# Changes in soil hydraulic conductivity, runoff, and soil loss due to irrigation with different types of saline–sodic water

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**6 authors**, including:

Uttam Mandal
Central Soil Salinity Research Institute


A. K. Bhardwaj
Central Soil Salinity Research Institute


David Warrington
Northwest A & F University


Asher D Bar-Tal
Agricultural Research Organization ARO

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# Changes in soil hydraulic conductivity, runoff, and soil loss due to irrigation with different types of saline–sodic water

Uttam Kumar Mandal [a,*], A.K. Bhardwaj [b], D.N. Warrington [c], Dina Goldstein [c], A. Bar Tal [c], G.J. Levy [c]

[a] *Central Research Institute for Dryland Agriculture, Santoshnagar, P.O.-Saidabad, Hyderabad-500059, India*
[b] *Department of Soil Science, NCSU, 100 Derieux Street, box 7619, Raleigh, NC-27695, USA*
[c] *Institute of Soil, Water and Environmental Sciences, Agricultural Research Organization, The Volcani Center, Bet Dagan 50250, Israel*

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**Abstract**

Irrigation with saline–sodic water causes sodic conditions in the soil which reduces soil productivity. We evaluated the changes in a number of important indices related to soil structural stability when treated wastewater (TWW), albeit with higher loads of organic matter and suspended solids, was used instead of more saline–sodic irrigation water, under different degrees of aggregate slaking. We studied soil saturated hydraulic conductivity (HC) using disturbed samples packed in columns, and soil infiltration rate, runoff and erosion under simulated rainfall. Aggregate slaking was manipulated by wetting the samples prior to all tests at either a slow (1–2 mm $h^{-1}$) or a fast (50 mm $h^{-1}$) rate. Samples of a calcareous silty clay (Typic Calciorthids) from the Bet She'an Valley, Israel, were taken from plots irrigated for three years with either TWW, saline–sodic Jordan River water (JRW), or moderately saline–sodic spring water (SPW), and also from a non-cultivated area (control). With little or no aggregate slaking (use of slow wetting), higher HC values and lower amounts of total runoff and soil loss were measured compared to when more severe aggregate slaking was induced (use of fast wetting). The HC values for the TWW treatment were similar to, or lower than, those for the control and significantly higher than those for the JRW treatment. For the runoff and soil loss data, differences among the water quality treatments were, generally, more pronounced when aggregate slaking was substantially reduced, and were related to soil sodicity. Under the latter condition, runoff and soil loss from the TWW treatment were comparable with those from the control and significantly lower than those from the JRW treatment. Our results suggested that replacing saline–sodic irrigation water with TWW could have favorable effects on soil structural stability, especially under conditions where aggregate slaking can be reduced (e.g., in regions with low to moderate rain intensities; and/or use of low intensity irrigation systems).


*Keywords:* Treated wastewater; Sodium adsorption ratio (SAR); Electrical conductivity (EC); Exchangeable sodium percentage (ESP); Wetting rate; Aggregate slaking

## 1. Introduction

Land application of treated wastewater (TWW) in cultivated fields may serve as a viable means of effluent disposal and of sustaining agricultural production, especially in regions experiencing shortages of fresh water. However, irrigation with TWW can have risks for both crop production and the soil environment. Among the potential risks associated with irrigation using TWW, is degradation of the soil structure. This can manifest itself in deterioration of aggregate stability resulting in decreased soil hydraulic conductivity; increased susceptibility to surface sealing, runoff and soil erosion problems; soil compaction; and decreased soil aeration.

When compared to more temperate or wet tropical regions, fresh water in Israel has a higher total salt concentration (8–9 $mmol_c\ l^{-1}$) and sodium adsorption ratio (SAR) (0.5–2 $mmol^{1/2}\ l^{-1/2}$). TWW mainly differs from this fresh water source in having a higher total salt concentration (17–20 $mmol_c\ l^{-1}$) and SAR (~6 $mmol^{1/2}\ l^{-1/2}$), in addition to dissolved organic matter and suspended solids (Feigin et al., 1991). Irrigation with water of moderate SAR (~6 $mmol^{1/2}\ l^{-1/2}$) leads to an exchangeable

* Corresponding author. Tel.: +91 40 24530161; fax: +91 40 24531802.
*E-mail address:* uttamkm@crida.ernet.in (U.K. Mandal).

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sodium percentage (ESP) of a comparable value in the soil (US Salinity Laboratory Staff, 1954) and adversely affects soil physical properties, e.g., soil hydraulic conductivity (HC), due to $Na^+$-induced clay dispersion (Halliwell et al., 2001). Numerous studies have also shown that the presence of dissolved organic matter in TWW, coupled with its higher sodicity, increases clay dispersion and results in higher flocculation values for both specimen and soil clays (Durging and Chaney, 1984; Frenkel et al., 1992; Tarchitzky et al., 1993, 1999).

The effects of irrigation with TWW on soil structural stability and its hydraulic properties have commonly been compared to those of irrigation with fresh water. Such studies have shown that irrigation with TWW, containing a high load of organic matter, decreased soil HC ascribed to pore blockage by the suspended solids present in the TWW (Rice 1974; Vinten et al., 1983; Magesan et al., 2000) and by the excessive growth of microorganisms due to the presence of nutrients in the TWW (Magesan et al., 1999). In studies that used TWW which had a higher degree of treatment and was thus of better quality, results were inconsistent. For example, Levy et al. (1999) showed that TWW had a comparable effect to fresh water on the HC whereas Tarchitzky et al. (1999) reported a reduction in the HC following leaching with TWW.

In other experiments (Shainberg et al., 2001; Levy et al., 2005; Bhardwaj et al., 2007), changes in soil HC during leaching with deionized water were compared for soils subjected to long term irrigation either with TWW or with fresh water. Similar studies compared the changes in infiltration rate, runoff and erosion during natural or simulated rainfall on such soils (Mamedov et al., 2001; Agassi et al., 2003; Lado et al., 2005). Results were found to vary due to differences in TWW quality, soil texture, $CaCO_3$ content, intensity of cultivation, irrigation method, and other conditions, e.g., antecedent moisture content, existing in the soil.

A notable exception to these studies was that of Bhardwaj et al. (2008) who tested the hypothesis that replacing saline–sodic irrigation water, that had been in use for many years, with the considerably less saline–sodic TWW, albeit with higher loads of organic matter and suspended solids, may help the soil regain its structure and hydraulic conductivity. Bhardwaj et al. (2008) examined the HC of undisturbed soil cores and the aggregate stability of samples taken from soils irrigated with different water qualities. He found significantly higher HC and aggregate stability in the TWW-irrigated samples than in those that were subjected to long term irrigation with saline–sodic water.

In most of the earlier studies, effects of sodicity on soil HC, permeability, and seal formation were determined for dry soils that were subjected to rapid wetting, either from below or from above, prior to their exposure to leaching or simulated rain. Fast wetting leads to aggregate slaking (Panabokke and Quirk, 1957; Loch, 1994). Recent studies showed that substantially reducing aggregate slaking, i.e., by using slow wetting rates (commonly ~2 mm $h^{-1}$), lessened the susceptibility of the soil to seal formation and to lower infiltration rates (Levy et al., 1997), and maintained higher HC values (Moutier et al., 2000) in comparison to cases where severe aggregate slaking occurred when using much faster wetting (e.g., ~50 mm $h^{-1}$). It has also been demonstrated that the importance of aggregate slaking, in determining soil susceptibility to permeability deterioration, depends on both soil sodicity and clay content (Shainberg et al., 2001; Mamedov et al., 2001).

Considering the importance of aggregate slaking immediately before determining soil HC and seal formation, our objectives were to compare the impact of irrigation with either TWW or saline–sodic water on a number of soil structural stability indices, e.g., HC, infiltration rate, and soil loss, under conditions of slaked and non-slaked aggregates. Aggregate slaking was manipulated by wetting soil samples at different rates immediately prior to different tests.

## 2. Materials and methods

This study was part of a long term water management and water quality experiment established in 2002 in the Bet She'an Valley, Israel. The soil of the experimental site is a calcareous silty clay (Typic Calciorthids), alkaline in nature with an average pH of 8.03 in the upper 40 cm soil layer. The dominant clay minerals in the soil (Neaman et al., 1999) were smectite (31%), calcite (27%) and palygorskite (22%). The soil particle size distribution was 172, 400 and 428 g $kg^{-1}$ of sand, silt and clay, respectively. Mean $CaCO_3$ and organic matter content, and cation exchange capacity were 546±14 g $kg^{-1}$, 17.3±0.8 g $kg^{-1}$, and 23.2±0.6 $cmol_c$ $kg^{-1}$, respectively. Three irrigation water types were used in the study: (1) saline–sodic water from the Jordan River, south of Lake Kinneret (JRW); (2) less saline–sodic, local spring water (SPW); and (3) treated wastewater (TWW) from a secondary treatment plant situated in the eastern Bet She'an Valley. Following a basic secondary treatment (oxidation ponds), the TWW was subjected to chlorination before filtration through a 125 μm pore sized screen prior its use for irrigation at the experimental site. Relevant chemical characteristics of the three irrigation waters are presented in Table 1.

Two crops were grown during the study period: corn (*Zea mays*) in the summer and winter of 2003–4 and cotton (*Gossypium*

Table 1
Important chemical characteristics of the three water types used during the last two years of irrigation cycles (mean±one standard deviation)

| Parameter | Irrigation water type[a] | | |
|---|---|---|---|
| | TWW | SPW | JRW |
| pH | 8.20±0.64 | 7.35±0.78 | 7.80±1.27 |
| EC (dS $m^{-1}$) | 1.53±0.12 | 3.55±0.49 | 5.00±0.85 |
| Cl ($mmol_c$ $l^{-1}$) | 4.30±0.61 | 27.10±6.65 | 40.75±10.96 |
| Na ($mmol_c$ $l^{-1}$) | 4.40±0.74 | 19.75±3.75 | 32.50±4.10 |
| Ca+Mg ($mmol_c$ $l^{-1}$) | 10.70±0.98 | 21.75±7.57 | 23.35±6.29 |
| SAR (mmol $l^{-1})^{0.5}$ | 1.91±0.2 | 6.05±0.07 | 9.55±0.07 |
| K ($mmol_c$ $l^{-1}$) | 3.10±0.4 | 0.15±0.07 | 0.46±0.47 |
| $HCO_3$ ($mmol_c$ $l^{-1}$) | 13.40±1.2 | 3.85±1.63 | 5.00±0.57 |
| $HSO_4$ ($mmol_c$ $l^{-1}$) | 1.20±0.16 | 1.90±0.28 | 5.85±0.64 |
| TSS[b] (mg $l^{-1}$) | 47.0±6.0 | – | – |
| BOD[c] (mg $l^{-1}$) | 78.0±11.0 | – | – |
| COD[d] (mg $l^{-1}$) | 368±56 | – | – |

[a] TWW = treated wastewater; SPW = spring water; JRW = Jordan River water.
[b] TSS = total suspended solids.
[c] BOD = biological oxygen demand.
[d] COD = chemical oxygen demand.

*hirsutum*) during the summers of 2004 and 2005. For the summer corn (2003), 428 mm of water was applied as drip irrigation with a total fertilizer dose of 320 kg N $ha^{-1}$ and 140 kg $P_2O_5$ $ha^{-1}$ applied as fertigation. For the winter corn of 2003–4, 339 mm of water was applied as irrigation and the total fertilizer dose was 250 kg N $ha^{-1}$ and 100 kg $P_2O_5$ $ha^{-1}$. During the cotton growing periods, the annual winter rainfall was 175 and 250 mm for 2004 and 2005, respectively. In both years, 800 mm of water was used for irrigation and 210 kg N $ha^{-1}$ was applied as fertilizer. No phosphate fertilizer was applied for the cotton crop. Fertilizer doses were applied evenly across all irrigated plots.

The experimental plan comprised of a complete randomized block design with 4 replicates. The plots were irrigated using a drip irrigation system. Disturbed soil samples for various laboratory studies were taken from each plot. In addition, soil samples were taken from a strip of land that had never been cultivated, which separated the experimental field from an adjacent field. These samples were referred to as the control treatment, as this uncultivated soil was assumed to represent the natural condition of the soil with respect to its structural stability. Apart from a higher organic matter content (22 g $kg^{-1}$), all other soil properties mentioned above were similar for both the control and the soil from the experimental site. All of the soil sampling was carried out at the end of the third irrigation season in Nov., 2005.

### 2.1. Hydraulic conductivity

As an indicator of soil structure, changes in saturated HC were measured using disturbed samples and deionized water. Soil columns were prepared by uniformly packing 120 g of air-dried, crushed and sieved soil (<2.0 mm) into Perspex cylinders (5.4 cm internal diameter and 10.4 cm length) to a dry bulk density of $1.165 \pm 0.015$ Mg $m^{-3}$. The bottom of each cylinder contained a 5.0 cm layer of acid washed sand over a fine metal sieve to facilitate drainage. A filter paper covered the soil surface to minimize disturbance during wetting and leaching.

The soil columns were wetted from below with deionized water; 0.004 dS $m^{-1}$ electrical conductivity (EC). The degree of slaking of the aggregates was manipulated by wetting the soil in the columns at a rate of 2 mm $h^{-1}$ (almost no slaking, designated as slow wetting) or 50 mm $h^{-1}$ (severe slaking, designated as fast wetting) using a peristaltic pump. When the water level reached the top of the soil columns, the flow direction was reversed and the columns were leached, from the top, with deionized water from a Mariotte bottle with a constant head of 10 cm. The leachate was collected continuously over fixed time intervals using a fraction collector, and the HC was then calculated. Leachate was also periodically analyzed for pH and EC, and the dispersed clay content was determined by absorbance at 420 mm with a Uvikon 933 spectrophotometer (Kontron Instruments, Milan, Italy).

### 2.2. Infiltration and erosion

Infiltration rate and soil erosion were investigated using a rotary disc rainfall simulator (Morin et al., 1967). The mechanical parameters of the simulated rainfall were: raindrop mean diameter, 1.9 mm; median drop velocity, 6.2 m $s^{-1}$; and kinetic energy, 18.1 J $mm^{-1}$ $m^{-2}$. The duration of each simulated rainstorm was 120 min, at a rainfall intensity of 37 mm $h^{-1}$ using deionized water.

Air-dried soil samples, crushed and sieved to <4 mm, were packed in $50 \times 30 \times 2$ cm perforated trays that were placed on an 8.0 cm layer of coarse sand in boxes positioned, at a slope of 9%, on a rotating framework under the rotary disc rainfall simulator. Prior to applying the rain, the soil in the trays was saturated by a mist type rain using tap water (EC = 0.9 dS $m^{-1}$; SAR = 2.5 $mmol^{1/2}$ $l^{-1/2}$) to minimize clay swelling and dispersion during the wetting process. The degree of aggregate slaking was manipulated by using two wetting rates: slow (1 mm $h^{-1}$) and fast wetting (50 mm $h^{-1}$). Slow wetting was achieved using mist at 50 mm $h^{-1}$ applied in pulses of 6 s duration every 3 min for 17 h; fast wetting was carried out by a continuous application of mist at 50 mm $h^{-1}$ for 27 min.

Water percolating through the soil, during each 70 mm rainstorm, was collected from a drainage tube set in the base of each box during alternate rotations of the framework (1 rotation took 79 s), the volume was measured, and the infiltration rate calculated. Runoff for the entire rainstorm was collected continuously and the total volume of runoff was measured. Soil loss was determined by oven drying (105 °C) and weighing the sediments contained in the runoff water. Soil loss due to splash from the trays was not measured. Young and Wieresma (1973) reported that soil carried by splash is positively correlated with soil removed by runoff water. Therefore, the amount of soil removed in runoff water serves as an indicator of soil detachment. Eroded sediments were considered to result from interrill erosion since the length of tray was only 50 cm (Meyer and Harmon, 1984).

### 2.3. Statistical analysis

Each of the three water quality treatments (TWW, SPW, and JRW) and the uncultivated control were studied using both fast and slow wetting rates; altogether 8 treatments were studied, each in four replicates. Mean values of the initial HC (defined below), total volume of runoff, and total soil loss data were subjected to the Tukey–Kramer HSD test using a significance level of 0.05 (SAS Institute, 1995). In cases where ratios were considered, the standard deviations of the ratios were used for direct comparison, rather than the Tukey–Kramer HSD test, because no normal distribution of these variables could be assumed.

## 3. Results and discussion

### 3.1. Hydraulic conductivity

The measured HC curves for the fast and slow wetted samples show (Fig. 1) that leaching the soils with deionized water resulted in a continuous decrease in the HC, albeit at different rates, irrespective of the rate of wetting and the different treatments. Two parameters were used to evaluate the effects of the treatments studied on soil HC: (i) initial HC (HCi), i.e., the HC measured at the beginning of the leaching from the first leachate sample equivalent to ~0.1 pore volumes; and (ii) the relative apparent

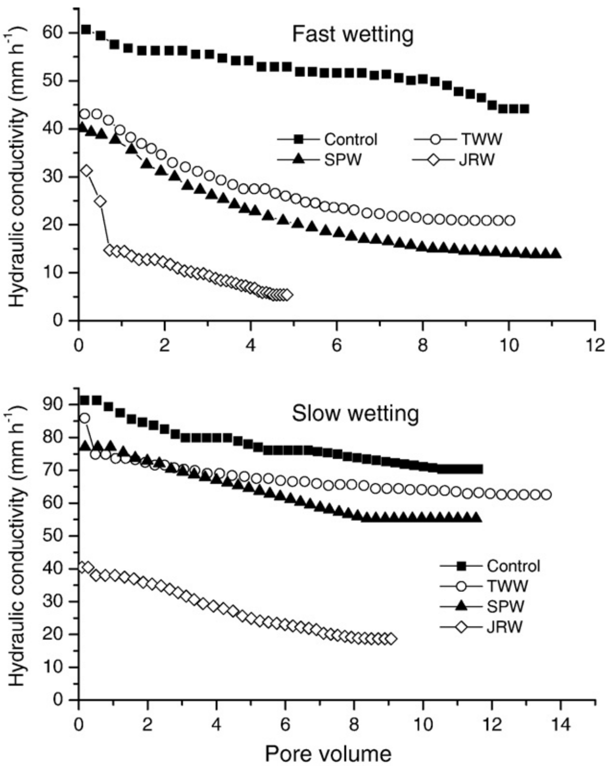


Fig. 1. Saturated hydraulic conductivity as a function of leachate pore volume for the different water quality treatments (control; treated wastewater (TWW); spring water (SPW); Jordan River water (JRW)) and the two rates of wetting.

steady state HC (RHCss) calculated as the ratio of the apparent steady state HC (i.e., the HC that was approaching asymptotically a steady state value) to the HCi.

### 3.1.1. Initial HC (HCi)

Within a given wetting treatment, HCi values for the control, TWW and SPW were comparable (Fig. 1 and Table 2). The HCi for the JRW was significantly lower ($P<0.05$) than the other water quality treatments when slow wetting was used. However, it was similar to the SPW and significantly lower than both the TWW and the control when fast wetting was used (Table 2). The HCi values for the water quality treatments were ranked in the order of control > TWW > SPW > JRW. This ranking order was the reverse of the given water quality treatments for both the EC and SAR of their soil saturated paste extracts (Table 2). The HCi was assumed to depend on clay swelling and aggregate slaking. Clay susceptibility to swelling increases with an increase in the SAR and a decrease in the EC of the soil solution (Quirk and Schofield, 1955). By contrast, aggregate slaking depends on both entrapped air and differential swelling. Separation between the effects of the two processes on aggregate slaking is, however, very difficult (Ruiz-Vera and Wu, 2006). The observed relations of the HCi with the EC and SAR suggested that the increase in SAR, i.e., enhancement of conditions favoring clay swelling that accompanied the decrease in HCi (Table 2), had a greater impact on the HCi than the expected mitigating effects of the increased EC on suppressing clay swelling.

Fast wetting resulted, in general, in significantly lower HCi values than with slow wetting (Table 2). Bulk densities, determined for the soil columns after wetting, were lower (1.09±0.014 Mg $m^{-3}$) than those for the dry soil columns (1.165±0.015 Mg $m^{-3}$). This decrease in the bulk density indicated that the soils in the columns swell during wetting. However, the bulk densities were comparable for the 4 water quality treatments and also for the two wetting rates, which means a similar magnitude of swelling in all the treatments. It was, therefore, concluded that the lower HCi of the fast- compared with the slow-wetted columns resulted from the adverse effects of more severe aggregate slaking, during the fast wetting, causing a shift in pore size distribution towards relatively finer water conducting pores. In the JRW treatment, the HCi for fast and slow wetting were not significantly different although the HCi for fast wetting was 13.23 mm $h^{-1}$ lower than the slow wetting. Apparently, because of the high SAR of the JRW treatment, slaking of aggregates was substantial even during the slow wetting procedure. Consequently, the HCi in the slow wetted JRW was comparable to that obtained by fast wetting and significantly lower than the HCi values for the other treatments subjected to slow wetting (Table 2).

### 3.1.2. Relative steady state HC (RHCss)

As shown in Fig. 1, a decrease in the HC occurred during the leaching with deionized water and, therefore, all the RHCss values for all the treatments were < 100% (Table 2). The magnitude of the RHCss values depended on both the rate of wetting and the water quality treatment (Table 2). The decrease in the HC was accompanied by the appearance of dispersed clay in the leachate

Table 2
Electrical conductivity (EC), sodium adsorption ratio (SAR) and pH of the saturated paste, and initial and relative steady state hydraulic conductivity (HCi and RHCss, respectively)

| Irrigation treatment[†] | EC | SAR | pH | HCi | | RHCss | |
|---|---|---|---|---|---|---|---|
| | (dS $m^{-1}$) | ($mmol_c$ $l^{-1}$)$^{0.5}$ | | (mm $h^{-1}$) | | (%) | |
| | | | | Fast wetting | Slow wetting | Fast wetting | Slow wetting |
| Control | 1.3±0.15[‡] | 1.0±0.15 | 7.66±0.10 | 60.7 a[§] B[¶] | 91.3 a A | 72.1±11.5 | 79.9±19.6 |
| TWW | 5.3±0.64 | 6.2±0.92 | 7.55±0.08 | 43.1 a B | 87.6 a A | 48.5±9.4 | 71.5±12.9 |
| SPW | 8.5±0.92 | 8.9±1.1 | 7.57±0.09 | 40.1 ab B | 77.1 a A | 34.4±6.9 | 71.7±15.4 |
| JRW | 15.9±1.1 | 14.2±1.9 | 7.32±0.11 | 31.3 b A | 41.3 b A | 17.3±4.6 | 17.7±8.1 |

[†]Control = uncultivated field; TWW = treated wastewater; SPW = spring water; JRW = Jordan River water.
[‡]Mean±one standard deviation.
[§]Within a column, numbers followed by the same lowercase letter are not significantly different at $P<0.05$ level.
[¶]Within an irrigation treatment, numbers followed by the same uppercase letter are not significantly different at $P<0.05$ level.

collected from the columns. However, concentrations of the dispersed clay were small, generally peaking at <0.5 g $l^{-1}$. By comparison, studies where a discharge of dispersed clay was observed and associated with a decrease in HC, the reported peak clay concentrations were of 1–2 orders of magnitude higher than those which we measured (Frenkel and Rhoades, 1978; Bagarello et al., 2006). We considered, therefore, that the small concentrations of the discharged clay have only a minor, if any, effect on the measured HC. It was further postulated that the observed dispersed clay probably originated from aggregate slaking during the wetting process and was not due to clay dispersion during leaching with deionized water, because the EC of the leachate at the peak clay discharge (after the second pore volume) was >1 dS $m^{-1}$ (i.e., >10 $mmol_c$ $l^{-1}$), which is well above the flocculation value (7 $mmol_c$ $l^{-1}$) of pure smectite having SAR 20 $mmol^{1/2}$ $l^{-1/2}$ (Oster et al., 1980).

For both fast and slow wetting, the highest RHCss, indicating the smallest relative change in HC during leaching with deionized water, was observed for the control (Table 2). The sodicity level in the control (uncultivated soil) was low (SAR ~1 $mmol^{1/2}$ $l^{-1/2}$), hence the small change in the HC during leaching could be explained by some constriction of water conducting pores due to minor swelling of the silty clay soil in the column as the EC of the leaching water was reduced, and possibly, in the case of fast wetting, to some limited clogging of the water conducting pores by fine particles released from aggregates' slaking during the wetting process.

The RHCss of the three water types was in the order of TWW>SPW>JRW with fast wetting (Table 2). This ranking was found to be associated with an opposite trend for the soil SAR (Table 2); i.e., the higher the SAR, even though coupled with a higher EC, the lower the RHCss. This relationship was also noted for the HCi data and suggested that, with an increase in soil SAR, the soil was more susceptible to (i) slaking during wetting, as demonstrated especially by its lower HCi, and (ii) a greater degree of clay swelling during leaching with deionized water. Both mechanisms led to lower RHCss values (Table 2).

With slow wetting, TWW and SPW had comparable RHCss values which were higher than that of the JRW (Table 2). It is postulated that slow wetting at the rate of 2 mm $h^{-1}$ caused only a limited slaking of aggregates. Thus, the reduction in the HC during deionized water leaching was mainly due to clay swelling. This assumption is supported by the higher HCi and RHCss values for slow wetting compared with fast wetting (Table 2). The similarity in the RHCss values for the TWW and SPW in the slow wetting case could, therefore, be explained by the relatively small difference between their soil SAR levels, which were nearly half of that for the JRW.

### *3.2. Simulated rainfall studies*

#### *3.2.1. Infiltration and runoff*

The infiltration curves for all treatments (Fig. 2) show that the infiltration rate decreased with increasing rain depth, due primarily to seal formation. Differences, if any, were small among the infiltration curves for the fast wetting treatment. However, the slow wetting treatment produced some notable differences due to the different water quality treatments (Fig. 2). It was also observed that the rate of wetting had an effect on the depth of rain to ponding, i.e., the depth of rain needed to initiate runoff (Fig. 2). In the fast wetting treatment, infiltration started to decrease and runoff was initiated after ~4 mm of rain whereas, in slow wetting, runoff initiation occurred after at least 12 mm of rain. This difference in the depth of rain to ponding suggested that aggregate slaking, which occurred during the fast wetting, enhanced seal development by increasing the quantity of finer soil material which could obstruct the water conducting pores of the surface soil layer. Furthermore, smaller aggregates, with their higher specific surface areas, are more susceptible to the physical and chemical processes responsible for dispersion and seal formation (Agassi et al., 1981). Thus, after fast wetting, a seal can form more rapidly than after slow wetting. In the latter treatment, where limited slaking, if any, took place during the wetting of the aggregates, the energy of the rain itself (expressed by rain depth) was needed to break down the surface aggregates to the extent achieved by aggregate slaking in the fast wetting treatment. Thus, with slow wetting, a greater rain depth was needed to develop the seal to the point where runoff was initiated.

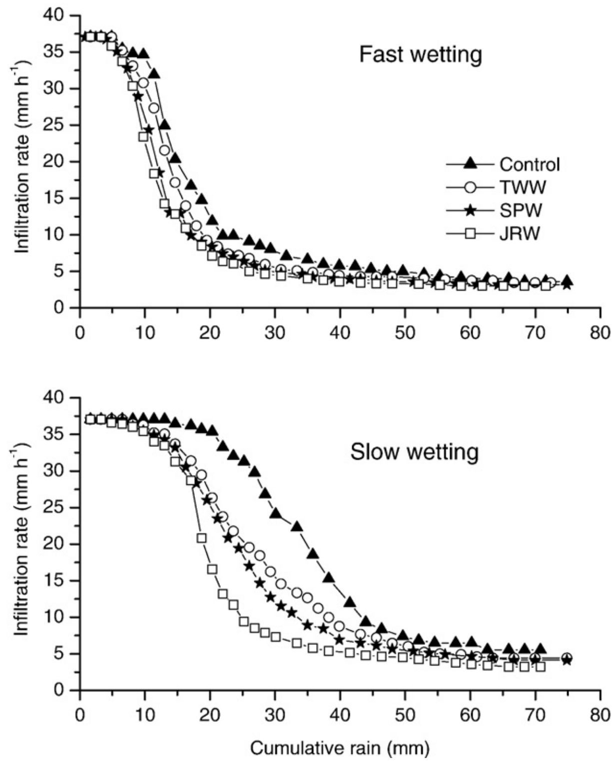


Fig. 2. Infiltration rate as a function of cumulative rainfall for the different water quality treatments (control; treated wastewater (TWW); spring water (SPW); Jordan River water (JRW)) and the two rates of wetting.

Infiltration curves are not suitable for quantitative comparison among treatments, especially due to the similarity in the apparent steady state infiltration rates at the end of the 70 mm storms for all treatments (Fig. 2). Therefore, we used the total volume of runoff generated by each storm (expressed as runoff depth in mm), which is an integrated value that reflects changes in the infiltration rate during the entire storm, as the indicator for testing the effects of our treatments on seal formation.

Total runoff data are presented in Fig. 3. For the fast wetting treatment, as expected on the basis of the similarity in the infiltration curves (Fig. 2), comparable runoff amounts were observed for all the water quality treatments. This result was in agreement with the findings of Mamedov et al. (2001), who reported similar runoff amounts for clay soils having ESP levels in the range of 1.6 to 17 which had been subjected to fast wetting. Evidently, when the seal was formed, due to the combined effects of aggregate slaking induced by fast wetting and the mechanical impact of raindrops, the degree of clay dispersivity induced by different soil sodicities did not appear to enhance the formation of the seal, or the subsequent production of runoff.

In the slow wetting treatment, the difference in total runoff depth between the JRW (39.1 mm) and the control (25.4 mm) was significant ($P<0.05$); all other comparisons among the water quality treatments indicated similar runoff depths (Fig. 3). However, a trend was noted in the slow wetting treatment whereby runoff depth increased in the order of control<TWW<SPW<JRW. This order demonstrated a close relationship with the soil SAR (Table 2). A similar trend, relating increase in runoff volume with increase in soil sodicity, was also noted by Mamedov et al. (2001). These observations suggested that, in the absence of aggregate slaking, an increase in soil sodicity probably contributed to the weakening of the aggregates making them more susceptible to (i) breakdown by the impact of the raindrops, and (ii) clay dispersion; both of which led to faster development of the seal (Fig. 2) and to greater runoff depths (Fig. 3).

It was also noted that runoff depths in the treatments subjected to slow wetting were lower than those with fast wetting. Nevertheless, differences were significant only in the control and the TWW treatments ( Fig. 3). Therefore, in the case of TWW irrigation alone, it may prove beneficial to use management practices to significantly reduce water losses due to runoff, e.g., by controlling the initial irrigation rate to induce slow wetting of dry soil prior to predicted rainstorm events.

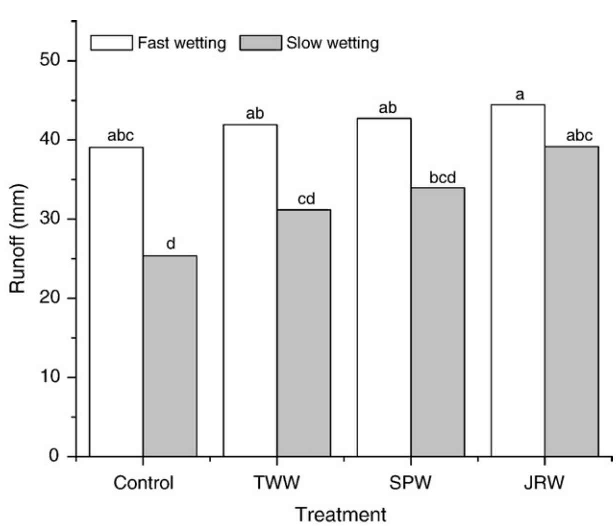


Fig. 3. Total runoff volume for the different water quality treatments (control; treated wastewater (TWW); spring water (SPW); Jordan River water (JRW)) and the two rates of wetting. Columns labeled with the same letter are not significantly different at $P<0.05$.

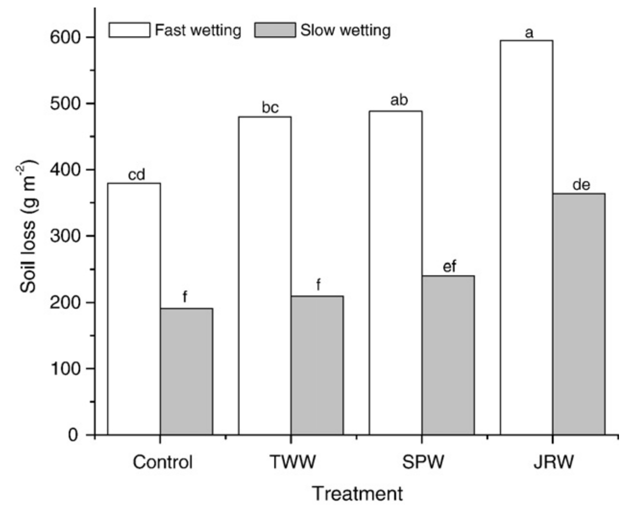


Fig. 4. Total soil loss for the different water quality treatments (control; treated wastewater (TWW); spring water (SPW); Jordan River water (JRW)) and the two rates of wetting. Columns labeled with the same letter are not significantly different at $P<0.05$.

### 3.2.2. Soil loss

Data for total soil loss ( Fig. 4) ranged from 190 for the control to 364 g $m^{-2}$ for JRW for slow wetting treatments and were significantly lower than those for the corresponding fast wetting treatments which varied in the range of 380 to 595 g $m^{-2}$. This clearly demonstrated the importance of minimizing aggregate slaking for reducing soil susceptibility to erosion.

Differences in total soil loss, due to water quality treatment for each of the two wetting rates, were relatively small compared to those between wetting rates and, in most cases, were not statistically significant ( Fig. 4). This indicated that the impact of increased soil sodicity on both runoff and soil loss, for a given degree of aggregate slaking, was comparable. However, a careful examination of the impact of soil sodicity on the relation between soil loss and runoff for each wetting rate revealed a somewhat different picture. In the case of fast wetting, a linear relation between runoff and soil loss best fitted the experimental data ( Fig. 5a). Mamedov et al. (2002) postulated that this type of relation indicates that erosion may be limited by the carrying capacity of the runoff water; i.e., the runoff is carrying the maximum sediment load even though detachment could theoretically supply more material for erosion. The reason that there is an excess amount of erodible material in the case of severe aggregate slaking could be due to the production of a greater quantity of fine sized soil material that is more readily transported in runoff water. Therefore, it can be concluded that, under fast wetting, the actual impact of sodicity on soil loss, is determined by its effect on the amount of runoff.

An exponential type curve better described the relation between runoff and soil loss when slow wetting was used ( Fig. 5b), suggesting that in this case the carrying capacity of the runoff was no longer the limiting factor for soil loss. With little or no slaking, the amount of available smaller sized erodible soil material is less than in the case of fast wetting. The exponential relation indicated that, when aggregate slaking was prevented, soil susceptibility to detachment by raindrop impact and runoff flow became greater with the increase in soil sodicity level ( Fig. 5b). Increasing sodicity

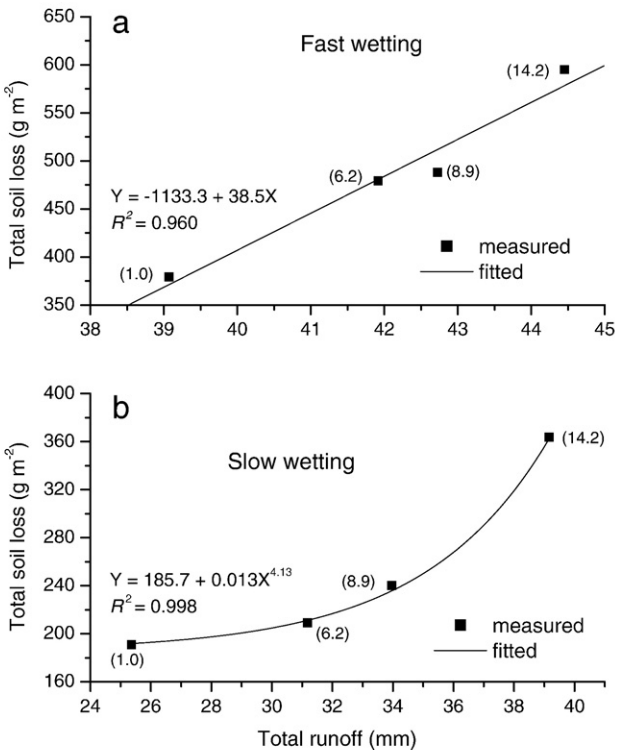


Fig. 5. Total soil loss as a function of total runoff. Numbers in brackets next to plotted points indicate the SAR relevant to the soil sample used.

increases the physico-chemical clay dispersion and weakens aggregates (Agassi et al., 1981). This adverse effect of sodicity on soil loss was noticeable, however, only in the case where most, if not all, of the mechanical breakdown of the aggregates was due to raindrop impact alone, following little or no slaking during wetting, resulting in seal formation. Evidently, when the aggregates are exposed to both types of force involved in mechanical breakdown, producing amounts of finer soil material in excess of the carrying capacity of the runoff water, the impact of sodicity on soil loss is indirect, transpiring through its effects on the amount of runoff.

## 4. Conclusions

Our results clearly demonstrate the importance of preventing aggregate slaking in maintaining high HC and low runoff and soil loss levels. Comparison of the results for the three water quality treatments indicated that effects of soil sodicity on these indices were more pronounced in the absence of aggregate slaking (using slow wetting) than in the case where aggregate slaking took place (using fast wetting). Irrespective of aggregate slaking, irrigation with TWW had a consistently more favorable effect on HC, runoff and soil loss than irrigation with the saline–sodic JRW. Comparison of the TWW treatment with the moderately saline–sodic SPW treatment showed, at times, comparable values for the measured parameters. This could be ascribed to the small difference in soil sodicity between the two treatments.

Involvement of aggregate slaking in the processes of water flow within the soil was found to be detrimental to both the soil structure and its water transmitting properties even at the sodicity levels found in TWW. Hence, our results suggest that replacing saline–sodic irrigation water by TWW, with significantly lower salinity and sodicity levels, may prove beneficial in improving soil structural stability and could also mitigate problems associated with high levels of runoff and soil erosion, particularly in regions of low to moderate rainfall intensities. There was no direct evidence that the presence alone of higher loads of organic matter and suspended solids in the TWW led to any deterioration or preservation of soil structural stability in this study.

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