g l}^{-1}$), and (b) B (mg l^{-1}).

particularly near the soil surface (i.e., 0–0.3 m) in the southwest corner, are areas of concern with regard to the reuse of drainage water. Because of the high affinity for the adsorption of Mo and B onto clay surfaces, the reuse of drainage waters high in Mo and B will elevate surface concentrations (in both adsorbed and solution phases) to points where they may exceed harmful levels for some plants (e.g., soil solution B $>10 \text{ mg l}^{-1}$) and livestock (e.g., soil solution Mo $>5000 \mu\text{g l}^{-1}$). In irrigated agriculture, it has been recommended that concentrations of Mo not exceed 0.01 mg l^{-1} for continuous application on all soils, or 0.05 mg l^{-1} for short-term use on soils that react with Mo (NAS, 1973). More recently, Albasel and Pratt (1989) recommended a guideline of 0.05 mg l^{-1} maximum Mo concentration for saline SO_4^- dominated waters used on alkaline soils of the WSJV for the protection of bovine animals.

Compared to other locations in the WSJV (Deverel et al., 1984; Deverel and Fujii, 1988; Deverel and Millard, 1988; Fujii and Deverel, 1989; Fujii and Swain, 1995), field-averaged Se and As levels are low while average B and Mo levels are representative with salinity (EC_c) and SAR reaching high levels. However, there are specific locations within the study site with large amounts of B and Mo. From a global perspective (U.S. Salinity Laboratory Staff, 1954; Lindsay, 1979; Kabata-Pendias and Pendias, 1992), the salinity, SAR, B, and Mo tend to be above average to high.

Organic C and total N are very low (Tables 1 and 2), which is typical of WSJV soils. Soil organic matter tends to rapidly decompose due to the high year-round temperatures. In

general, NO_3^- levels tend to be quite low with the NO_3^- concentration for the top 0–1.2 m of soil averaging $0.65 \text{ mmolc l}^{-1}$ for the entire study area. The highest level of nitrate occurs at the 0.3–0.6 m depth increment (see Tables 1–4). Spatially, the highest nitrate levels occur at the 0.3–0.6 m depth increment in isolated locations at the north end in the middle of paddock 8 and at the south end with a high pocket extending through the mid-portions of paddocks 1, 2, 3, and 4 (Fig. 8a).

Fig. 8b shows that, in general, the clay content tends to decrease with depth. The average percentage of clay for the 0–0.3 m depth increment is 35.9% and gradually decreases to 23.3% at the 0.9–1.2 m depth increment (see Tables 1–4). The higher clay content at the surface will enhance the formation of surface cracks, which will likely serve as conduits for water flow before the clay swells and closes the cracks. The averages of %sand, %silt, and %clay for the top 0–1.2 m are 38.6%, 32.4%, and 29.0%, respectively, making the soil a clay loam. The silt content tends to remain fairly constant through the root zone with values of 32.8%, 31.1%, 31.3%, and 34.5% for the respective depths of 0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m. However, sand content tends to increase with depth with values of 31.3%, 38.5%, 42.5%, and 42.2% for the respective depths of 0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m. The increase in the sand fraction with depth should be conducive to drainage.

A comparison of the coefficients of variation (CV) for various chemical and physical properties at each of the four depth increments shows a range of spatial variability with

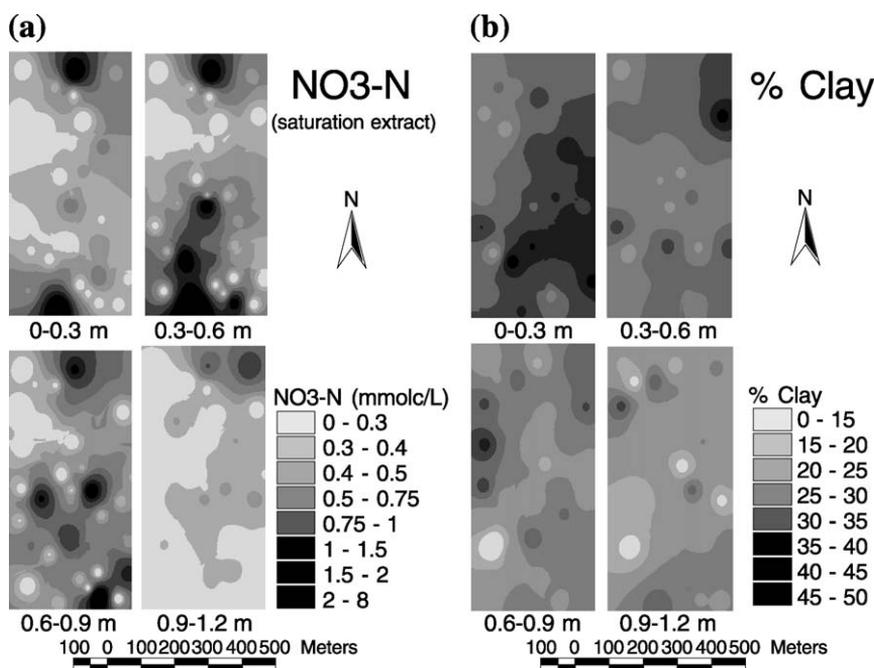


Fig. 8. IDW interpolated maps of measurements taken at 384 sites for the depths of 0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m for (a) $\text{NO}_3\text{-N}$ (mmolc l^{-1}) and (b) %clay.

some general trends (see [Tables 1–4](#)). The properties of %CaCO₃, NO₃⁻ in the saturation extract, exchangeable Ca²⁺, Se, and As consistently have the highest CVs. Anions and cations in the saturation extract, except for NO₃⁻ and Ca²⁺, and exchangeable cations, except for Ca²⁺ and Mg²⁺, tend to have low to high CVs (i.e., CVs ranging from 20 to 80). SAR, ESP, EC_e, B, Mo have low to moderate CVs (i.e., CVs from 20 to 50). Clay percentage and θ_v have low CVs (i.e., CVs from 10 to 20) with pH_e, ρ_b , and Ca²⁺ in the saturation extract consistently having the very lowest CVs (CVs generally less than 10). The chemical property of pH_e and the physical properties of ρ_b , and SP are the only soil properties consistently normally distributed.

The partitioning of the local- and global-scale variability using one-way ANOVA on duplicate composite (0–1.2 m) soil samples indicates that the greatest local-scale variability is for the properties of pH_e, Se, and %clay, while the lowest is for %sand, exchangeable Na⁺, and Mo. The local-scale variability of pH_e is the highest at 16.5%, which indicates that 16.5% of the variability of the entire 32.4-ha field can be found within 1 m. At the other extreme, the local-scale variability of %sand is only 1.2%. It should be noted that the local-scale variability includes variation in reproducibility in methodology.

Correlations between EM EC_a (both EM_h and EM_v EC_a) and various soil properties show that EC_e; SO₄⁻, Na⁺, and Mg²⁺ in the saturation extract; and SAR are highly correlated ($r > 0.7$), while θ_v ; Cl⁻, NO₃⁻, and K⁺ in the saturation extract; ESP; B; Se; and Mo are less positively correlated, but still significant at the $P < 0.05$ level ([Table 5](#)). A modest negative correlation exists between EM EC_a and the properties of CaCO₃, and inorganic and organic C. These correlations indicate that the EC_a-based sample design utilized in this study will provide the best spatial representation for these properties, whereas a random or stratified random sample design probably would have been better for the remaining properties. Because the properties correlated to EC_a are also those of greatest significance when evaluating the resultant spatio-temporal changes due to drainage water reuse on an arid zone soil, the sample design is not likely to compromise the evaluation of the sustainability of drainage water reuse at the study site, but sample design modifications would likely improve the spatial characterization of the poorly correlated properties (i.e., ρ_b ; %clay; pH_e; SP; HCO₃⁻, and Ca²⁺ in the saturation extract; exchangeable Na⁺, K⁺, and Mg²⁺; As; CEC; gypsum; and total N).

K_s was determined at only five sites, so the spatial characterization of K_s is not as comprehensive as the other physicochemical properties. The extensive spatial database of physicochemical properties suggests that K_s values are not likely to be significantly different at other points within the study area. The values obtained provide a reasonable initial representation of K_s that can be used in future temporal comparisons over the course of the drainage water reuse experiment.

[Table 6](#) provides the K_s data and associated properties (i.e., texture, ρ_b , EC_e, and SAR) that influence K_s . Some measured K_s values were purposely omitted because their high outflow rates were suspect, indicating preferential water flow between plastic PVC liner and soil. SAR and EC_e data indicate that the relatively high sodium concentration does not cause dispersion because of the high concentration of salinity in the samples. The high solution salinity, despite using a 0.01 M calcium solution for the hydraulic conductivity measurements, is a consequence of the extremely low water flow rates during the K_s measurements, draining less than 1 pore volume. Therefore, it is expected that the CaCl₂

Table 5

Correlation coefficients between EM EC_a (both EM_h and EM_v) and soil properties measured over 0–1.2 m

Soil property	EM _h EC _a	EM _v EC _a
θ_v (m ³ /m ³) ^a	0.62**	0.64**
ρ_b (Mg/m ³) ^a	– 0.35	– 0.31
Clay (%) ^b	0.29	0.25
EC _e (dS/m)	0.74**	0.78**
pH _e	0.01	– 0.01
SP (%)	0.25	0.22
Anions in saturation extract (mmolc/l)		
HCO ₃ [–]	0.05	0.06
Cl [–]	0.34*	0.43**
NO ₃ [–]	0.47**	0.34*
SO ₄ [–]	0.81**	0.82**
Cations in saturation extract (mmolc/l)		
Na ⁺	0.76**	0.80**
K ⁺	0.69**	0.69**
Ca ²⁺	0.18	0.23
Mg ²⁺	0.74**	0.65**
Exchangeable cations (mmolc/kg)		
Na ⁺	0.19	0.21
K ⁺	– 0.15	– 0.15
Ca ²⁺	– 0.34*	– 0.36*
Mg ²⁺	0.14	0.14
SAR	0.64**	0.70**
ESP (%)	0.31*	0.32*
B (mg/l)	0.48**	0.42**
Se (μg/l)	0.56**	0.51**
As (μg/l)	0.15	0.15
Mo (μg/l)	0.58**	0.45**
CEC (mmolc/kg)	– 0.12	– 0.12
CaCO ₃ (g/kg)	– 0.44**	– 0.37**
Gypsum (g/kg)	0.19	0.18
Inorganic C ^c (g/kg)	– 0.53**	– 0.49**
Organic C ^c (g/kg)	– 0.51**	– 0.48**
N ^c (g/kg)	– 0.13	– 0.10

N = 48.

^a N = 31.^b N = 41.^c Measured over 0–0.6 m.

* Significant (test for |r| = 0) at P < 0.05 level.

** Significant (test for |r| = 0) at P < 0.01 level.

solution did not affect the laboratory-measured soil hydraulic properties. To further ensure that this was indeed the case, the K_s was determined for an additional three soil samples, using calcium chloride solution of both 0.01 and 0.1 M. The increase in molar concentration increased the K_s a little for one soil sample, but slightly reduced the K_s for the two other samples.

The laboratory-measured K_s values were very low, indicating a general lack of soil structure. The lowest reported measured K_s value (site 32, depth 0.25 m) corresponds to the sample with the highest measured ρ_b . With the pressurized falling head method, K_s

Table 6
Soil hydraulic and associated properties (paddock 1)

Sample-site depth (cm)	Saturated hydraulic conductivity (cm h ⁻¹)			ρ_b (Mg m ⁻³)	Sand (%)	Silt (%)	Clay (%)	EC _c (dS m ⁻¹)	SAR
	Constant	Falling	Pressurized falling						
10–25	–	0.00194	0.00148	1.13	29	31	40	12.0	8
10–50	–	–	–	1.28	30	45	25	10.0	8
19–50	0.0456	0.0232	–	1.35	30	41	29	17.1	30
32–25	–	–	0.000705	1.44	28	41	31	13.8	22
32–50	0.0689	–	–	1.36	25	43	32	14.9	23
45–25	–	–	0.00707	1.30	27	44	29	21.1	33
45–50	0.00311	0.00146	0.00171	1.29	31	54	15	17.1	29
47–25	–	–	0.0000846	–	–	–	–	–	–
47–50	–	0.0798	–	1.34	28	34	38	16.2	24

(–) Not measured.

values of 0.0001 cm h⁻¹ and smaller could be measured, although the structural integrity of the soil was affected by the higher air pressures, as evidenced by the presence of suspended soil particles in the effluent.

Limited field experiments were conducted to estimate in situ K_s . Successful measurements were obtained for sites 264, 271, and 312 (see Fig. 2), with in situ K_s values of 1.79, 0.5, and 0.49 cm h⁻¹, respectively. These values are much larger than measured in the laboratory (Table 6). The hydraulic conductivity results of the in situ method pertain to a much larger soil volume than the laboratory measurements, although the exact size of the effective soil volume is uncertain (Youngs, 2001), and includes the effects of anisotropy of the conductivity field as well. Moreover, the in situ measurements represent the effective K_s between 0.9 and 1.5 m below the soil surface, whereas the laboratory measurements were determined from soil cores from the first 0.5 m.

The extremely low K_s values are typical of soils in the WSJV and are certainly a limiting factor in the management of these soils. However, farmers have developed a means of working around the permeability of WSJV soils. Soil of the type found at the Westlake Farm study site forms surface cracks as it dries. These cracks create prismatic columns of soil that extend deeper into the soil profile as the soil continues to dry by evapotranspiration. Historically, farmers working this type of soil have used these cracks as a conduit for getting moisture into the soil profile because of the low permeability of the soil in the absence of the cracks. After the soil dries and the cracks have formed, the farmer will irrigate the field. Irrigation water moves down the cracks and then moves laterally into the prismatic columns of soil. The expanding lattice clay swells from the presence of water. This swelling gradually seals the cracks, which decreases the permeability to a level near the K_s . Once sealed, very little irrigation water will penetrate or move through the soil.

3.2. Bermuda grass establishment and yield

Results are reported from sampling in fall 2000 and spring 2001 on paddocks 1 to 4 only. Bermuda grass is a halophytic, C4 species with a large degree of salt tolerance (Ayers

and Westcott, 1976). Grass yields declined with increasing soil salinity in the first 0.3 m of soil (Fig. 9). Above approximately 22 dS m^{-1} in the saturation extract, little to no Bermuda grass was able to establish. This response matched estimates of salinity tolerance reported by Ayers and Westcott (1976) and Maas and Hoffman (1977). In addition to the influence of soil chemical properties like EC_e , forage yield is also a function of soil moisture content and other factors not specifically measured. At some sites with low EC_e values, forage yields may be depressed due to uneven water distribution. The data presented in Fig. 9 represent the upper limit of forage yields at the time of sampling from well-irrigated locations in the first four paddocks. These paddocks represent the full range of conditions found at the site and forage responses reflect the initial performance of Bermuda grass to site conditions during the establishment phase of the crop, before long-term irrigation with drainage water has occurred.

Both B and Mo are also present at the site and are positively correlated with EC_e ; consequently, in locations with the largest amounts of soil B and Mo, which were also those locations high in salinity, there was little to no forage to collect, and none for cattle to graze. Means, standard errors, and other statistics for a subsample of forages from the first four paddocks for two sample events are reported in Table 7. The data are mostly right-skewed with a majority of the samples distributed near the mean and a few samples collected with high mineral content values. Most of the site has soil EC_e values in the moderate range, while some areas are much more saline. The distribution of forage-quality values reflects those conditions.

Forage-quality values are similar to others commonly used for average estimates (Table 7). Only limited amounts of data have been reported on trace element uptake by Bermuda grass so comparisons cannot be made with literature values. In certain locations, large amounts of soil Mo and B are found at this site. However, forage Mo levels were not

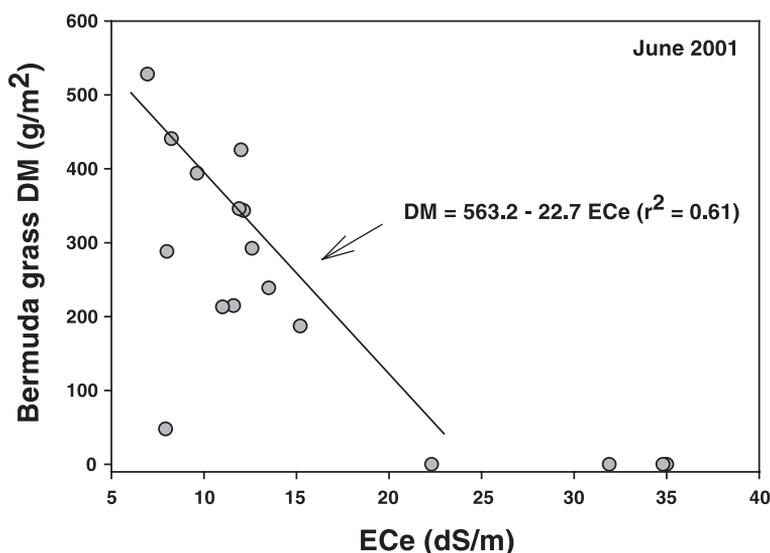


Fig. 9. Forage yield (standing biomass) in June 2001 in paddocks 1 to 4 compared to soil EC_e .

Table 7

Selected forage quality and mineral contents from forage harvests in fall, 2000 (paddocks 1 to 4; $N=60$)

Label	Mean	Minimum	Maximum	Range	S.D.	S.E.	CV	Skewness	Pr < W**
CP (g/kg)*	160.0	93.1	221.0	127.9	29.3	3.8	18.3	0.05	0.73
Ash (g/kg)	131.0	82.7	230.0	147.3	29.8	0.4	22.8	0.97	0.00
ADF (g/kg)	294.0	221.0	364.0	143.0	32.1	4.1	10.9	-0.36	0.25
P (g/kg)	2.2	1.5	3.4	1.9	0.4	0.1	17.6	0.77	0.01
K (g/kg)	19.8	10.7	34.1	23.4	4.6	0.6	23.4	0.86	0.01
S (g/kg)	7.4	3.8	9.3	5.5	1.0	0.1	13.9	-0.49	0.02
Ca (g/kg)	4.9	3.5	7.7	4.2	1.0	0.1	20.9	1.02	0.00
Mg (g/kg)	2.5	1.6	5.6	4.0	0.8	0.1	32.7	2.00	0.00
Na (g/kg)	7.9	3.6	23.9	20.3	3.4	4.4	43.4	2.82	0.00
Cl (g/kg)	8.8	3.6	33.1	29.5	4.6	0.6	52.6	3.12	0.00
B (mg/kg)	133.2	44	257	213	39.4	5.09	29.6	1.24	0.00
Zn (mg/kg)	35.5	17	58	41	9.5	26.8	1.23	0.18	0.71
Mn (mg/kg)	80.8	46	132	86	18.5	2.4	22.9	0.54	0.24
Fe (mg/kg)	667.1	175	4714	4539	817	105.5	122.5	4.03	0.00
Cu (mg/kg)	8.1	4.2	13.7	9.5	1.7	0.2	21.6	1.04	0.01
Mo (mg/kg)	2.4	1.4	5.3	3.9	0.8	0.1	31.7	1.98	0.00
Se ($\mu\text{g/kg}$)	88.5	16	328	312	60	7.7	67.8	2.17	0.00

*Dry matter basis.

**Shapiro–Wilk probability level for test of normality; values <0.05 indicate non-normality.

particularly high, likely because Bermuda grass did not grow in locations with the largest amount of soil Mo. Reports of grass species Mo concentrations often cite 1–4 mg kg⁻¹ as commonly observed values (Vlek and Lindsey, 1977; Albasel and Pratt, 1989; McBride et al., 2000; O'Connor et al., 2001). In particular, Mo may be toxic to cattle if consumed in large amounts, primarily by interfering with Cu metabolism (Suttle, 1991). Some samples are high in Mo, but on average, Mo concentrations reported are not excessive, especially in comparison to concentrations reported as typical for legumes like alfalfa and clover, which are usually two to four times as enriched as grass species growing under comparable experimental conditions (O'Connor et al., 2001). The ratio of forage Cu to forage Mo (3.3:1) in these samples is above the ratio often cited for concern (2:1), but Suttle (1991) has proposed that the critical ratio declines as forage Mo increases. Cu levels of less than 5 mg kg⁻¹ in the presence of soil Mo have been cited (Johansen et al., 1997). Average Cu levels are greater than 8 mg kg⁻¹ in these samples (Table 7). CP, ADF, K, P, Ca, and Mg levels all are close to standard values used in ratio formulation tables, in the absence of specific forage analyses (NRC, 1989). Ash contents are higher on average than those considered typical. At the highest ash levels found in these samples (>20%), some soil contamination likely occurred. Ash content is not harmful to cattle per se, but reduces the nutritional value of forages and may reduce intake by livestock. High ash contents may reflect the saline-sodic character of the site, but there is little information available about these relationships to date. Most of the recent concern for Mo toxicosis is associated with Mo uptake by forages from soils amended with sewage sludge (O'Connor et al., 2001; McBride et al., 2000). There are fewer studies discussing the uptake of Mo by forages from soils naturally abundant in Mo (Vlek and Lindsey, 1977), so the ability to make comparisons is limited. In many instances, concern over the effects of Mo in forages is influenced by the concen-

trations of other elements like S (Suttle, 1991). Few feeding studies and fewer actual grazing studies under such conditions have been reported, so further work on livestock performance and health on these pastures will be of interest.

4. Summary and conclusions

A soil-quality assessment was conducted on a research site located in the WSJV, which is representative of land that has been taken out of agricultural productivity because profitable crop yields were unobtainable due to poor soil quality. The soil-quality information can be used in comparison to other arid locations throughout the world. The site was found to have high levels of soil salinity and sodicity, and sporadically high levels of the trace elements B and Mo. Nevertheless, a halophytic crop (bermuda grass) was easily established across the site except where the soil EC_e was greater than approximately 22 dS m^{-1} . Overall, such sites appear suited for the production of adapted forage crops, if adequate infiltration and leaching can be maintained, and excessive amounts of trace elements do not accumulate in forages.

The concept of soil quality has grown out of concern about the sustainability of agriculture (Warkentin, 1995). Sustainability implies that soil quality must be maintained over time (Aon et al., 2001). The initial assessment of soil quality and future spatio-temporal changes in soil quality are the basis for evaluating the sustainability of drainage water reuse in the WSJV.

At the research site, water flow restricted by the low hydraulic conductivity of the predominantly montmorillonitic clay soils of the WSJV and the influence of dispersion on infiltration due to Na^+ accumulation are the most crucial soil-related factors affecting the sustainability of drainage water reuse. Due to their elevated levels, temporal changes in salinity, SAR, B, and Mo levels are long-term chemical concerns. The sustainability of drainage water reuse depends on maintaining a LF that prevents the accumulation of excessive salinity, B, and Mo to prevent the occurrence of toxic effects upon forage and grazing livestock, and yet low enough to meet the objectives of minimizing drainage volumes and the dissolution of additional salts and minerals.

Although the soil at the research site has high SAR values with low measured K_s , there are mitigating factors that make adequate leaching achievable: (1) the reused drainage water is high in salinity, (2) water flow through surface cracks is a common infiltration pathway of WSJV soils, (3) the presence of an efficient drainage system, and (4) a dense forage cover over most of the site. The high salinity of the drainage water should compensate for increased levels of exchangeable Na^+ that would otherwise reduce hydraulic conductivity and soil aeration to detrimental levels. Although the heavy clay soils of the WSJV have low K_s values, they develop surface cracks when dry, which break the surface soil into columns of prismatic aggregates and provide a conduit for water to get into the soil profile before sealing closed. Historically, this has been the means by which these soils were managed to achieve leaching. The root system of the forage crop should help maintain these conduits for water flow by drying the soil and assisting in aggregate formation. Finally, the drainage system should reduce the likelihood of a rising water table and provide a drainage outlet so leaching is maintained.

Understanding how to manage this saline-sodic soil using drainage water for irrigation exceeds well-established soil and crop management principles. Whether or not this system is sustainable will become apparent as the drainage water reuse experiment proceeds. However, if drainage water reuse is sustainable at the Westlake Farm site, it is likely sustainable throughout the WSJV because this site represents an extreme case.

To conduct the soil-quality assessment, methodology was adapted from the response surface sample design strategy presented by Lesch et al. (1995, 2000) and the EC_a survey guidelines of Corwin and Lesch (2003). The methodology uses GPS-based mobile EC_a equipment to establish a soil sample design scheme to characterize the spatial variability of soil properties associated, either directly or indirectly, with the EC_a measurement. These properties (e.g., salinity, texture, and water content) are generally associated with the soil quality of most arid zone soils.

The methodology has strengths and limitations, which became apparent. Lesch et al. (1995) have previously shown the value of using a response surface sample design based on geo-referenced EC_a measurements to minimize the number of sample sites at field scale necessary to spatially characterize soil salinity. The utility of this approach to assess soil quality of an arid zone soil was demonstrated. The methodology provides a practical means of assessing arid-zone soil quality at field scale when the soil quality properties are correlated with EC_a . At the Westlake Farm site, the correlated properties (i.e., θ_v ; EC_e ; Cl^- , NO_3^- , SO_4^- , Na^+ , K^+ , and Mg^{2+} in the saturation extract; SAR; ESP; B; Se; Mo; $CaCO_3$; and inorganic and organic C) include most of the soil properties associated with soil quality of arid zone soils, particularly in the WSJV. However, a number of soil properties (i.e., ρ_b ; %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. As a result, the spatial distribution of these poorly correlated properties would not be well represented with a response surface sample design. This suggests that a combined response surface and random (or stratified random) sample design using geo-referenced EC_a measurements would improve results. Subsequently, all future soil samples used to evaluate the spatio-temporal effects of drainage water reuse will be taken at the initial 40 selected sites plus additional sites selected with a stratified random sample design.

MacBratney and Webster (1983) suggested that the number of samples required to characterize soil variability can be minimized if it is possible to account for the spatial patterns. Without a priori information about the spatial dependence of each soil property, random or regular interval sampling will likely be necessary to evaluate soil quality (Halvorson et al., 1997). The methodology presented uses rapid, easy, low-cost mobile equipment to obtain EC_a data that provides a priori spatial information about several soil physicochemical properties (e.g., salinity, water content, texture) crucial to the soil quality of arid soils. From this spatial information, a soil sample design is developed that reflects the spatial patterns, but minimizes clustering of samples. The methodology lends itself to assessing spatio-temporal change in soil quality since the rapid measurement of EC_a with mobile equipment provides an intensive spatial database for comparative analysis. Forage sampling and remote sensing data can also be readily linked to soil quality characteristics through this method.

Acknowledgements

The authors wish to acknowledge the University of California Kearney Foundation of Soil Science for the funds that supported the purchase of field instrumentation and funded the chemical analyses needed for the initial soil-quality assessment, and the University of California Salinity-Drainage program for additional financial support for site preparation and for costs associated with sample collection and site management. Ceil Howe, Jr., and Ceil Howe III provided the site and substantial amounts of labor and the use of equipment to help prepare and manage the site. The authors thank Nahid Vishteh and Harry Forster for their analytical technical support. Finally, the authors acknowledge the conscientious work and diligence of Clay Wilkinson, Derrick Lai, Linda Lam, Jon Edwards, Jennifer Palarca, and Samantha Chang, who performed the physical and chemical analysis of the soil samples.

References

- Albasel, N., Pratt, P.F., 1989. Guidelines for molybdenum in irrigation waters. *J. Environ. Qual.* 18, 259–264.
- Aon, M.A., Sarena, D.E., Burgos, J.L., Cortassa, S., 2001. (Micro)biological, chemical and physical properties of soils subjected to conventional or no-till management: an assessment of their quality status. *Soil Tillage Res.* 60, 173–186.
- Ayers, R.S., Westcott, D.W., 1976. *Water Quality for Agriculture*. FAO 29. Food and Agriculture Organization of the United Nations, Rome. 97 pp.
- Barshad, I., 1948. Molybdenum content of pasture plants: I. Nature of soil molybdenum, growth of plants, and soil pH. *Soil Sci.* 71, 297–313.
- Bradford, S., Letey, J., 1993. Cyclic and blending properties for using saline and non-saline waters for irrigation. *Irrig. Sci.* 12, 123–128.
- Carter, L.M., Rhoades, J.D., Chesson, J.H., 1993. Mechanization of soil salinity assessment for mapping. *Proc. 1993 ASAE Winter Meetings, Chicago, IL*, 12–17 Dec. 1993.
- Cervinka, V., 1994. Agroforestry farming systems for the management of selenium and salt on irrigated farmlands. In: Frankenberger Jr., W.T., Benson, S. (Eds.), *Selenium in the Environment*. Marcel Dekker, New York.
- Corwin, D.L., Lesch, S.M., 2002. Application of soil EC to precision agriculture: theory, principles, and guidelines. *Agron. J.* 95 (3), in press.
- Criscimanno, E., Iovino, M., Provenzano, G., 1995. Influence of salinity and sodicity on soil structural and hydraulic characteristics. *Soil Sci. Soc. Am. J.* 59, 1701–1708.
- Cunniff, P. (Ed.), 1995. *Official Methods of Analyses of AOAC International*. vol. I, 16th ed. AOAC International, Gaithersburg, MD. Method 942.05.
- Deverel, S.J., Fujii, R., 1988. Processes affecting the distribution of selenium in shallow ground water of agricultural areas, western San Joaquin Valley, California. *Water Resour. Res.* 24, 516–524.
- Deverel, S.J., Millard, S.P., 1988. Distribution and mobility of selenium and other trace elements in shallow ground water of the western San Joaquin Valley, California. *Environ. Sci. Technol.* 22, 697–702.
- Deverel, S.J., Gilliom, R.J., Fujii, R., Izbicki, J.A., Fields, J.C., 1984. Areal distribution of selenium and other inorganic constituents in shallow ground water of the San Luis Drain Service Area, San Joaquin Valley, California: a preliminary study. *USGS Water Resources Investigations Report* 84-4319.
- Doran, J.W., Parkin, T.B., 1994. Defining and assessing soil quality. In: Doran, J.W., Coleman, D.C., Bezdicek, D.F., Stewart, B.A. (Eds.), *Defining Soil Quality for a Sustainable Environment*. SSSA Spec. Publ. No. 35 SSSA, Madison, WI, pp. 3–21.
- Fujii, R., Deverel, S.J., 1989. Mobility and distribution of selenium and salinity in groundwater and soil of drained agricultural fields, Western San Joaquin Valley of California. In: Jacobs, L.W. (Ed.), *Selenium in Agriculture and the Environment*. SSSA Special Publ. No. 23. SSSA, Madison, WI, pp. 195–212.

- Fujii, R., Swain, W.C., 1995. Areal distribution of selected trace elements, salinity, and major ions in shallow groundwater, Tulare Basin and southern San Joaquin Valley. USGS Water Resources Investigations Report 95-4048, Sacramento, CA.
- Gee, G.W., Bauder, J.W., 1986. Particle size analysis. In: Klute, A. (Ed.), *Methods of Soil Analysis: Part 1. Physical and Mineralogical Methods*, 2nd ed. Monograph No. 9, Agronomy Series. ASA-CSSA-SSSA, Madison, WI, pp. 383–411.
- Gotway, C.A., Ferguson, R.B., Hergert, G.W., Peterson, T.A., 1996. Comparison of kriging and inverse-distance methods for mapping soil parameters. *Soil Sci. Soc. Am. J.* 60, 1237–1247.
- Grattan, S.R., Rhoades, J.D., 1990. Irrigation with saline groundwater and drainage water. In: Tanji, K. (Ed.), *Agricultural Salinity Assessment and Management*. ASCE Manuals and Reports on Engineering Practice #71. ASCE, New York, pp. 433–449.
- Halvorson, J.L., Smith, J.L., Papendick, R.I., 1997. Issues of scale for evaluating soil quality. *J. Soil Water Conserv.* 52, 26–30.
- Johansen, C., Kerridge, P.C., Sultana, A., 1997. Responses of forage legumes and grasses to molybdenum. In: Gupta, U.C. (Ed.), *Molybdenum in Agriculture*. Cambridge Univ. Press, New York, pp. 202–228.
- Johnson, C.M., Ulrich, A., 1959. Analytical methods for use in plant analysis. *Bulletin*, vol. 766. Agricultural Experiment Station, University of California, Berkeley, pp. 26–78.
- Kabata-Pendias, A., Pendias, H., 1992. *Trace Elements in Soils and Plants*, 2nd ed. CRC Press, Boca Raton, FL.
- Kaffka, S.R., Corwin, D.L., Oster, J.D., Hopmans, J., Mori, Y., van Kessel, C., van Groenigen, J.W., 2002. Using forages and livestock to manage drainage water in the San Joaquin Valley: initial site assessment. Kearney Foundation Report. University of California, Berkeley, CA, pp. 88–110.
- Klute, A., Dirksen, C., 1986. Hydraulic conductivity and diffusivity: laboratory methods. In: Klute, A. (Ed.), *Methods of Soil Analysis: Part 1. Physical and Mineralogical Methods*, 2nd ed. Monograph No. 9, Agronomy Series. ASA-CSSA-SSSA, Madison, WI, pp. 687–734.
- Komarek, A.R., Robertson, J.B., Van Soest, P.J., 1993. A comparison of methods for determining ADF using the filter bag technique versus conventional filtration. *J. Dairy Sci.* 77 (Suppl. 1), 114.
- Komarek, A.R., Robertson, J.B., Van Soest, P.J., 1994. Comparison of the filter bag technique to conventional filtration in the Van Soest NDF analysis of 21 feeds. Presented at National Conference on Forage Quality, Evaluation and Utilization Proceedings. University of Nebraska, Lincoln, NE.
- Kravchenko, A., Bullock, D.G., 1999. A comparative study of interpolation methods for mapping soil properties. *Agron. J.* 91, 393–400.
- Laslett, G.M., McBratney, A.B., Pahl, P.J., Hutchinson, M.F., 1987. Comparison of several spatial prediction methods for soil pH. *J. Soil Sci.* 38, 325–341.
- Leenaers, H., Okx, J.P., Burrough, P.A., 1990. Comparison of spatial prediction methods for mapping floodplain soil pollution. *Catena* 17, 535–550.
- Lesch, S.M., Strauss, D.J., Rhoades, J.D., 1995. Spatial prediction of soil salinity using electromagnetic induction techniques: 2. An efficient spatial sampling algorithm suitable for multiple linear regression model identification and estimation. *Water Resour. Res.* 31, 387–398.
- Lesch, S.M., Rhoades, J.D., Corwin, D.L., 2000. ESAP-95 Version 2.0 User Manual and Tutorial Guide. Research Report No. 146. George E. Brown, Jr. Salinity Laboratory, Riverside, CA.
- Lima, L.A., Grismer, M.E., Nielsen, D.R., 1990. Salinity effects on Yolo loam hydraulic properties. *Soil Sci.* 150, 451–458.
- Lindsay, W.L., 1979. *Chemical Equilibria in Soils*. Wiley, New York.
- Maas, E.V., Hoffman, G.J., 1977. Crop salt tolerance—current assessment. *J. Irrig. Drain. Div.*, ASCE 103 (IR2), 115–134.
- McBratney, A.B., Webster, R., 1983. How many observations are needed for regional estimation of soil properties? *Soil Sci.* 135, 177–183.
- McBride, M.B., Richards, B.K., Steenhuis, T., Spiers, G., 2000. Molybdenum uptake by forage crops grown on sewage sludge-amended soils in the field and greenhouse. *J. Environ. Qual.* 29, 848–854.
- McNeill, J.D., 1980. Electromagnetic terrain conductivity measurement at low induction numbers. Tech. Rpt. TN-6. Geonics, Mississauga, Ontario, Canada.
- National Academy of Sciences, 1973. *Water quality criteria 1972*. Committee on Water Quality Criteria, EPA R3-73-033 (1973). U.S. Government Printing Office, Washington, DC.

- NRC, 1989. Nutrient Requirements of Dairy Cattle, 6th ed. National Academy Press, Washington, DC. 157 pp.
- O'Connor, G.A., Brobst, R.B., Chaney, R.L., Kincaid, R.L., McDowell, L.E., Pierzynski, G., Rubin, A., van Riper, G.G., 2001. A modified risk assessment to establish molybdenum standards for land application of biosolids. *J. Environ. Qual.* 30, 1490–1507.
- Oster, J.D., 1997. Impact of alternative crops on salt disposal strategies. Proc. Calif. Plant and Soil Conference, Visalia, California.
- Oster, J.D., Jayawardane, N.S., 1998. Agricultural management of sodic soils. In: Sumner, M.E., Naidu, R. (Eds.), *Sodic Soils: Distribution, Processes, Management and Environmental Consequences*. Oxford Univ. Press, New York. Chapter 8.
- Posnikoff, J.F., Knapp, K.C., 1996. Regional drain water management: Source control, agro-forestry, and evaporation ponds. *J. Agric. Resour. Econ.* 21, 277–293.
- Rhoades, J.D., 1982. Soluble salts. In: Page, A.L. (Ed.), *Methods of Soil Analysis: Part 2. Chemical and Microbiological Properties*, 2nd ed. Monograph, vol. 9. ASA, Madison, WI, pp. 167–179.
- Rhoades, J.D., 1989. Intercepting, isolating, and reusing drainage waters for irrigation to conserve water and protect water quality. *Agric. Water Manag.* 16, 37–53.
- Rhoades, J.D., 1992. Instrumental field methods of salinity appraisal. In: Topp, G.C., Reynolds, W.D., Green, R.E. (Eds.), *Advances in Measurement of Soil Physical Properties: Bring Theory into Practice*. SSSA Special Publ. No. 30. ASA-CSSA-SSSA, Madison, WI, pp. 231–248.
- Rhoades, J.D., 1993. Electrical conductivity methods for measuring and mapping soil salinity. In: Sparks, D.L. (Ed.), *Advances in Agronomy*, vol. 49. Academic Press, San Diego, CA, pp. 201–251.
- Rhoades, J.D., Corwin, D.L., Lesch, S.M., 1999. Geospatial measurements of soil electrical conductivity to assess soil salinity and diffuse salt loading from irrigation. In: Corwin, D.L., Loague, K., Ellsworth, T.R. (Eds.), *Assessment of Non-Point Source Pollution in the Vadose Zone*. AGU Geophysical Monograph, vol. 108. American Geophysical Union, Washington, DC, pp. 197–215.
- Sah, R.N., Miller, R.O., 1992. Spontaneous reaction for acid dissolution of biological tissues in closed vessels. *Anal. Chem.* 64, 230–233.
- Shainberg, I., Letey, J., 1984. Response of soils to sodic and saline conditions. *Hilgardia* 52, 1–57.
- Suttle, N.F., 1991. The interactions between copper, molybdenum, and sulphur in ruminant nutrition. *Annu. Rev. Nutr.* 4, 121–140.
- Sweeney, R.A., 1989. Generic combustion method for determination of crude protein in feeds: collaborative study. *J. Assoc. Off. Anal. Chem.* 72, 770–774.
- Tracy, M.L., Moller, G., 1990. Continuous flow vapor generation for inductively coupled argon plasma spectrometric analysis: Part 1. Selenium. *J. Assoc. Off. Anal. Chem.* 73, 404–410.
- Trangmar, B.B., Yost, R.S., Uehara, G., 1985. Application of geostatistics to spatial studies of soil properties. *Adv. Agron.* 38, 45–93.
- USDA, 1986. Soil Survey of Kings County, CA. Sheet No. 7, p. 43.
- U.S. Salinity Laboratory Staff, 1954. U.S.D.A. Handbook 60—Diagnosis and Improvement of Saline and Alkali Soils. U.S. Government Printing Office, Washington, DC.
- Vlek, P.L.G., Lindsey, W.E., 1977. Molybdenum contamination in Colorado soils. In: Chappel, W.R., Peterson, K.K. (Eds.), *Molybdenum in the Environment*, vol. 2. Marcel Dekker, New York, pp. 619–650.
- Warkentin, B.P., 1995. The changing concept of soil quality. *J. Soil Water Conserv.* 50, 226–228.
- Warrick, A.W., Zhang, R., El-Harris, M.K., Myers, D.E., 1988. Direct comparisons between kriging and other interpolators. In: Wierenga, P.J., Bachelet, D. (Eds.), *Validation of Flow and Transport Models for the Unsaturated Zone*. Proceedings. New Mexico State University, Las Cruces, NM.
- Weber, D.D., Englund, E.J., 1992. Evaluation and comparison of spatial interpolators. *Math. Geol.* 24, 381–391.
- Wienhold, B.J., Hendrickson, J.R., Karn, J.F., 2001. Pasture management influences on soil properties in the Northern Great Plains. *J. Soil Water Conserv.* 56, 27–31.
- Wollenhaupt, N.C., Wolkowski, R.P., Clayton, M.K., 1994. Mapping soil test phosphorus and potassium for variable-rate fertilizer application. *J. Prod. Agric.* 7, 441–448.
- Youngs, E.G., 2001. Hydraulic conductivity of saturated soils. In: Smith, K.A., Mullins, C.E. (Eds.), *Soil and Environmental Analysis, Physical Methods*, 2nd ed. Marcel Dekker, New York, pp. 141–182.