Soil & Water Management & Conservation

Effects of the Dissolved Organic Carbon of Treated Municipal Wastewater on Soil Infiltration as Related to Sodium Adsorption Ratio and pH

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The increasing scarcity of fresh water in arid and semiarid regions means we must use alternative supplies for irrigation, such as treated municipal wastewaters. Generally, only their salinity level, the Na adsorption ratio (SAR), and the microbial content are evaluated. These waters contain higher concentrations of Na and thus higher SAR, and higher pH and dissolved organic C (DOC) than traditional supplies. We examined the effect of DOC in treated wastewater on soil physical properties, specifically the infiltration rate, aggregate stability, penetration resistance, and dispersibility of the soil. Wastewater and prepared waters (DOC-free) at SAR 4, 7, and 10 and at pH 7.0 and 8.0 were applied to a sandy loam soil outdoors as discrete events for 153 d, allowing soil to dry to −50 kPa before irrigation. Independent of Na level and pH, the treated wastewater reduced the infiltration rate, reduced aggregate stability, and increased dispersibility. The effect of DOC was equivalent to an increase in SAR of two to three units. An increase in the SAR (from 4 to 7) and pH (from 7 to 8) decreased infiltration and aggregate stability. The increased pH from 7.0 to 8.0 reduced infiltration and aggregate stability, with and without DOC, for all SAR values examined. We conclude that the SAR, DOC, and pH of wastewaters are high enough to cause a loss of infiltration and deterioration of soil physical properties. The pH and DOC of the water, plus salinity and SAR, should be considered when evaluating suitability for irrigation.

Abbreviations: DI, dispersion index; DOC, dissolved organic C; EC, electrical conductivity; ESP, exchangeable Na percentage; ET_0, potential evapotranspiration; SAR, Na adsorption ratio.

Core Ideas

- **DOC in treated wastewater decreased infiltration relative to prepared waters of the same composition.**
- **The effect of the DOC was equivalent to an increase in SAR of 2-3 units.**
- **The SAR of typical wastewaters is sufficiently high to cause loss of infiltration.**
- **DOC, SAR and elevated pH all had detrimental effects on soil physical properties.**
- **Sustained use of wastewaters for irrigation requires treatment to lower pH below 8.0 and SAR below 4.**

Limited fresh water supplies in arid and semiarid regions are being depleted by overuse and excessive extraction. The sustainability of irrigated agriculture in these regions, including California is in jeopardy unless new water supplies are developed. Agriculture is the major water user but as it typically operates on a low profit margin, it cannot compete with industrial and domestic users in a rising water price market. As fresh water shortages have developed, many growers are considering recycled wastewater as a potential new resource for crop irrigation. Many regions are implementing wastewater utilization for crop irrigation and maintaining landscape vegetation. In Israel, recycled water comprises more than 50% of the total water used for crop irrigation (Israel Water Authority, 2015). On a world basis, it is estimated that at least 20 million hectares of agricultural land are irrigated with wastewater (Corcoran et al., 2010).

The reuse of wastewater instead of using fresh water for irrigation may have several potentially negative impacts on soils and crop production; among these are: (i) increased soil salinity (including Cl and Na ions, which may have specific toxicity to many crops) and (ii) increased soil Na concentrations with attendant

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Soil Sci. Soc. Am. J. 81:602–611
doi:10.2136/sssaj2016.09.0310
Received 27 Sept. 2016.
Accepted 3 Feb. 2017.
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effects on soil physical properties. The effects of increased salinity and potentially toxic ions on crop production can be evaluated on the basis of existing crop salt tolerance and ion toxicity studies (Grieve et al., 2012). The Na hazard, characterized by the SAR [Na/(Ca+Mg)$^{0.5}$, where concentrations are expressed in mmol L^{-1}), and elevated pH have an adverse effect on soil physical properties and are used to evaluate the suitability of water for irrigation (Suarez, 2012). More recently, concerns have been raised about the survival of pathogens even in tertiary-treated wastewater (Rosenberg Goldstein et al., 2014), as well as contaminants such as pharmaceuticals (Petrie et al., 2015) and dissolved organic matter (Michael-Kordatou et al., 2015).

There is an extensive body of literature on the impact of treated wastewater on soil physical properties, mostly infiltration and hydraulic conductivity, but the results are not entirely consistent. Sparling et al. (1999) found no detrimental effects on unsaturated hydraulic conductivity after 3 yr of irrigation with tertiary-treated wastewater at one site and no detrimental effects after 7 yr of irrigation with secondary-treated wastewater at another site. Mathan (1994) found an increase in hydraulic conductivity after irrigation with municipal wastewater for 10 to 15 yr, compared with an adjacent site irrigated with well water of comparable electrical conductivity (EC) and SAR. Tunc and Sahin (2015) found an increase in the aggregate stability of soils irrigated for 2 yr with wastewater compared with fresh water with lower EC and SAR.

However, most researchers have found wastewater to have a detrimental effect on soil properties. Among these, Schacht and Marschner (2015) compared the hydraulic conductivity at four locations where adjacent fields were using fresh water and treated wastewater. On average, the hydraulic conductivity at the wastewater sites was 50% of that of the freshwater sites. The DOC was not characterized but the effluent SAR ranged from 2.5 to 3.2. The pH of the treated wastewater ranged from 7.9 to 8.4. Assouline and Narkis (2011) found a decrease in saturated hydraulic conductivity, sorptivity, and infiltration rate after 15 yr of irrigating with treated wastewater compared with freshwater. They attributed the effects primarily to increases in SAR and clay swelling (smectitic clayey soil). Levy et al. (1999) examined the relative saturated hydraulic conductivity of three soils after application of two wastewaters with differing chemical O_2 demands and SAR (53 mg L^{-1} and 5.6 respectively versus 313 mg $L^{-1}$and 6.4) in a laboratory column study. They determined that increased organic matter resulted in decreased hydraulic conductivity. Levy et al. (2014) determined that long-term use of treated wastewater with SAR 4 to 6 increased soil SAR and exchangeable Na percentage (ESP) to levels that could be detrimental to soil physical properties. The SAR of tertiary-treated wastewaters in southern California typically ranges from 4 to 5 (Goldberg and Suarez, 2016). Although these SAR levels are considered safe for use according to some irrigation water criteria (e.g., Ayers and Westcot 1985), they are sufficiently elevated relative to fresh water values that they adversely impact infiltration (Suarez et. al., 2006). An analysis of the long-term use of treated wastewater has raised concerns about declines in crop productivity over time (Assouline and Narkis, 2013). These declines may be related to increasing soil salinity, reductions in infiltration, the resultant aeration problems, or a combination of these. Bedbabis et al. (2014) reported that soil irrigated with treated wastewater had decreasing water infiltration over a 4-yr time period. Sensitivity to individual variables such as EC, SAR, and pH cannot be readily quantified in most field studies, as there is generally only one fresh water and one wastewater composition.

Part of the discrepancy in the results among the studies may be related to differences in the original properties of the fields being compared. Few of the long term studies have a characterization of the initial conditions and the effects of the various processes are rarely isolated.

In addition to SAR effects, there have been some indications that long-term use of wastewater results in adverse effects, including blockage of pores (Bouwer and Chaney, 1974; De Vries, 1972). Irrigation with water with a dissolved organic matter concentration of 70 mg L^{-1} resulted in reduced infiltration as a result of clogging of soil pores and thus reduced drainage and aeration (Gharaibeh et al., 2016). Humic substances also influence clay dispersion. Tarchitzky et al. (1993) observed that even relatively small concentrations of humic substances and fulvic acid increased the flocculation value for Na montmorillonite clay (but not Ca montmorillonite) and that the effect was greater at pH 8 than at pH 4.

Municipal wastewaters inevitably have elevated alkalinity because of decomposition of organic matter and thus elevated pH once the CO_2 degasses. The increased alkalinity and resulting increased pH are rarely considered when evaluating the impact of wastewaters on soil physical properties. In most of the earlier studies, it is not possible to separate the effects caused by SAR, pH, and DOC because comparisons were made between one fresh water and one wastewater and they differed in all three variables. Suarez et al. (1984) established that elevated pH at the same EC and SAR decreased saturated hydraulic conductivity and increased the electrolyte level needed for flocculation. This was attributed to changes in the variable edge charge with increasing pH (a change from a positive to a negative edge charge of the clays with increasing pH). Tarchitzky et al. (1999) determined that for wastewaters at SAR 3.2, EC 1.3 dS m^{-1}, and DOC of 8 mg L^{-1}, there was an increase in the flocculation value compared with that for prepared water with similar EC and SAR. Also, packed columns of a sandy soil (predominantly montmorillonite clay) experienced up to a 80% reduction in relative hydraulic conductivity in the presence of wastewater compared with a solution of comparable electrolyte content. It is not clear whether the results from short-term laboratory column studies under continuously saturated conditions and relatively high water flow will accurately reflect those contained in the field.

Our objective was thus to evaluate and isolate the effect of DOC, pH, and elevated SAR in treated wastewater on infiltration and other soil physical properties under simulated field conditions.

We examined the effects of variable pH, SAR, and DOC using wastewater with a moderate DOC and prepared wastewater with comparable inorganic ion composition in a 5-mo outdoor experiment under conditions of wetting and drying.

MATERIALS AND METHODS

Infiltration

Surface samples (0–10 cm) of an Arlington sandy loam soil were collected from the Agricultural Experiment Station on the campus of the University of California, Riverside, CA. The soil is classified as coarse-loamy, mixed, active Thermic Haplic Durixeralf. Soils were crushed and passed through a 5-mm screen, air dried, and analyzed for texture and chemical characteristics.

Plastic containers (35.5 cm tall with a 26-cm base diameter and a 28.5-cm top diameter) were fitted with two ceramic extractors (2 cm diameter by 7 cm long; Model SSAT, Cat. No. A4, Irrometer Company, Riverside, CA) at the bottom of the containers in a 7 cm thick layer of No. 90 fine quartz sand. After mixing and dividing the soil for the individual containers, each container was prepared by adding soil in 5-cm increments. A total of 20 cm of soil was added above the sand. The column was lightly packed during the filling process to achieve a bulk density of 1.4 Mg m^{-3}. A schematic representation of the design is presented in Fig. 1.

Using a completely randomized design, all containers were placed in an open outdoor area in four rows. We applied tap water (EC = 0.6 dS m^{-1} and SAR < 0.4 mmol$^{0.5}$ L$^{-0.5}$) to enable soil settling before the initiation of the treatments. A vacuum of −30 kPa (−0.3 bars) was applied to the extractors before, during, and after each water application. The vacuum was shut off when drainage ceased. We irrigated the containers eight times with sequences of tap water and subsequent drying until the infiltration rates were relatively uniform from irrigation to irrigation. This process took 24 L of water per column and eight cycles of wetting and drying over 4 mo. After these initial irrigation events, the infiltration rates stabilized and we initiated the 153-d experiment with application of the first irrigation water treatments.

We collected a large container of secondary-treated municipal wastewater from the city of Carlsbad, which is located on the southern California coast. Portions of the wastewater were adjusted to three levels of SAR (4, 7, and 10) and pH values of 7.0 and 8.0 by addition of salts (NaCl, $CaCl_2{\cdot}2H_2O$, and $MgCl_2{\cdot}6H_2O$) and pH adjustment with HCl. The initial wastewater as sampled had a SAR of 4 (mmol$^{0.5}$ L$^{-0.5}$). The adjustment and balance of the treatment SAR and pH values were obtained with the Extract Chem version 2.0 program (Suarez and Taber, 2012). We also prepared synthetic waters with the same macroion chemical composition, EC, and pH by adding various salts [$CaCl_2{\cdot}2H_2O$; $MgCl_2{\cdot}6H_2O$; NaCl; $NaHCO_3$; Na_2SO_4; $(NH_4)_2SO_4$; K_2SO_4] to tap water (ground water from the San Bernardino basin, California), constrained by the solubility of calcium carbonate (up to threefold supersaturation can be maintained without precipitation (Suarez and Rhoades, 1982). The initial wastewater had a SAR of 4 and an EC of 1.7 dS m^{-1}. Thus the experimental design consists of three levels of SAR, two pH values, two types of water, and three repetitions for a total of 36 experimental units, distributed in a random design (12 treatments and three repetitions). The EC of the analyzed solutions was relatively constant at 2.7 dS m^{-1}.

All plots were treated by alternating events of manual irrigation with the treatment waters and subsequent drying periods. We avoided the interactive effects of rainfall and irrigation by covering the containers during the infrequent winter rains (250 mm average in Riverside, CA). The irrigation waters were applied manually on the surface as flood irrigation events with applications of 3.00 L (5.0 cm height) per container. Infiltration in cm d^{-1} was calculated for each container. The soil was subjected to a vacuum system in the bottom sand layer of each container via two ceramic extractors (Model SSAT, Cat. No. A4, Irrometer Company) connected to a reservoir and a vacuum pump. In this manner, we maintained a bottom boundary

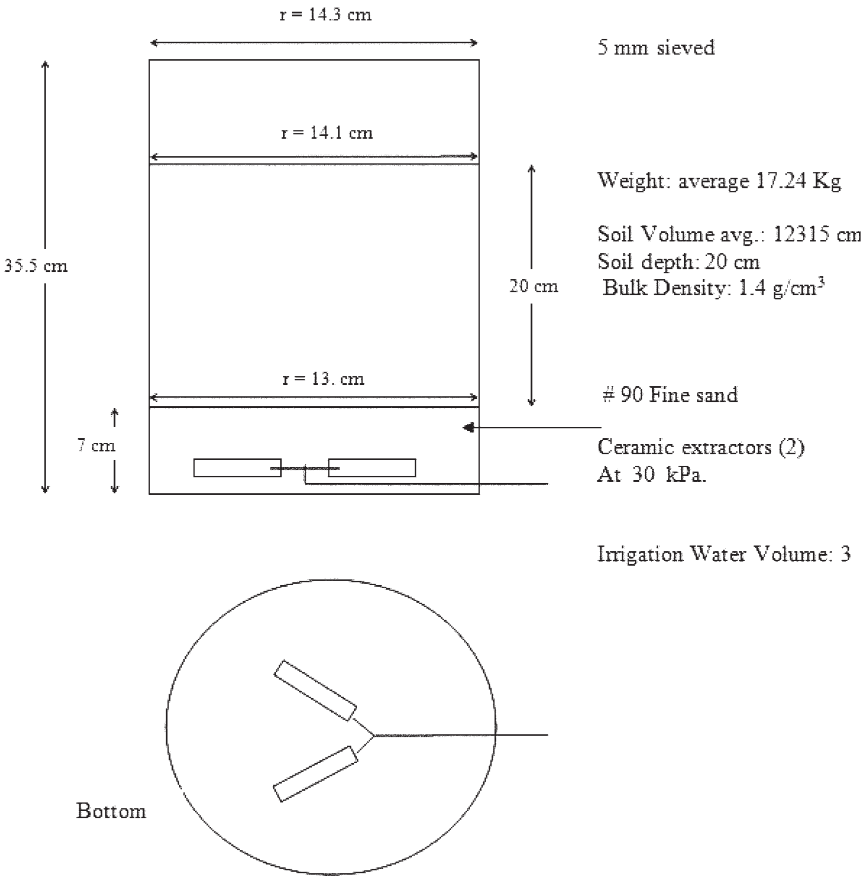

Fig. 1. Schematic representation of the construction of the soil-filled containers.

matric pressure condition of −30 kPa. The vacuum system was turned on at the time of irrigation and maintained until there was no further water drainage. The soil surface was allowed to dry to −50 kPa, at which time the new irrigation was applied. Drying was monitored using tensiometers (Model R, Cat. No. 112. Irrometer Company), installed at 5 cm depth in containers. The climatic conditions in Riverside, CA were such that it took approximately 2 wk for the soil to dry between irrigation events. Local potential evapotranspiration (ET_0) was determined from an on-site weather station; total water applied was recorded to ensure that the ratio of irrigation water to ET_0 was within a range that was appropriate for an irrigated field.

Infiltration rates were measured as the time required for infiltration of the 5 cm of water. This measurement is more stable than typical field infiltration rates because we have a fixed pressure head of −30 kPa at the lower boundary and we dried the soil to a consistent initial water content before each irrigation, using a tensiometer placed at 5 to 8 cm depth in a reference container adjacent to the experiment. We evaluated the initial infiltration rate of each container to ensure uniformity, ensuring that infiltration rates with tap water before the start of the experiment were all within 7% of the mean. The final infiltration measurements were made on Day 153. The experiment was terminated when the changes in infiltration rates between the irrigation events of most treatments appeared to be minimal. The relative infiltration rate for each container was calculated as the ratio of the final infiltration rate (from the last irrigation event) divided by the initial infiltration rate of that container before the initiation of the water treatments.

Penetrometer Resistance

Undisturbed soil cores were collected from the containers for laboratory measurement of penetrometer resistance at the conclusion of the infiltration experiment. Before collection of undisturbed soil cores, the containers were irrigated once more with the treatment waters and a drying period was allowed until the soil water content reached −50 kPa, for optimum sampling. For each sample, a brass core sampler 5.4 cm in diameter and 2.5 cm in height was pressed into the soil. The soil adjacent to the sample was removed and a flat plastic tool was inserted below the bottom of the core to hold the core. The soil core was slowly lifted to ensure that the sample did not slide or separate. The cores were then all equilibrated at −0.30 MPa matric potential. Penetration resistance to a 30° semiangle probe 3.4 mm in diameter was measured using an electric penetrometer machine at a rate of penetration of 15 mm min^{-1} (the system was constructed by Lausheng Wu, Dept. of Environmental Sci. UC Riverside, Riverside CA). The readings for each core were collected via an automated system of data acquisition throughout the 0 to 1.5 cm depth. The measurements were averaged with depth of each sample. The samples were then oven-dried at 105°C for 24 h and the gravimetric water content was determined.

Aggregate Stability

Aggregate stability of the samples collected at the end of the experiment was measured using a modification of the method of Kemper and Rosenau (1986), adapted by Nimmo and Perkins (2002). Air-dried aggregates of 1 to 2 mm were obtained and weighed. Aggregates were prewetted for 30 min using a humidifier chamber and then placed on the sieves. The stability of the 1- to 2-mm aggregates from each treatment was measured with a wet-sieving machine (wet sieving apparatus, #08.13, Eijkelkamp Soil & Water, Giesbeek, the Netherlands) that allowed us to raise and lower the sieves by 1.3 cm within a set of cans filled with the solution. We used a stroke frequency of 35 strokes min^{-1} for 3 min. In the first set of measurements, we used solutions in the cans corresponding to the irrigation water used in the treatments from which we obtained the aggregates. This is a modification of the Kemper and Rosenau (1986) method that we made to enable measurement of stability with the irrigation water rather than with deionized water. We then reacted the aggregates with a second set of cans, all containing a dispersing solution of 2 g L^{-1} $(NaPO_3)_6$ (as per Kemper and Rosenau, 1986). Sieving was resumed until only sand particles were left on the sieve. Both sets of cans were oven-dried and the dry weight of the material in each can was determined. The stable fraction was equal to the weight of the soil contained in the dispersing solution cans divided by the sum of the weights in both cans (Kemper and Rosenau, 1986).

Dispersion Index

To calculate the dispersion index (DI), we used a modified Emerson dispersion test (Loveday and Pyle, 1973). Three air-dried aggregates about 3 to 5 mm in diameter were placed into 50 mL of the corresponding irrigation water treatment in a Petri dish. The temperature was kept constant (± 1.0°C), and the Petri dish was protected from disturbance. At 2 and 20 h after immersing the aggregates, a visual judgment of the degree of dispersion was made; at each time, a score of 0, 1, 2, 3, or 4 was awarded. According to the method, the scale from 0 to 4 indicates an increasing degree of dispersion, thus a score of 0 represented no dispersion and was checked by comparing the appearance of aggregates in a 0.01-M $CaCl_2$ solution in which no dispersion took place; a score of 1 is a slight dispersion, recognized by a slight milkiness of the water adjacent to the aggregate; a score of 2 is moderate dispersion with obvious milkiness; a score of 3 is strong dispersion with considerable milkiness and about half of the original volume dispersed; a score of 4 is complete dispersion that leaves only sand grains in a cloud of clay (Loveday and Pyle, 1973). For those samples scoring 0, a further estimate of dispersion was made after remolding 5- by 5-mm aggregates cubes at −10.3 kPa (−100 cm) matric potential. For those soils where the air-dried aggregates did not disperse after remolding at −10.33 kPa matric water potential, the 2- and 20-h scores were added together. The range of possible values for DI in this case is 0 to 8. For those soils where the air-dried aggregates did disperse, the 2 and 20 dispersion scores were added together and then added to 8 so that the range of possible values was 9 to 16. The complete range of values for DI was therefore 0 to 16, capturing both degree and rate of dispersion.

Table 1. Composition of wastewater and treatments.†

SAR	pH	EC	Ca^{2+}	Mg^{2+}	Na^+	NH_4^+	K^+	Alkalinity	SO_4^-	Cl^-
			$\text{mmol}_c\ L^{-1}$							
					Original wastewater					
4.42	7.2	1.74	4.06	2.3	7.9	1.61	0.5	4.2	4.69	7.31
					Wastewater treatments					
4	6.96	2.65	9.08	4.66	10.6	1.5	0.49	0.14	4.81	19.8
7	7.03	2.60	5.72	3.59	15.2	1.5	0.60	0.14	4.80	19.5
10	6.87	2.68	4.00	2.48	17.9	1.5	0.48	0.14	4.82	18.1
4	7.88	2.60	6.00	6.88	10.4	1.5	0.46	1.50	4.55	19.0
7	7.94	2.67	4.65	4.37	15.5	1.5	0.47	1.52	4.67	20.2
10	8.01	2.65	4.00	2.46	18.0	1.5	0.48	1.52	4.63	18.8
					Synthetic treatments					
4	6.99	2.67	9.16	4.31	10.8	1.5	0.52	0.14	4.62	23.5
7	6.87	2.68	5.89	3.15	15.2	1.5	0.52	0.14	4.64	23.3
10	7.93	2.70	4.11	2.00	15.2	1.5	0.52	0.14	4.65	23.2
4	8.0	2.75	6.83	7.29	12.0	1.5	0.57	1.51	5.18	20.1
7	7.89	2.69	5.38	4.28	15.8	1.5	0.58	1.52	5.27	20.5
10	8.03	2.67	4.44	2.39	20.0	1.5	0.56	1.52	5.13	19.7

†SAR, Na adsorption ratio; EC, electrical conductivity.

Chemical Analysis

At the end of the infiltration experiment, we collected soil samples from the containers for chemical analyses. Saturation paste and extracts (US Salinity Laboratory Staff, 1954) were obtained for each depth. We measured EC, pH, and SAR. Treatment solutions were analyzed before and during the initiation of the experiment. Soluble Na, Mg, and Ca concentrations for calculation of SAR were determined using inductively coupled plasma optical emission spectrometry (Optima 3300 DV, PerkinElmer, Norwalk, CT). Total alkalinity was determined by titrating known aliquots of samples with 1.00 $\text{mmol}_c\ L^{-1}$ HCl (standardized against potassium biiodate) to the inflection point (pH of ~4.40) (American Public Health Association, 2012). We used a digital chloridometer [a coulometric titrator (4425000. Labconco. Kansas City, MO) using Ag wire] to determine Cl ion concentrations. Irrigation water DOC was determined by first acidification with H_2SO_4 to below pH 2, sparging with N_2 (removing dissolved inorganic C) then combustion at 900°C, in a Model 5120 UIC Carbon Analyzer (UIC Inc., Joliet, IL). The DOC concentration was determined to be 13 mg L^{-1} in the treated wastewater. Statistical analysis using Tukey- and *t*-test was performed using the SAS package (SAS Institute, 2011)

RESULTS AND DISCUSSION

The cumulative ET_0 during the conduct of the experiments (153 d from first irrigation water treatment) was 41.5 cm and the corresponding amount of applied water was 65 cm. This corresponds to a calculated leaching fraction of 0.36, which is relatively realistic for field conditions in terms of water applications and ET_0. Most controlled studies use very large volumes of water relative to ET_0. The analyses of the treatment solutions are shown in Table 1. The EC and SAR values were close to the target values and can be considered the same for the synthetic waters and wastewaters.

As shown in Fig. 2, the initial infiltration rates for the synthetic waters ranged from 66 to 73 cm d^{-1}. The infiltration rates of the individual water applications decreased with time for all SAR and pH treatments. The treatment effects of SAR were even observed in the first irrigation event and, as expected, the SAR 4 treatments had the smallest decrease in infiltration. As also shown in Fig. 2, the higher SAR treatments had a rapid loss in infiltration relative to the SAR 4 treatment. The decrease in infiltration over time is expected for two reasons: (i)The soil SAR is near zero before the first infiltration event and cation exchange ($2Na^+ + CaX_2 \rightarrow Ca^{2+} + 2\ NaX$) buffers the change in soil water SAR so it gradually increases as successive irrigations are applied; (ii) the cumulative effects of soil dispersion over time can also be expected. This soil does not contain swelling clays and thus we consider that the infiltration losses were associated primarily with soil dispersion.

The major effect of reduced infiltration was related to an increase in irrigation water SAR. However, an adverse effect of increased pH (from 7.0 to 8.0) on infiltration was also observed. The pH 8.0 treatments consistently had a lower infiltration than the pH 7.0 treatments at every SAR value (Fig. 2). At the end of 153 d, the infiltration rates were relatively stable. On the basis of these data, we note that the reduction in infiltration of the pH 8.0 versus pH 7.0 irrigation water at SAR 4 is approximately intermediate between the curves for pH 7.0 SAR 4 and pH 7.0

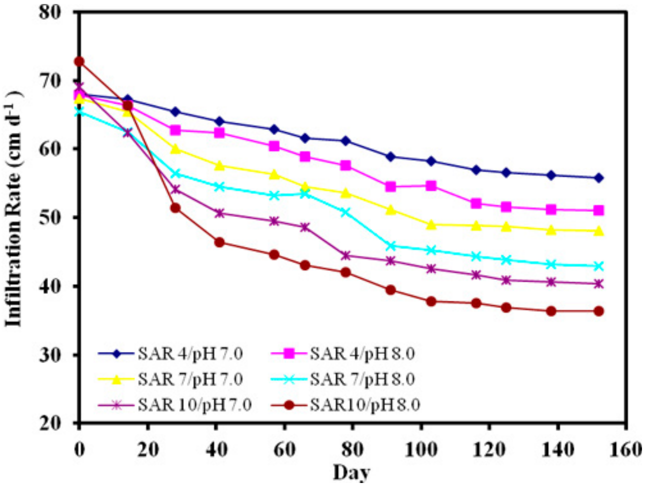

Fig. 2. Infiltration rate versus time for the prepared synthetic water treatments relative to Na adsorption ratio (SAR) and pH for the 13 irrigation events.

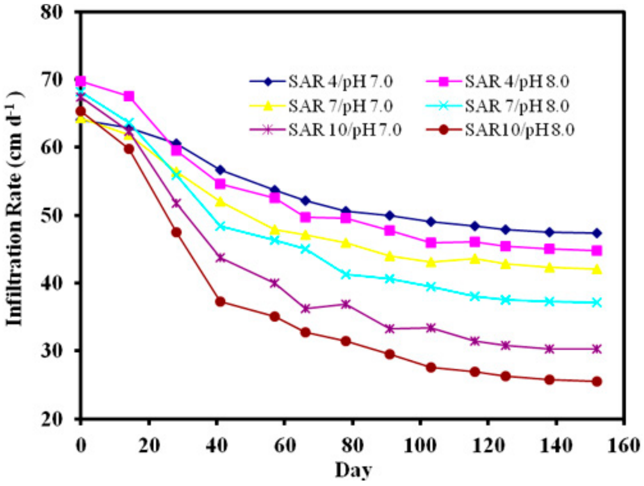

Fig. 3. Infiltration rate versus time for the wastewater treatments as related to Na adsorption ratio (SAR) and pH for the 13 irrigation events.

SAR 7. This means that the pH effect on infiltration caused by an increase in pH (8.0 vs. 7.0) was equivalent to the expected rate at pH 7 SAR 5.5. In a similar manner we can see that the pH 8.0 rates are always lower than the pH 7.0 rates and are approximately intermediate between the pH rate at that SAR and the pH 7 rate at the next SAR value.

The infiltration rates of the wastewaters were initially comparable with those of the synthetic waters (Day 0), as shown in Fig. 3. Comparable effects are seen for the actual wastewaters compared with the synthetic waters both in terms of time effects, pH, and SAR effects. A comparison of the data in Fig. 3 with those in Fig. 2 reveals that the infiltration rate declined faster in the presence of the DOC (wastewater). The differences at the end of the experiment are also noticeable; for example, at SAR 10 pH 8.0, the wastewater infiltration rate was reduced to 26 cm d^{-1}, but for the synthetic water at the same SAR and pH, the rate was 37 cm d^{-1}.

The final relative infiltration rate (final divided by initial) for all treatments as a function of the irrigation water's SAR is shown in Fig. 4a; all treatments had reduced infiltration under increasing SAR. If we use Tukey's test, there were significant differences in the final relative infiltration rate with SAR for pH 7 and 8 for both synthetic and wastewaters ($p < 0.05$). Differences between synthetic and wastewater infiltration using *t*-test were significant only at SAR 10 and pH 7, and SAR 10 pH 8 ($p < 0.05$). For all other treatments, the relative infiltration rates of wastewater treatments were lower than those of the synthetic waters (Fig. 4a) but were not statistically different (at $p < 0.05$). Despite the lower mean infiltration rate for all pH 8 versus pH 7 treatments (Fig. 4a), only the synthetic water at SAR 10 had a statistically different infiltration rate between the two pH treatments ($p < 0.05$). Comparing the synthetic waters with the wastewaters at the same SAR and pH, we observe an additional loss of 10% in the infiltration rate of the wastewater for all the different SAR and pH treatments. In this instance, the adverse effect of DOC on infiltration rate is comparable to an additional increase of 2 to 3 units of SAR. The differences in infiltration rate with an increase in pH from 7 to 8.0 ranged from a decrease of 7% at SAR 4 to 8.5% at SAR 10 for the synthetic waters and a relative decrease from 9.6 to 6.0% for the wastewaters again at SAR 4 and 10, respectively. Thus the one-unit increase in pH caused an average decrease of 8% in the final infiltration rate.

The final infiltration rates, expressed as a function of the SAR values of the soil saturation extracts collected at the termination of the experiment, are presented in Fig. 4b. All saturation extract values were greater than the irrigation water SAR (compare the values in Fig. 4b with the treatment values in Fig. 4a). This is expected because as the water evaporates, the soil solution is concentrated and the SAR increases (even without calcite precipitation). The synthetic water saturation extract samples from the pH 7.0 treatments were very close to

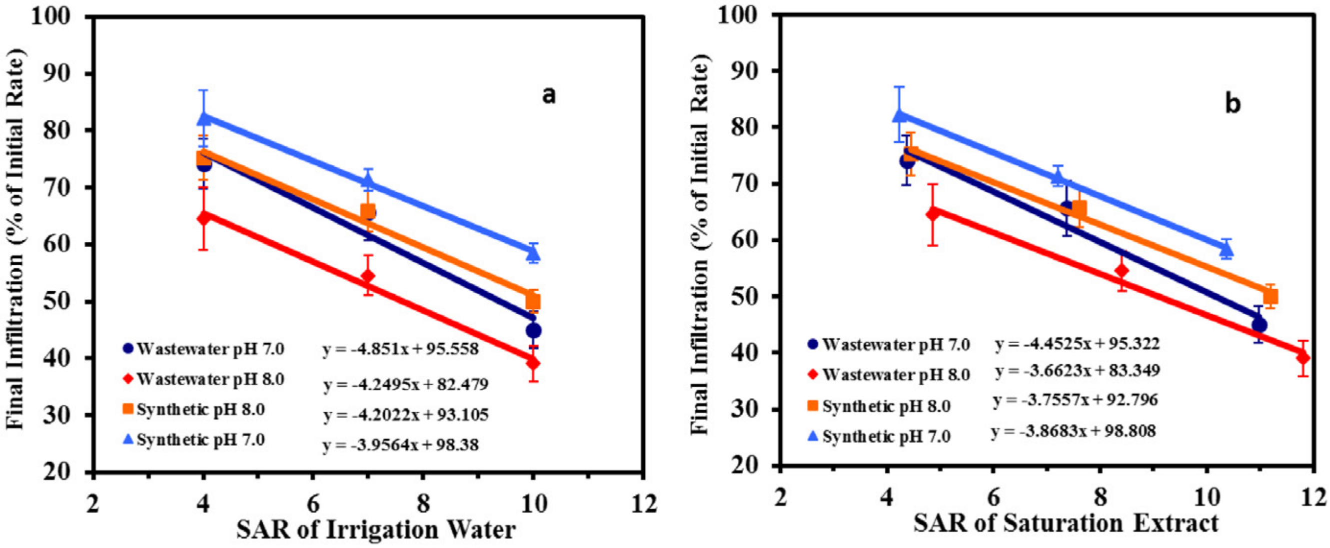

Fig. 4. Relative infiltration rate (final divided by initial) as a percentage for the prepared (synthetic) waters and wastewater treatments relative to pH and (a) Na adsorption ratio (SAR) of the irrigation water and (b) SAR of the soil saturation extract at the end of the experiment. Error bars represent the SE of the means.

the SAR values in the irrigation water, only 0.27 SAR units above the irrigation water SAR, on average, indicating that the soil had essentially equilibrated with the irrigation water by the end of the experiment (data not shown). The saturation extracts from the synthetic water treatments at pH 8.0 were, on average, 0.75 SAR units above the irrigation water SAR. We note that the data for the wastewaters show a greater shift toward higher SAR relative to that observed in the synthetic waters, with an average increase of 0.57 units at pH 7 and an average increase of 1.36 SAR units at pH 8.0. We consider that the elevated SAR values in the wastewater treatments relative to the synthetic water treatments is likely to be caused by the decomposition of DOC over time, producing additional alkalinity in these treatments and precipitating some of the Ca in the solution as calcite, thus raising the SAR of the soil solution. The decomposition of organic matter, producing net alkalinity and causing calcite precipitation, has been documented previously (Amrhein and Suarez, 1987). Levy et al. (2014) observed that the SAR and ESP in soils irrigated with wastewater for 10 yr were higher than expected. They did not provide details but attributed it to cycles of irrigation and rainwater. Alternatively, it can be explained in a similar manner to what we observed for both the concentration of the water by evapotranspiration and decomposition of the DOC, namely production of additional alkalinity and subsequent $CaCO_3$ precipitation, thus raising the SAR.

Penetrometer resistance decreased with depth and increased with SAR for all four treatments. The changes in penetrometer resistance related to SAR and depth are shown in Fig. 5a and Fig. 5b for synthetic waters at pH 8.0 and wastewater at pH 8.0, respectively. Similar relations were observed for both waters at pH 7.0 (data not shown). As shown in Fig. 5a for the synthetic waters at pH 8.0, penetrometer resistance was highest at the surface and lowest at 15 to 17.5 cm depth, indicating that the surface is more sensitive to increases in SAR. At SAR 4.0, the resistance at depth was 0.24 MPa, increasing to 0.45 MPa at the surface. This is in contrast to most studies, which show increasing resistance with depth under field conditions (Bauder et al., 2005). However, our soils were not initially compacted, as they were uniformly packed into soil containers. The best reference point for the subsequent discussion is the value of 0.24 MPa at 15 to 17.5 cm depth (Fig. 5a). One of the adverse effects of sodic soils is an increase in penetrometer resistance. This is expected, as dispersive forces are greater and aggregate stability is lower at the surface (Kemper and Koch, 1966). Of interest is that, at least in these controlled studies, the resistance measurements are sensitive enough to detect changes associated with a small increase in SAR (from 3 to 7 and from 7 to 10).

Qualitatively, resistance values below 1.38 MPa are considered "good" under field conditions (Bauder et al., 2005). For the synthetic waters, we determined a value of 1.15 MPa for the surface soil from the SAR 10 treatment (Fig. 5a), almost a threefold increase from the surface value at SAR 4 and approaching the 1.38 MPa value where the resistance rating changes to "fair". In contrast, the 15 to 17.5 cm depth sample for the same treatment only increased to 0.48 MPa. Our penetrometer data from soil samples collected from the wastewater treatments at pH 8 are shown in Fig. 5b. We measured a value of 1.6 MPa for the surface soil from the SAR 10 treatment (Fig. 5b), which is thus within the "fair" ranking. The value represents a twofold increase from the surface resistance value at SAR 4. In contrast, the resistance of the 15 to 17.5 cm depth sample for the same treatment only increased to 0.50 MPa.

The resistance measurements for the surface soils of all treatments are given in Fig. 6, enabling a comparison of the water and pH effects. When we used the Tukey test, there were significant differences in the surface penetrometer measurements (each treatment had three replicates) with SAR for pH 7 and 8 for both synthetic and wastewaters ($p < 0.05$). In all instances, SAR 4 had significantly lower resistance measurements than SAR 10. For wastewater at pH 8 SAR 4 was significantly different from

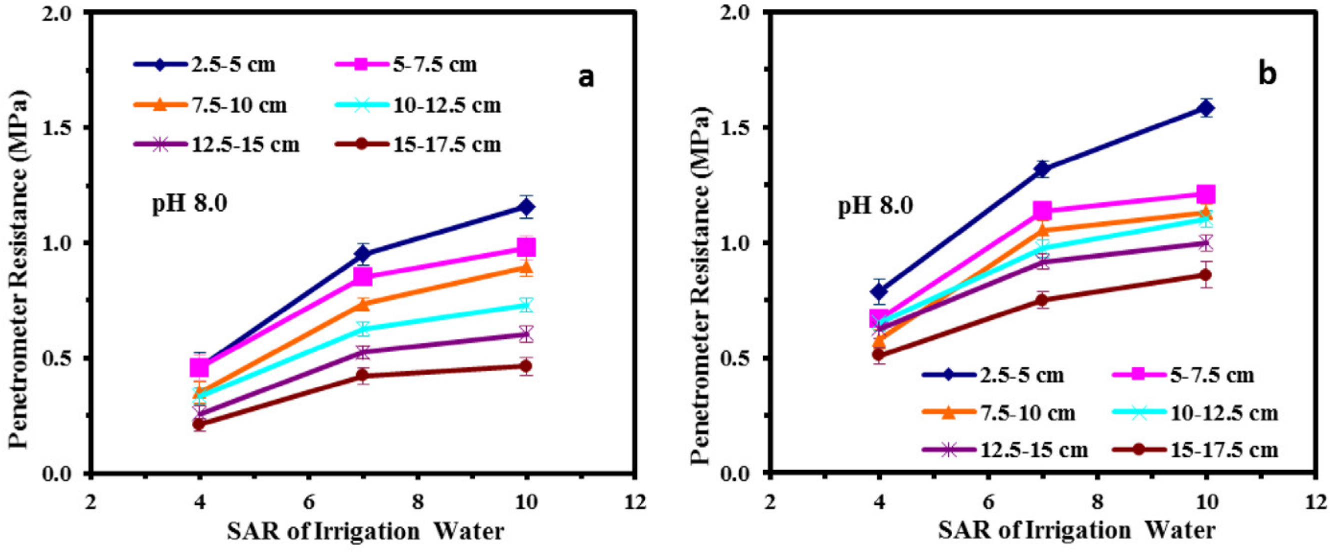

Fig. 5. Penetrometer resistance (MPa) from soil collected at the end of the experiment from the treatments with the (a) prepared synthetic waters and (b) wastewaters, relative to Na adsorption ratio (SAR) and depth. Data are from the pH 8.0 treatments. Error bars represent the SE of the means.

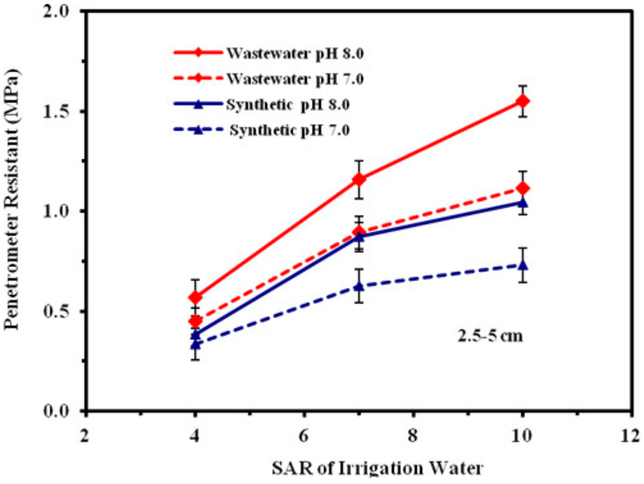

Fig. 6. Penetrometer resistance of soils taken at 2.5 to 5 cm depth at the end of the experiment from the wastewater and synthetic water treatments at pH 7.0 and 8.0. Error bars represent the SE of the means.

SAR 7 and SAR 10 but SAR 7 and SAR 10 were not significantly different. For wastewater at pH 8, all SAR treatments were significantly different from each other. For the synthetic waters at pH 7, the SAR 4 treatment was statistically different from SAR 10, whereas at pH 8, SAR 4 was significantly different from SAR 7 and SAR 10. The surface soil penetrometer data at pH 7.0 for both wastewater and synthetic water show similar trends to the data discussed above at pH 8.0 (Fig. 5a,b), but with lower resistance values. The synthetic water at pH 7.0 only had an increase in resistance to 0.70 MPa at SAR 10, in contrast to the value of 1.15 MPa for the pH 8.0 treatments (Fig. 5a). The resistance increased relatively little with wastewater at pH 7.0 at SAR 4, relative to our reference resistance from synthetic water at pH 7.0 at depth (0.24 MPa).

Comparisons of the pH 7.0 and pH 8.0 treatments in for surface soils Fig. 6 show that for both waters, the pH 8.0 treatments consistently had a greater resistance at the same SAR. In terms of resistance, the effect of wastewater was equivalent to the effect of an increase in pH from 7.0 to 8.0 or an increase in SAR of about 0.8 units at SAR 4 and 4 units of SAR at SAR 7. The pH effect on resistance corresponded to an effective increase of 0.5 SAR units at SAR 4. At SAR 7, the increase in pH was equivalent to an increase in SAR of 14.5 units. These data indicate that the adverse effects of increased pH were more pronounced at higher SAR and also more pronounced than in the infiltration experiments.

The aggregate stability of the surface soils (2.5–5 cm) for all the treatments is shown in Fig. 7. All treatments showed a relatively rapid decrease in aggregate stability with increasing SAR. As with the infiltration and penetrometer measurements, the wastewaters had a lower aggregate stability than the DOC-free synthetic waters at all SAR values. In addition, the aggregate stability of the soils treated with pH 8.0 water was lower than those treated with pH 7.0 water at all SAR values (Fig. 7). According to the results of Tukey's test, all treatments (synthetic water at pH 7, synthetic water at pH 8, wastewater at pH 7, and wastewater at pH 8) had significant differences for all the SAR values. The aggregate stability was also significantly different between pH 7 and pH 8 for all SAR values and between synthetic and wastewater treatments. Differences between waste and synthetic waters were statistically different for all treatments (pH, 7 and 8; SAR 4,7, and 10). The aggregate stability of the soil treated with pH 7.0, SAR 4 water was 82% (aggregates remaining after 3 min), just slightly lower than the value given in Kemper and Rosenau (1986) for an irrigated loam soil from Colorado (86%), where the EC and SAR of the irrigation water was not specified but was presumably low.

Our values are considerably higher than those reported by Schacht and Marschner (2015) for soils with a high clay content, with and without application of wastewater. They observed values ranging from 45 to 70% for fresh water treatments and 35 to 60% for soils collected from locations irrigated with

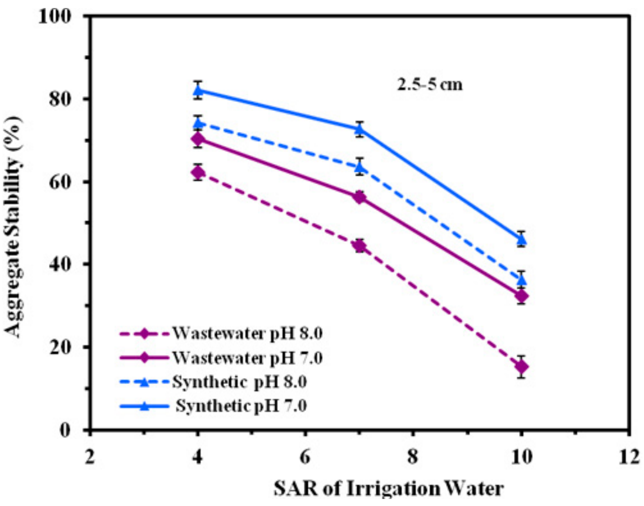

Fig. 7. Aggregate stability (%) of soils taken at 2.5 to 5 cm depth at the end of the experiment from the wastewater and synthetic water treatments at pH 7.0 and 8.0 relative to irrigation water Na adsorption ratio (SAR). Error bars represent the SE of the means.

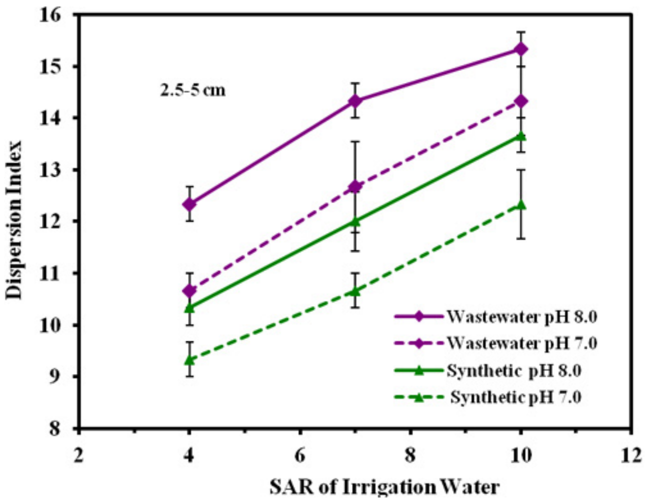

Fig. 8. Dispersion index of soil samples taken at 2.5 to 5 cm depth at the end of the experiment from the wastewater and synthetic water treatments at pH 7.0 and 8.0 relative to irrigation water Na adsorption ratio (SAR). Error bars represent the SE of the means.

wastewater. Higher clay content should be reflected in greater aggregate stability (Ruiz-Vera and Wu, 2006). However, the freshwaters used by Schacht and Marschner (2015) had a lower EC (0.49−1.1 dS m^{-1}) than our treatments (2.7 dS m^{-1}) and thus the aggregates would be more dispersive. They did not separate the effects and thus their differences in stability relating to fresh water versus wastewater would be related to the combined effects of EC, SAR, DOC (not reported), and pH. Differences depending on methodology, including prewetting rate and clay mineralogy in addition to texture, can produce differences in results (Ruiz-Vera and Wu, 2006). Our soils were prewetted very slowly and thus we might expect greater stability than otherwise. The results of aggregate stability are thus best interpreted relative to those in the same experiment, in which case, the test is excellent in terms of distinguishing the differences among treatments. However, aggregate stability may not be the optimal method of comparing the adverse effects of DOC, pH, and SAR for different soils.

The soil samples that we collected from the pH 7 synthetic water treatment at SAR 4 had the highest aggregate stability, followed by synthetic water at pH 8.0 then wastewater at pH 7, then wastewater at pH 8.0. The position of the treatments in terms of aggregate stability at SAR 7 and SAR 10 was maintained relative to the other treatments at higher SAR (Fig. 7). We noted a very rapid decrease in stability relating to SAR from SAR 4 to SAR 7 and from SAR 7 to SAR 10. The essentially parallel lines in Fig. 7 confirm the lack of an interaction between pH and water type (synthetic or wastewater) across the three SAR levels. This response is similar to that observed in the infiltration measurements (Fig. 4a, b). Similar results are seen when we compare the wastewater and the synthetic water treatments. The very sharp decrease in aggregate stability with SAR was also observed by Ruiz-Vera and Wu (2006) for two of the three soils that they examined.

The DI data were obtained using a modification of the Emerson test (Loveday and Pyle, 1973), shown in Fig. 8 for the surface soil (2.5–5.0 cm) of all treatments. The DI increased with increasing SAR and again, the relative ranking was synthetic water at SAR 4 and pH 7.0 being the least dispersive, followed by synthetic water at pH 8.0, wastewater at pH 7.0 and wastewater at pH 8.0, the latter being the most dispersive. Similar relationships occurred at SAR 7 and SAR 10 (Fig. 8). Using Tukey's test, we determined that DI was significantly different for all SAR values for wastewater treatments at pH 7; for pH 8 treatments, the DI of SAR 4 was significantly different from SAR 7 and 10. For synthetic waters at pH 7 and 8 SAR 4 was significantly different from SAR10. Differences between synthetic waters and wastewaters using the *t*-test were significant ($p < 0.05$) for most SAR and pH treatments. Differences in the response to pH were not significant at $p < 0.05$ except for wastewater at SAR 4 and pH 7. All other treatments were significant at $p < 0.15$.

The effect of ESP on DI in distilled water has been reported by Loveday and Pyle (1973) for two Australian soils. They observed increasing DI with increasing ESP and fitted a regression line where $ESP = 1.15(DI) - 3.24$. If we assume that at a SAR less than 30, ESP $\approx$ SAR (US Salinity Laboratory Staff, 1954), our expected DI using their relationship and using the soil obtained from the prepared water treatments at pH 7.0 would be 6.3, 8.9, and 11.5 for SAR 4, 7, and 10, respectively. This is in good agreement with our results for the Arlington sandy loam soil: a DI of 9.2, 10.5 and 12.2 at pH 7.0 for SAR of 4, SAR 7, and SAR 10, respectively. Using the slope of the ESP–DI relationship of Loveday and Pyle (1973), the increase in DI associated with increase in pH to 8.0 in our treatments is equivalent to an increase in SAR (or ESP) of 1.3, whereas the effect of the wastewater at pH 7.0 on our soil is equivalent to an increase in SAR of 1.7. The combined effects of pH 8 and wastewater correspond to an increase of SAR of 3.7. The relatively parallel relationship between DI and either SAR and pH effects in our data (Fig. 8) suggest no interaction between the effect of pH and/or organic matter and the effect of SAR. The changes in DI with changes in SAR are in agreement with the effects of pH and DOC on infiltration and aggregate stability, as discussed above.

CONCLUSIONS

The results of this study indicate that short-term studies with several infiltration events do not represent the longer-term consequences of irrigating with degraded water. After 153 d of irrigation with wetting and drying cycles, we determined that the DOC in treated wastewater caused a decrease in infiltration rate, a decrease in aggregate stability, and an increase in dispersion relative to prepared waters of the same composition but free of DOC. The effect of the DOC was equivalent to an increase in SAR of 2 to 3 units. The SAR of typical wastewaters is high enough to cause loss of infiltration and increasing SAR had a detrimental effect on infiltration and soil physical properties. The pH of the water also has to be considered when evaluating their suitability for irrigation. We determined that an increase in pH of irrigation water from 7.0 to 8.0, the latter being a value equal or lower than that typical of treated wastewater in arid regions, reduced infiltration and aggregate stability, but increased soil resistance, with and without DOC and for all three SAR values examined. Our recommendations for the sustained use of wastewaters for irrigation are that the pH be lowered to below 8.0 and the SAR be reduced to below 4, possibly by addition of acid and a Ca source.

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