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Evaluation of the Impact of Different Soil Salinization Processes on Organic and Mineral Soils

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Abstract Soil salinization is a worldwide problem of which secondary salinization is increasingly more frequent, threatening agricultural production. Salt accumulation affects not only plants but also the physio-chemical characteristics of the soil, limiting its potential use. Climate change will further increase the rate of salinization of soil and groundwater as it leads to increased evaporation, promotes capillary rise of saline groundwater as well as increased irrigation with brackish water. Episodic seawater inundation of coastal areas is likely to increase in frequency as well. This work analyzed three types of salinization: seawater inundation (by irrigating soils with a 54 dS m^{-1} NaCl solution), saline groundwater capillary rise (soil contact with a 27 dS m^{-1} NaCl solution), and irrigation with two types of brackish water with different residual sodium carbonate (RSC). Two soils were used: a mineral soil (7.0 % clay; 0.7 % organic matter) and an organic soil (2.7 % clay; 7.4 % organic matter). The tested soils had different resilience to salinization: The mineral soil had higher sodium adsorption ratio (SAR) due to low levels of calcium + magnesium but had higher leaching efficiency and more limited effects of RSC. The organic soil however was more prone to capillary rise but seemingly more structurally stable. Our results suggest that short-term inundation with seawater can be mitigated by leaching although soil structure may be affected and that capillary rise of brackish groundwater should be carefully monitored. Also, the impact of irrigation with brackish water with high RSC can be inferior in soils with higher exchangeable acidity.

Keywords Soil salinization · Seawater inundation · Brackish irrigation · Capillary rise

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1 Introduction

Soil salinization is a worldwide problem threatening agricultural production; therefore, limiting its expansion is needed in order to support the projected increase in human population. As much as 10 billion hectares of soil are, in varying degrees of severity, already affected by salinization (Yensen and Biel 2006) with a potential increase of up to 16 % per year (Aydemir and Sünger 2011). Secondary salinization due to poor irrigation practices is responsible for 24.6 to 61.5 % of all irrigated salt affected soils (Pitman and Läuchli 2004; Mateo-Sagasta and Burke 2012), as well as for an estimated 25 % of all saline groundwater (Weert et al. 2009).

Excess salt accumulation affects not only plants (Aslam et al. 2011) but also the chemical and physical characteristics of the soil. Excessive concentrations of sodium (expressed by sodium adsorption ratio (SAR) or exchangeable sodium percentage (ESP)) affect soil structural stability and particle aggregation due to slaking, swelling, and dispersion effects. This leads to hydraulic and drainage problems which may culminate in water logging and lateral dispersion of salts (Qadir and Schubert 2002). High salinity also affects soil pH and nutrient concentrations as well as heavy metal mobility and bioavailability (Khodaverdiloo and Taghlidabad 2013).

Climate change will further increase the rate of salinization of both soil and groundwater. There are several mechanisms by which climate change can impact soil salinity, namely (1) the increased occurrence of seawater inundation in coastal areas (either episodic or permanent), (2) the increased capillary rise of saline groundwater due to increased evaporation, and (3) the expansion of irrigated lands and the increasing use of poor quality brackish water as a response to the scarcity of freshwater sources (Szabolcs 1974; Jesus et al. 2015). These mechanisms can be interconnected in numerous ways. For instance, seawater inundation can directly impact soil but also leach and/or intrude into a groundwater aquifer turning it brackish, and also, either through capillary rise (Chagué-Goff et al. 2014) or use in irrigation affect soil salinity (Violette et al. 2009) (Rengasamy 2006). In addition, irrigation with brackish water can degrade groundwater resources which, in turn, could be used again as irrigation water and thusly potentially trigger a vicious cycle at a regional level (Schofield et al. 2001).

All of these mechanisms congregate mostly in coastal areas, particularly in dry or seasonally dry climates where groundwater levels are usually shallow. However, although salinization of groundwater is certainly an unfavorable process, it seems inevitable that irrigation with this poor quality water will have to be faced as a potential mitigation strategy for climate change effects due to the decrease in fresh water resources.

It is clear, therefore, that an assessment is needed of how these increasingly common salinization processes affect soil quality. However, the periodic and short-term seawater inundation of agricultural and coastal soils is rarely analyzed in the literature, except in the most extreme events such as tsunamis (McLeod et al. 2010; Yoshii et al. 2013), or is focused on its negative impact on coastal wetlands (Kirwan and Guntenspergen 2012) rather than on the physio-chemical impacts on agricultural soils. Regarding saline water capillary rise, there are some works that evaluate and model (Jorenush and Sepaskhah 2003) this phenomenon, but few analyze the specific ion distribution over time and depth. Concerning brackish water irrigation, studies rarely consider the effect of residual sodium carbonate (RSC) (Prasad et al. 2001; Minhas et al. 2007), and there is a lack of consideration of different soil types and characteristics, which might affect the soil response to varying types of brackish water irrigation.

Finally, the use of leaching with poor quality water to control or remediate salinized soils has been thoroughly tested and applied in the field (Murtaza et al. 2009), but the impact of the application of this low-quality water on soil quality has yet to be fully explored.

The objectives of this work are to simulate three different types of salinization processes (seawater inundation, saline groundwater capillary rise, and irrigation with two brackish waters of different composition) and to evaluate their respective effects on the quality of two types of soils: a mineral and an organic soil.

2 Material and Methods

2.1 Soil Characteristics

This study involved the use of two different soils: a clear white soil, herein denoted as mineral soil (MS), and a dark brown soil, herein denotedas organic soil (OS). These two soils were specifically chosen due to their differences in organic matter and particle size compositions. The soils were air dried before being stored in the dark at room temperature. Representative samples for analysis were collected based on the Portuguese norms NP EN 932-1 and 932-2 2002 for sample collection and reduction techniques. The soils were sieved according to ASTM norms using the following sieve sizes: 25, 20, 12.5, 10, 8, 6.3, 5, 4, 2, 1 mm and 500 and 250 μm. Wet sieving, due to the high organic content of the OS, was avoided, and instead, the fraction under 250 μm was analyzed by a laser diffraction particle size analyzer (Malvern MasterSizer Model MS2000). A combination of agitation in water and ultrasound was used to disperse microaggregates, instead of using chemical dispersant and wet sieving. Texture was then assessed based on the feret diagram.

Soil density was calculated using a pycnometer (for aggregates between 0.063 to 31.5 mm) using the method described in NP EN 1097-6 2003, while bulk density was calculated using a graduated beaker of 100 mL. Soil saturation percentage was obtained by method 27a of U.S. Salinity Laboratory (1954), derived from a saturated soil paste prepared by method 2 of the same reference. The qualitative lime content test was prepared based on method 23a effervescence test of U.S. Salinity Laboratory (1954). Loss on ignition tests were performed in accordance to Heiri et al. (2001). Cation exchange capacity (CEC) was tested using the methylene blue adsorption method described in Aprile and Lorandi (2012).

Soil water extractions were performed in accordance to U.S. Salinity Laboratory (1954) for saturated soil paste of 1:1, 1:2, or 1:5 soil to water ratios, followed by filtration with Whatman® 2.5 μm filter paper. To convert from soil to water ratios to saturated values, the following formula was used: denominator value of ratio (for example, 5 in a 1:5 extraction)$\times 100/$saturation percentage.

Electrical conductivity (EC) was determined with a WTW Tetracon® 325 conductivity electrode while pH was determined using a WTW pH electrode SenTix 21. pH was determined using ratios of 1:1 soil to water extractions, 1:2 soil to 0.01 M $CaCl_2$ solution, and 1:1 soil to 1 M KCl solution. Calcium + magnesium were determined simultaneously following EPA method (# 130.2 Hardness, Total (mg L^{-1} as $CaCO_3$) (Titrimetric, EDTA). Sodium was analyzed using a HANNA FC 300 B Na^+ electrode connected to a HANNA HI 4214 benchtop measuring unit after daily calibration. Important soil physical and chemical characteristics are shown in Table 1.

Screening tests for possible heavy metal contamination were done using of a Portable Analytical X-Ray Dispersive Energy Fluorescence Spectrometer (Innov-X System), and no abnormal level of metals was detected in either type of soil (data not shown).

2.2 Experimental Setup

Soils were filled into acrylic columns with a height of 25.5 cm and an internal diameter of 5 cm. The bottom of each column was packed with glass wool to a height of 5 cm for drainage. The soils were subsequently added to the columns up to 10 cm height above the glass wool, allowing them to fall freely to avoid undue compaction and to retain bulk densities similar to the original values referred in Table 1. The columns were perforated at the bottom to allow for drainage and were set vertically in metal stands at 10 cm above the lab benchtop for the tests with seawater inundation. Soil-loaded columns were placed in an oven at 40 °C (for the capillary rise tests) or at 60 °C (for the irrigation tests). Although used for practical reasons, these temperatures simulate salt accumulation at a much faster rate than what happens in the field. Nevertheless, these temperatures remained within acceptable values according to Berglund et al. (2010).

2.3 Seawater Inundation

The objective of this experiment was to simulate the effects of seawater inundation into the two soils (MS and OS, each one having different physical and chemical characteristics) to see how these soils responded. Two sets of columns were prepared in triplicate for each type of soil. Before testing, the soils were wetted with distilled water to saturation and allowed to drain. This was done in order to avoid hydrophobicity problems from using dry soils and corresponding drainage problems, as observed in preliminary tests. The experiment

Table 1 Characterization of the organic soil (OS) and mineral soil (MS) used in this work

	OS	MS		OS	MS
Sand (%)	75.29	60.0[a]	EC (dS m^{-1})	0.90	1.28
Silt (%)	22.61	28.0[a]	Ca^{2+}+Mg^{2+} (meq L^{-1})	3.05	4.18
Clay (%)	2.11	7.0[a]	Bulk density (kg L^{-1})	1.10	1.16
Texture	Loamy sand	Sandy loam	Soil density (kg L^{-1})	2.14	2.68[a]
Saturation percentage (%)	33.0	46.3	pH water (1:1)	7.22	7.49
Lime content	Not detected	Not detected	pH 0.01 M $CaCl_2$ (1:2)	5.76	5.88
CEC (meq100 g^{-1})	42.3	179.7	pH 1 M KCl (1:1)	4.80	4.09
OM (%)	7.4	0.7[a]	Exchangeable acidity (meq100 g^{-1})	0.25	1.85

[a] Results obtained from Carvalho, M. 2014, unpublished data

was divided in two phases: Phase 1 salinized the soil using a saline solution (herein referred to as samples of "soil after salt addition") and the saline leached solutions (herein referred to as "salt solution leached") were collected and tested, and phase 2, in which the salinized soil was leached twice with distilled water.

For the salt addition phase, 2 pore volumes of a prepared solution using NaCl (Merck; ≥99.5 %) at seawater salinity level (EC=54 dS m^{-1}, pH 6.02, sodium levels: 13,950 mg L^{-1}) were added. Due to differences in porosity, the volumes added were different: 157.6 and 235.2 mL, for OS and MS, respectively. The leachate obtained was collected and analyzed. Contact time between the soil and saline solution was dependent on the infiltration rate of each replicate and therefore varied in each column. The EC and pH of the leachate and soil samples were assessed, along with calcium + magnesium and sodium levels.

During the second phase of the test (leaching phase), leaching with distilled water was performed to simulate heavy rain events. Distilled water was added at 2 pore volumes (157.6 mL for the OS and 235.2 for the MS). The volume of distilled water was allowed to leach through the soil once, and then, leaching was repeated with the same volume (as listed above) of new distilled water. Both the first and the second washing leachates were collected and analyzed at the end of the experiment. The EC and pH of the leachates were assessed, along with calcium + magnesium and sodium levels.

2.4 Capillary Rise of Saline Groundwater

A test designed to simulate soil salinization due to capillary rise was conducted using the same experimental setup described above but with a soil height of approximately 20 cm in each column. Two sets of columns were prepared in triplicate with each soil type and were placed in a container with a 27 dS m^{-1} saline solution (prepared with NaCl (Merck; ≥99.5 %), at a height corresponding to that of the glass wool in the columns. The value of salinity used was similar to the one reported for saline groundwater in Fan et al. (2012). The level of water in the container was maintained by the addition of the 27 dS m^{-1} saline solution whenever needed. If there was no visible capillary rise to the top of the 20 cm soil height in the columns, the unsaturated soil was removed and not replaced.

Initially, column top soil samples were collected at different times (every 2 days) in order to assess the evolution of salinity over time and tested for moisture content, EC, pH, and calcium + magnesium levels. After 5 days, all of the soil was removed from the columns but carefully separated into two soil depths. Samples of 0–10 and 10–20 cm depths were acquired to assess variability of saline groundwater effects with depth, with EC, pH, calcium + magnesium, and sodium analyses performed.

2.5 Irrigation with Brackish Water

The objective of this test was to simulate a moderately severe case of irrigation of two types of soils (MS and OS) with brackish water (secondary salinization) and to assess the leaching potential. A column setup similar to the one previously described was used. This test was also composed of two phases: the salt addition phase, in which two different irrigation solutions were added to the soils at a rate simulating normal irrigation practices (the volume added was just enough to compensate for evaporation and limited to no leaching), and a leaching phase, in which the same irrigation solution was added in excessive volume to promote leaching.

The irrigation solutions used have similar EC values, resembling mildly brackish water but with varying RSC. One of the irrigation solutions used was composed of NaCl 1 g L^{-1}, $CaCl_2 \cdot 2H_2O$ 1.325 g L^{-1}, and $NaHCO_3$ 1.85 g L^{-1}, resulting in a theoretical EC of 5.73 dS m^{-1} (actual EC of 5.20 dS m^{-1} or 8.8 % lower), SAR of 13.08, and RSC of 4.00. The other irrigation solution was composed of NaCl 2.3 g L^{-1} and $CaCl_2 \cdot 2H_2O$ 1.325 g L^{-1}, resulting in a theoretical EC of 5.77 dS m^{-1} (actual EC of 6.10 dS m^{-1} or 5.7 % higher), SAR of 13.21, and RSC <0. All chemicals used were p.a. grade. Irrigation was performed by adding a few drops at a time to each triplicate column, more closely resembling surface drip irrigation than ponding as an irrigation method.

In the first phase (or the salt addition period), 67 mL of each solution were initially added to all soil columns (approximately 75 % of the saturation percentage of the MS) and three subsequent irrigations were made during 4 days to compensate for evaporation. Salt buildup was monitored by small sample collections (approximately 7 g of soil at 1:5 soil to water extraction) which were then used to determine EC and pH, as well as soil moisture (data not shown). Final soil samples were collected (65 g) to evaluate EC, pH, calcium + magnesium, and sodium concentration (salt addition phase soil samples).

In the second phase of this test, leaching was performed at 1.2 pore volumes with the two different solutions used above for irrigation (which had similar EC and SAR, but with different RSC values). This resulted in the addition of 96 mL of irrigation solution in the OS and 145 mL in the MS. Soil samples were collected (designated "soil after leaching"), as well as the leaching solutions, and monitored for pH, EC, calcium + magnesium, and sodium.

2.6 Statistical Analysis

All statistical tests were performed using the Stastistica 8.0 software (StatSoft, Inc, Tulsa, USA). Normality and homogeneity of variances were tested by Shapiro–Wilk test and by Levene's test and Brown–Forsythe test, respectively. *T* tests were performed for comparison between two independent sets of data at α=0.05. When normality was not verified, Mann–Whitney *U* test was used (with respective formula for effect size, *r*). The tests performed with Mann–Whitney *U* are indicated with asterisks in the tables and use α=0.10. Dependent paired *t* tests were performed to compare soil samples before and after leaching.

3 Results

3.1 Seawater Inundation

The results obtained for the seawater inundation test are shown in Table 2. It can be seen that the EC values of both soil samples after salt addition are similar to that of the salt solution used, as well as the respective sodium levels. Despite this, the SAR level in the MS is almost double that of the OS, likely due to differences in calcium + magnesium concentrations between the soils. Despite having higher initial values of soluble calcium + magnesium and pH, the MS was the most affected by the simulated salinization process. In this phase, the EC of the salt solution leached is within expected levels, considering the dilution with the distilled water that was previously added to the soils (which would result in predicted values of 23.4 and 24.5 dS m^{-1} for the MS and OS, respectively).

Similar trends in sodium and calcium + magnesium values can be observed in these leachates, but calcium + magnesium is likely to be higher in the OS leachate. Nevertheless, in this case, the values were not statistically different due to the presence of a large outlier. In addition, there is a reduction in pH in both cases (due to the application of simulated seawater), but it is significantly higher in the solution obtained from the MS (p=0.006), with a large size effect of over 0.5.

The distilled water leaching phase, repeated twice to simulate heavy rain, was able to reduce sodium levels in both soils. However, it also leached out calcium + magnesium, leaving these soils more vulnerable to a rapid increase in SAR if a salt addition event reoccurred. As expected, leaching of calcium + magnesium was more visible in the OS due to its higher initial concentration in these ions.

A simplified mass balance demonstrates that the value for total salts removed by both leaching events in the MS was, on average, 57.5 %, of which over 88 % occurred in the first leaching event. In the OS, this value is slightly lower, with 50.6 % of salts removed, 85 % of which occurred in the first leaching event. In the case of sodium, there was a 53 % removal in the MS (93 % in the first leaching event) and 38.7 % in the OS (88 % in the first leachate).

3.2 Saline Groundwater Capillary Rise

This test was designed to simulate the effects of capillary rise of saline water through the soil profile, and a summary of the results obtained can be found in Table 3.

Due to its physical characteristics, the OS had a higher capillary fringe (20 cm) than the MS (17 cm) and reached the maximum level of soil depth tested (20 cm), although it is possible that it would have been higher if more soil height was available. These soil samples were small and tested at high dilution rates, which explains the high variability observed in this phase, in particular for calcium + magnesium results. Nevertheless, in both extracts, both the EC and calcium + magnesium values were superior in the OS than in the MS, and an increase over time is clearly visible, as expected.

When all of the soil was removed from the columns and analyzed at two different depths, several differences between the soil types in terms of EC, sodium, and calcium + magnesium levels were observed in the top 10 cm of soil. The OS, due to its higher capacity for capillary rise, showed higher values for all variables, except SAR. Lack of a statistically significant difference between the two soils for SAR values (despite large differences in calcium + magnesium, as well as in

Table 2 Seawater inundation tests

	Salt addition phase							
	Soil after salt addition				Salt solution leached			
	MS	OS	Statistics		MS	OS	Statistics	
			P value	*r*			*p* value	*r*
EC (dS m^{-1})	56.6±12.4	55.5±24.3	0.823	0.01	29.3±22.7	24.1±15.4	0.297	0.26
pH	4.7±2.5	7.0±2.8	0.000	0.99	4.1±2.9	5.73±12.3	0.006	0.87
Ca^{2+}+Mg^{2+} (meq L^{-1})	2.2±17.9	9.9±55.3	0.071	0.60	19.4±5.7	21.9±39.0	0.631	0.06
Na^{+} (meq L^{-1})	548.8±13.3	490.9±27.3	0.540	0.10	271.5±19.5	215.4±22.2	0.225	0.34
SAR	537.7±22.5	245.3±44.4	0.036	0.71	87.2±17.4	70.9±47.8	0.491	0.13
	Leaching phase							
	First distilled water leached solution				Second distilled water leached solution			
EC (dS m^{-1})	21.9±18.7	19.8±44.0	0.719	0.04	2.9±35.5	3.49±33.3	0.587	0.08
pH	4.0±2.1	5.8±5.2	0.001	0.96	5.0±10.4	6.9±5.5	0.007	0.86
Ca^{2+}+Mg^{2+} (meq L^{-1})	5.6±5.7	11.5±28	0.035	0.71	0.3±27.7	1.7±103.6	0.081*	−0.71
Na^{+} (meq L^{-1})	205.3±34.1	138.6±43.1	0.278	0.28	16.3±38.9	18.6±29.8	0.658	0.05
SAR	121.7±31.0	57.0±35.0	0.058	0.63	43.3±49.3	24.2±22	0.206	0.36

Salt addition phase refers to the addition of a 54 dS m^{-1} solution and monitoring soil samples and leached salt solutions. Leaching phase refers to washing with distilled water after salt addition and assessment of leached solutions (*Mann–Whitney *U* test, all remaining statistical analysis use independent *t* tests; r indicates effect size; the results are expressed as units±% relative standard deviation)

sodium) may have been due to concurrent increases of sodium and calcium + magnesium in such a way that it prevented any difference in the ratio between these concentrations when expressed as SAR.

Table 3 Saline groundwater capillary rise simulation results with two types of soils: *Top soil samples—different times*, refers to capillary rise evolution; *Soil samples—different depths*, refers to assessment of vertical distribution of salts (*Mann–Whitney *U*, all remaining statistical results are independent *t* tests; *r* indicates effect size; the results are expressed as units±% relative standard deviation)

	Top soil samples—different times							
	First extract				Second extract			
	MS	OS	Statistics		MS	OS	Statistics	
			p value	*r*			*p* value	*r*
EC (dS m^{-1})	0.9±35.8	4.7±39.6	0.025	0.75	7.2±39.5	20.9±64.6	0.161	0.43
pH	6.9±11.0	6.3±1.1	0.08*	0.71	5.1±16.7	6.5±2.3	0.055	0.64
$Ca^{2+}Mg^{2+}$ (meq L^{-1})	5.3±31.1	62.2±40.0	0.017	0.80	60.6±43.3	256.0±60.6	0.098	0.54
	Soil samples—different depths							
	0–10 cm				10–20 cm			
EC (dS m^{-1})	12.6±18.9	22±18.6	0.026	0.75	21.5±17.2	36.6±23.2	0.048	0.67
pH	3.5±4.4	5.8±5.3	0.000	0.97	3.5±3.1	6.2±4.0	0.08*	−0.71
$Ca^{2+}Mg^{2+}$ (meq L^{-1})	57.0±7.2	123.6±11.0	0.001	0.94	22.8±25.2	43.1±36	0.100	0.53
Na+ (meq L)	107.0±10.7	141.8±8.7	0.023	0.76	88.7±13.9	102.3±9.1	0.201	0.37
SAR	16.1±24.4	23.8±38.2	0.254	0.31	26.5±13.8	24.7±15.6	0.592	0.08

At a higher soil depth (10–20 cm), the differences for all parameters are less pronounced, certainly because the forced evaporation created in the experimental design was not enough to reach this depth. This would explain lower calcium + magnesium and sodium levels observed at this depth, when compared to the first 10 cm of soil, resulting in the existence of few statistically different cases between soil types.

The pH of the MS, as previously observed in the seawater inundation tests, is significantly lower than the value for the OS at both soil depths.

Further statistical analysis (paired t tests, data not shown in Table 3) comparing different soil depths for the same soil type reveal that only calcium + magnesium content is significantly different between the two soil layers in the OS (p=0.0033; r=0.99), although differences in pH and sodium were close to be nearly significant (p=0.051, r=0.90; p=0.069, r=0.87, for pH and sodium, respectively), while SAR is very similar between soil depths for this soil (p=0.869, r=0.02). Concerning the MS, the same statistical test shows that there are more visible differences at different soil depths, mainly in calcium + magnesium (p=0.008; r=0.98) and SAR (p=0.033; r=0.94), with no other variables assayed having p values close to the 0.05 limit of significance.

3.3 Irrigation with Brackish Water

Irrigation tests with two different brackish water solutions (EC=5 dS m^{-1} but different RSC) were conducted in both MS and OS in order to simulate salt accumulation due to poor irrigation practices. Table 4 summarizes the main results obtained for the two experimental phases, salt addition, and leaching.

In the salt addition phase of the test, the irrigation of the MS and OS with either RSC or no RSC solutions yielded similar results. Sodium levels were 34 and 61 % higher (MS and OS, respectively) when irrigation with no RSC was performed. However, as expected, calcium + magnesium concentrations were the parameter that changed more significantly when irrigation with no RSC was applied, with increases of 605 and 654 % for MS and OS, respectively. This was accompanied by an EC increase of 79 and 76 %, respectively, when compared with irrigation with high RSC. It is worthy to note that calcium + magnesium levels increase in the MS irrigated with RSC after leaching.

In the leaching phase, however, results obtained for the RSC and no RSC leachate solutions varied considerably. EC and calcium + magnesium levels were statistically different in both soils for samples with high RSC versus no RSC, although only the amount of sodium removed was significant in the MS. Since leaching occurred at such a different rate between soil types, with different removal of contaminants, the soil samples acquired after leaching reflected that difference: In the MS, SAR was not statistically different between high RSC and no RSC treatments, while in the OS, only sodium content was not statistically different between treatments. Despite this, the trends are similar to the results obtained before leaching: EC, with calcium + magnesium and sodium are higher after irrigation with no RSC solution, while pH and SAR are lower when compared with levels obtained after irrigation with high RSC water.

Furthermore, a statistical comparison between soil types (data not shown in Table 4) was performed. In the salt addition phase of the test, only the pH is significantly different when comparing the soil types with no RSC (p=0.0005, r=0.96), which may indicate that the increase of pH due to carbonate addition compensates for the potential pH drop normally seen in the MS. In the leachate solution samples, every single parameter was different in the MS treated with no RSC solution when compared with the OS in the same situation. For the high RSC solution, sodium and SAR levels in the leachate were not statistically different between the different soil types. Comparing the two soils, based on the soil samples obtained after leaching, it can be observed that the EC (p=0.02, r=0.76 for high RSC and p=0.049, r=0.66 for low RSC) and pH (p=0.0002, r=0.98 for high RSC and p=0.0020 r=0.93 for low RSC) are statistically different in both low and high RSC level treatments.

4 Discussion

This study examined the impact of three different but often interconnected salinization processes: short-term seawater inundation, capillary rise of brackish groundwater, and irrigation with brackish water of varying qualities. The impacts of these processes on two different soil types (MS and OS), as well as the implications of the obtained results, are analyzed herein.

Detailing the soil response to specific tests, the MS showed a higher SAR value than the OS after initial salt

Table 4 Results of irrigation with brackish water tests: *Salt addition phase* refers to the utilization of two different solutions ("RSC" and "no RSC" are solutions with or without residual sodium carbonate); *Leaching phase* refers to washing the soils with the same solutions as above (*Mann–Whitney U, all remaining statistical results are independent *t* tests; *r* indicates effect size, results expressed as units±% relative standard deviation)

	Salt addition phase							
	MS				OS			
	RSC	no RSC	Statistics		RSC	no RSC	Statistics	
			p value	*r*			*p* value	*r*
EC (dS m^{-1})	4.8±5.5	8.6±17.5	0.012	0.83	6.1±25.3	10.7±8.9	0.012	0.83
pH	7.2±3.9	5.3±2.4	0.000	0.97	7.4±2.5	7.1±4.0	0.194	0.38
Ca^{2+}+Mg^{2+} (meq L^{-1})	7.4±4.1	52.2±26.9	0.005	0.88	9.5±19.1	71.6±21.6	0.002	0.92
Na^{+} (meq L^{-1})	25.6±2.4	34.3±20.0	0.093	0.55	29.3±14.9	47.2±23.8	0.061	0.62
SAR	13.3±3.0	6.7±7.1	0.081*	0.71	13.4±6.0	8.00±27.6	0.016	0.80
	Leaching phase—leachate solutions							
EC (dS m^{-1})	4.9±5.3	6.4±2.3	0.001	0.95	6.16±6.7	7.7±4.4	0.008	0.86
pH	5.0±3.8	4.8±1.9	0.134	0.47	6.1±1.7	6.0±2.2	0.157	0.43
Ca^{2+}+Mg^{2+} (meq L^{-1})	41.6±3.7	66.4±3.2	0.081*	−0.72	58.9±18.0	82±4.3	0.023	0.76
Na^{+} (meq L^{-1})	34.6±7.7	41.2±1.2	0.014	0.82	31.4±16.3	34.6±10.6	0.424	0.17
SAR	7.6±8.6	7.2±0.50	0.190*	0.53	5.9±23.1	5.4±12.9	0.633	0.06
	Leaching phase—soil after leaching							
EC (dS m^{-1})	3.4±3.2	5.1±6.5	0.001	0.95	3.8±4.1	6.5±12.9	0.005	0.88
pH	6.8±1.3	5.5±6.1	0.003	0.91	7.6±0.9	7.1±2.1	0.004	0.90
Ca^{2+}+Mg^{2+} (meq L^{-1})	15.3±39.6	41.5±7.0	0.003	0.92	8.7±9.3	42.4±11.4	0.000	0.97
Na^{+} (meq L^{-1})	22.0±12.7	28.0±4.1	0.026	0.75	27.3±13.5	31.1±15.3	0.343	0.22
SAR	8.5±35.7	6.2±1.5	0.663*	0.18	13.2±18.0	6.7±11.7	0.011	0.83

addition in the seawater inundation test. However, leaching with distilled water (simulating heavy rain) removed less calcium + magnesium and more sodium in MS, resulting in a larger SAR reduction (indicated by large values of SAR in the two leachates) with an estimated value of SAR similar to that of the OS at the end of the test (data not shown). These results may be due to the lower infiltration rate already mentioned for the MS, which increases the contact time of both the salt solution and the leaching solution. However, if this infiltration rate is significantly reduced, it may lead to higher salt content due to evaporation, and leaching would become increasingly less effective. An example of the impact of this situation can be found in (McLeod et al. 2010) for instance, where drainage problems prevented the natural restoration of adequate soluble salts levels in soils affected by the 2004 tsunami, despite the large amount of available rainfall for leaching.

In this test, the first leaching event removed the largest quantity of salts, as previously reported by other authors (Qadir et al. 2003): In the MS, the first leaching removed an estimated 14 kg of dissolved salts and 4.7 kg of sodium for every cubic meter of leaching water applied, while the second leaching event only removed 1.9 kg of total salts and 0.38 kg of sodium per applied cubic meter. In the OS, a similar reduction was found (12.7 kg of total salts and 3.2 kg of sodium in the first and 2.2 kg of total salts and 0.43 kg of sodium per applied cubic meter in the second leaching event). However, leaching may not always be desirable, since nutrients may also be leached away as indicated in our experiment by the values of dissolved calcium + magnesium that were effectively reduced by 36 and 82 % in the MS and OS, respectively. Natural leaching due to heavy rain can lead to further salinization of groundwater resources after inundation events like tsunamis (Violette et al. 2009). To improve leaching and prevent water logging, the installation of a drainage system may be required (Ritzema et al. 2008). In extreme cases where inundation is frequent, biosaline agriculture of halophytes with high salinity and flooding tolerance may be an option (Glenn et al. 2013).

In the capillary rise test, the capillary fringe was higher in the OS, reaching the top 20 cm, while in the MS, it only reached 16 or 17 cm. This is likely due to the different pore sizes of both soils (macropores but especially micropores). Furthermore, organic content is likely to aid in capillary rise. This can be seen in other studies in which organic matter addition increased the extent of capillary rise (Eusufzai and Fujii 2012) as well as water retention (Pandey and Shukla 2006). In the soils tested, the accumulation of calcium + magnesium observed on the surface layer of the soil is caused by the addition of sodium, which displaced the calcium + magnesium from the exchange sites deep in the soil. Afterwards, the capillary rise transported these ions upwards (since they are now dissolved in the soil solution) onto the soil surface, where constant evaporation and compensatory rise of saline water in cycles further contributed to their accumulation. Since the OS has more total calcium + magnesium, the expected larger concentration of these ions on the soil surface was in fact observed. Ibrahimi et al. (2013) used a similar lysimeter setup and observed the same salt accumulation pattern due to capillary rise, but they additionally observed that groundwater table variations may lead to further accumulation, as opposed to a stable groundwater level (as simulated in our work). Potential mitigation solutions for capillary rise include the control of the water table by planting deep rooted and salt tolerant trees with high water consumption (Herron et al. 2003) and by applying physical capillary barriers made up of coarse materials, which can retard capillary rise (Ityel et al. 2014).

Irrigation with brackish water with high levels of RSC and salinity was also tested in this work with results that agree with those that can be found in the literature. Irrigation with brackish water with high RSC resulted in increased soil EC, pH, and SAR, as well as in reduced infiltration rates as seen in Choudhary et al. (2010). In Minhas et al. (2007), similar trends were found: ESP increased likely due to the precipitation of calcite and gypsum. However, in an experiment in which the saturated soil paste extract was concentrated multiple times through evaporation, some sodium precipitation was detected, possibly contributing to ESP reduction. This fact may also help to explain why there was less sodium in soils irrigated with a higher RSC solution than in our work, as some portion of sodium might have precipitated.

Based on our results obtained after leaching, it is not clear whether the RSC affected salt leaching or if leaching increased the solubility of potentially precipitated calcium or magnesium carbonates, which resulted in a simultaneous reduction in EC levels. However, irrigation with RSC seems to have negatively impacted sodium mobility, which points more strongly to a problem of infiltration rate rather than precipitation.

Application of organic matter could be employed to decrease pH and dissolve calcium and/or magnesium carbonates (Qadir et al. 2005), therefore limiting the negative effects of the applied irrigation water with high levels of RSC. However, this organic matter would also increase the cation exchange capacity of the soil and likely lead to further sodium accumulation. Furthermore, the positive impact of organic matter is only likely to be significant when the soil is calcareous, and pH reduction mobilizes significant amounts of calcium to assist in the displacement of sodium on the cation exchange sites of the soil (Choudhary et al. 2011). In the case of the two soils tested in this work, the addition of organic matter would not make sense in the case of OS. For the MS, the exchangeable acidity would be much more significant in reducing pH than the addition of organic matter.

Comparing the initial irrigation solutions with the resulting leachate, the latter actually had a lower SAR value than the original. Consequently, a reuse of this leachate could prove beneficial in different aspects: (1) It would ensure that salt leaching, as well as nutrient and water loss, into groundwater is minimized (Glenn et al. 2009); (2) in the case of the MS leachate, the low pH could prove to be beneficial in mobilizing calcium, particularly in calcareous soils (Sadiq et al. 2007); (3) the required drainage system would also prevent water logging due to flooding and/or structural damages (as seen in the MS) caused by sodium (Ritzema et al. 2008). However, the main problems associated with this potential approach are the high cost of an adequate drainage system and that the leaching rate would likely have to be less frequent than the one simulated in this work. This would result in leachates with higher EC than observed. Several options could be chosen to ensure good soil quality for agricultural production depending on the level of the resulting EC. These include traditional physical desalinization treatments (Abulnour et al.

2003; Bunani et al. 2015), a focus on salt tolerant plant production or fiber crop production (Barbosa et al. 2015) or even, if appropriately managed and studied, mixing the saline leachate with high-quality water to increase available water quantity at the expense of quality (Barnes 2012).

Ultimately, the different responses of the two soils to salinization were related to their initial characteristics throughout all of the three tests performed: The high exchangeable acidity and low exchangeable bases of the MS helped to explain the pH reduction and higher SAR developed with salt addition, while the OS, with its high organic content, had qualitatively higher structural stability, as well as higher capillary strength and lower SAR values.

5 Conclusions

In this work, the two different soils tested had different responses to the salinization processes, with important consequences to potential mitigation strategies. The MS was more prone to SAR accumulation due to low levels of native calcium + magnesium but had higher leaching efficiency and more limited effects of RSC due to its high exchangeable acidity. The OS, on the other hand, was more prone to capillary rise but seemingly was more structurally stable than the MS.

Short-term inundation with seawater can be partially mitigated by leaching with high-quality water and/or rainfall whenever it is available, although soil fertility and structure may be negatively affected. Capillary rise of brackish groundwater should also be carefully monitored in coastal agricultural areas, where perched water tables are frequent. In soils not yet affected by salinity, sodium and calcium + magnesium were mobilized deeper in the soil and transported upwards. This led to the accumulation of these cations at the surface of the soil and/or at the limit of the capillary fringe as demonstrated in this work.

In the tested conditions, irrigation can be maintained within acceptable limits of EC and SAR for salt tolerant crops using brackish water with high RSC, provided that frequent leaching is maintained as well. Soils with high exchangeable acidity are able to counteract the negative effects of carbonates in the irrigation water, although this capacity is likely limited over time.

Future research on soil salinization should focus on remediation options and how their efficiency might be affected by different soil types.

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References

Abulnour, A. G., Sorour, M. H., & Talaat, H. A. (2003). Comparative economics for desalting of agricultural drainage water (ADW). *Desalination, 152*(1–3), 353–357. doi:10.1016/S0011-9164(02)01083-4.

Aprile, F., & Lorandi, R. (2012). Evaluation of cation exchange capacity (CEC) in tropical soils using four different analytical methods. *Journal of Agricultural Science, 4*(6), 278.

Aslam, R., Bostan, N., Nabghae, A., Maria, M., & Safdar, W. (2011). A critical review on halophytes: salt tolerant plants. [Review]. *Journal of Medicinal Plants Research, 5*(33), 7108–7118. doi:10.5897/JMPRx11.009.

Aydemir, S., & Sünger, H. (2011). Bioreclamation effect and growth of a leguminous forage plant (*Lotus corniculatus*) in calcareous saline sodic soil. *African Journal of Biotechnology, 10*(69), 115571–115577.

Barbosa, B., Costa, J., Fernando, A. L., & Papazoglou, E. G. (2015). Wastewater reuse for fiber crops cultivation as a strategy to mitigate desertification. *Industrial Crops and Products*. doi:10.1016/j.indcrop.2014.07.007.

Barnes, J. (2012). Mixing waters: the reuse of agricultural drainage water in Egypt. *Geoforum, 57*, 181–191. doi:10.1016/j.geoforum.2012.11.019.

Berglund, Ö., Berglund, K., & Klemedtsson, L. (2010). A lysimeter study on the effect of temperature on CO2 emission from cultivated peat soils. *Geoderma, 154*(3–4), 211–218. doi:10.1016/j.geoderma.2008.09.007.

Bunani, S., Yörükoğlu, E., Yüksel, Ü., Kabay, N., Yüksel, M., & Sert, G. (2015). Application of reverse osmosis for reuse of secondary treated urban wastewater in agricultural irrigation. *Desalination* (0), doi:10.1016/j.desal.2014.07.030.

Chagué-Goff, C., Wong, H. Y., Sugawara, D., Goff, J., Nishimura, Y., Beer, J., et al. (2014). Impact of tsunami inundation on soil salinisation: up to one year after the 2011 Tohoku-Oki Tsunami. In Y. A. Kontar, V. Santiago-Fandiño, & T. Takahashi (Eds.), *Tsunami Events and Lessons Learned* (Vol. 35, pp. 193-214, Advances in Natural and Technological Hazards Research): Springer Netherlands.

Choudhary, O. P., Ghuman, B. S., Dhaliwal, M. S., & Chawla, N. (2010). Yield and quality of two tomato (Solanum lycopersicum L.) cultivars as influenced by drip and furrow irrigation using waters having high residual sodium carbonate. *Irrigation Science, 28*(6), 513–523. doi:10.1007/s00271-010-0211-y.

Choudhary, O. P., Ghuman, B. S., Bijay, S., Thuy, N., & Buresh, R. J. (2011). Effects of long-term use of sodic water irrigation, amendments and crop residues on soil properties and crop yields in rice–wheat cropping system in a calcareous soil. *Field Crops Research, 121*(3), 363–372. doi:10.1016/j.fcr.2011.01.004.

Eusufzai, M., & Fujii, K. (2012). Effect of organic matter amendment on hydraulic and pore characteristics of a clay loam soil. *Open Journal of Soil Science, 2*(4), 372–381. doi:10.4236/ojss.2012.24044.

Fan, X., Pedroli, B., Liu, G., Liu, Q., Liu, H., & Shu, L. (2012). Soil salinity development in the yellow river delta in relation to groundwater dynamics. *Land Degradation & Development, 23*(2), 175–189. doi:10.1002/ldr.1071.

Glenn, E. P., McKeon, C., Gerhart, V., Nagler, P. L., Jordan, F., & Artiola, J. (2009). Deficit irrigation of a landscape halophyte for reuse of saline waste water in a desert city. *Landscape and Urban Planning, 89*(3–4), 57–64. doi:10.1016/j.landurbplan.2008.10.008.

Glenn, E. P., Anday, T., Chaturvedi, R., Martinez-Garcia, R., Pearlstein, S., Soliz, D., et al. (2013). Three halophytes for saline-water agriculture: an oilseed, a forage and a grain crop. *Environmental and Experimental Botany, 92*, 110–121. doi:10.1016/j.envexpbot.2012.05.002.

Heiri, O., Lotter, A., et al. (2001). Loss on ignition as a method for estimating organic and carbonate content in sediments: reproducibility and comparability of results. *Journal of Paleolimnology, 25*(1), 101–110.

Herron, N., Davis, R., Dawes, W., & Evans, R. (2003). Modelling the impacts of strategic tree plantings on salt loads and flows in the Macquarie River Catchment, NSW, Australia. *Journal of Environmental Management, 68*(1), 37–50. doi:10.1016/S0301-4797(02)00230-X.

Ibrahimi, M., Miyazaki, T., Nishimura, T., & Imoto, H. (2013). Contribution of shallow groundwater rapid fluctuation to soil salinization under arid and semiarid climate. Arabian Journal of Geosciences 1-11, doi:10.1007/s12517-013-1084-1.

Ityel, E., Ben-Gal, A., Silberbush, M., & Lazarovitch, N. (2014). Increased root zone oxygen by a capillary barrier is beneficial to bell pepper irrigated with brackish water in an arid region. *Agricultural Water Management, 131*, 108–114. doi:10.1016/j.agwat.2013.09.018.

Jesus, J., Danko, A. S., Fúza, A., & Borges, M.-T. (2015). Phytoremediation of salt affected soils: a review of processes, applicability and the impact of climate change. *Environmental Science and Pollution Research*. doi:10.1007/s11356-015-4205-4.

Jorenush, M. H., & Sepaskhah, A. R. (2003). Modelling capillary rise and soil salinity for shallow saline water table under irrigated and non-irrigated conditions. *Agricultural Water Management, 61*(2), 125–141. doi:10.1016/S0378-3774(02)00176-2.

Khodaverdiloo, H., & Taghlidabad, R. H. (2013). Phytoavailability and potential transfer of Pb from a salt-affected soil to *Atriplex verucifera, Salicornia europaea* and *Chenopodium album*. Chemistry and Ecology, 1-11, doi:10.1080/02757540.2013.861827.

Kirwan, M. L., & Guntenspergen, G. R. (2012). Feedbacks between inundation, root production, and shoot growth in a rapidly submerging brackish marsh. *Journal of Ecology, 100*(3), 764–770. doi:10.1111/j.1365-2745.2012.01957.x.

Malvern MasterSizer Model MS2000. Hydro G, Malvern instruments. UK.

Mateo-Sagasta, J., & Burke, J. (2012). Agriculture and water quality interactions: a global overview. *SOLAW Background Thematic Report - TR08*.

McLeod, M. K., Slavich, P. G., Irhas, Y., Moore, N., Rachman, A., Ali, N., et al. (2010). Soil salinity in Aceh after the December 2004 Indian Ocean tsunami. *Agricultural Water Management, 97*(5), 605–613. doi:10.1016/j.agwat.2009.10.014.

Minhas, P. S., Dubey, S. K., & Sharma, D. R. (2007). Effects on soil and paddy–wheat crops irrigated with waters containing residual alkalinity. *Soil Use and Management, 23*(3), 254–261. doi:10.1111/j.1475-2743.2007.00090.x.

Murtaza, G., Ghafoor, A., Owens, G., Qadir, M., & Kahlon, U. Z. (2009). Environmental and economic benefits of saline-sodic soil reclamation using low-quality water and soil amendments in conjunction with a rice–wheat cropping system. *Journal of Agronomy and Crop Science, 195*(2), 124–136. doi:10.1111/j.1439-037X.2008.00350.x.

Pandey, C., & Shukla, S. (2006). Effects of composted yard waste on water movement in sandy soil. *Compost Science & Utilization, 14*(4), 252–259. doi:10.1080/1065657X.2006.10702293.

Pitman, M., & Läuchli, A. (2004). Global impact of salinity and agricultural ecosystems. In A. Läuchli & U. Lüttge (Eds.), *Salinity: environment - plants - molecules* (pp. 3–20). Netherlands: Springer.

Prasad, A., Kumar, D., & Singh, D. V. (2001). Effect of residual sodium carbonate in irrigation water on the soil sodication and yield of palmarosa (*Cymbopogon martinni*) and lemongrass (*Cymbopogon flexuosus*). *Agricultural Water Management, 50*(3), 161–172. doi:10.1016/S0378-3774(01)00103-2.

Qadir, M., & Schubert, S. (2002). Degradation processes and nutrient constraints in sodic soils. *Land Degradation & Development, 13*(4), 275–294. doi:10.1002/ldr.504.

Qadir, M., Steffens, D., Yan, F., & Schubert, S. (2003). Sodium removal from a calcareous saline–sodic soil through leaching and plant uptake during phytoremediation. *Land Degradation & Development, 14*(3), 301–307. doi:10.1002/ldr.558.

Qadir, M., Noble, A. D., et al. (2005). Driving forces for sodium removal during phytoremediation of calcareous sodic and saline–sodic soils: a review. *Soil Use and Management, 21*(2), 173–180. doi:10.1079/SUM2005312.

Rengasamy, P. (2006). World salinization with emphasis on Australia. *Journal of Experimental Botany, 57*(5), 1017–1023. doi:10.1093/jxb/erj108.

Ritzema, H. P., Satyanarayana, T. V., Raman, S., & Boonstra, J. (2008). Subsurface drainage to combat waterlogging and salinity in irrigated lands in India: lessons learned in farmers' fields. *Agricultural Water Management, 95*(3), 179–189. doi:10.1016/j.agwat.2007.09.012.

Sadiq, M., Hassan, G., Mehdi, S. M., Hussain, N., & Jamil, M. (2007). Amelioration of saline-sodic soils with tillage implements and sulfuric acid application. *Pedosphere, 17*(2), 182–190. doi:10.1016/s1002-0160(07)60024-1.

Schofield, R., Thomas, D. S. G., & Kirkby, M. J. (2001). Causal processes of soil salinization in Tunisia, Spain and Hungary. *Land Degradation & Development, 12*(2), 163–181. doi:10.1002/ldr.446.

Szabolcs, I. (1974). Salt affected soils in Europe. Research Institute for Soil Science and Agricultural Chemistry of the Hungarian Academy of Sciences: Martinus Nijhoff.

U.S. Salinity Laboratory. (1954). *Diagnosis and improvement of saline and alkali soils*. Washington, D.C.: US Dept. of Agriculture.

Violette, S., Boulicot, G., & Gorelick, S. M. (2009). Tsunami-induced groundwater salinization in southeastern India. *Comptes Rendus Geoscience, 341*(4), 339–346. doi:10.1016/j.crte.2008.11.013.

Weert, F. V., Gun, J. V. D., & Reckman, J. (2009). Global overview of saline groundwater occurrence and genesis. *Utrecht/Paris, IGRAC/UNESCO*. http://www.un-igrac.org/dynamics/modules/SFIL0100/view.php?fil_Id=135.

Yensen, N. P., & Biel, K. Y. (2006). Soil remediation via salt-conduction and the hypotheses of halosynthesis and photoprotection ecophysiology of high salinity tolerant plants. In M. A. Khan, & D. J. Weber (Eds.), (Vol. 40, pp. 313-344, Tasks for vegetation science 34): Springer Netherlands.

Yoshii, T., Imamura, M., Matsuyama, M., Koshimura, S., Matsuoka, M., Mas, E., et al. (2013). Salinity in soils and tsunami deposits in areas affected by the 2010 Chile and 2011 Japan tsunamis. *Pure and Applied Geophysics, 170*(6–8), 1047–1066. doi:10.1007/s00024-012-0530-4.