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Soil Chemistry after Irrigation with Treated Wastewater in Semiarid Climate

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ABSTRACT: Soil irrigation using treated wastewater in the Brazilian semiarid region is a promising practice as this area currently faces water scarcity and pollution of water resources by domestic sewage. The aim of this study was to evaluate the use of treated wastewater in drip irrigation and its effect on the chemistry of soil cultivated with squash (*Cucurbita maxima* Duch.) Coroa IAC and to verify whether there was an increase in soil salinity under a semiarid climate. The experiment was conducted for 123 days on a farm close to the sewage treatment plant, in a randomized block design with five treatments and four replications. The treatments consisted of two irrigation water depths (100 and 150 % of the evapotranspiration), two applications of gypsum to attenuate wastewater sodicity (0 and 5.51 g per plant), and a control treatment with no application of wastewater or gypsum. During the experiment, treated wastewater and soil gravitational water, at a depth of 0.40 m, were collected for measurement of Na^+, K^+, Ca^{2+}, Mg^{2+}, NO_3^-, NH_4^+, Cl^-, alkalinity, electrical conductivity, pH and sodium adsorption ratio. At the end of the experiment, soil samples were collected at depths of 0.00-0.10, 0.10-0.20, and 0.20-0.40 m; and pH, total N, organic C, exchangeable cations and electrical conductivity of the saturation extract (CEs) were analyzed. Besides an increase in pH and a reduction in total N, the irrigation with wastewater reduces soil salinity of the naturally salt-rich soils of the semiarid climate. It also led to soil sodification, in spite of the added gypsum, which indicates that irrigation with wastewater might require the addition of greater quantities of gypsum to prevent physical degradation of the soil.

Keywords: salinity, soil sodification, gypsum application.

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INTRODUCTION

The use of treated wastewater for agriculture has many theoretical advantages such as reduction in the consumption of high quality water (e.g., from rivers, lakes, wells) for irrigation; reduction in the amount of agricultural inputs required through the use of organic matter, P, and N from wastewater; and reduction in or even elimination of the risk of eutrophication by reducing discharge of wastewater directly into water bodies. These advantages become especially salient in semiarid climates. Essentially, a lack of water for irrigation makes usable wastewater an extremely important agricultural resource, even though it is not usually considered as a viable option. Furthermore, the nutrients present in wastewater can significantly improve the fertility of sandy soils that are commonly found in the pediplains of semiarid regions. Finally, the agricultural use of treated wastewater avoids impacts of its release in rivers with low discharge or under intermittent condition. In addition to these indicators favorable to the use of wastewater in agricultural irrigation in semiarid regions (in contrast with more humid climates), strong solar radiation on the soil greatly reduces the pathogen load that may persist in the wastewater after treatment.

However, an intensification in soil salinization and sodification, naturally occurring processes in semiarid climate, are major disadvantages. One of the main problems associated with irrigation using wastewater is an increase in soil exchangeable Na, as Na is present in high concentrations in wastewater (Hayes et al., 1990; Papadopoulos and Stylianou, 1991; Bond, 1998; Magesan et al., 1999; Jnad et al., 2001). Sodification processes cause disaggregation of the soil clay fraction due to an interlayer cationic replacement by Na^+ ions (Halliwell et al., 2001). The monovalency of the Na ion and its large hydration sphere further facilitate dispersion of the clay (Balks et al., 1998), which may then lead to a reduction in hydraulic conductivity (Magesan et al., 1999; Shainberg et al., 2001; Bagarello et al., 2006) and decrease permeability (Meenner et al., 2001), drainage, and soil aeration (McBride, 1989). High concentrations of Na also compete with Ca and alter the ion exchange properties of the soil (Katerji et al., 1996; Fonseca et al., 2005). Some studies regarding the Na-Ca system in soils have been conducted (So and Aylmore, 1993; Bond, 1998), but it is not clear how the Na-Ca system behaves when wastewater is used for irrigation.

There are some agricultural practices that result in increased production of crops grown in saline and sodic soils while minimizing negative environmental side effects (Qadir and Oster, 2004; Murtaza et al., 2006). The most common methods are phytoremediation, addition of gypsum, and leaching by drainage. Phytoremediation can be carried out only after establishing soil salinization or/and sodification and verifying that reversion of these processes is possible (van Hoorn et al., 1997). Qadir and Oster (2004) reviewed 15 studies on the usefulness of bioremediation for salinity or high sodicity and found that chemical remediation was most effective.

The application of preventive measures, such as gypsum ($CaSO_4$) input, which introduces Ca in the soil system to compete with the Na, would constitute an ideal scenario, where a continuous control of sodification is possible. This also enables soil quality improvement, such as an increase in permeability and saturated and unsaturated hydraulic conductivity (Scotter, 1985; Balks et al., 1998). However, this practice also leads to an increase in soil salinity, which must then be controlled by leaching (Quirk, 2001); this is also a good method to control soil sodification (Beltrán, 1999; Qadir and Oster, 2004). One of the problems associated with this practice, which could limit its use with wastewater, is the unwanted leaching of other chemical species (nutrients or pollutants) into the saturated zone, which then necessitates quality monitoring of the deep soil and the groundwater (Domínguez-Mariani et al., 2003; Kass et al., 2005).

In order to verify a possible soil salinization or salt leaching by drainage under semiarid climate, this works aimed to evaluate the use of treated wastewater in irrigation of squash (*Cucurbita maxima* Duch.) Coroa IAC and its effect on the chemistry of soil and soil water.

MATERIALS AND METHODS

This study was carried out from February to May 2011 in the municipality of Cravolândia, state of Bahia, Brazil, in an area close to a wastewater treatment plant operated by the Empresa Baiana de Águas e Saneamento. The area is situated at an altitude of 477 m, with geographic coordinates of 13° 21’ 32’’ S latitude and 39° 48’ 54’’ W longitude. Average annual rainfall and temperature are 800 mm and 22 °C, respectively. According to the Köppen classification, the climate is type Aw (hot and humid with summer rains); however, according to Ministerial Decree No. 1 of March 9, 2005, this municipality was classified as lying in the semiarid region of Bahia after taking into consideration updated criteria for annual average rainfall (less than 800 mm), aridity index (up to 0.5), and drought risk (greater than 60 %).

A randomized block experimental design with five treatments and four replications was used, for a total of 20 plots. The treatments were: T1 – irrigation depth = 150 % of the maximum water demand of the crop; T2 - irrigation depth = 150 % of the maximum water demand of the crop and the addition of Ca as gypsum; T3 - irrigation depth = 100 % of the maximum water demand of the crop; T4 - irrigation depth = 100 % of the maximum water demand of the crop and the addition of Ca as gypsum; and T5 - no irrigation as control. Other experimental details have been previously published (Oliveira et al., 2013).

Three squash seeds were sown in previously fertilized soil at a depth of 0.02 m within a 3.0 × 2.5 m plot area. Two rows of three plants, a total of six plants, were maintained in each plot. Thinning was carried out after germination, leaving two plants per row. The experimental area was approximately 900 m^2. All recommended agricultural practices and other necessary natural plant health treatments were carried out throughout the crop cycle.

Drip irrigation with self-compensating drippers was used, placing one line of drippers per row at a spacing of approximately 0.86 m for a ratio of three drippers per plant and an average maximum flow rate of 4 L h^{-1}. The pumping system consisted of a solar electric pumping system that captured wastewater from the last aerobic lagoon of the wastewater treatment plant. A disk filter and a screen filter were fitted to the pumping pipe to prevent clogging of the drippers from material suspended in the treated wastewater. Requisite water depth was calculated using the Blaney Cridlle equation, with monthly average temperature data acquired from the Itaquara, Jaguaquara, and Itiruçu weather stations located about 20-40 km from the experimental area. During the experiment (123 days), the total irrigation depth and the total rain depth were 264.1 and 204.5 mm, respectively.

Application of Ca as calcium sulfate at a rate of 5.5 g per plant was carried out twice a month by adding the chemical to the soil surface close to the drippers in accordance with the respective treatments. The amount of Ca used was calculated to reduce the Na adsorption ratio ($SAR = [Na^+]/([Ca^{2+}]+[Mg^{2+}])^{0.5}$, concentrations in $mmol_c$ L^{-1}) of the wastewater by 50 %, using an irrigation management estimate that each plant received 18 L of water during the course of the experiment.

Soil

Before implementation of the experiment, the soil physical and chemical properties in the 0.0-0.2 m (20 sub-samples) soil layer were analyzed before experiment with the following results: sand, silt and clay: 802; 15 and 183 g kg^{-1}, respectively; pH(H_2O) 5.4; P (Mehlich-1) 7.0 mg dm^{-3}; K^+, Ca^{2+}, Mg^{2+}, Na^+, Al^{3+}, and H+Al: 0.31, 2.20, <0.01, 0.28, <0.01, and 1.76 $cmol_c$ dm^{-3}, respectively; sum of bases 2.79 $cmol_c$ dm^{-3}; cation exchangeable capacity 4.55 $cmol_c$ dm^{-3}; base saturation 61.3 %; exchangeable sodium percentage 6.15 %; electrical conductivity 267 µS cm^{-1}. Soil manure was calculated according to Ramos (2010).

At the end of the experiment, composite soil samples (three sub-samples) were collected at depths of 0.00-0.10, 0.10-0.20, and 0.20-0.40 m at a distance of 0.20 m from a dripper and always in the same direction from the irrigation line. Samples were air dried and sieved through a 2.0 mm mesh, and the following analyses were carried out: pH, EC, organic C, total N, exchangeable K, Ca, Mg, Na, Al, and total acidity (H+Al); subsequently, sum of bases, cation exchange capacity, base saturation, and exchangeable sodium percentage were calculated as recommended by Raij et al. (2001) and Claessen (1997). Sodium, potassium, and calcium were extracted with ammonium chloride (NH_4Cl) and measured in a flame photometer. Total acidity was determined by extraction with calcium acetate (1 mol L^{-1} at pH 7.0) and subsequent titration with NaOH (0.025 mol L^{-1}). Al and Ca+Mg were determined from potassium chloride (KCl) extracts, which were subsequently titrated with NaOH (0.025 mol L^{-1}) for Al and with EDTA (0.005 mol L^{-1}) for Ca+Mg.

Soil water

In order to collect soil drainage water, trenches were opened and eight PVC gutters installed in the T1 and T2 treatment plots (two in each plot) at a depth of 0.40 m. They were connected to HDPE collecting bottles by a PVC pipe.

Wastewater and soil water were sampled twice a month during the experiment at three different points: (a) at the last treatment lagoon, (b) directly from the dripper, and (c) from the soil drainage collecting bottles. The samples were transported under refrigeration to the laboratory, where they were vacuum filtered through pre-washed 0.47 µm cellulose ester membranes and stored under refrigeration in two separate bottles: vial A for anion analysis (4 °C in the dark) and vial B for metal analysis (brought to pH 2 by adding ultrapure HNO_3^- and stored at 4 °C). The pH and EC were determined directly using a potentiometer and a conductivity cell. Sodium, K, and Ca were analyzed by flame photometry; nitrate and ammonia were analyzed by spectrophotometry; while titration with EDTA (0.005 mol L^{-1}), $AgNO_3$ (0.0141 mol L^{-1}), and H_2SO_4 (0.025 mol L^{-1}) was used for determining Mg, Cl, and alkalinity, respectively. The statistical analysis was performed for comparing the soil chemistry among the five treatments after experiment, by the mean of Tuckey test at 5 % level of probability.

RESULTS AND DISCUSSION

Electrical conductivity (EC), pH, and sodium adsorption ratio (SAR) of the wastewater obtained directly from the treatment lagoon or from the dripper showed little variation during the experiment (Table 1); however, an increase in the SAR of samples obtained from the dripper (in April) and in the pH of samples from the treatment lagoon (in May) were also observed. Whereas EC values remained stable throughout the experiment, the pH of the samples collected from the treatment lagoon ranged from 8.5 to 9.2 (Table 1); this was higher than the normal range for irrigation water, which is between 6.5 and 8.4 (Ayers and Westcot, 1999). Thus, in May, even though samples from both the lagoon and the dripper had pH levels greater than the recommended maximum, the degree of restriction for using this wastewater was classified as “low to moderate” for all samples obtained, in accordance with the criteria given by Ayers and Westcot (1999).

All drainage water samples showed similar results, except for Ca content, which increased due to its displacement by excess Na from the wastewater, and nitrate content, which increased due to nitrification of the ammonium and organic matter present in the wastewater. However, it is also possible that the Ca added in the form of gypsum was not leached but rather was being taken up by the plants or adsorbed onto the soil. Furthermore, irrigation and percolation through 0.40 m of soil resulted in reducing ten times the content of fecal coliforms, which was close to the upper limit of 1,000 NPM/100 mL for class 2 freshwater (Conama Resolution 357/2005), rendering this water useful for irrigation of certain crops, such as vegetables.

Acidity and aluminum content

Soil pH profiles were similar across irrigation treatments, i.e., an elevation in pH at depths from 0.05 to 0.15 m but a reduction in the deeper layers (Figure 1). The lowest soil pH recorded was from the unirrigated control plot (T5, Table 2). These results are in accordance with previous studies, which also recorded an elevation in soil pH after irrigation with wastewater (Herpin et al., 2007; Leal et al., 2009b; Andrade Filho et al., 2013).

This increase is related to the high pH of the wastewater (Stewart et al., 1990), the presence of additional HCO_3^-, and changes in N dynamics due to denitrification and/or reduction of NO_3^-, which, in turn, releases OH^- (Stewart et al., 1990; Fonseca et al., 2005; Firme, 2007; Gloaguen et al., 2007). However, Speir et al. (1999) affirmed that such changes are, in general, of low magnitude, with values typically lower than 1 pH unit, and therefore did not significantly affected the availability of nutrients. Notably, the addition of Ca had no significant effect on soil pH.

More homogenous results for total acidity were observed between plots, irrespective of whether they were irrigated or not. Total acidity decreased with all treatments at depths from 0.05 to 0.30 m.

A consequence of the increase in soil pH was a decrease in Al availability as these two parameters clearly showed opposing trends (Figure 1). The crop and soil management practices used stimulated the release of soil Al from these soils, which are naturally feldspar enriched. However, the irrigated soil had lower Al content, attesting to the beneficial effect of irrigation with treated wastewater on aluminum toxicity under these soil and climatic conditions.

Table 1. Properties of domestic wastewater used for irrigation and of soil drainage water

Property	Wastewater[(1)]	Drainage water[(2)]	
		Treatment: I-150 %	Treatment: I-150 % + gypsum
pH	8.45	7.35	7.30
EC (dS m^{-1})	1.55	1.80	1.55
SAR ($[mmol_c L^{-1}]^{1/2}$)	7.80	6.40	7.80
Ca^{2+} (mg L^{-1})	2.50	4.35	4.75
Mg^{2+} (mg L^{-1})	0.35	0.70	0.45
K^+ (mg L^{-1})	1.40	0.60	0.55
Na^+ (mg L^{-1})	10.00	11.75	12.75
Cl^- (mg L^{-1})	6.30	6.15	6.10
HCO_3^- (mg L^{-1})	33.50	39.80	42.2
NO_3^- (mg L^{-1})	0.85	2.45	2.55
NH_4^+ (mg L^{-1})	1.60	0.60	1.55
Fecal coliforms (NPM 100 mL^{-1})	9,400	1,010	

[(1)] Sampled from the dripper, average of three samples over the growing period; [(2)] Sampled at 0.40 m depth, average of three samples over the growing period. EC: electrical conductivity; SAR: sodium adsorption ratio.

Table 2. Soil chemistry for the 0.00-0.40 m layer after the experiment

Treatment[(1)]	pH(H_2O)	CE	K^+	Ca^{2+}	Mg^{2+}	Na^+	Al^{3+}	Al+H	SB	CEC	ESP	V
		dS m^{-1}	cmol$_c$ dm^{-3}								%	
T1	6.3 a	0.14 a	0.4 a	2.6 a	1.0 a	0.9 a	0.1 a	2.1 a	4.9 a	7.0 a	12.8 a	70.0 a
T2	6.2 a	0.14 a	0.4 a	2.7 a	1.2 a	1.0 a	0.2 a	2.4 a	5.3 a	7.7 a	13.0 a	68.8 a
T3	6.3 a	0.17 a	0.4 a	2.3 a	0.9 a	1.1 a	0.2 a	2.2 a	4.7 a	6.9 a	15.9 a	68.1 a
T4	6.0 a	0.15 a	0.3 a	2.3 a	1.0 a	1.0 a	0.1 a	2.3 a	4.6 a	6.9 a	14.5 a	66.7 a
T5	5.1 b	0.25 b	0.4 a	2.5 a	1.5 a	0.3 b	0.3 a	2.3 a	4.7 a	7.0 a	4.3 b	67.1 a

[(1)] T1: irrigation depth = 150 % of the evapotranspiration (ET); T2: irrigation depth = 150 % of the ET + application of gypsum; T3: irrigation depth = 100 % of the ET; T4: irrigation depth = 100 % of the ET + application of gypsum; T5: no irrigation and no application of gypsum (control). SB: sum of bases; CEC: cation exchange capacity; ESP: exchangeable sodium percentage; V: base saturation. Mean values followed by the same letter in the column do not differ from each other by the Tukey test at the 5 %.

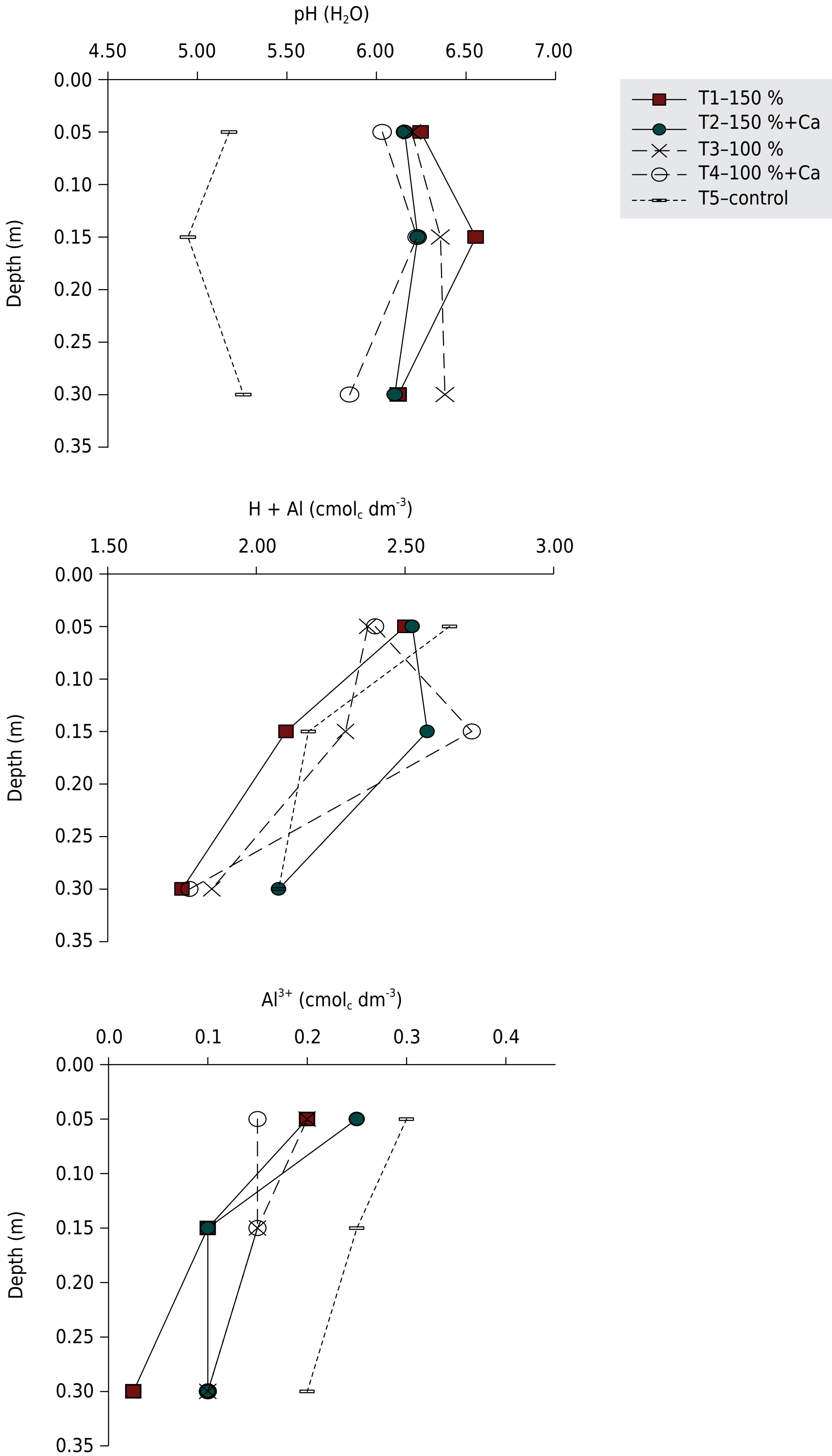

Figure 1. Average values of soil pH in water, potential acidity (H+Al) and exchangeable aluminum (Al^{3+}) in the 0.00-0.10, 0.10-0.20, and 0.20-0.40 m layers after the experiment.

Nitrogen and soil organic matter

Irrigation with treated wastewater, regardless of the treatment type, led to a maximum reduction of 30.7 % in total soil N compared to the non-irrigated plots (Figure 2), with an average decrease of 20.2 % in the surface layer (0.00-0.10 m) and a decrease of 11.8 % in the subsurface layer (0.10-0.20 m).

Such a decrease was not observed in the 0.20-0.40 m layer probably because the biodynamic processes involved are restricted to the surface soil layers, where the irrigation effect is more apparent. Results with respect to changes in organic matter content were not very clear, although it was possible to observe an accumulation in organic matter, probably due to algal decomposition of wastewater in the subsurface layer. In such scenario, greater water availability, combined with the addition of fertilizers and N and P from the wastewater, would stimulate soil microbial activity that could then lead to increased mineralization of the organic matter already in the soil surface layer, while the fine particulate organic matter would accumulate in the subsurface (Nogueira et al., 2005; Gloaguen et al., 2007; Duarte et al., 2008). Thus, Gloaguen (2006) observed a decrease in soil organic matter during the second year of wastewater irrigation, while Firme (2007) also concluded that the levels of organic matter decreased over time and through soil layers.

Cations and cation exchange capacity of the soil

Soil Ca content showed a similar trend of decrease in the 0.00-0.20 m layer in all the irrigated plots (Figure 3). These results contradict the observations made by Firme (2007), who noted an increase in soil Ca that was proportional to the increase in the depth of applied irrigation within the first few months of sugarcane irrigation. We observed that the addition of Ca via wastewater or fertilizer did not lead to an increase in the Ca content of the surface layer of the soil probably because the Ca leached into the deeper layers of the soil (0.20-0.40 m), where it was possible to identify an accumulation. Soil drainage, already prominent in this region due to the sandy nature of the soil, was further enhanced in plots that received an irrigation depth of 150 % of the crop water demand, which then increased even more in plots that received gypsum application.

Given the soil dynamics observed above, greater leaching of magnesium would be expected, as Mg forms weaker bonds with the soil exchange complex. However, we observed that Mg systematically decreased in both the cultivated and the irrigated soil, indicating that the addition of Mg via wastewater was not sufficient for the plants, and

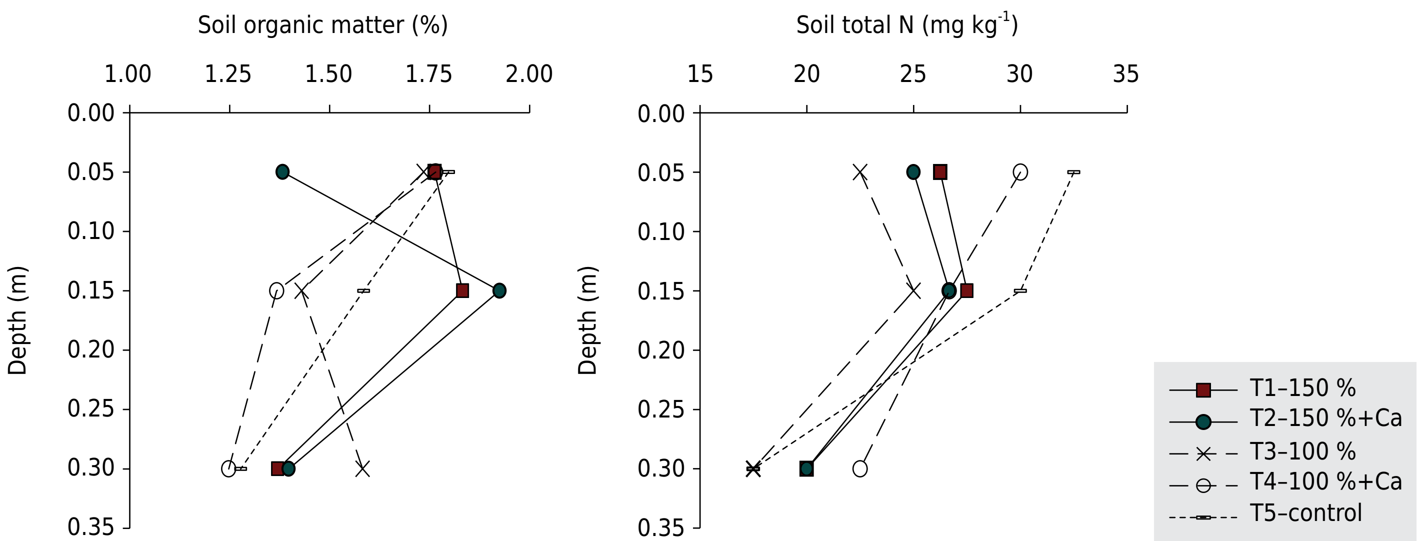

Figure 2. Average values of soil organic matter and soil total nitrogen in the 0.00-0.10, 0.10-0.20, and 0.20-0.40 m layers after the experiment.

that they used the soil Mg, which decreased by an average of 33.7 % in the 0.00-0.40 m layer (Figure 3). Nonetheless, a vertical movement of Mg was identified in plots with a higher irrigation depth (150 %) where the Mg content in the deepest layer (0.20-0.40 m) was greater than that in the plots with a lower irrigation depth (100 %).

Among the elements analyzed, K had the lowest concentration, at 0.45 $cmol_c\ dm^{-3}$ (Figure 3). Soil K concentrations were remarkably similar within the various plots (including the control plot) and in all three soil layers tested, with little variation in the 0.00-0.10 m layers and smaller values in the 0.10-0.20 m layer (0.3 $cmol_c\ dm^{-3}$).

While no significant increase in soil K was observed in this study, several other studies have reported higher leaching of K due to its displacement by the excess Na present in the treated wastewater (Al-Nakshabandi et al., 1997; Gloaguen et al., 2007). However, this lack of difference in K concentrations across the samples tested made further characterization of the soil dynamics impossible.

The cation exchange capacity (CEC) of the soil was different among the soil layers tested, and this variability increased with depth (Figure 4). The lower CEC values found in the soil surface layers (average of irrigated plots compared to the control plot: -8.5 %) were compensated by increased values with increasing depth (+7.9 % and +7.1 % in the 0.10-0.20 and 0.20-0.40 m layers, respectively). This phenomenon could be explained

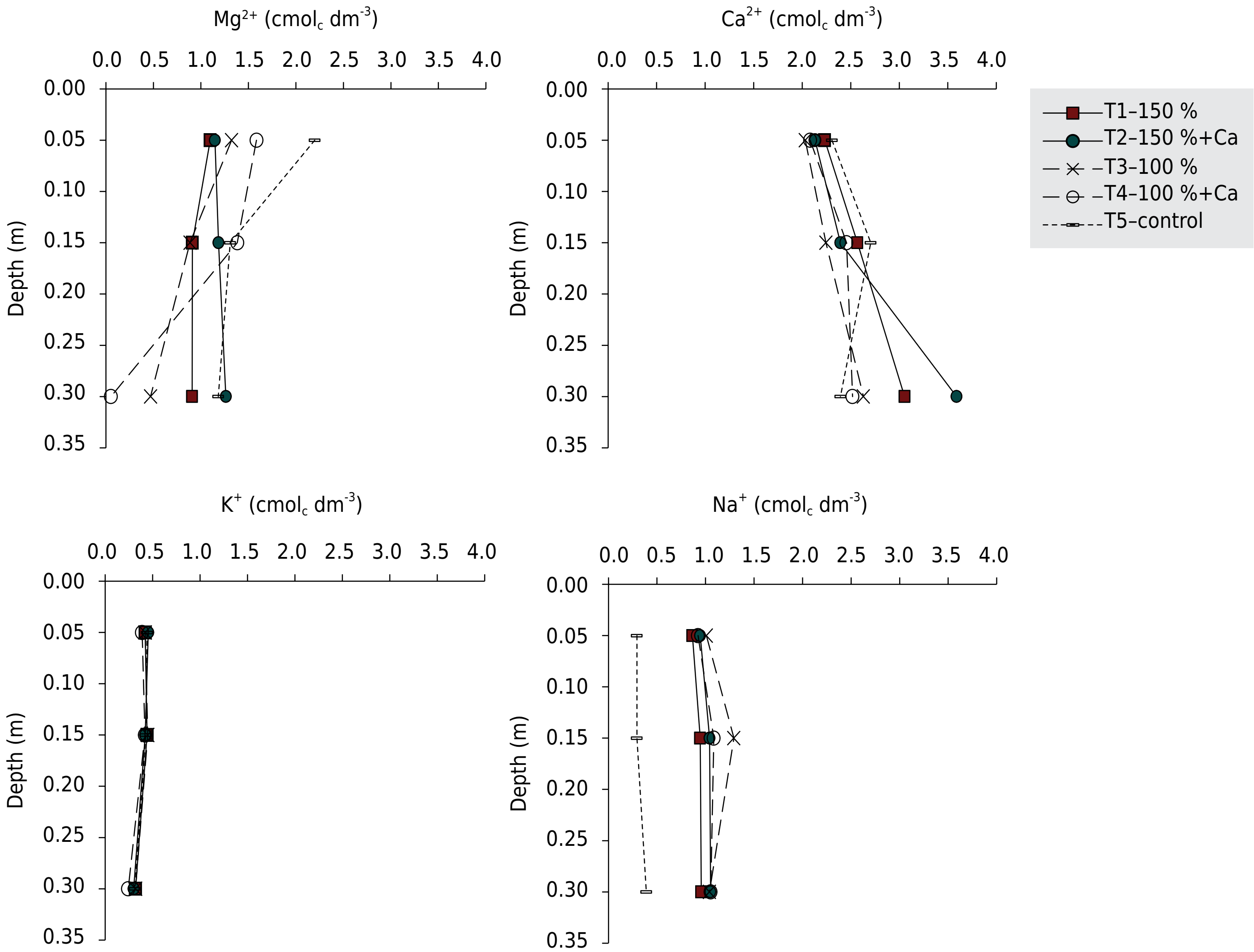

Figure 3. Average values of soil exchangeables magnesium, calcium, potassium and sodium, in the 0.00-0.10, 0.10-0.20, and 0.20-0.40 m layers after the experiment.

by excess Na in the wastewater that would cause dispersion and illuviation of clay in the deeper layers; however, as the addition of Ca in treatment plots did not attenuate this phenomenon, this hypothesis was discarded. Particularly, as it was noted that higher CEC values were more pronounced in the deeper layers of plots that received higher irrigation depth, a hypothesis of two cumulative effects is proposed. First, the labile wastewater organic matter percolates into the soil, and a further addition of N (C:N ratio is low in the wastewater) accelerates the decomposition of soil organic matter, which then leads to a decrease in the surface CEC.

Soil salinity and soil sodicity

The electrical conductivity of the wastewater used for irrigation was higher (1.55 dS m^{-1}, Figure 4) compared to the values generally recorded for effluents from wastewater treatment plants, ranging from 0.5 to 1 dS m^{-1} (Sandri, 2003; Medeiros et al., 2005; Gloaguen, 2006; Firme, 2007; Tarchouna et al., 2010). This difference is due to the dry tropical climate (semiarid) of this region and the water treatment system used, which consists of a sequence of seven anaerobic, aerobic, and facultative lagoons. These two factors lead to an increase in the salinity of the wastewater due to high evapotranspiration. Surprisingly, irrigation with this water did not lead to soil salinization in any of the irrigated plots, and the average electrical conductivity of the soil was 40.3 % lower in the irrigated soil compared to the non-irrigated soil.

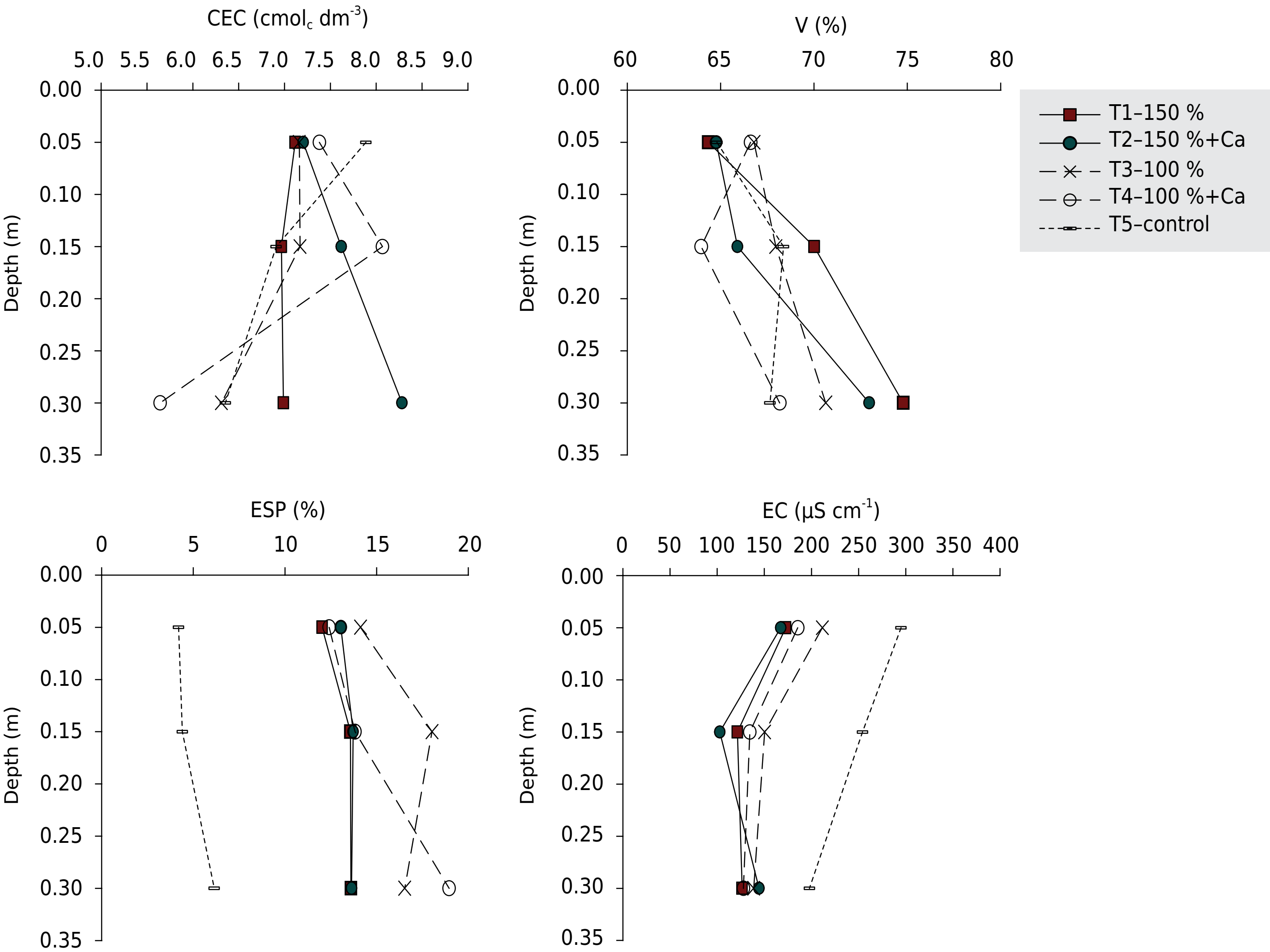

Figure 4. Average values of soil cation exchange capacity (CEC), base saturation (V), exchangeable sodium percentage (ESP), and electrical conductivity (EC) in the 0.00-0.10, 0.10-0.20, and 0.20-0.40 m layers after the experiment.

Soils of semiarid regions tend to have greater salt content due to water deficit, arising from the fact that the annual evapotranspiration rate exceeds the precipitation rate. A lack of salt drainage from the soil and an elevation in brackish groundwater levels can also cause soil salinization. In this context, irrigation, even with low quality brackish water, causes leaching of the salts (Figure 4 and Table 2). Higher soil moisture in the irrigated plots also facilitates soil drainage and salt leaching, especially during rains. Thus, the observation that the leaching of salts was more pronounced in the plots with greater irrigation depth (salinity 44 % lower compared to control plots) compared to those with lower irrigation depth (salinity 36 % lower compared to control plots) further confirms the occurrence of this process. Finally, it is important to emphasize here that the addition of Ca, essentially used to decrease soil sodification, did not increase soil salinization.

Considering this fact and the pH and EC soil values observed (lower than 6.56 and 0.30 dS m^{-1}, respectively), which are much lower than the levels established for saline soils [EC = 4 dS m^{-1} and pH = 7 (Chhraba, 2004)], it can be affirmed that irrigation with wastewater under semiarid climatic conditions did not cause soil salinization in the short term, especially when combined with a drainage control system. Under tropical humid and sub-humid climatic conditions, soil salinization occurred during irrigation with wastewater, as mentioned by Gloaguen et al. (2007), Fonseca (2007), and Leal et al. (2009a).

Irrigation with treated wastewater caused a clear increase in the concentration of soil Na, being 2.5 to 4.4 higher than the one of the control plot; the average was 3.1 times greater for the entire soil profile (Figure 4). Thus, even if lower concentrations of Na were found in the soil through drainage and addition of Ca, it was not sufficiently effective in attenuating soil sodification. The same trend was observed for the exchangeable sodium percentage (ESP), with values sometimes exceeding 15 %. Such a steep increase can permanently damage the soil (Chhraba, 2004) as it leads to the dispersion and possible illuviation of the clays, and to loss of soil hydraulic conductivity (Gonçalves et al., 2007). The reduction observed in Ca, Mg, and general soil salinity only intensifies this concern because excess Na is more deleterious in the fresh environment (increased SAR), especially because divalent cations are responsible for flocculation of colloids.

A method of remediation, as demonstrated by Vasconcelos et al. (2013), is the application of gypsum to sodic-saline soils, which significantly increases the water infiltration rate by as much as three times when applied at the required gypsum level (RPL) of 250 %, and thus helps leach the excess Na. Santos et al. (2014) also concluded that the leaching of Na, and salts in general, takes place after the application of mineral or residual gypsum associated with the leaching of 1 to 2.5 pore volumes. This implies that the application of larger quantities of $CaSO_4$ in this experiment should have led to more effective attenuation of soil sodification; however, particular attention must be given to the fact that excess RPL also leads to the leaching of colloids with the adsorbed pollutants coming from the wastewater, as observed by Vasconcelos et al. (2013).

CONCLUSIONS

Four months of irrigation with treated wastewater under a semiarid climate led to a decrease in soil total nitrogen and soil acidity. The excess salts contained in the wastewater induced an increase in the exchangeable sodium content with percent exchangeable sodium exceeding 15 %.

Applying $CaSO_4$ to reduce the sodium adsorption ratio of the wastewater by half permitted re-equilibration of soil pH. This was, however, insufficient to eliminate the risk of soil sodification, which requires higher levels of gypsum application.

Irrigation also induced salt leaching from the soil, making the salt content in the irrigated soil lower than that of the non-irrigated soil.

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