

Co-pyrolyzed biochar alters salt mobility and soil chemistry under saline irrigation

Biocarvão co-pirolisado altera a mobilidade de sais e química do solo sob irrigação salina

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ABSTRACT - Salinity is a major constraint to agricultural sustainability, particularly in drylands where irrigation with saline or brackish water is often inevitable. Biochar has been proposed as a strategy to improve soil properties and mitigate salt stress; however, the mechanisms by which co-pyrolyzed biochar rates influence salt mobility and soil chemical dynamics remain poorly understood. This study evaluated the effects of biochar derived from the co-pyrolysis of sewage sludge and cashew pruning residues on salt mobility and soil chemical attributes under saline irrigation in a 30-day soil column experiment. Five biochar rates (0, 5, 10, 20, and 40 Mg ha⁻¹) were applied, and leachate and soil samples were analyzed to assess ion dynamics and chemical changes along the soil profile. Biochar application enhanced salt mobilization and leaching, as indicated by increased electrical conductivity and ion concentrations in the leachate, particularly at higher application rates. In the soil, biochar reduced electrical conductivity and increased pH, especially in the surface layer. However, exchangeable Na⁺ and exchangeable sodium percentage (ESP) were also higher in this layer, indicating surface accumulation of sodium. These results demonstrate that biochar influences salt redistribution within the soil-leachate system rather than promoting a simple reduction in salinity. The findings highlight the dual role of biochar in saline systems, simultaneously enhancing salt leaching while potentially increasing surface sodium contents. Therefore, its application should be carefully managed, particularly under continuous irrigation using saline or brackish water.

Keywords: Ion redistribution. Soil conditioner. Sodium adsorption ratio.

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RESUMO - A salinidade impõe limitações à sustentabilidade agrícola, especialmente em regiões áridas e semiáridas, onde o uso de água salina ou salobra para irrigação é inevitável. O biocarvão tem sido proposto como uma estratégia para melhorar as propriedades do solo e mitigar o estresse salino; no entanto, os mecanismos pelos quais biocarvões influenciam a mobilidade de sais e a dinâmica química do solo ainda são pouco compreendidos. Este estudo avaliou os efeitos de um biocarvão produzido pela co-pirólise de lodo de esgoto e resíduos de poda de cajueiro sobre a mobilidade de sais e os atributos químicos do solo sob irrigação salina, em experimento com colunas de solo. Foram aplicadas cinco doses de biocarvão (0, 5, 10, 20 e 40 Mg ha⁻¹), e amostras de lixiviado e solo foram analisadas para avaliar a dinâmica iônica e as alterações químicas ao longo do perfil. A aplicação de biocarvão aumentou a mobilização e a lixiviação de sais, evidenciadas pela elevação da condutividade elétrica (CE) e das concentrações de íons no lixiviado. No solo, o biocarvão reduziu a CE e elevou o pH, principalmente na camada superficial. Entretanto, os teores de Na⁺ trocável e a porcentagem de sódio trocável (PST) aumentaram nessa camada, indicando acúmulo superficial de sódio. Os resultados demonstram que o biocarvão promove a redistribuição de sais no sistema solo-lixiviado, em vez de uma simples redução da salinidade. Assim, embora favoreça a lixiviação de sais, seu uso pode aumentar o risco de sodicidade superficial, exigindo manejo criterioso, sob irrigação salina contínua.

Palavras-chave: Redistribuição de íons. Condicionador de solo. Razão de adsorção de sódio.

INTRODUCTION

Salinity is a major challenge for agricultural sustainability, particularly in arid and semi-arid regions where irrigation with saline water is often necessary (HUANG et al., 2019). Excessive salt accumulation in the root zone can compromise soil structure, alter its chemical balance, and reduce plant growth and productivity (FOULADIDORHANI et al., 2024). With the increasing scarcity of freshwater resources, it is essential to develop effective strategies to mitigate the negative impacts of salinity and enhance soil resilience to irrigation with lower-quality water (ZHANG et al., 2021).

Among the management strategies for salt-affected soils, biochar application has gained attention due to its potential to improve soil properties and reduce salt stress (ZHANG et al., 2021). Biochar is a carbon-rich material produced through the pyrolysis or co-pyrolysis of organic residues (NASCIMENTO et al., 2023). In co-pyrolysis, two or more feedstocks are combined during carbonization, yielding biochars with enhanced properties compared to those derived from a single feedstock. These improvements include reduced ash content, greater chemical stability, increased porosity, enhanced adsorption capacity, and improved cation exchange properties (CHEN et al.,

2020). Co-pyrolysis has also been considered a promising approach for the final treatment of sewage sludge, enabling the production of materials with lower toxicity potential, higher specific surface area, and abundant surface functional groups, which may favor its safer application as a soil amendment (D'ÁVILA et al., 2026). In addition, the use of lignocellulosic biomass, such as cashew pruning residues, represents a sustainable strategy, as these materials are widely available and renewable, particularly in drylands, contributing to agro-industrial waste valorization and circular economy practices

In this context, studies indicate that biochars produced by co-pyrolysis can reduce dissolved salt concentrations in soil and increase the leaching of undesirable ions, promoting environmental improvements and contributing to salinity mitigation (CAO et al., 2024). Co-pyrolyzed biochar effects vary with the temperature applied during the process, as well as with factors such as feedstock composition, soil type, and application rate, directly influencing soil chemical properties and nutrient availability (CHEN et al., 2024). Despite growing

interest in co-pyrolyzed biochars, it remains unclear how rates of biochar derived from sewage sludge and cashew pruning residues influence salt mobility, ion redistribution, and soil chemical properties in saline systems.

Therefore, this study hypothesizes that increasing rates of biochar derived from the co-pyrolysis of sewage sludge and cashew pruning residues modify salt dynamics by enhancing ion mobility and redistribution along the soil profile, thereby altering soil chemical properties. Thus, this study aimed to evaluate the effects of different rates of co-pyrolyzed biochar on salt mobility and soil chemical attributes in a short-term soil column experiment under saline irrigation.

MATERIALS AND METHODS

The study was conducted from April to May 2024 in the Soil Fertility Laboratory of the Department of Soil Science at the Federal University of Ceará (UFC), PICI Campus, Fortaleza, CE, Brazil (3° 44' 45" S, 38° 34' 56" W, Figure 1).

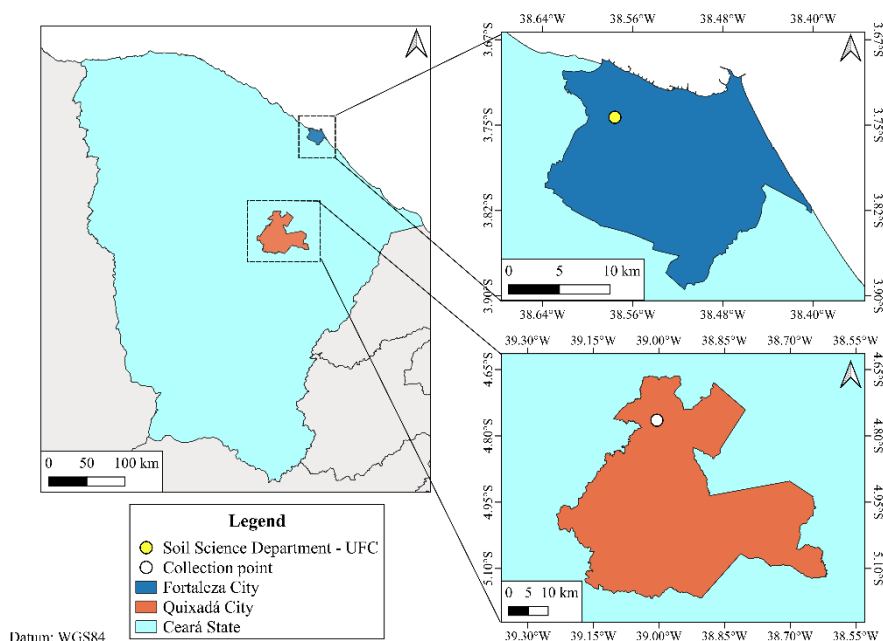


Figure 1. Location of the soil collection site and experimental site.

The soil used in the experiment was collected in the municipality of Quixadá, Ceará, Brazil (4°45'52.4" S, 39°00'13.5" W; Figure 1), and classified as a LUVISSOLO CRÔMICO Pálico saprolítico, corresponding to a Typic Ustalfs according to the USDA Soil Taxonomy (SOIL SURVEY STAFF, 2022). The climate of the region is classified as tropical wet-dry (Aw) according to Köppen's classification.

Soil samples with disturbed structure were collected from the 0–60 cm depth, homogenized, air-dried, and passed through a 2-mm-mesh sieve for the determination of physical and chemical attributes (Table 1).

Sewage sludge for biochar production was obtained from a domestic wastewater treatment plant equipped with an

Upflow Anaerobic Sludge Blanket (UASB) reactor in Fortaleza, Ceará, Brazil. Cashew pruning residues (leaves and branches) were collected from the same UASB site. After collecting, the biomass was air-dried and mixed in a 1:1 (w/w) ratio. This proportion was selected to optimize sludge utilization and to improve the physicochemical properties of the resulting biochar through the incorporation of lignocellulosic biomass, contributing to reduced ash content and enhanced structural characteristics (CAO et al., 2024). Thus, this ratio was adopted for biochar production in a pyrolytic reactor with a capacity for 40 kg of biomass, heated to 500 ± 4.3 °C for 97 min, under moderate nitrogen flow, ensuring inert heating conditions (SPPT Technological Research Company).

Table 1. Chemical and physical attributes of the soil used in the experiment before application of the treatments.

pH	OM	P	K ⁺	Na ⁺	Ca ²⁺	Mg ²⁺	Al ³⁺	H+Al ³⁺
H ₂ O	g dm ⁻³	mg dm ⁻³	cmol _c dm ⁻³					
7.15	7.3	4.61	0.19	0.06	9.23	4.9	0	3
EC _{se}	CEC	ESP	Particle-size			FC	Bulk density	
			(g kg ⁻¹)					
dS m ⁻¹	cmol _c dm ⁻³	%	Sand	Silt	Clay	(m ³ m ⁻³)	(kg m ⁻³)	
0.052	17.44	0.351	751.3	12.20	236.43	0.26	1530	

pH - Hydrogen potential; OM - Organic Matter: Walkley-Black Wet Digestion; P extracted with Mehlich 1; Ca²⁺ and Mg²⁺ extracted with 1 M KCl at pH 7.0; Na⁺ and K⁺ extracted with 1 M NH₄OAc at pH 7.0; H+Al³⁺ extracted with 0.5 M CaOAc at pH 7.0; EC_{se} - Electrical conductivity of the water extract; CEC - Cation exchange capacity; ESP - Exchangeable sodium percentage; FC - Moisture at Field capacity. The soil is considered non-saline and non-sodic.

The selected biochar rates were based on ranges widely reported in the literature, typically varying between 5 and 50 Mg ha⁻¹ (MAJOR, 2010). The lowest rate (5 Mg ha⁻¹) was considered a conservative baseline, and the subsequent treatments (10, 20, and 40 Mg ha⁻¹) were established through a doubling sequence to better capture changes in soil properties and salt dynamics across the rate range. In addition, previous studies conducted using the same biochar (MONIZ et al., 2025; SANTOS et al., 2026), which demonstrated its effectiveness as a soil conditioner influencing multiple soil attributes, further support the selection of these rates. The higher application rates (20 and 40 Mg ha⁻¹) were included as upper-bound scenarios to allow the evaluation of potential adverse effects, such as excessive ion release and salt accumulation. After assembling the soil columns, the upper 0–10 cm layer was amended with biochar derived from the co-pyrolysis of sewage sludge and cashew pruning residues.

The nominal field rates were normalized to the conventional 0–0.20 m soil layer. For the column experiment, the amount applied to the treated 0–0.10 m layer was calculated as $m = R \times A_{\text{column}} \times (0.10/0.20)$, where R is the field rate expressed in kg m⁻² and A_{column} is the cross-sectional area of the column. This calculation resulted in 0, 1.96, 3.93, 7.85, and 15.71 g per column for the nominal rates of 0, 5, 10, 20, and 40 Mg ha⁻¹, respectively. Therefore, the reported rates refer to the 0–0.20 m normalization, whereas biochar was physically incorporated only into the upper 0–0.10 m of the packed soil.

This methodological approach was intentionally adopted to enable a more accurate assessment of vertical ion mobility and redistribution throughout the soil profile. Accordingly, the amount of biochar corresponding to each treatment was thoroughly homogenized with the soil within the 0–0.10 m layer, whose mass was approximately 1.18 kg, as estimated from soil bulk density and the volume of the application layer.

The experimental units consisted of PVC (polyvinyl chloride) tubes measuring 0.10 m in diameter and 0.50 m in height. The columns were filled with soil to achieve a bulk density of 1500 kg m⁻³, and the assembly considered only the soil volume corresponding to the lower 0.40 m of the tube. The remaining upper 0.10 m was left unfilled to serve as a containment zone during the application of the saline solution. A porous cotton fabric (Murin) was affixed to the bottom section of each column, and sealed with a PVC cap. An outlet

valve was installed at the center of the cap to allow the drainage of the leachate. A fine gravel layer was also placed at the bottom of each column to facilitate drainage.

The soil and soil-biochar mixtures were moistened with distilled water to a moisture content slightly above field capacity (0.26 m³ m⁻³). Each column was covered with plastic to prevent evaporation. Field capacity was assumed when drainage ceased, and the leachate was collected in plastic pots.

After reaching field capacity, each column received a daily application of 50 mL of saline solution for 30 consecutive days. The plastic pots used for collection were replaced daily to obtain fresh leachate samples for subsequent analysis.

The saline solution was prepared using distilled water with the addition of sodium chloride (NaCl), calcium chloride (CaCl₂·2H₂O), and magnesium chloride (MgCl₂·6H₂O) in an equivalent ratio of 7:2:1, representing the typical ionic composition of brackish groundwater reported for semi-arid regions of northeastern Brazil (PALÁCIO et al., 2011). This proportion reflects the predominance of Na over divalent cations (Ca and Mg), which is commonly observed in regional irrigation waters and is relevant for assessing sodium-related soil processes.

The solution was adjusted to an electrical conductivity of 3.0 dS m⁻¹, corresponding to a high salinity hazard for irrigation water. This classification is based on widely accepted criteria for irrigation water quality, in which electrical conductivity (EC_w) is used as a key indicator of dissolved salt concentration and salinity risk (GHEYI et al., 2016). According to these guidelines, irrigation waters with EC values close to or exceeding 3.0 dS m⁻¹ are considered to impose moderate to severe restrictions for agricultural use due to their potential to induce osmotic stress, salt accumulation, and soil chemical imbalances. Therefore, the selected EC level represents a realistic saline irrigation scenario commonly observed in semi-arid regions and is suitable for evaluating salt-related soil processes under controlled experimental conditions. The salt concentration (SC) required to obtain the desired electrical conductivity was estimated according to Equation (1).

$$SC = EC \times 10 \quad (1)$$

where: SC is the salt concentration (mmol_c L⁻¹) and EC is the pre-established electrical conductivity (dS m⁻¹).

The leachate samples collected daily over the 30-day experimental period (a total of 600 samples) were analyzed for calcium (Ca) and magnesium (Mg) contents using atomic absorption spectrometry (Thermo Scientific ICE 3000 Series). Sodium (Na) and potassium (K) were determined using flame photometry (BARMAN et al., 2012), while EC was measured using a pipette-type conductivity meter. The pH was measured with a digital benchtop pH meter. The sodium adsorption ratio (SAR) was calculated using Equation 2. Daily measurements over the 30 days were used as an experimental factor, enabling the assessment of temporal variations in leaching in soil treated with biochar under saline conditions.

$$SAR = \frac{Na}{\sqrt{\frac{[Ca] + [Mg]}{2}}} \quad (2)$$

where: SAR is the sodium adsorption ratio; Na, Ca and Mg are expressed in mmol_c L⁻¹.

One day after the 30th application of the saline solution, soil samples were collected from each experimental unit at four depths: P1 (0–10 cm), P2 (10–20 cm), P3 (20–30 cm), and P4 (30–40 cm), with each depth beginning immediately after the previous one. After air-drying, soil extracts were prepared to evaluate Ca, Mg, Na, K, and H+Al concentrations. The pH, EC, and cation exchange capacity (CEC), and exchangeable sodium percentage (ESP) were subsequently calculated.

Exchangeable Na and K in the soil were extracted using Mehlich-1 solution and determined by flame photometry. Exchangeable Ca and Mg were extracted with 1 mol L⁻¹ KCl and measured by atomic absorption spectrometry. Aluminum (Al) was extracted using 1 mol L⁻¹ KCl and quantified by titration with 0.025 mol L⁻¹ NaOH. Soil pH was measured in water (1:2.5, soil:solution ratio) using a pH meter. Electrical conductivity (EC) was determined in a 1:2 (v:v) soil–water extract, following the method described by Raij et al. (2001).

The temporal dynamics of the response variables were analyzed using linear mixed-effects models to account for repeated observations collected over time from the same experimental units. Each soil column was treated as an experimental unit, and repeated measurements collected over time within each column were modeled accordingly. In the model, treatment (Rate × Time interaction, in days) was specified, with rate treated as a fixed effect and time as a repeated factor. The covariance structure of the residuals was selected by comparing candidate models using the Akaike Information Criterion (AIC), and the first-order autoregressive structure [AR(1)] was chosen as it provided the best fit, reflecting the temporal dependence among observations, in which correlations decrease as the time interval increases (LITTELL et al., 2006).

The use of mixed models with appropriate covariance structures is recommended for longitudinal data analysis because it accounts for the lack of independence among repeated observations, improving parameter estimation and controlling Type I error rates. Denominator degrees of

freedom were estimated using the Kenward–Roger method, which provides more accurate inference, particularly in studies with small sample sizes or unbalanced data (LUKE, 2017).

Treatment effects were evaluated based on estimated marginal means (least squares mean, LSM_{means}), which represent the overall average response across all time points. Pairwise comparisons among treatments were performed using Tukey's post hoc test ($p < 0.05$). Therefore, the statistical comparisons reflect differences in overall response levels across time, rather than pointwise differences at individual time points.

Soil chemical attributes were analyzed in a completely randomized design in a 5 × 4 factorial arrangement (biochar rates × sampling depths), with four replicates. Data were subjected to analysis of variance (ANOVA), and when significant, means for sampling depth were compared using Tukey's test ($p \leq 0.05$). The effects of biochar rates were evaluated by regression analysis. These analyses were performed using SISVAR – ESAL, version 5.7 (FERREIRA, 2019), following standard procedures for factorial experiments in soil science. Mixed-model analyses for repeated measures over time were conducted using PROC MIXED in SAS® OnDemand for Academics (version 9).

RESULTS AND DISCUSSION

The chemical characterization of sewage sludge, the mixture of sewage sludge and cashew pruning residues, and the resulting biochar (Table 2) was performed to assess the potential and limitations of these materials for soil application.

Since biosolids derived from sewage sludge may contain potentially toxic elements, the concentrations of Cd, Cr, Cu, Ni, Pb, and Zn were determined in both the sewage sludge–cashew residue mixture and the biochar produced by co-pyrolysis. The concentrations of potentially toxic elements were compared with the regulatory limits established by the Brazilian National Environmental Council (CONAMA) Resolution N°. 498/2020 (BRASIL, 2020), which governs the agricultural use of biosolids.

The maximum allowable concentrations defined by this regulation are 39 mg kg⁻¹ for Cd, 1000 mg kg⁻¹ for Cr, 1500 mg kg⁻¹ for Cu, 450 mg kg⁻¹ for Ni, 300 mg kg⁻¹ for Pb, and 2800 mg kg⁻¹ for Zn. The concentrations observed in the biochar (Cd = 1.0, Cr = 40, Cu = 170, Ni = 23, Pb = 15.9, and Zn = 139 mg kg⁻¹) were substantially lower than these limits for all evaluated elements.

Although pyrolysis can concentrate certain elements due to mass reduction, the results indicate that the biochar produced from the co-pyrolysis of sewage sludge and cashew pruning residues complies with national regulatory standards and poses low environmental risk with respect to heavy metal content.

A significant interaction between biochar rates and evaluation time (Rate × Day) was observed for pH, electrical conductivity (EC), Na, Ca, Mg, and sodium adsorption ratio (SAR), indicating that the temporal response of these variables differed among treatments (Figure 2).

Table 2. Chemical characterization of the sewage sludge, the mixture of sewage sludge with cashew pruning residues, and the biochar from the mixture.

Attributes	Sewage sludge	Sewage sludge +cashew pruning residues	Biochar
CEC (mmol _c kg ⁻¹)	341.79	409.35	108.62
EC (mS cm ⁻¹)	11.23	3.16	0.8
pH	5.55	5.14	9.1
Moisture content (%)	21.80	8.03	1.9
Volatile solids (%)	50.23	64.93	31.2
Ash content (%)	27.33	20.20	51.2
Fixed carbon (%)	0.63	6.83	15.6
Cellulose (%)	7.20	15.16	2.2 ¹
Hemicellulose (%)	6.20	12.5	4.9 ¹
Lignin (%)	12.50	23.7	65.1 ¹
OC (g kg ⁻¹)	322.3	370.6	348.0
N (total) (g kg ⁻¹)	38.84	23.96	24.45
P (g kg ⁻¹)	8.50	5.03	17.7
K (g kg ⁻¹)	2.09	1.27	6.1
Ca (g kg ⁻¹)	21.44	13.92	19.3
Mg (g kg ⁻¹)	9.41	6.36	7.3
Na (g kg ⁻¹)	1.85	0.98	4.1
Zn (mg kg ⁻¹)	-	620.0	139
Cu (mg kg ⁻¹)	-	70.0	170
Cd (mg kg ⁻¹)	-	0.5	1.0
Cr (mg kg ⁻¹)	-	20.0	40
Ni (mg kg ⁻¹)	-	1.0	23
Pb (mg kg ⁻¹)	-	1.0	15.9

The characterization of sewage sludge and the mixture (sludge + pruning residues) was performed based on EC and pH (BRASIL, 2017), and nutrients quantified by flame spectrometry (Na and K) and atomic absorption (Ca and Mg). Organic carbon and nitrogen were quantified via elemental analysis. For biochar, nutrients were extracted using the Modified Dry Ash method (ENDERS; LEHMANN, 2012) and quantified as follows: P by colorimetry, Ca and Mg by ICP-OES, and K and Na by flame photometry. EC and pH were determined according to Rajkovich et al. (2012), and total organic carbon was analyzed using the Walkley-Black method. Total nitrogen was obtained by the Kjeldahl method.

In the leachate from the treatment without biochar application (rate 0), subjected to daily application for 30 days of a saline solution containing Na, Ca, Mg, and Cl, a progressive increase in EC was observed over time, associated with the accumulation of ions in solution, particularly Na and Cl. Under these conditions, increases in Na, Ca, Mg, and K concentrations, as well as in SAR, were observed throughout the experimental period (Figures 2C, 2E, 2F, 2G).

The Na concentration curve exhibited a continuous increasing trend, whereas Ca, Mg, and K concentrations showed greater variability over time (Figures 2C, 2E, 2F, 2G). This pattern indicates that irrigation with saline water progressively altered the composition of the leachate, with effects intensifying over time, particularly for Na⁺. This behavior is attributed to its higher proportion in the applied

solution relative to divalent cations (Ca and Mg), as well as its greater mobility in the soil–water system due to its monovalent nature (ZHU et al., 2025). Consequently, this contributed to the increase in SAR over time (Figure 2G). These temporal patterns are consistent with previous findings, as Barman et al. (2012) reported distinct leaching behaviors for Na, Ca, and K in soil column experiments. However, from a soil management perspective, the increase in SAR in the drainage water indicates a potential limitation for its reuse in irrigation systems, particularly in poorly drained environments. Therefore, while biochar application contributes to the removal of salts from the soil profile, it should be interpreted as a process of salt export via leaching, with implications for drainage water quality.

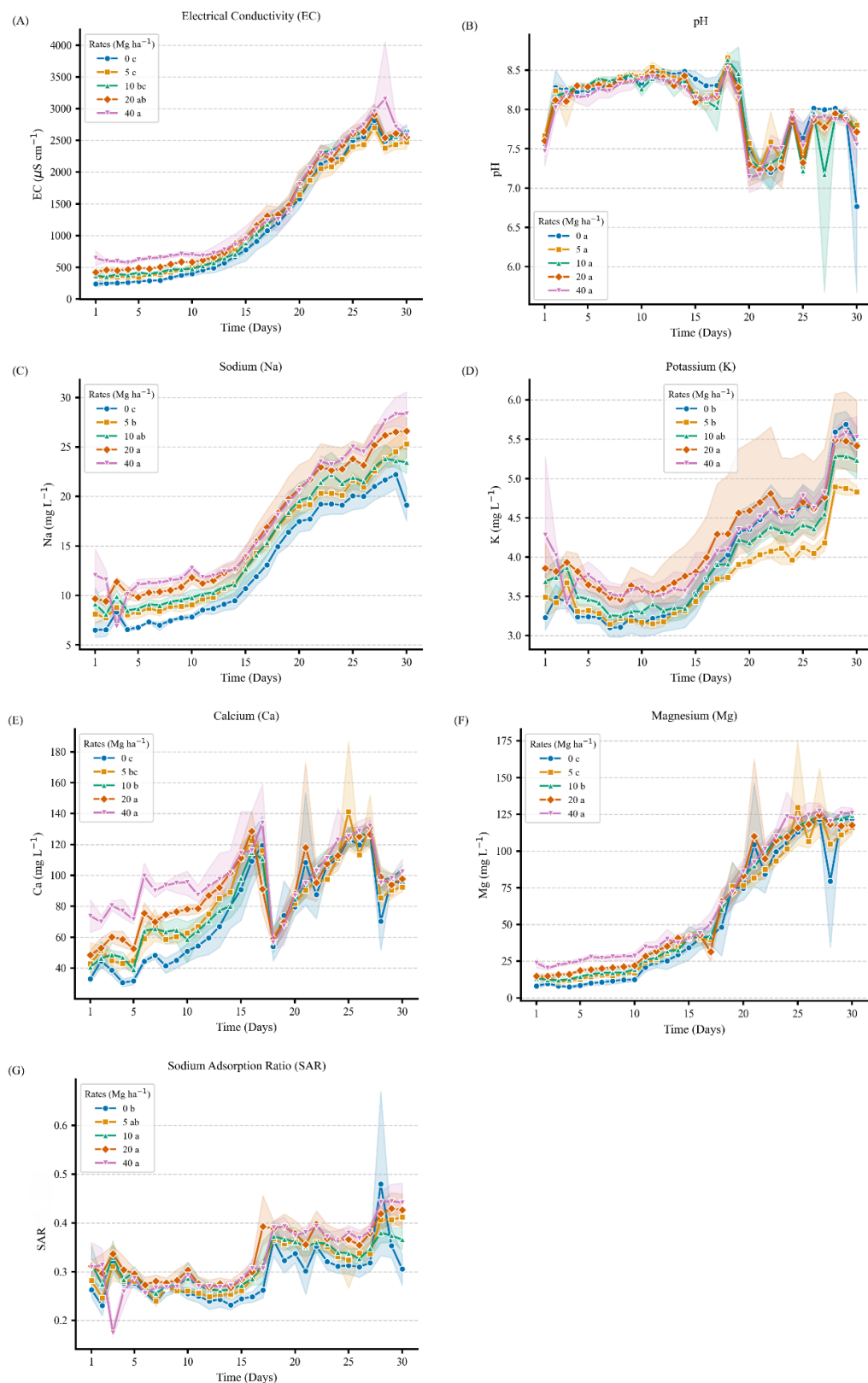


Figure 2. Temporal dynamics of leachate chemical attributes as a function of biochar application rates under saline irrigation over 30 days after application. Panels represent: electrical conductivity - EC (A), pH (B), sodium - Na (C), potassium - K (D), calcium - Ca (E), magnesium - Mg (F), and sodium adsorption ratio (SAR) (G).

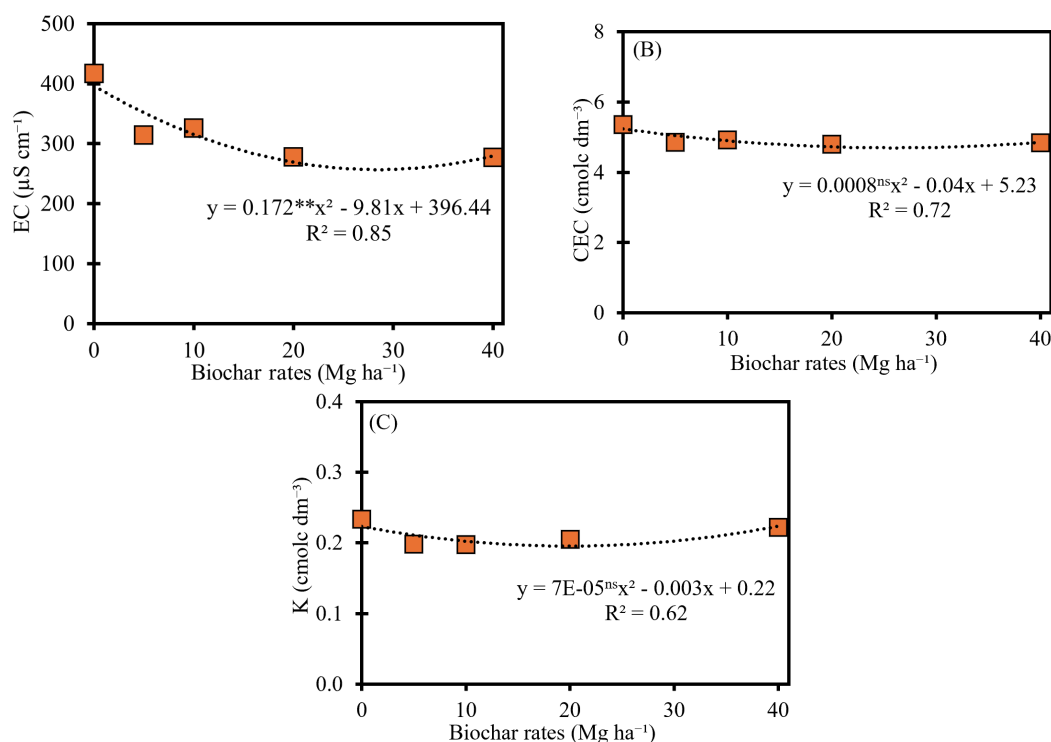
The increase in K^+ concentration in the leachate over time was less pronounced compared to Na, Ca, and Mg (Figure 2D), with noticeable variation occurring mainly at later stages (after approximately day 20), indicating a delayed response. This behavior can be attributed to the absence of K in the applied saline solution, resulting in its lower concentration in the soil solution. In general, ions not supplied via irrigation water tend to contribute less to leachate composition, as their dynamics are strongly dependent on the ionic composition of the applied solution (LIRA et al., 2018).

With biochar application, particularly at rates $\geq 20 \text{ Mg ha}^{-1}$, the effects observed in the leachate were intensified over time, resulting in higher EC values and greater concentrations of Na, Ca, Mg, and K^+ compared to the treatment without biochar (rate 0) (Figure 2). This behavior may be associated with both the release of ions from the biochar itself and its influence on soil properties, such as cation exchange capacity and pH, which can alter the dynamics and mobility of ions introduced via irrigation water (LEE et al., 2022; NASCIMENTO et al., 2023; ZHU et al., 2025). However, due to the recalcitrant nature of biochar (Table 2), the release of ions into the soil solution shortly after application is likely less significant than the contribution from brackish irrigation water.

Regarding ion release from biochar, it is noteworthy that Ca, Mg, K, Na, and other elements are present in its

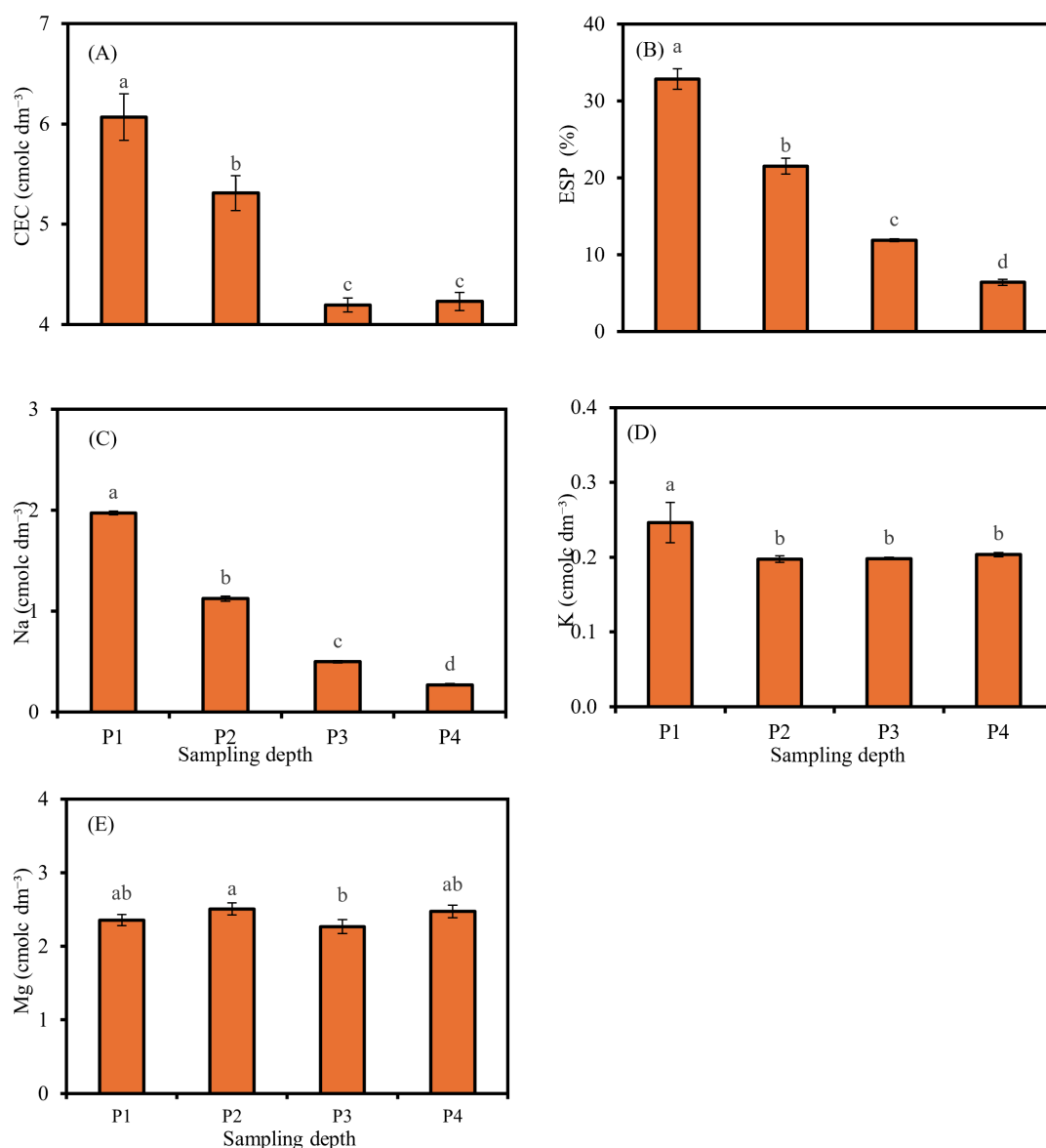
composition (Table 2). In the characterization analysis (Table 2), total elemental contents were determined by dry digestion combined with strong acid addition and heating to promote complete sample decomposition and nutrient release, considering the highly recalcitrant nature of biochar materials (NASCIMENTO et al., 2023). These results represent total elemental contents and do not imply that all nutrients are readily available after soil application. Therefore, not all nutrients present in biochar are immediately subject to leaching; however, partial release is expected over time, as evidenced by the positive effects of biochar application on plant development in degraded soils (SILVA et al., 2026).

In this context, the observed increase in K concentration in the leachate over time (Figure 2D) is also noteworthy. Since K was not included in the irrigation water, its increase in the leachate is likely associated with biochar application. Although soil K contents (Figure 3C) did not increase markedly with biochar rates, higher values were observed in the surface layer (P1), where the material was incorporated (Figure 4D). Similarly, higher Na contents and exchangeable sodium percentage (ESP) were observed in P1 (Figures 4B and 4C), indicating the contribution of these elements to the soil via biochar. This aspect deserves attention, as biochar application should not contribute to increased risks of soil salinity and/or sodicity (LEE et al., 2022).



* - Significant at $p \leq 0.05$; ** - Significant at $p \leq 0.01$

Figure 3. Electrical Conductivity - EC (A), Cation Exchange Capacity - CEC (B), and Potassium Content - K (C) in soil as a function of biochar rates.



Means followed by the same letter do not differ according to Tukey's test at a $p \leq 0.05$ significance level. Bars represent the standard error of the mean. Sampling depth: P1 (0–10 cm), P2 (10–20 cm), P3 (20–30 cm) and P4 (30–40 cm)

Figure 4. Cation exchange capacity - CEC (A), exchangeable sodium percentage - ESP (B), soil contents of Na (C), K (D), and Mg (E) as a function of sampling depth.

Another observation in the leachate was a decrease in pH starting from approximately day 20 (Figure 2B) across all treatments. Since pH is inversely related to H^+ concentration, this behavior indicates an increase in hydrogen ion concentration in the leachate. This may be associated with the release of low-molecular-weight organic acids, which contribute to solution acidification (LIU et al., 2025). In this case, the release of organic acids likely originated not only from the biochar but also from native soil organic matter, as acidification was also observed in the treatment without biochar (rate 0).

Overall, the results indicate that biochar application modifies both the magnitude and timing of salt mobilization in the soil. Higher application rates (20–40 Mg ha⁻¹) were particularly effective in enhancing solute transport and salt leaching. However, this process should be interpreted as a redistribution of salts along the soil profile rather than a

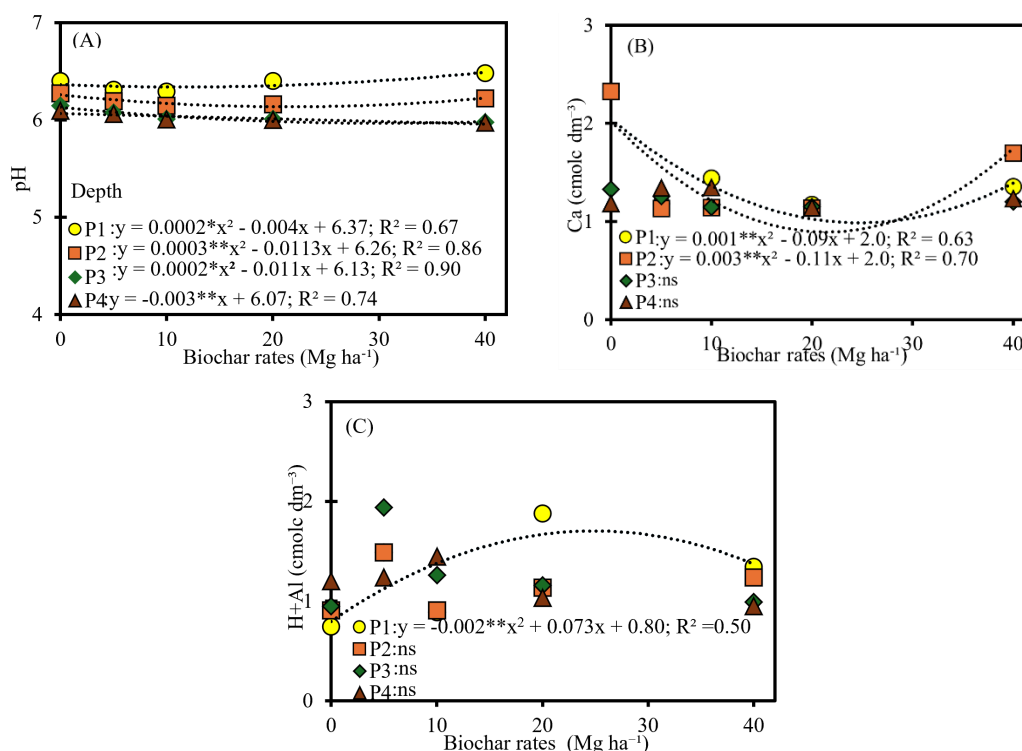
simple reduction in salinity, highlighting the importance of considering both soil and drainage water quality in the management of saline systems.

The influence of biochar application rates on soil properties and ion dynamics may be attributed to the introduction of negative charges, which have the potential to retain cations. This retention is more pronounced for divalent elements; however, it does not prevent sodium leaching, as Na is a monovalent ion and is therefore retained with lower affinity (ZHU et al., 2025). These charges also do not prevent chloride leaching, since Cl, being an anion, is not retained by the negatively charged surfaces of the soil or the biochar itself. Although soil cation exchange capacity (CEC) did not increase with biochar rates (Figure 3B), it was higher in layers P1 and P2 (Figure 4A), illustrating the influence of the carbonaceous material in the soil layer where it was applied.

The distribution of exchangeable Na varied

significantly with soil depth (Figure 4C), as previously described, with accumulation in the surface layer (P1) and an average reduction of 83.3% between P1 and the deepest layer (P4). This pattern reflects the surface application of the saline solution combined with the potential release of sodium from the biochar. The effectiveness of biochar in mitigating salinity depends on its specific properties and chemical composition. In this study, the biochar contained 4.09 g kg^{-1} of Na (Table 2), indicating a potential contribution of sodium, as also discussed for the leachate. Biochars with high salt content may contribute to surface salinization when applied at excessive rates (LEE et al., 2022), highlighting the importance of proper feedstock selection and application rate.

In the context of the results presented thus far, it is important to examine the soil attributes evaluated. In the absence of biochar application, it would be expected that, after 30 days of irrigation with saline water, the contents of Ca and Mg would increase relative to the initial soil characterization. However, considering the soil chemical attributes that responded significantly to the treatments, the Ca content ($2 \text{ cmol}_e \text{ dm}^{-3}$) in the absence of biochar, after 30 days, was lower than that observed in the initial soil characterization ($9 \text{ cmol}_e \text{ dm}^{-3}$), as shown in Table 1. Soil Ca content varied with biochar application rates (Figure 5B), indicating the influence of the carbonaceous material on Ca sorption processes.



* Significant at $p \leq 0.05$; ** Significant at $p \leq 0.01$. Depths correspond to P1 (0–10 cm), P2 (10–20 cm), P3 (20–30 cm), and P4 (30–40 cm)

Figure 5. Soil pH (A), exchangeable Ca (B), and potential acidity (H+Al) (C) as affected by biochar application rates across soil depths. Lines represent fitted models where significant.

For Mg, in addition to the lower soil contents observed after 30 days of evaluation ($2 \text{ cmol}_e \text{ dm}^{-3}$) compared to the initial characterization ($4.9 \text{ cmol}_e \text{ dm}^{-3}$) (Table 1), greater variability was observed across soil depths (Figure 4E). This suggests downward movement of Mg from the surface to subsurface layers, likely favoring its leaching.

After 30 days, soil EC decreased in response to increasing biochar rates (Figure 3A). This result supports the enhancement of ion leaching from the soil following the application of carbonaceous material. These findings are consistent with the increased mobilization and leaching of salts reported in the literature due to biochar application (LEE et al., 2022; FOULADIDORHANI et al., 2024). Similar responses have been reported, including in a meta-analysis by Wang et al. (2024), which showed an average reduction in soil EC following biochar application. However, this reduction should be interpreted alongside the increase in salt

concentrations observed in the leachate, indicating that salts were redistributed within the system rather than removed.

Biochar application increased exchangeable K in the soil, particularly in the surface layer (Figures 3C and 4D). However, this increase should not be interpreted solely as improved K retention. The observed pattern likely results from both the direct input of K from the biochar and changes in soil cation exchange dynamics. Therefore, the results indicate an overall increase in K availability but do not allow for clear separation between retention mechanisms and direct nutrient supply. This increase may be beneficial for plant nutrition in saline environments, as demonstrated by HUANG et al. (2019).

For soil active and potential acidity (Figures 5A and 5C), the most pronounced effects were observed in the surface layer (P1), where biochar was applied. After 30 days of incubation, soil pH values ranged between 6 and 7, which

were lower than those observed in the initial soil characterization (Table 1). This decrease in pH can be attributed to the leaching of base cations, which reduces the soil buffering capacity and contributes to changes in potential acidity (Figure 5C).

The simultaneous increase in pH and $H^+ + Al$ with increasing biochar rates suggests a redistribution of acidity forms, as well as dynamic changes in soil chemical equilibria, rather than a simple neutralization process. Although the alkaline components of biochar (e.g., ash and basic cations) may contribute to increases in soil pH (NASCIMENTO et al, 2023), part of these bases may have moved downward through the soil profile. As a result, some exchange sites may become exposed and available for occupation by acidic cations, such as aluminum, thereby increasing exchangeable acidity even under conditions of higher pH (active acidity). In addition, the contribution of organic functional groups in biochar, such as carboxylic and phenolic groups, may influence proton exchange reactions and partially explain the observed behavior (GEZAHEGN; SAIN; THOMAS, 2019).

CONCLUSIONS

Under the short-term soil column conditions evaluated in this study (30 days), biochar application altered soil chemical properties and salt dynamics under saline irrigation. Increasing biochar rates promoted the mobilization and leaching of soluble ions, as evidenced by higher ion concentrations in the leachate over time, thereby reducing soil electrical conductivity (EC).

The increase in Na concentrations in the leachate with higher biochar rates suggests a contribution of this element from the biochar itself, although the specific mechanisms governing its release were not directly assessed in this study. Therefore, higher application rates (20–40 Mg ha⁻¹), while effective in promoting salt leaching, may also increase Na inputs to the system and, consequently, elevate the risk of soil sodicity. These findings indicate that the effects of biochar depend on its chemical composition, particularly its Na content, highlighting the importance of carefully selecting feedstocks and application rates for use under saline irrigation conditions.

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