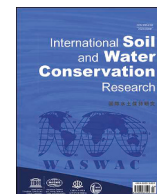




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Original Research Article

Integrated organochemical – Microbial solutions remediate degraded saline-sodic soils

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ABSTRACT

Projections of global population growth underscore the urgent need to restore degraded saline-sodic soils to meet rising demands for food, feed, and fiber. This study evaluated the individual and combined effects of gypsum (G), elemental sulfur (S), vermicompost (VC), biochar (B), and microbial inoculation on soil remediation. A comprehensive soil degradation index (CSDI) was developed with total (CSDI-T) and minimum datasets (CSDI-M) using 13 soil properties. All treatments significantly improved soil health ($p < 0.05$), with G + VC and S + VC combinations reducing CSDI-T by 84–85 % and 65–71 % and CSDI-M by 84–87 % and 66–70 %, respectively. Soil remediation rates correlated directly with crop yield, with CSDI models explaining 29–87 % of the variance in wheat yield. These findings highlight G/S + VC treatments as cost-effective, environmentally sustainable solutions for soil restoration and productivity enhancement, with CSDI models offering robust tools for evaluating amendment strategies.

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1. Introduction

Remediating degraded soils is crucial for meeting the increasing demands of a growing global population and ensuring long-term food security. As both global population and income levels rise, food production must increase by 70 % worldwide—and by as much as 100 % in developing countries—by 2050 (Dubois, 2011). This increased demand for agricultural commodities will exert additional pressure on soil and water resources, exacerbating land degradation. Soil degradation undermines the soil's capacity to deliver vital agroecosystem services, including nutrient cycling, water regulation, and sustained crop productivity (Rezapour et al., 2022). Among the leading contributors to land degradation are salinity and sodicity, which stem from both natural processes and

human activities. These factors critically impair the sustainability, productivity, and resilience of agricultural systems under environmental stress (Alcívar et al., 2018; Dubois, 2011).

Saline-sodic soils represent a particularly severe challenge due to their impaired structure and diminished functionality. These soils are characterized by poor structural integrity, high pH, electrical conductivity (EC) greater than 4 dS m^{-1} , and/or high levels of sodium (Brady, 1984; Chen et al., 2023; Garcia-Franco et al., 2021). In more intensive cases, the accumulation of excessive soluble salts and high exchangeable sodium percentage ($\text{ESP} > 15 \%$) leads to clay dispersion, thereby reduced aggregate hierarchy and stability, impaired water infiltration, and reduced soil porosity (Outbakat et al., 2022; van de Craats et al., 2020). As a result, saline-sodic soils exhibit high erodibility, reduced water storage, and poor crop yield (Kim et al., 2017; Rezapour et al., 2022). Nutrient imbalances and ion toxicities in these soils can further decline soil fertility, declining crop growth and productivity (Brady, 1984). The economic toll of soil salinity is estimated \$27.3 billion in annual global losses due to reduced crop yields (Qadir et al., 2014).

Globally, salinity and sodicity affect approximately 1.257 billion

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hectares across 118 countries (FAO, 2022), with expanding degradation due to climate change and unsustainable land use practices. Saline and sodic soils occupy around 34 million hectares, representing more than 20 % of the country's total land area (FAO, 1979). Continued changes in climate patterns is expected to exacerbate soil salinization and sodicity, particularly through rising sea levels, increased evapotranspiration, and altered precipitation patterns (Teron et al., 2024). In coastal regions, sea level rise leads to saltwater intrusion into freshwater systems, which results in increased soil salinity (Hu et al., 2022). Similarly, higher temperatures elevate evaporation rates, leaving salts behind in the soil profile (Wong et al., 2008). Studies suggest that the intensification of these climate-driven processes could increase the area of salt-affected soils by 50 % by 2050, particularly in vulnerable regions like the Middle East, North Africa, and South Asia. As a result, climate change could severely undermine soil remediation efforts and food security (Wong et al., 2008).

The expansion of these degraded soils poses a serious threat to food security by reducing arable land and negatively impacting crop yields (Rezapour et al., 2022; Wang et al., 2021). Therefore, rehabilitating saline-sodic soils is essential for optimizing agricultural production and achieving sustainable development goals, especially in regions where expanding cropland is not feasible.

Restoring saline-sodic soils, however, presents significant challenges due to their compromised structure, unfavorable chemical composition, and low organic matter content (Brady, 1984; Lin et al., 2025). Various remediation methods, including physical (e.g., plowing, subsoiling, leaching), chemical (e.g., gypsum, sulfuric acid, phosphogypsum), and organic amendments (e.g., farm manure, compost, biochar), have been applied to improve soil properties (Alcivar et al., 2018; Chen et al., 2023; Kim et al., 2017; Miranda et al., 2018; Rezapour et al., 2022; Singh et al., 2022). Organic amendments such as biochar and poultry manure have shown significant potential in improving soil structure, fertility, and biological activity in saline-sodic soils. Biochar, a carbon-rich material derived from the pyrolysis of organic matter, has been proven to enhance soil physical properties by improving aggregate stability, water retention, and porosity (Lehmann & Joseph, 2024). Additionally, biochar helps in cation exchange, reducing the negative impact of sodium on soil structure. Poultry manure, rich in organic matter and nutrients, improves soil microbial activity, promotes nutrient cycling, and enhances soil water-holding capacity (Yuan et al., 2024). Incorporating organic amendments not only improves soil structure but also aids in long-term carbon sequestration and enhances soil resilience to climate change. However, research on the synergistic effects of combined organochemical remediation techniques remains limited. Furthermore, it is unclear how different soil degradation indicators respond to these integrated approaches in saline-sodic soils.

Many current studies on soil degradation focus on isolated physical or chemical indicators, such as bulk density, organic matter content, or aggregate stability. However, these individual measures may not capture the complex interactions between soil physical, chemical, and biological processes that drive degradation in saline-sodic soils (McBratney et al., 2014). Comprehensive soil degradation indices, such as the Total Soil Quality Index (TSQI) and the Comprehensive Soil Degradation Index (CSDI), offer a more holistic approach by integrating multiple soil parameters. These indices allow for a more accurate assessment of soil health and the effectiveness of remediation strategies. Despite their potential, few studies have fully developed or implemented such indices in saline-sodic soils, particularly in the context of combined remediation approaches. In addition to the lack of combined remediation studies, there is no established framework for evaluating CSDI in saline-sodic soils. The CSDI is an essential tool for assessing soil

health, as it integrates multiple soil degradation indicators (e.g., soil texture, bulk density, organic matter content, aggregate stability, and the K factor) into a single index value, allowing for a more holistic evaluation of soil quality and sustainability (Chen et al., 2022; Wang et al., 2018). Developing a reliable CSDI can provide a more accurate assessment of soil erodibility and the effectiveness of remediation practices.

This study aims to (1) evaluate the effects of combined remediation techniques on individual soil degradation indices in saline-sodic soils, (2) identify the soil traits most critical to degradation in these soils, and (3) develop comprehensive soil degradation indices using both total and minimum dataset approaches to assess the impact of amendments effectively. We hypothesize that the combined application of organochemical - microbial amendments will synergistically enhance saline-sodic soil remediation by improving individual soil degradation indices and CSDs.

2. Materials and methods

2.1. Study region and field campaign

Composite soil samples (five subsamples) were collected from 0 to 20 cm soil profile. The sampling site was selected within the Jebel Kandi series, identified as one of the most significant and extensive series of saline-sodic soils in the study region. A soil profile was excavated within the series, which was designated as the primary and central sampling points. Composite soil samples were then collected from a distance of 10–15 m from the central point, extending in the north-south and east-west directions. The sampling area encompasses a significant portion of saline-sodic soils, which is characterized by a semi-arid climate, with an average annual precipitation of approximately 330 mm and a mean annual temperature of around 13 °C. These soils are generally classified as Sodic Calcixerepts and Calcic Solonetz according to the U.S. Soil Taxonomy and the World Reference Base (WRB) system, respectively. Soil analyses indicated high values for pH, calcium carbonate (CaCO₃), EC, and sodium adsorption ratio (SAR), along with moderate cation exchange capacity (CEC) and low levels of organic carbon and micronutrients (Fe, Mn, Zn, and Cu) (Table 1).

2.2. Experimental design

A Randomized Complete Block Design (RCBD) with factorial treatments was employed. Factor 1 consisted of seven remediation treatments with microbial inoculation (T + MI), while Factor 2 involved the same treatments without microbial inoculation (T - MI), with each treatment replicated three times. The seven treatments included control (C), gypsum (G), elemental sulfur (S),

Table 1
The chemical properties of the study soil and biochar and vermicompost.

Parameter	Unit	Soil	Biochar	Vermicompost
pH		9.2	8.2	7.9
EC	dSm ⁻¹	10.4	0.79	1.6
SAR		18.1	3.2	2.8
OC	%	0.36	65	25
Total N	%	0.036	0.68	2.03
P	mg kg ⁻¹	5.7	0.28	0.56
K	mg kg ⁻¹	154.8	0.41	0.40
Fe	mg kg ⁻¹	2.2	330.1	897.4
Mn	mg kg ⁻¹	2.6	33	537
Zn	mg kg ⁻¹	0.36	62.1	170
Cu	mg kg ⁻¹	0.16	38.7	48

EC: electrical conductivity, SAR: sodium adsorption ratio, OC: organic carbon.

vermicompost (VC), biochar (B), gypsum + vermicompost (G + VC), and elemental sulfur + vermicompost (S + VC). Biochar and vermicompost were applied at a rate of 90 kg ha⁻¹. The biochar was produced through the slow pyrolysis of grape wood at 400 °C under oxygen-limited conditions, while vermicompost was generated from plant residues (e.g., deciduous leaves) and cow manure, maintained at 80 % moisture content during the vermicomposting process (Gong et al., 2018). The major chemical properties of both biochar and vermicompost are presented in Table 1. To estimate the gypsum requirement (GR), the Eq. (1) was utilized:

$$GR = \left(\frac{ESP_i - ESP_f}{100} \right) \times CEC \quad (1)$$

where ESP_i and ESP_f represent the initial (20.2 %) and desired (8 %) values of the exchangeable sodium percentage, respectively, and CEC represents the cation exchange capacity (cmolc kg⁻¹). To reduce the initial soil ESP, gypsum and elemental sulfur were applied at rates of 2.0 g kg⁻¹ and 0.4 g kg⁻¹, respectively.

2.3. Soil incubation and leaching column experiment

For all treatments, soil and amendments were thoroughly mixed in separate plastic bags. The treated soils were incubated in covered plastic containers at a temperature of 25–27 °C and maintained at field capacity moisture for 5 months. Following incubation, the soils were packed uniformly into leaching columns made of PVC cylinders (40 cm in length and 20 cm in inner diameter). To prevent clogging of the drainage hole, each cylinder's base was filled with acid-washed sand (< 0.2 mm) to a height of 2.5 cm. Soil was packed uniformly into the cylinders to minimize air pockets and achieve a bulk density representative of field conditions (1.3 g cm⁻³) (Chaganti & Crohn, 2015; Sundha et al., 2020). Leaching was initiated by supplying water from the top (steady-state water flow), and the effluent was collected continuously using a Mariot device. Electrical conductivity (EC) and sodium adsorption ratio (SAR) were regularly measured after collecting each pore water sample. The leaching process continued until the effluent stabilized, and cumulative pore volume was recorded once EC and SAR levels dropped below critical thresholds (EC < 4 dS m⁻¹, SAR < 13). The concentration of soluble salts in the pore water was also measured, and breakthrough curves for each soil were plotted during the drainage process. A mean leaching requirement (LR) of 0.36 was determined for all treatments after the column experiment, which was subsequently applied at each stage of wheat irrigation during pot cultivation.

2.4. Pot experiment

Following the column experiment, the processed soil was transferred to 3 kg pots for a greenhouse study involving 42 pots over five months. In each pot, 20 spring wheat seeds were planted, and after approximately two weeks, five seedlings were maintained per pot. The soil in the pots was fertilized (e.g., with urea and superphosphate) according to soil testing results and regularly irrigated to maintain near field capacity moisture. For treatments in Factor 1 (with microbial inoculation), seeds were inoculated with a microbial solution (*Thiobacillus* sp.) containing 2 % concentration and a minimum bacterial population of 10⁸ CFU per gram. Based on earlier research (Rouydel et al., 2021), these

bacteria were extracted from salt-affected soils in the vicinity of Lake Urmia and screened by their plant-growth promoting characteristics (PGPs), which encompass indole acetic acid (IAA), hydrogen cyanide (HCN), siderophore, and exopolysaccharide (EPS). The bacteria were cultivated in a nutrient broth (NB) medium to reach a colony-forming unit (CFU)/mL concentration of 10⁸. After 150 days, the plants were harvested, and root length and diameter were measured. The wet and dry biomass yields per pot were recorded. Soil samples from the pots were carefully collected, air-dried, crushed, and passed through a 2-mm mesh sieve.

2.5. Soil analyses

Soil samples collected from all pots were analyzed for various soil degradation indicators using standard methods. These indicators included soil particle size distribution, bulk density (BD), mean weight diameter (MWD) of soil aggregates, soil stability index (SSI), cation ratio of soil structural stability (CROSS), soil crusting index (SCI), clay dispersion ratio (CDR), K factor of soil erodibility (K-factor) (Renard et al., 1994), EC, organic matter, SAR, exchangeable sodium percentage (ESP), and exchangeable sodium (Na). The sand, clay, and silt content of the samples were measured by the Bouyoucos hydrometer method (Gee & Or, 2002). Soil BD and MWD were estimated using the core method (Grossman & Reinsch, 2002) and the wet-sieving method (Kemper & Rosenau, 1986), respectively. Soil pH and EC were determined from a saturated paste and its extract (Sparks et al., 2020). Soil organic carbon (SOC) was estimated by the wet oxidation method using K₂Cr₂O₇ and H₂SO₄ (Nelson & Sommers, 1982). To convert SOC percentage to soil organic matter (SOM) percentage, SOC was multiplied by 1.75. Soil cation exchange capacity was determined using 1 N NH₄OAc buffered at pH 8.2 (Sparks et al., 2020). The saturation extract was used to extract soluble cations (Sparks et al., 2020), with concentrations specified using titration methods for calcium and magnesium analysis and a flame photometer for sodium analysis. Sodium absorption ratio (SAR), exchangeable sodium, and exchangeable sodium percentage (ESP) were estimated using methodologies suggested by the U.S. Salinity Laboratory (Richards, 1954).

The SSI, CROSS, SCI, and CDR were respectively computed using Eqs. (2)–(5) as follows (Dong et al., 1983; FAO, 1979; Pieri, 2012; Rengasamy & Marchuk, 2011).

$$SSI = \frac{1.72 \times OC}{P_c + P_{Si}} \times 100 \quad (2)$$

$$CROSS = \frac{Na^+ + 0.56K^+}{\sqrt{Ca^{2+} + 0.6Mg^{2+}/2}} \quad (3)$$

$$SCI = \frac{1.5 Si_f + 0.75 Si_c}{Cl + (10 \times OM)} \quad (4)$$

$$CDR = \frac{Clay_w}{Clay_c} \quad (5)$$

Where OC, P_c, and P_{Si} are the percentage of organic carbon, clay, and silt, respectively, in Eq. (2). In Eq. (3) the concentrations of Na, K, Ca and Mg are expressed in millimoles of charge L⁻¹. In Eq. (4), Si_f, Si_c, Cl, and OM are the percentage of fine silt, coarse silt, clay, organic matter. In Eq. (5), Clay_w and Clay_c are clay content in water-dispersed and Calgon – dispersed samples, respectively.

Following equation (EPIC model) was used to calculate K factor of soil erodibility (Williams et al., 1984).

$$K = \left\{ 0.2 + 0.3 \exp \left[-0.0256 S_a \left(\frac{1 - S_i}{100} \right) \right] \right\} \left(\frac{S_i}{Cl + S_i} \right)^{0.3} \left(1.0 - \frac{0.25 OC}{OC + \exp(3.72 - 2.95 OC)} \right) \left(1.0 - \frac{0.7 SN1}{SN1 + \exp(-5.51 + 22.9 SN1)} \right) \quad (6)$$

Where K is the K factor calculated by the EPIC model ($t \ ha \ h \ ha^{-1} \ MJ^{-1} \ mm^{-1}$). The S, Si, Cl, and OC are the percentage of sand, silt, clay, and organic carbon, respectively, and SN1 is equals to $1 - S_a/100$.

2.6. Soil degradation index

In order to comprehensively evaluate soil degradability, a comprehensive soil degradation index (CSDI) was computed using both total dataset (TDS) and minimum dataset (MDS) approaches as follows (Chen et al., 2022; Wang et al., 2018; Wang & Zhang, 2021).

$$CSDI = \sum_{i=1}^n W_i \times S_i \quad (7)$$

where S_i is the score each soil degradability indicator, W_i is the weighting coefficient of soil degradation indicators derived from the factor analysis, and n is the number of selected soil degradation indicators determined. For both TDS and MDS approach, factor analysis was employed to assign weights to the variables in accordance with their communalities. Each factor's communality accounted for a specific percentage of the variance observed in the TDS and MDS. The weight assigned to each indicator was calculated by determining the ratio of the communality of that indicator to the total sum of all communalities assessed within the TDS and MDS approaches (Andrews et al., 2004; Mamehpour et al., 2021). The score (ranging from 0 to 1) of soil degradation indicator was calculated by the membership function related to the indicators and soil degradability as the following equations (Wang et al., 2018)

$$S(x) = \begin{cases} 1 & x \geq b \\ \frac{x-a}{b-a} & a < x < b \\ 0 & x \leq a \end{cases} \quad (8)$$

$$S(x) = \begin{cases} 1 & x \leq b \\ \frac{x-b}{b-a} & a > x > b \\ 0 & x \geq a \end{cases} \quad (9)$$

where $S(x)$ is the membership function, x is the value of the soil degradation indicator, and a and b are the lower and upper limits of the soil degradation indicator, respectively. The indicators that had positively correlated with soil degradation (e.g., BD and SCI) were calculated using Eq. (8), whereas those that had negatively correlated with soil degradation (e.g., OM and MWD) were calculated by Eq. (9). The soil degradation indicators were selected based on their influence and impact on soil degradation as well as

the productivity of agroecosystems (Mamehpour et al., 2021; Wang et al., 2018). A principal component analysis (PCA) was

performed on the standardized data matrix of the TDS to pinpoint significant indicators within the dataset and to determine the MDS. Key indicators were identified as principal components (PCs) with eigenvalues ≥ 1 , which accounted for at least 5 % of the total variance. For each PC, indicators with factor loading values that were within 10 % of the highest loading were included in the MDS dataset (Andrews et al., 2004). Where multiple soil indicators were significantly important within a PC, the Pearson correlation coefficient was employed to assess the redundancy of these variables.

2.7. Statistical analyses

Mean comparison was conducted using Duncan's significant difference test at the 5 % confidence interval. Principal Component Analysis (PCA) was utilized to determine the weights of soil degradation indicators for assessing CSDI. The Kaiser-Meyer-Olkin (KMO) test and Bartlett's test of sphericity were applied to evaluate the suitability of PCA for effectively sorting the soil degradation indicators for factor analysis. All statistical analyses were performed using SPSS (version 16.0, SPSS Inc.).

3. Results

3.1. Chemical indicators of soil degradation

The majority of soil degradation chemical indicators, including pH, electrical conductivity (EC), sodium adsorption ratio (SAR), exchangeable sodium (Na), exchangeable sodium percentage (ESP), and organic matter (OM), showed significant improvement in response to soil remediation practices (Table 2). Both inoculated and non-inoculated treatments significantly reduced pH, EC, SAR, and ESP compared to the control. The greatest pH reduction occurred in the gypsum + vermicompost (G + VC) and sulfur + vermicompost (S + VC) treatments, with pH decreasing by 1.42–1.45 and 1.32–1.38 units, respectively, compared to the control. The EC values decreased significantly in the order of $G + VC > S + VC > G > S > \text{biochar (B)}$, with reductions ranging from 130 to 142 % for combined treatments (e.g., G + VC) and 59–119 % for individual treatments (e.g., vermicompost (VC)). SAR, exchangeable Na, and ESP were also reduced across all treatments, with the most significant reductions observed in combined treatments. SAR decreased by 1.83–1.87 times (G + VC) and 1.5 to 1.63 times (S + VC), while ESP decreased by 1.72–1.77 times (G + VC) and 1.34 to 1.53 times (S + VC) compared to the control.

Additionally, VC and B treatments significantly increased OM levels, with increases ranging from 0.42 to 1.7 times for individual treatments, and 2.2 to 2.4 times for the G + VC treatment. Microbial inoculation further enhanced the performance of the treatments in regulating pH, EC, SAR, exchangeable Na, ESP, and OM, though the differences were not statistically significant across all treatments (Fig. 1).

Table 2
Soil degradation chemical indicators of saline-sodic soils across treatments.

Experimental factor	Experimental tretment	pH	EC (dSm ⁻¹)	SAR	Na - Ex (cmol kg ⁻¹)	ESP (%)	OM
With microbial inoculation	C	9.35 a	4.6 a	12.9 a	1.70 a	14.7 a	0.66 e
	G	8.07 b	2.1 c	5.1cd	0.88 c	6.4 b	0.75 de
	S	8.04 b	2.7 b	5.4 c	0.89 c	6.6 b	0.86 de
	VC	8.10 b	2.1 c	6.2 b	0.88 c	6.0 bc	1.79 b
	B	8.52 ab	2.8 b	6.2 b	0.98 b	7.3 b	0.93 c
	G + VC	7.90 bc	1.9 d	4.5 e	0.80 e	5.3 e	2.23 a
	S + VC	7.97 bc	2.0 d	4.9 cd	0.85 cd	5.8 d	1.77 b
Without microbial inoculation	C	9.35 a	4.8 a	13.0 a	1.72 a	15.0 a	0.63 e
	G	8.17 b	2.3 c	5.4 c	0.90 c	6.6 c	0.68 de
	S	8.23 b	2.8 b	5.5 c	0.91 c	7.2 cd	0.72 de
	VC	8.30 b	2.4 c	6.3 b	0.89 cd	7.8 b	1.64 b
	B	8.69 a	2.9 b	6.6 b	1.10 b	8.3 b	0.85 cd
	G + VC	7.93 bc	2.0 d	4.6 e	0.82 e	5.5 f	2.05 a
	S + VC	8.03 bc	2.0 d	5.2 d	0.86 d	6.4 e	1.68 b

C: control, G: gypsum, S: elemental sulfur, VC: vermicompost, B: biobhar, EC: electrical conductivity, SAR: sodium absorption ratio, Na-Ex: Na- Exccangeable, ESP: exchangeable sodium percentage, OM: organic matter. For each factor, different letters in each column show significant differences at $p < 0.05$ according to the Fisher's LSD.

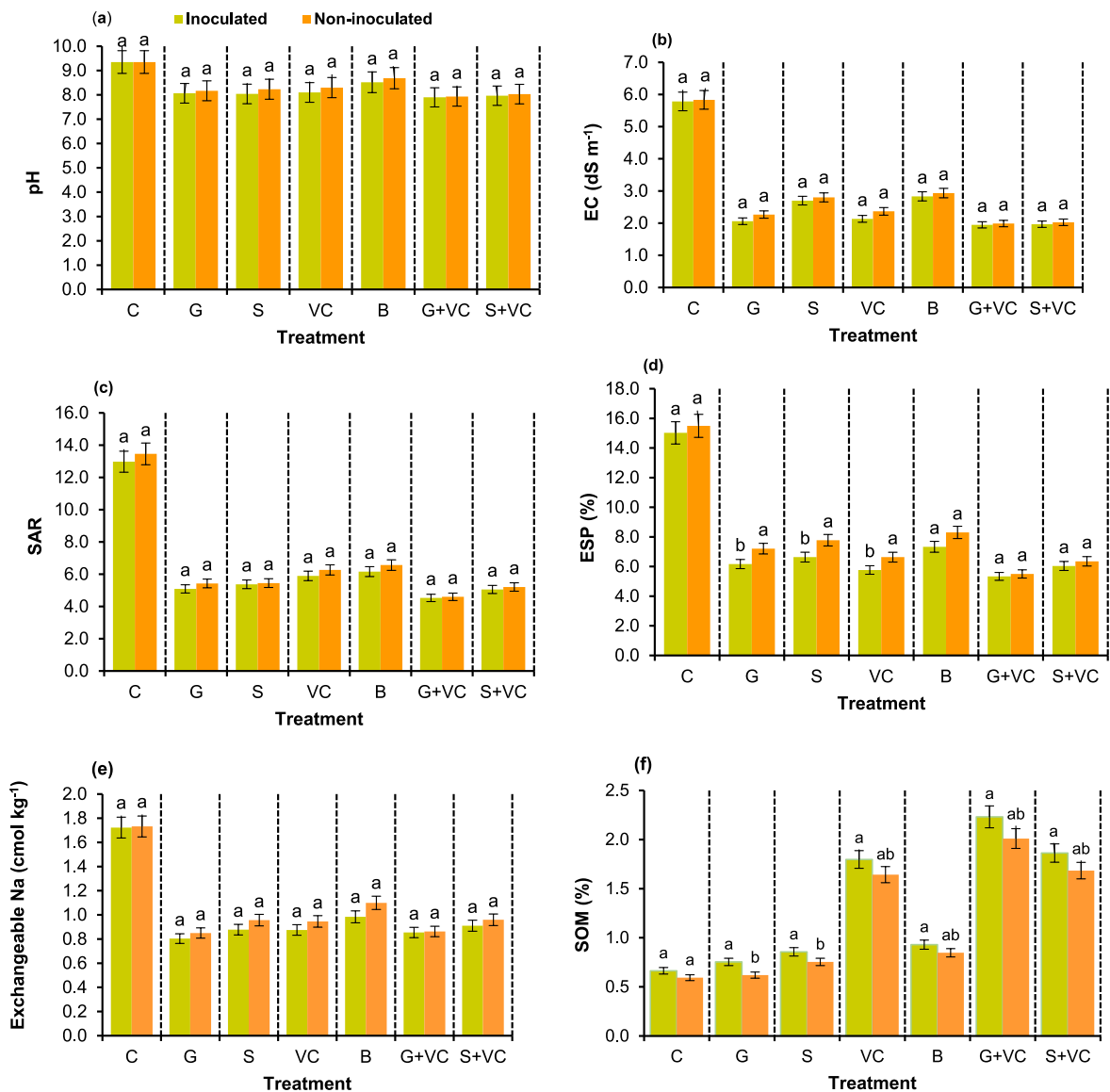


Fig. 1. The comparison of the mean values of soil degradation chemical indicators between experimental factors. Different letters in each treatment represent the significant differences at $p < 0.05$ between the inoculated and non-inoculated factors according to Fisher's LSD.

Table 3
Soil degradation physical indicators of saline-sodic soils across treatments.

Experimental factor	Experimental tretment	BD gcm ⁻³	MWD mm	SSI %	CROSS	SCI	CDR	K
With microbial inoculation	C	1.58 a	0.41 f	1.66 f	27.33 a	0.84 a	11.99 a	0.052 a
	G	1.50 b	0.42 f	2.14 d	10.20 de	0.82 a	4.11cd	0.051 a
	S	1.55 a	0.44 f	1.88 e	10.81 d	0.79 a	4.39 c	0.051 a
	VC	1.48 b	0.61 c	4.47 c	12.52 c	0.60 b	5.18 b	0.047 b
	B	1.49 b	0.46 d	2.32 d	13.28 b	0.77 a	5.32 b	0.050 a
	G + VC	1.43 b	0.70 a	5.58 a	8.98 f	0.53 c	3.55 e	0.044 c
	S + VC	1.45 b	0.63 b	4.62 b	10.12 e	0.58 b	4.07 d	0.047 b
Without microbial inoculation	C	1.59 a	0.40 d	1.48 gh	28.39 a	0.87 a	12.48 a	0.053 a
	G	1.51 b	0.40 d	1.88 g	10.93 c	0.86 a	4.45 cd	0.052 a
	S	1.56 a	0.42 d	1.55 f	10.98 c	0.82 a	4.47 cd	0.052 a
	VC	1.49 b	0.59 c	4.11 c	12.75 b	0.62 b	5.29 b	0.049 b
	B	1.51 b	0.44 d	2.12 d	13.41 b	0.79 a	5.59 b	0.051 a
	G + VC	1.45 b	0.66 a	5.11 a	9.13 d	0.56 c	3.62 e	0.046 c
	S + VC	1.46 b	0.60 b	4.45 b	10.45 c	0.61 b	4.23 d	0.048 b

BD: bulk density, MWD: mean weight diameter of soil aggregates, SSI: soil stability index, CROSS: the cation ratio of soil structural stability, SCI: soil crust index, CDR: clay dispersion ratio, K: K factor of soil erodibility.

3.2. Physical indicators of soil degradation

The mean comparisons of selected soil degradation physical indicators, including bulk density (BD), mean weight diameter (MWD), soil stability index (SSI), cation ratio of soil structural stability (CROSS), soil crusting index (SCI), and clay dispersion ratio (CDR) for different treatments are presented in Table 3. All treatments, except sulfur (S), reduced BD values compared to the control, with reductions occurring in the order: G + VC > S + VC > VC > B > G > S.

Similarly, all treatments significantly decreased CDR, CROSS, and SCI compared to the control, with combined treatments (G + VC and S + VC) showing the greatest effectiveness in regulating these indicators. Reductions in CDR, CROSS, and SCI ranged from 2.9 to 3.45 times, 2.7 to 3.1 times, and 1.42 to 1.55 times, respectively. Additionally, treatments significantly increased MWD and SSI compared to the control, with the highest improvements seen in combined treatments, followed by individual organic treatments (VC and B) and chemical treatments (G and S).

For the soil erodibility factor (K-factor), all treatments showed a reduction compared to the control, with greater improvements seen in inoculated treatments. The synergistic treatments (G + VC and S + VC) had the most substantial effects, with improvements ranging from 9.5 % to 17 %. Microbial inoculation further decreased K-factor values across all treatments, though these changes were not statistically significant (Fig. 2).

3.3. Developing soil degradation index

A comprehensive soil degradation index (CSDI) was developed using both the total dataset (TDS) and minimum dataset (MDS) approaches to assess soil the degradability comprehensively. The CSDI was categorized into five degradation levels (I to V), where higher values indicated greater soil degradation (Table 4).

3.3.1. The TDS index approach

The CSDI based on the TDS approach (CSDI-T) used 12 soil degradation indicators, including MWD, CROSS, CDR, BD, SCI, SSI, K-factor, OM, EC, SAR, ESP, and exchangeable Na. The highest contributions to the CSDI-T were from OM and K-factor, while EC and exchangeable Na had the lowest contributions (Table 5). A mean reduction of 37.1 %–85.4 % in CSDI-T values was observed for inoculated treatments, and 34.8 %–84.5 % for non-inoculated treatments compared to the control (Table 6). Treatments

improved CSDI-T grades, moving from Grade V (very high degradation) in the control to Grade IV (high degradation) for individual treatments and to Grade II (low degradation) for combined treatments (Fig. 3). Combined treatments (e.g., G + VC, S + VC) showed the most significant impact (see Fig. 4) (see Table 7).

3.3.2. The MDS index approach

Principal component analysis (PCA) was used to develop the MDS dataset, reducing the number of soil indicators to five: MWD, K-factor, OM, CROSS, and ESP. Based on the MDS dataset, the following CSDI-M models were formulated:

For inoculated treatments:

$$\begin{aligned} CSDI - M = & 0.211(OM) + 0.207(K - factor) + 0.196MWD \\ & + 0.194ESP + 0.192CROSS \end{aligned} \quad (10)$$

For non-inoculated treatments:

$$\begin{aligned} CSDI - M = & 0.205(K - factor) + 0.202OM + 0.201ESP \\ & + 0.199CROSS + 0.193MWD \end{aligned} \quad (11)$$

Inoculated treatments showed a reduction in CSDI-M values ranging from 35 % to 87 %, with the greatest improvements observed in combined treatments (G + VC and S + VC) (Table 6). These treatments shifted the soil degradation grade from IV (control) to II and III for synergistic and individual treatments, respectively (Fig. 3) (see Table 8).

3.4. Wheat growth attributes and CSDI

Wheat growth attributes, including wet biomass weight (WBW), dry biomass weight (DBW), root length (RL), and root volume (RV), showed significant improvements across most treatments due to soil remediation techniques (Table 9). The increases were as follows: RL (120–264 %), RV (4–28 %), WBW (20–400 %), and DBW (15–403 %) compared to the control. The order of effectiveness for the amendments was: G + VC > S + VC > VC > G > B > S.

Inoculated treatments, particularly G + VC and S + VC combinations, further enhanced wheat growth parameters, with improvements of 6–16 % for RL, 2–5 % for RV, 10–76 % for WBW, and 13–77 % for DBW compared to non-inoculated treatments, although the changes in root length and volume were not

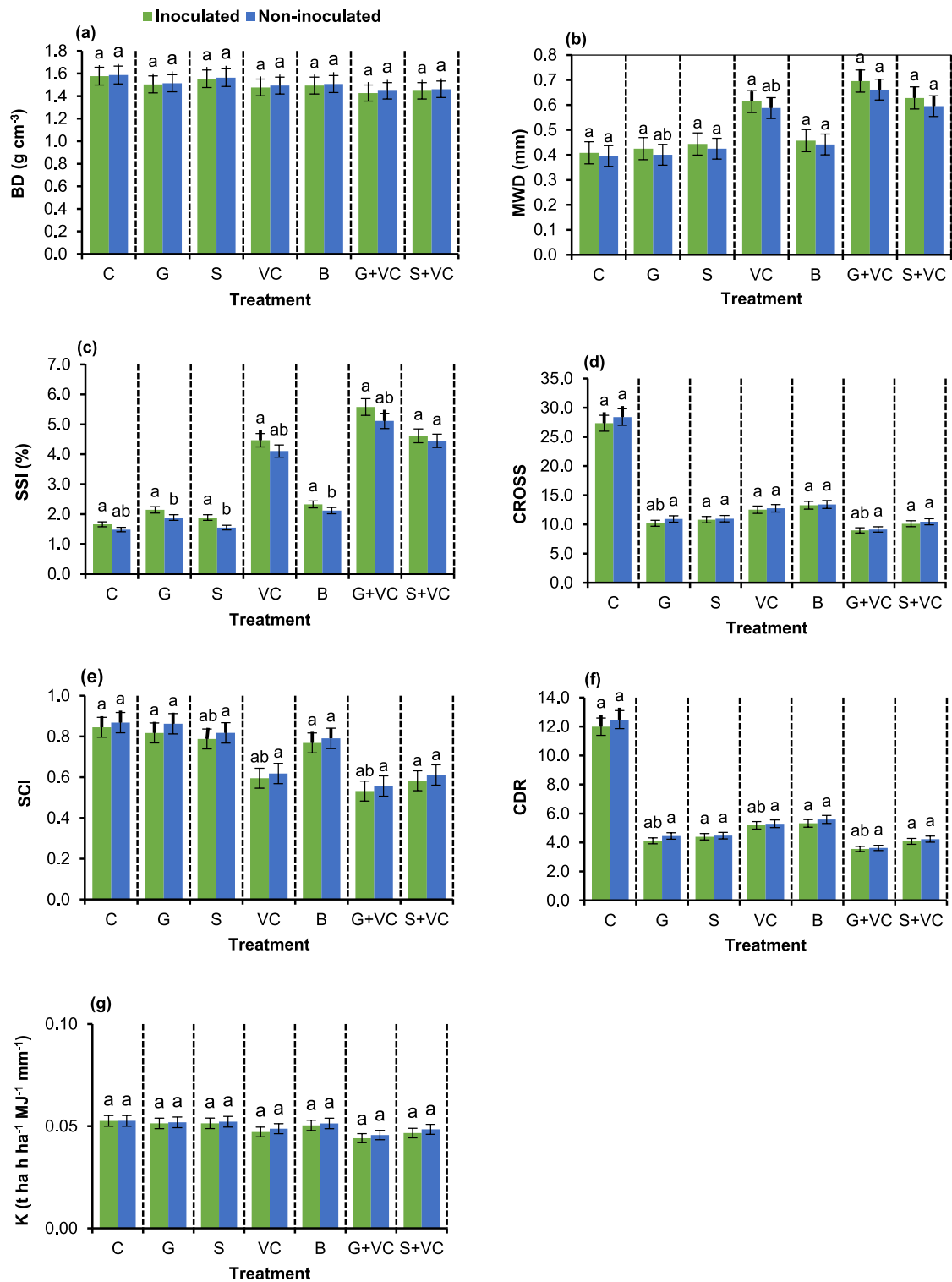


Fig. 2. The comparison of the mean values of soil physical degradation indicators between experimental factors. Different letters in each treatment represent the significant differences at $p < 0.05$ between the inoculated and non-inoculated factors according to Fisher's LSD.

Table 4
Categorization of comprehensive soil degradation index.

CSDI model	V (Very high)	IV (High)	III (Moderate)	II (Low)	I (Very low)
CSDI - T	>0.63	0.46–0.63	0.29–0.46	0.12–0.29	<0.12
CSDI - M	>0.64	0.47–0.64	0.3–0.47	0.13–0.30	<0.13

Table 5
Commonality and weight of soil degradation indicators for TDS and MDS approaches across experimental factors.

Soil degradation indicator	With microbial inoculation				Without microbial inoculation			
	TDS approach		MDS approach		TDS approach		MDS approach	
	Commonality	Weight	Commonality	Weight	Commonality	Weight	Commonality	Weight
MWD	0.912	0.086	0.912	0.196	0.913	0.084	0.913	0.193
CROSS	0.895	0.084	0.895	0.192	0.939	0.086	0.939	0.199
CDR	0.901	0.085			0.969	0.089		
BD	0.868	0.081			0.826	0.076		
SCI	0.923	0.087			0.952	0.087		
SSI	0.948	0.089			0.946	0.087		
K-factor	0.962	0.090	0.962	0.207	0.969	0.089	0.969	0.205
OM	0.98	0.092	0.98	0.211	0.957	0.088	0.957	0.202
EC	0.809	0.076			0.731	0.067		
SAR	0.903	0.085			0.969	0.089		
ESP	0.847	0.079	0.847	0.194	0.949	0.087	0.949	0.201
Na-Ex	0.708	0.066			0.804	0.074		

Table 6
CSDI models of saline-sodic soils across treatments.

Experimental factor	Model	C	G	S	VC	B	G + VC	S + VC
With microbial inoculation	CSDI -T	0.89 a	0.51 c	0.55 b	0.39 d	0.56 b	0.13 f	0.29 e
	CSDI -M	0.92 a	0.58 b	0.55 b	0.38 c	0.57 b	0.13 e	0.28 d
Without microbial inoculation	CSDI -T	0.92 a	0.51 c	0.56 b	0.45 d	0.60 b	0.14 f	0.32 e
	CSDI -M	0.92 a	0.60 b	0.55 c	0.44 d	0.57 bc	0.15 f	0.31 e

C: control, G: gypsum, S: elemental sulfur, VC: vermicompost, B: biobhar, MI: Microbial Inoculation. For each factor, different letters in each row show significant differences at $p < 0.05$ according to the Fisher's LSD.

statistically significant in most treatments (Fig. 5) (see Fig. 6). All wheat growth parameters exhibited significant relationships with both CSDI-T and CSDI-M models, with the strongest correlation observed for RV, followed by RL, DBW, and WBW. The regression equations linking CSDI models to wheat growth (Table 10). The CSDI models accounted for 29 %–87 % of the variability in wheat growth attributes, with CSDI-T showing slightly better performance than CSDI-M in predicting wheat growth. A strong positive relationship between CSDI-T and CSDI-M was observed ($\text{CSDI-T} = 0.94 \times \text{CSDI-M} + 0.024$, $R^2 = 0.94$, $p < 0.01$), suggesting that CSDI-M can provide adequate information for assessing soil degradation similar to CSDI-T.

4. Discussion

4.1. Soil degradation chemical indicators

The results showed that combined treatments, particularly G + VC and S + VC, were highly effective in reducing pH, EC, SAR, and ESP in saline-sodic soils. The reduction in pH observed with these treatments aligns with previous studies that have demonstrated the effectiveness of chemical amendments (e.g., gypsum and sulfur) in improving soil chemical properties when combined with organic amendments like vermicompost. The addition of VC likely enhanced the dissolution of calcium-rich sources, promoting calcium availability for cation exchange, thus reducing soil pH and sodium-related indices (Sheikhi Shahriver & Khademi, 2018). The significant reduction in EC across all treatment combinations suggests that soil amendments not only improved soil physical properties (e.g., aggregate stability, porosity, and permeability) but also promoted the leaching of excess salts from the soil profile (Kim et al., 2017; Liu et al., 2020). This finding is consistent with earlier research, which highlighted that organic matter can increase the soil's buffering capacity and enhance its ability to

retain and adsorb soluble salts (Darwish et al., 2005; Zhang et al., 2020). The reductions in SAR and ESP observed in the combined treatments are likely the result of increased availability of calcium (Ca^{2+}) and magnesium (Mg^{2+}), which replaced sodium ions (Na^+) at cation exchange sites, improving soil structure and reducing salinity and sodicity indicators (Rezapour & Kalashypour, 2019; Sappor et al., 2017). These findings support the notion that the symbiotic use of chemical and organic amendments can effectively restore degraded saline-sodic soils (Alcívar et al., 2018; Schultz et al., 2017). The observed increases in OM, particularly in treatments involving VC and biochar, are consistent with the literature, as organic amendments are known to enhance soil organic matter content, which plays a critical role in improving soil structure, nutrient cycling, and microbial activity (Ding et al., 2021; Rezapour & Kalashypour, 2019). The increase in OM further improves soil health by enhancing physical, chemical, and biological properties, promoting sustainable ecosystem functions (Arora et al., 2017; Brady & Weil, 2004). Finally, although microbial inoculation did enhance the effectiveness of the treatments in regulating pH, EC, SAR, exchangeable Na, ESP, and OM, the effects were not statistically significant across all treatments. This suggests that while microbial inoculation can improve soil remediation processes by producing organic acids and increasing nutrient availability, its overall impact may depend on specific soil conditions and the interactions between microbial communities and soil amendments (Sappor et al., 2017).

4.2. Indicators of soil physical degradation

The observed reductions in BD across all treatments, except sulfur, suggest that soil amendments, particularly those involving organic matter (VC and B), significantly improve soil structure and health. Lower BD values indicate better soil porosity and a

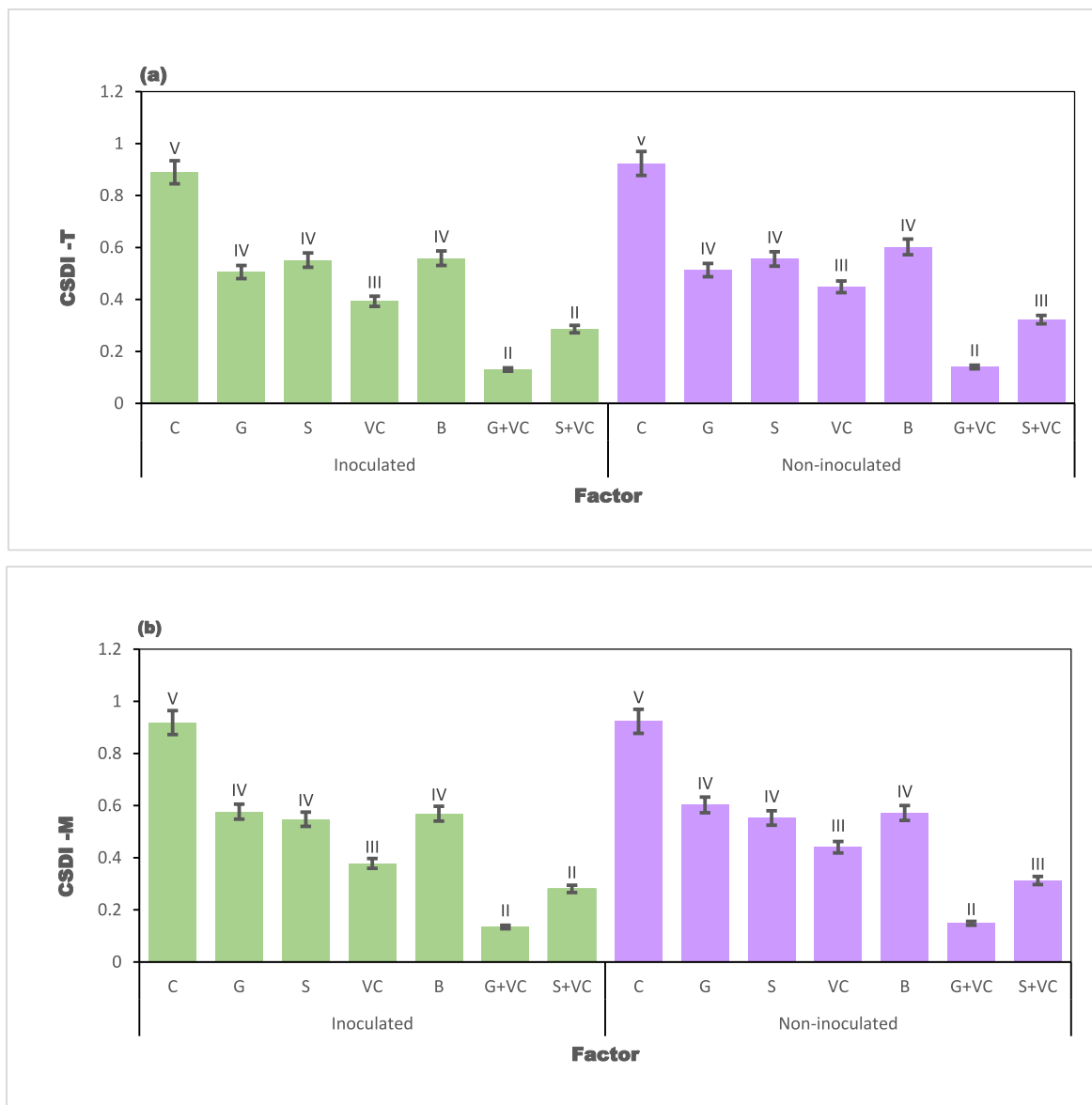


Fig. 3. The comparison of CSD-T (a) and CSDI-M (a) grades across inoculated and non-inoculated treatments. Symbols I, ..., IV indicate the categorization of CSDIs grades (V, very high grade; IV, high grade; III, moderate grade; II, low grade; I, very high grade).

reduction in compaction, which are essential for improved water infiltration and root growth (Brady et al., 2008). Combined treatments like G + VC and S + VC were particularly effective in lowering BD, likely due to the synergistic effects of organic matter and calcium, which enhance aggregate stability and soil structure.

The significant reductions in CDR, CROSS, and SCI highlight the role of combined chemical-organic amendments in improving soil structural stability. Lower values of these indicators suggest higher aggregate stability, which contributes to reduced soil erosion and enhanced resistance to degradation. The combined treatments facilitated colloid flocculation, promoted by the increased availability of Ca^{2+} from gypsum and organic amendments. This process replaces Na^{+} ions, which are detrimental to soil structure, thereby improving the soil's ability to maintain stability (Alcívar et al., 2018; Ding et al., 2021).

The increase in MWD and SSI with organic and combined treatments further supports the idea that organic matter plays a critical role in improving soil aggregation. Organic matter acts as a

binding agent, forming organo-mineral complexes that enhance the stability of soil aggregates. This effect is particularly evident in the combined treatments, which show the highest improvements in MWD and SSI, indicating better soil structure and reduced vulnerability to erosion (Kaplan et al., 2020; Nouri et al., 2019).

The reductions in the K-factor, especially with the addition of VC and microbial inoculation, suggest that soil erodibility decreases when organic matter and microbial activity are enhanced. Organic amendments increase soil porosity and promote aggregate formation, which reduces the susceptibility of the soil to erosion. The positive effects of microbial inoculation on reducing K-factor values may be attributed to the increased production of organic acids and exopolysaccharides by soil microorganisms, which help in the formation of stable soil aggregates (Kim et al., 2017; Rezapour et al., 2022).

Although the changes observed with microbial inoculation were not statistically significant, the trend indicates that microorganisms likely play a supportive role in improving soil physical

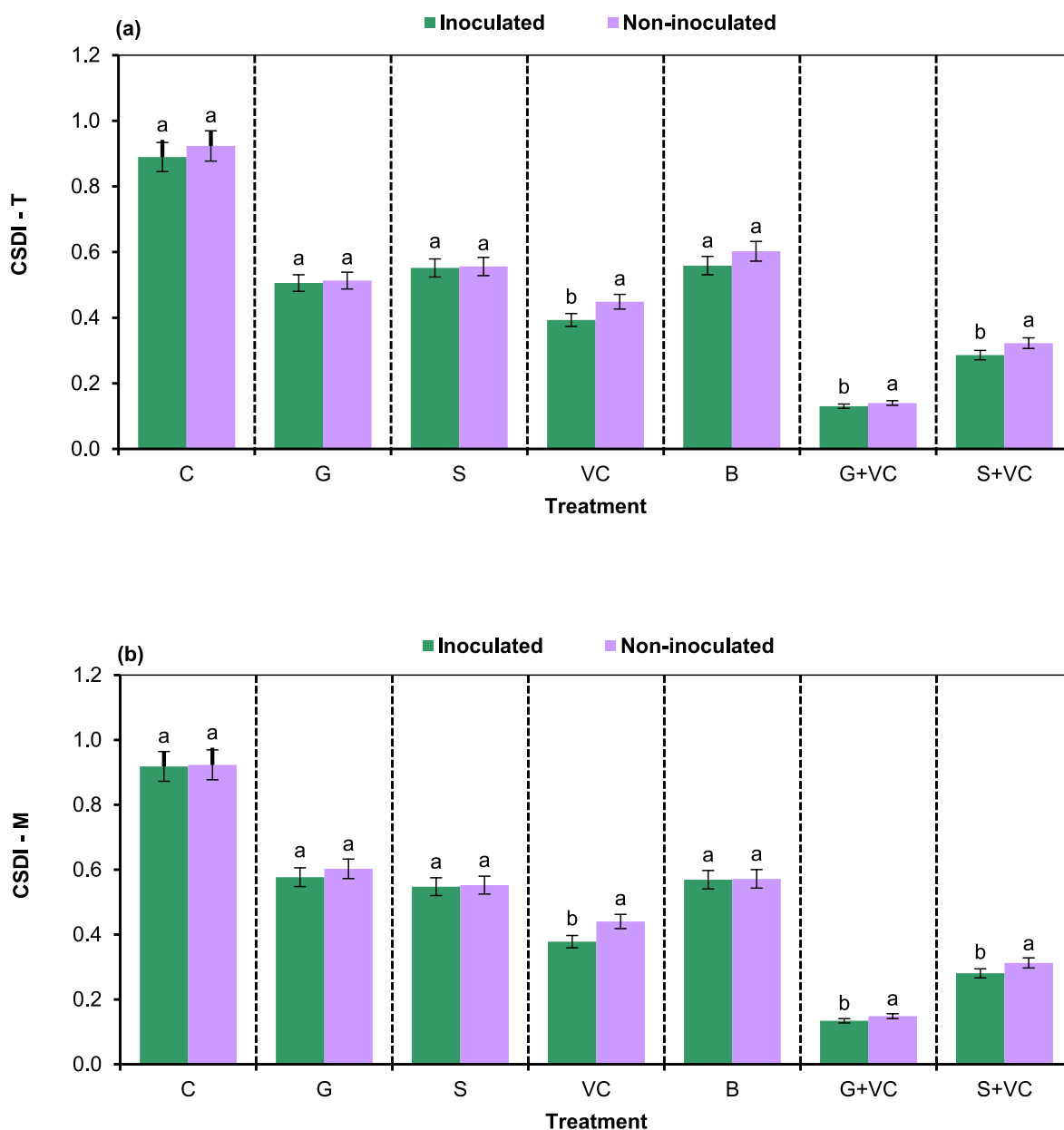


Fig. 4. The comparison of the mean values of CSDI -T (a) and CSDI -M (b) between experimental factors. Different letters in each treatment represent the significant differences at $p < 0.05$ between the inoculated and non-inoculated factors according to Fisher's LSD.

properties by enhancing soil particle rearrangement and increasing water-holding capacity. The formation of micro- and macro-aggregates, supported by microbial exudates, improves soil structure and reduces erosion risks (Nouri et al., 2018; Singh et al., 2020).

As a result, the process of microbial reclamation is comparatively slower than that of chemical amendments, such as gypsum or sulfur, and it may require several growing seasons before notable enhancements in soil health and crop yield can be observed. Furthermore, although microbial inoculation presents a sustainable option for the remediation of saline-sodic soils, its drawbacks including environmental stress, variable performance, and gradual effectiveness underscore the necessity for integrated strategies that combine more efficient microbial, organic, and chemical treatments to achieve optimal outcomes (Naamala & Smith, 2021; Kumawat et al., 2022). Therefore, future research

should prioritize the development of resilient, salt-tolerant microbial consortia and economical delivery systems to address these challenges.

4.3. Soil degradation index

4.3.1. The TDS indicator approach

The TDS approach provided a comprehensive assessment of soil degradation, with OM and K-factor contributing most significantly to the CSDI-T. OM plays a critical role in soil structure, water retention, and nutrient cycling, which are vital for reducing degradation in saline-sodic soils (Brady et al., 2008). The strong positive correlation between OM and CSDI-T ($R^2 = 0.72$) and the negative correlation between K-factor and CSDI-T ($R^2 = 0.75$) indicate that higher organic matter levels help mitigate degradation, while a high K-factor promotes degradation through

Table 7
Results of principal component analysis (PCA) of the selected indicators for the microbial inoculation treatments.

Component	PC1	PC2
Eigenvalue	6.12	4.53
Variability (%)	51.02	37.78
Cumulative (%)	51.02	88.8
Indicators	Eigen vector or factor loading	
MWD	−0.928	−0.224
CROSS	0.384	0.865
CDR	0.397	0.862
BD	0.852	0.376
SCI	0.913	0.299
SSI	−0.901	−0.371
K	0.918	0.345
OM	−0.939	−0.313
EC	0.557	0.706
SAR	0.448	0.838
ESP	0.589	0.841
Na - Ex	0.037	0.707

Table 8
Results of principal component analysis (PCA) of the selected indicators for the microbial non-inoculation treatments.

Component	PC1	PC2
Eigenvalue	5.62	5.29
Variability (%)	46.84	44.08
Cumulative (%)	46.84	90.92
Indicators	Eigen vector or factor loading	
MWD	−0.932	−0.21
CROSS	0.313	0.933
CDR	0.313	0.933
BD	0.807	0.417
SCI	0.883	0.414
SSI	−0.925	−0.301
K	0.916	0.362
OM	−0.947	−0.223
EC	0.53	0.671
SAR	0.312	0.894
ESP	0.341	0.933
Na - Ex	0.195	0.875

increased erodibility (Ding et al., 2021; Rezapour et al., 2022).
The reductions in CSDI-T across both inoculated and non-inoculated treatments demonstrate the effectiveness of organic-

chemical amendments. The most substantial improvements were seen in combined treatments (G + VC, S + VC), which improved soil health by enhancing aggregate stability, nutrient availability, and microbial activity. Vermicompost, rich in nutrients and beneficial microorganisms, likely played a key role in improving soil structure and reducing salinity and sodicity stress, as seen in other studies (Chatterjee & Singh, 2022; Manyanga et al., 2024).

4.3.2. The MDS indicator approach

The MDS approach, by reducing the number of indicators to five, demonstrated a more efficient yet still robust assessment of soil degradation. The key indicators (MWD, K-factor, OM, CROSS, and ESP) are widely recognized for their role in assessing soil physical properties, erodibility, and microbial dynamics (Wang et al., 2018; Zhou et al., 2023).

Eqs. (10) and (11) illustrate the weightings of each indicator in the CSDI-M model for inoculated and non-inoculated treatments. Inoculated treatments prioritized OM, K-factor, and MWD, which directly influence soil health, while non-inoculated treatments gave more weight to K-factor and OM, reflecting their importance in non-microbial environments. The reductions in CSDI-M values, particularly with G + VC and S + VC treatments, indicate that combined organic-chemical amendments effectively mitigate soil degradation (Liu et al., 2020).

The improved CSDI grades across treatments reflect better soil quality, with combined treatments showing the most significant improvements. The shift from Grade IV (high degradation) in the control to Grade II (low degradation) for combined treatments highlights the potential of integrated remediation techniques for restoring saline-sodic soils. These findings align with previous research, emphasizing the effectiveness of chemical-organic symbiosis in enhancing soil health and reducing degradation (Bilgili et al., 2019; Zhang et al., 2020).

4.4. Wheat growth attributes and CSDI

The significant improvements in wheat growth attributes, such as RL, RV, WBW, and DBW, across treatments confirm that soil remediation practices effectively enhance plant growth in saline-sodic soils. Combined treatments like G + VC and S + VC had the most substantial impact, as they improved soil physical attributes (e.g., aggregation and permeability), chemical properties (e.g., pH, EC, SAR), and nutrient availability. These findings are consistent

Table 9
Wheat growth attributes of saline-sodic soils across treatments.

Experimental factor	Treatment	Root length	Root volume	Wet biomass weight	Dry biomass weight
		mm	mm ³	g	
With microbial inoculation	C	11.8 f	52.3 f	1.4 g	0.7 f
	G	29.7 d	57.3 c	4.5 d	2.0 d
	S	25.7 e	54.3 d	1.6 f	0.9 ef
	VC	34.3 c	61.7 b	5.0 c	2.1 c
	B	27.3 e	56.3 c	2.5 e	1.0 e
	G + VC	38.5 a	67.0 a	7.2 a	3.7 a
	E + VC	35.5 b	63.3 ab	5.8 b	2.8 b
Without microbial inoculation	C	9.3 f	51.3 d	1.4 f	0.6 f
	G	28.0 c	55.7 c	2.9 d	1.4 d
	S	22.0 e	52.7 d	1.4 f	0.7 ef
	VC	31.2 b	58.7 b	3.2 c	1.5 cd
	B	25.3 d	54.7 cd	1.6 e	0.9 e
	G + VC	34.0 a	64.3 a	5.3 a	2.5 a
	E + VC	32.3 ab	61.3 ab	3.3 b	1.6 b

G: gypsum, S: elemental sulfur, VC: vermicompost, B: biochar. For each factor, different letters in each column indicate significant differences at $p < 0.05$ according to Fisher's LSD.

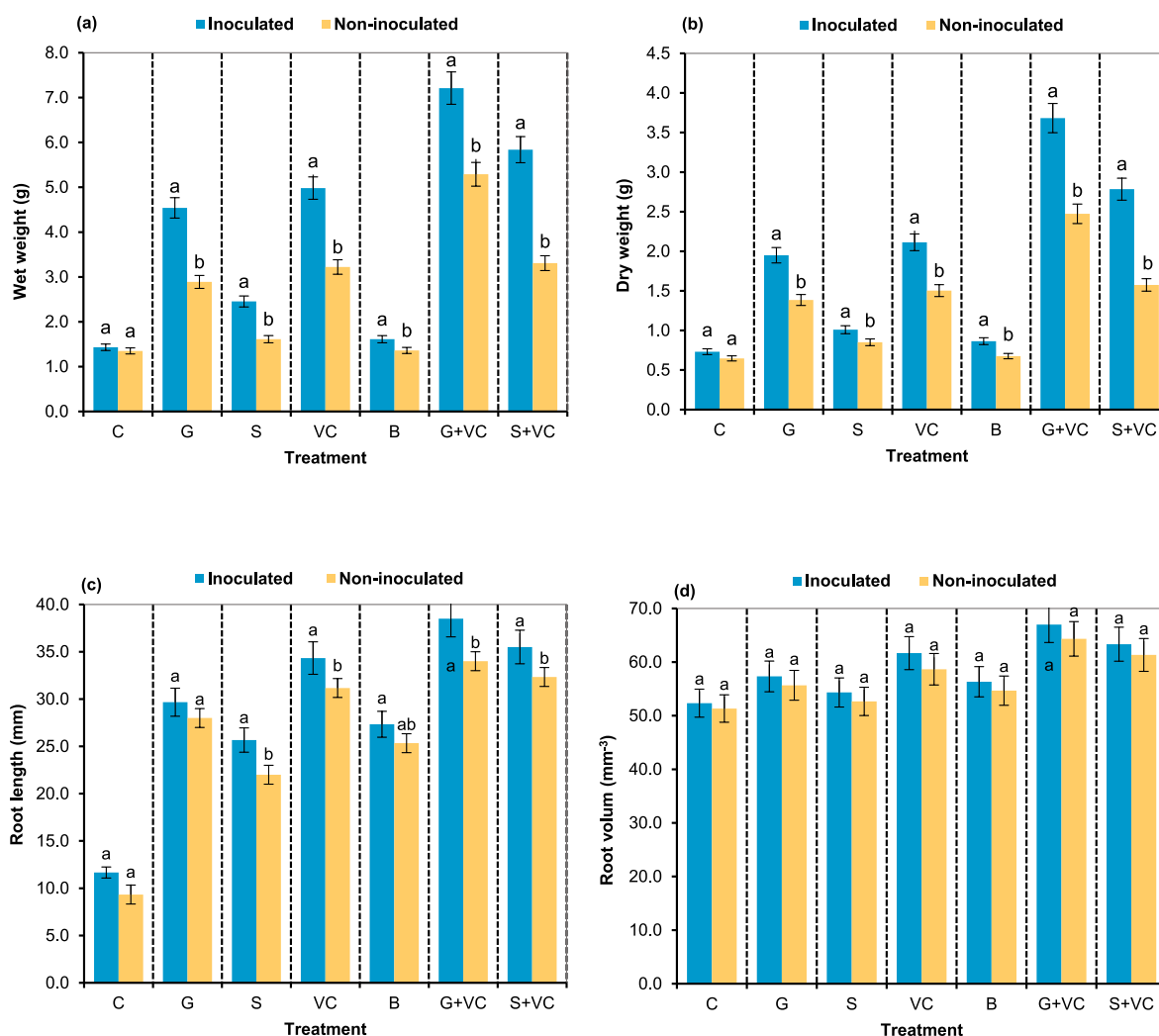


Fig. 5. The comparison of the mean values of wheat biomass and root characteristics between experimental factors. Different letters in each treatment represent the significant differences at $p < 0.05$ between the inoculated and non-inoculated factors according to Fisher's LSD.

with studies showing that chemical-organic amendments positively influence plant growth by enhancing soil structure and mitigating salinity stress (Alcivar et al., 2018; Sundha et al., 2020).

The enhancements observed in inoculated treatments, though not statistically significant for root length and volume, suggest that microbial inoculation supports wheat growth when combined with organic-chemical amendments. This effect can be attributed to the microbial communities' ability to alleviate abiotic stress and regulate nutrient availability through the production of phytohormones, siderophores, and beneficial metabolites (Arora et al., 2017; Garcia-Franco et al., 2021).

The significant relationships between wheat growth attributes and CSDI models (both CSDI-T and CSDI-M) highlight the importance of soil health in influencing plant growth. The strongest correlations were observed for RV, followed by RL, DBW, and WBW, indicating that root development is particularly sensitive to changes in soil degradation indicators. The regression equations (Table 10) illustrate that CSDI-T had a higher predictive accuracy than CSDI-M, in line with previous studies suggesting that the TDS approach (CSDI-T) is more comprehensive for assessing soil functions (Ding et al., 2021; Mamehpour et al., 2021).

However, the strong positive relationship between CSDI-T and CSDI-M ($R^2 = 0.94$) suggests that CSDI-M is still a practical and

efficient approach for evaluating soil degradation and plant growth, especially when time and cost constraints are considered. The reduction in the number of indicators with the MDS approach offers several advantages: fewer indicators to measure, faster laboratory analysis, and lower costs, while still providing reliable assessments of soil health and plant growth.

The use of combined treatments, particularly G + VC and S + VC, was the most effective strategy for improving wheat growth attributes and soil health in saline-sodic soils, highlighting their potential as cost-effective, environmentally friendly solutions for soil remediation (Liu et al., 2020).

However, to successfully implement this framework, it is essential to meticulously evaluate the specific characteristics of local soils, climate variables, and socio-economic aspects. By prudently modifying the CSDI framework to align with regional circumstances while upholding scientific standards, we can create more efficient mechanisms for evaluating and tackling soil degradation on a global scale. The critical approach is to find a balance between establishing standardized methods for comparison and tailoring strategies to ensure local applicability, all aimed at facilitating sustainable land management choices.

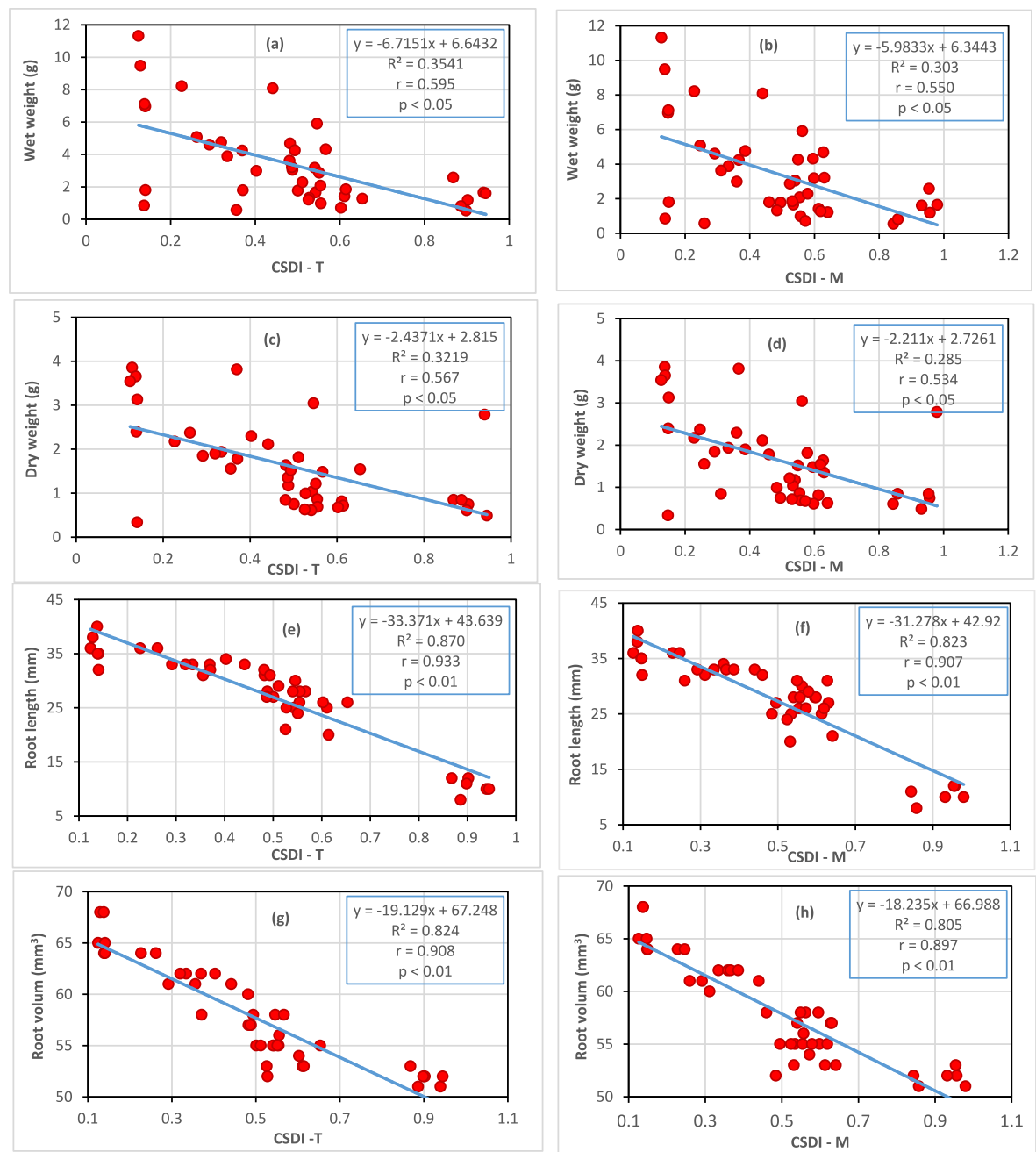


Fig. 6. The linear regression analysis of the relationship between CSDI -T (a, c, e, g) and CSDI -M (b, d, f, h) and wheat biomass and root characteristics.

Table 10
The regression equations linking CSDI models to wheat growth parameters.

Wheat Growth Attribute	Equation for CSDI-T	R ² (CSDI-T)	p-value (CSDI-T)	Equation for CSDI-M	R ² (CSDI-M)	p-value (CSDI-M)
Root Length (RL)	RL = −33.37 × CSDI-T + 46.64	0.87	<0.01	RL = −31.29 × CSDI-M + 42.92	0.83	<0.01
Root Volume (RV)	RV = −19.13 × CSDI-T + 67.25	0.82	<0.01	RV = −18.23 × CSDI-M + 66.99	0.81	<0.01
Wet Biomass Weight (WBW)	WBW = −6.72 × CSDI-T + 6.64	0.35	<0.05	WBW = −5.98 × CSDI-M + 6.34	0.30	<0.05
Dry Biomass Weight (DBW)	DBW = −2.44 × CSDI-T + 2.82	0.32	<0.05	DBW = −2.21 × CSDI-M + 2.73	0.29	<0.05

4.5. The challenges and perspectives of organochemical amendments on the soil –sodic soil remediation

The long-term implications of organochemical amendments on soil health and sustainability encompass both advantages and

possible drawbacks. While these amendments enhance soil fertility, inadequate management practices may result in nutrient depletion. For instance, applying organic amendments at elevated rates, particularly those characterized by low C/N such as fresh manure, can lead to nitrogen leaching, particularly in sandy soil

types (Lu et al., 2025). Likewise, the over-application of gypsum may cause sulfate leaching, posing a risk of groundwater contamination. Consequently, it is essential to regularly monitor soil nutrient levels to mitigate nitrogen leaching. Furthermore, elevated salt concentrations, when combined with the introduction of fresh organic matter, can initially promote the growth of halotolerant bacteria at the expense of more sensitive species (Zhao et al., 2025); therefore, it is crucial to maintain a steady supply of organic inputs to ensure the stability of microbial communities (Liu et al., 2020; Manyanga et al., 2024; Naamala & Smith, 2021; Singh et al., 2022; Sundha et al., 2020).

The application of organochemical amendments is a practical and cost-effective approach for smallholder farmers dealing with saline-sodic soils. By integrating organic materials with specific chemical amendments, this method provides synergistic advantages that enhance soil health, boost crop yields, and lower long-term input expenditures. Furthermore, it contributes to environmental sustainability by decreasing reliance on chemical fertilizers, fostering carbon capture, improving water quality, and promoting soil biodiversity. The evidence suggests (Zhang et al., 2020; Ding et al., 2021; Rezapour et al., 2022; Xiao et al., 2025) that integrated organochemical methods have notably enhanced the levels of both macro and micronutrients in soils affected by salinity and sodicity, which in turn supports the livelihoods of farmers and promotes the health of ecosystems through a decrease in the reliance on chemical fertilizers. Nevertheless, for these practices to be widely embraced by smallholder farmers, it is essential to address several critical factors, including the creation of locally tailored amendment recipes and the implementation of educational and demonstration initiatives for farmers.

Although organochemical amendments provide considerable agronomic and environmental advantages, their widespread adoption necessitates a careful consideration of cost, efficacy, and logistical practicality. A tailored approach that takes into account local resource availability, agricultural economics, and existing policy frameworks is crucial for sustainable implementation. Strategies to optimize these trade-offs may include integrating low-cost amendments such as crop residues with high-efficiency inputs like biochar, implementing government and market incentives such as carbon farming subsidies and organic premium pricing, establishing decentralized production facilities like local biochar kilns or composting hubs to minimize transportation expenses, and employing precision application techniques through soil testing and variable-rate technology to enhance the effectiveness of amendments (Alcívar et al., 2018; Elmeknassi et al., 2024; Xiao et al., 2025). Future advancements in cost-effective processing methods, streamlined distribution channels, and supportive incentive programs will be vital in addressing the existing trade-offs.

5. Conclusion

In this study, we investigated the individual and synergistic effects of various organic and chemical treatments on the remediation of calcareous saline-sodic soils. By developing a comprehensive soil degradation assessment framework using both total dataset (TDS) and minimum dataset (MDS) approaches, we quantitatively assessed soil improvement across a range of soil degradation indicators. Our results demonstrate that all treatments significantly enhanced soil quality, with combined treatments, particularly gypsum or sulfur with vermicompost (G + VC and S + VC), showing the greatest improvements, followed by vermicompost alone.

A notable reduction of 34 %–87 % in the Comprehensive Soil Degradation Index (CSDI) was observed across treatments

compared to the control, depending on the treatment type and modeling approach used. These improvements in CSDI were strongly correlated with key wheat growth parameters, including wet and dry biomass weight, root length, and root volume. The ability of these treatments to improve soil physical and chemical properties, as well as boost wheat growth, underscores their potential to significantly rehabilitate degraded saline-sodic soils in the study region and similar environments.

Our findings highlight the substantial role that organic amendments, particularly vermicompost, in combination with gypsum or sulfur, can play in improving soil structure, nutrient availability, and overall soil health. These combined amendments not only mitigate salinity and sodicity but also enhance plant growth, thus offering an effective solution for restoring degraded soils. This approach holds promise for reducing environmental risks associated with soil degradation, promoting sustainable agricultural production, and contributing to long-term soil resilience.

However, we recommend further field validation of CSDI frameworks in other regions to ensure broader applicability and to refine the use of these indicators in different environmental contexts. The insights gained from this study provide a foundation for future work aimed at optimizing soil management practices in saline-sodic regions and ensuring more sustainable agricultural systems globally.

CRedit authorship contribution statement

Salar Rezapour: Writing – review & editing, Writing – original draft, Validation, Supervision, Software, Resources, Project administration, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Amin Nouri:** Writing – review & editing, Writing – original draft, Visualization, Validation, Software, Methodology, Investigation, Conceptualization. **Farrokh Asadzadeh:** Writing – review & editing, Validation, Methodology, Formal analysis. **Ruijun Qin:** Writing – review & editing, Validation, Software, Methodology, Investigation. **Günay Erpul:** Writing – review & editing, Validation, Supervision, Methodology, Conceptualization.

Data availability

The datasets relevant to the current study will be available on an online repository.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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