Biochar application enhances soil quality by improving soil physical structure under particular water and salt conditions in arid region of Northwest China

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Highlights

- Deficit and brackish water irrigation significantly reduce soil quality index.
- Biochar enhances SQI by improving soil physical structure under full irrigation.
- Biochar shows positive effects on bacterial community structure.

Abstract

Exploring the suitability of biochar for improving soil quality under different water and salt conditions is important for maintaining soil health and productivity in the arid regions of Northwestern China. We compared the effects of biochar application practices on soil physical, chemical and biological properties under different irrigation and water salinity levels in a two-year field experiment in a mulched and drip-irrigated maize field in Gansu province, China. Eight treatments in total included the combination of two biochar addition rates of 0 t ha^{-1} (B0) and 60 t ha^{-1} (B1), two irrigation levels of full (W1) and deficit irrigation (W2; W2=1/2 W1) and two water salinity levels of fresh water (S0, 0.71 g L^{-1}) and brackish water (S1, 4.00 g L^{-1}). The minimum dataset method was used to calculate the soil quality index (SQI) under different treatments. Deficit and brackish water irrigation significantly reduced SQI by 3.80-9.80% through reducing some soil physical, chemical and biological properties. Biochar application significantly increased the SQI by 6.13 and 10.40% under full irrigation with fresh and brackish water, respectively. Biochar addition enhanced the relative abundance of beneficial bacteria (e.g., Proteobacteria, Patescibacteria) in the soil in all water-salt treatments. The partial least squares path model showed that biochar application significantly enhanced the SQI mainly by improving soil aggregation and pore structure under particular water-salt conditions. This research provides an important basis for utilizing biochar to improve soil quality in arid regions of Northwest China under various water-salt conditions.

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1. Introduction

Owing to population expansion and rapid economic progress, the disparity between water supply and demand has progressively widened across various regions of China. The disparity is particularly pronounced in the arid regions of Northwest China, causing increasingly serious fresh water scarcity (Cai *et al*. 2019). The region covers an area of 2.5 million km^2 and has about 110.9 billion m^3 of total water resources (Chen *et al*. 2016). The limited fresh water resources not only restrict stable economic development, but also has adverse impacts on the environment (Wei *et al*. 2022). Consequently, deficit irrigation (Yang *et al*. 2015, 2018; Zhou *et al*. 2017) and brackish water irrigation (Li *et al*. 2015; Yuan *et al*. 2019) with mulching have been widely adopted locally in agricultural production to effectively mitigate the scarcity of freshwater resources.

In areas with extreme water scarcity, deficit irrigation techniques can maximize water use efficiency and stabilize crop yields (Yang *et al*. 2018; Zou *et al*. 2021). However, long-term deficit irrigation not only significantly reduces soil bulk density, water-holding porosity space, and average weight diameter of aggregates (El Baroudy *et al*. 2013), but also decreases soil total organic carbon (TOC) and levels of major nutrients (e.g., nitrogen, phosphorus, potassium) (Blanco-Canqui *et al*. 2010; El Baroudy *et al*. 2013; Yaseen *et al*. 2014). In addition, deficit irrigation alters soil microbial community composition. Long-term deficit irrigation decreases the relative abundance of Acidobacteriota, and increases the relative abundance of Actinobacteriota (Rodriguez-Ramos *et al*. 2022). These findings suggest that deficit irrigation may impact soil quality. There is an abundance of brackish water resources at shallow depths in the Northwest region (Chen *et al*. 2023), and using brackish water irrigation is considered an effective strategy to alleviate the shortage of fresh water resources without significantly affecting crop yields (Jiang *et al*. 2012; Yuan *et al*. 2019). However, long-term use of brackish water irrigation may result in the accumulation of salinity in the crop root zone, which disrupts the water-salt balance, alters soil physical and chemical properties (e.g., by destroying soil aggregate structure, reducing TOC content and nutrient availability) (Guo H N *et al*. 2020; Dong *et al*. 2022; Wang *et al*. 2022), and markedly decreases soil microbial β-diversity (e.g., by reducing the relative abundance of Proteobacteria and Firmicutes) (Hu *et al*. 2020; Wu *et al*. 2021). These changes pose potential threats to both the soil environment and crop yields due to the ultimate degradation of soil quality (Liu B X *et al*. 2019; Zhang *et al*. 2020; Wang H *et al*. 2023).

In recent years, studies have demonstrated the potential of biochar for improving soil quality, which has been attributed to its well-developed porosity structure, large surface area, strong adsorption capacity and high stability (Leng *et al*. 2019, 2021; Lawal *et al*. 2021; Tan *et al*. 2021). For example, biochar application has been shown to promote the adsorption of organic compounds in the soil and enhance soil fertility (e.g., improve soil total nitrogen, soil total phosphorus, TOC, and soil available nitrogen levels) and enzyme activities (Abrishamkesh *et al*. 2016; Wu *et al*. 2024b). In addition, biochar application has been found to provide soil microbial communities with activated carbon and micro- or macro-nutrients, benefitting the environment of microbes (Xu *et al*. 2016). Moreover, biochar has a priming effect that can increase the mineralization of soil organic matter, providing favorable conditions for microbial growth (Wang *et al*. 2016). Thus, biochar has direct or indirect effects on microbial metabolic activities that ultimately alter soil microbial community richness and diversity. Further, biochar application has been found to effectively improve the soil environment and enhance soil quality (Palansooriya *et al*. 2019; Singh *et al*. 2021; Li *et al*. 2022). Several studies in recent years have explored the impact of biochar on soil properties under water and salt stress conditions. For instance, Agbna *et al*. (2017) indicated that biochar application significantly increased soil water content, TOC, and total nitrogen, while it significantly reduced soil inorganic nitrogen (NO_3^--N and NH_4^+-N) under deficit irrigation. Hale *et al*. (2021) found that biochar application did not significantly affect soil nutrient availability and microbial community biomass under deficit irrigation. Under brackish irrigation, biochar application significantly mitigated the adverse effects of salt stress on soil properties (biochar reduced soil electrical conductivity and increased soil total nitrogen, total phosphorus, and TOC levels, and increased soil cation exchange capacity) and enhanced biodiversity (Xu *et al*. 2023; Wu *et al*. 2024a, b; Xiao *et al*. 2024). Overall, biochar application may have great potential to improve soil quality and promote sustainable agricultural production under deficit and brackish water irrigation in the arid regions of Northwest China.

Soil quality refers to the ability of soil to maintain the sustainable functioning of an ecological service system. It depends on the soil's physical, chemical, and biological properties and their interactions (Ratcliffe *et al*. 2018; Jia *et al*. 2022). Effective soil quality assessment usually involves the selection of a range of indicators that are sensitive to soil management and responsive to the changes in soil functioning, and easy to measure (Armenise *et al*. 2013). These indicators include soil physical, chemical, and biological properties, and ultimately the measurements are synthesized into a simplified format to inform decision-making to help maintain sustainable agroecosystems (de Paul Obade and Lal 2016). Soil microorganisms have the ability to regulate the dynamics of soil physical and chemical properties (Gu *et al*. 2019). They can directly and indirectly be involved in fundamental ecosystem processes such as organic matter decomposition and nutrient cycling, ultimately affecting soil quality and productivity (Karhu *et al*. 2014; Naik 2019). Currently, most soil quality studies focus on the relationship between specific soil quality components (e.g., total phosphorus, total potassium) and the richness or diversity of soil microbial communities (Bello *et al*. 2021; Soothar *et al*. 2021). However, a single soil property does not fully reflect the interrelationship between the soil microbial community structure and its overall ecological functions (Guo J J *et al*. 2020). In recent years, more and more studies have shown that soil microorganisms respond rapidly to natural perturbations and environmental stresses, and thus have gradually become a powerful tool for monitoring soil quality (Chu *et al*. 2007; Fan *et al*. 2022). However, microbial effects on soil quality under different water-salt treatments with biochar application practices in the arid regions of Northwest China remain unknown.

Because of the potential of biochar to improve soil quality, a field experiment was conducted to investigate the effect of biochar application on soil quality and the microbial community under different water-salt conditions in the arid region of Northwest China. The objectives of this study were to: (i) evaluate the impacts of biochar application on soil quality and microbial community structure under varying water and salinity conditions, and (ii) investigate the key soil properties that regulate soil quality and microbial community structure after biochar application under different water and salinity treatments. We hypothesized that: (i) the ability of biochar application to change soil quality and microbial community structure varies significantly across different soil water and salt conditions, and (ii) biochar predominantly improves soil quality and changes microbial community structure through modification of physical, chemical and biological properties. This aim of the study was to provide a theoretical basis for exploring ways to improve soil quality and ecology under different water and salt conditions in an arid region of Northwest China.

2. Materials and methods

2.1. Experimental site description

The field experiment was located at the National Field Scientific Observation and Research Station on Efficient Water Use of Oasis Agriculture in Wuwei of Gansu Province of Northwest China (37°52′ N, 102°50′ E, 1,581 m altitude) (Fig. 1-A and B). The site is an arid inland region with a mean annual temperature of 8°C and frost-free period of 150 d. The mean annual precipitation and pan evaporation were 164 mm and 2,000 mm, respectively. The site has rich light and heat resources with a mean annual sunshine duration of over 3,000 h and an annual accumulated temperature (>0°C) of over 3,550°C. The average groundwater table depth is below 30 m.

2.2. Experimental design

The field experiment was carried out in 24 test pits (Fig. 1-C). Prior to the experiment, the basic soil properties of the site were determined. The average soil bulk density (BD), field capacity (FC), saturated water content (θ_s) and electrical conductivity (EC) of the soil profile (0-100 cm) were 1.50 g cm^{-3}, 0.31 cm^3 cm^{-3}, 0.38 cm^3 cm^{-3}, and 261.80 µS cm^{-1}, respectively. The TOC and total nitrogen (TN) concentrations and pH of the topsoil (0-20 cm) were 8.70 g kg^{-1}, 0.73 g kg^{-1}, and 8.37, respectively. The experiment included two irrigation rates, two water salinity levels and two biochar application rates (Table 1), which were arranged in a completely randomized block design (2×2×2) with three replicates to give a total of 24 plots. The two irrigation rates were full irrigation (W1) and deficit irrigation (W2). The irrigation rate for W1 was the difference between the mean observed soil water content (SWC) and the average FC over the wetting depth to the depth of the maize root-zone. The irrigation rate for W2 was 1/2 of that of W1. Irrigation was triggered when the average soil water content of 0-60 cm soil reached about 65-70% FC for the W1 treatment. A drip irrigation system was used for field irrigation. The total amount of irrigation water applied to the W1 treatment was 421 mm (six times) and 532 mm (seven times) in 2020 and 2021, respectively. The two water salinity levels were fresh water (S0) and brackish water (S1). Fresh water was obtained from local groundwater (0.71 g L^{-1}), and brackish water (4.00 g L^{-1}) was obtained through dissolving NaCl, $MgSO_4$ and $CaSO_4$ in groundwater at a mass ratio of 2:2:1 according to the salt ion composition of the local groundwater (Yuan *et al*. 2019). An amount of 40-mm fresh water was applied to each plot shortly after sowing to ensure seedling emergence, and the different irrigation treatments started from the jointing stage each year. The biochar was purchased from Yuhong Biomass Energy Development Company (Shanxi, China). It was derived from corn straw by pyrolyzing it at 500°C for 10 h. The pH, EC, TN, TOC and particle size of biochar were 9.47, 730.20 µS cm^{-1}, 9.90 g kg^{-1}, 689.92 g kg^{-1}, and<150.00 µm, respectively. The exchangeable Na^+, Mg^{2+}, Ca^{2+}, and K^+ concentrations of the biochar were 0.51, 0.10, 0.40, and 5.17 g kg^{-1}, respectively.

2.3. Field management

The local maize variety (Xianyu 335) was selected as the test crop in the two-year experiment. Maize was sown on 7 May and 1 May, and harvested on 26 September and 21 September in 2020 and 2021 respectively. The planting density was 55 plants per test plot with a row spacing of 40 cm and an along the row plant spacing of 30 cm. After seeding the plots were covered with white plastic film. Compound fertilizer, including 226.50 kg P_2O_5 ha^{-1}, 54 kg K_2O ha^{-1}, and 121.50 kg N ha^{-1}, was applied as basal fertilization during cultivation, and supplemental N fertilizer at 100 kg N ha^{-1} N was applied on 15 July 2020 and 26 July 2021 through the drip irrigation system. Other field management practices, such as thinning, spraying and weeding, followed local maize management practice. The maize grain yield in 2020 and 2021 is given in Chen *et al*. (2023).

2.4. Soil sample collection and analysis

Soil hydraulic characteristic parameters Soil samples were collected after the maize harvest in September 2021. The soil porosity parameters were determined according to soil water characteristic curves (SWCCs), which were measured by the centrifugal method (Xing *et al*. 2017; Fu *et al*. 2019). Firstly, undisturbed topsoil (0-20 cm) samples were collected from each plot using cutting rings (100 cm^3), and the samples were water saturated by capillary rise for 48 h. The saturated soil samples were dehydrated using a high-speed centrifuge (CR22N, Japan) with specific speeds at corresponding times to allow calculation of the soil matric potential. Soil cores were weighed at the end of each speed, and oven-dried (105°C for 24 h) after centrifugation to obtain soil mass water content and bulk density (BD), and using these parameters to calculate soil volumetric water content (SWC). Finally, SWC curves reflecting the relationship between soil matric potential and SWC were fitted according to the VG model (van Genuchten 1980).

$$\theta = \theta_r + \frac{\theta_s - \theta_r}{\left[1+\left(\alpha H\right)^n\right]^m} \qquad (1)$$

where H is the soil matric potential (hPa); θ is the soil volumetric water content (cm^3 cm^{-3}); θ_s and θ_r are the saturated and residual soil volumetric water contents (cm^3 cm^{-3}); α is the inverse of the air entry value; and m and n are model parameters (m=1-1/n); The θ_s, θ_r, α, n and m values were obtained during the process of SWCC fitting with Equation (1) using RETC software. The total porosity (TP) is numerically equal to the saturated soil volumetric water content. The field capacity (FC) was defined as the equilibrium soil volumetric water content at -10 kPa matric potential. The air capacity (AC) was calculated as the difference between TP and FC. The available water content (AWC) is numerically equal to soil capillary porosity, which was calculated as the difference between the FC and the PWP (PWP is the permanent wilting point calculated as the volumetric soil water content at -1,500 kPa matric potential) (Głąb *et al*. 2016; Fu *et al*. 2019).

The soil near-saturated hydraulic conductivity (K_{ns}) was determined by a mini disk infiltrometer (Mini Disk, Decagon Devices Inc., USA). An area with little disturbance was selected in each plot as the observation site. Firstly, the residual film and impurities on the soil surface were removed, then the surface was scraped with a knife and covered with fine sand. A mini disk infiltrometer filled with water was placed vertically on the observation site, and the scale of the infiltrometer was read every 30 s to determine cumulative infiltration. The infiltration model reflecting the change of cumulative infiltration amount (I) with time (t) can be approximated by the first two terms of Philip's expansion (Philip 1957), which was adopted by Zhang's model (Zhang 1997):

$$I = C_1\sqrt{t} + C_2 t \qquad (2)$$

where C_1 (cm $s^{-1/2}$) and C_2 (cm s^{-1}) are fitting coefficients that represent the capacities of soil sorptivity and hydraulic conductivity, respectively.

The soil near-saturated hydraulic conductivity was calculated as follows:

$$k_f = \frac{C_1}{A} \qquad (3)$$

where A is a dimensionless parameter related to the suction head according to the soil texture and the size of the infiltrometer.

$$A = \frac{11.65\left(n^{0.1}-1\right)\exp\left[2.92\left(n-1.9\right)\alpha h_0\right]}{\left(\alpha r_0\right)^{0.91}} \quad n \geqslant 1.9 \qquad (4)$$

$$A = \frac{11.65\left(n^{0.1}-1\right)\exp\left[7.5\left(n-1.9\right)\alpha h_0\right]}{\left(\alpha r_0\right)^{0.91}} \quad n < 1.9 \qquad (5)$$

where n and α are the van Genuchten model parameters of the soil; r_0 is the disk radius (2.25 cm); and h_0 is the suction head at the disk surface (-2 cm).

Soil water-stable aggregates The wet sieving method was used for determining the distribution of soil water stable aggregates (Elliott 1986). About 2 kg of undisturbed topsoil sample was excavated with a shovel from each plot and brought to the laboratory. The parts of the soil sample deformed due to contact between the shovel and soil block during soil sampling were stripped and discarded, and then the large soil block was stripped along natural fissures to obtain a small soil block of 10-12 mm and air-dried. Approximately 500 g of air-dried soil sample was sieved using a sieve shaker with mesh sizes of 10, 7, 5, 3, 2, 1, 0.50, and 0.25 mm (the samples larger than 10 mm were considered as soil blocks and removed). The dry weight of aggregates of different particle sizes was weighted using a high-precision balance. The air-dried soil aggregate samples (50 g) were prepared according to respective dry weight proportions.

Finally, the soil aggregate samples were soaked in a bucket for 5 min, and then subjected to automatic up-down oscillation for 2 min using a soil water-stable aggregation analyzer (the mesh sizes of sieves were 5, 2, 1, 0.50, 0.25, and 0.053 mm) (XY-100, Beijing Xiangyuweiye Instrument Equipment Co., Ltd., China) at an amplitude of 3 cm and frequency of 30 times min^{-1} (Fu *et al*. 2019; Ma *et al*. 2022). The aggregates remaining on the sieves were transferred into a clean aluminum box for oven drying and weighing. The water-stable aggregates were divided into large aggregates (>2 mm), small aggregates (0.25-2 mm), microaggregates (0.053-0.25 mm) and silt and clay fractions (<0.053 mm). Soil water-stable aggregate stability indices were determined from the mean weight diameter (MWD, mm), geometric mean diameter (GMD, mm) and percentage of water-stable aggregates that were greater than 0.25 mm (WR$_{0.25}$, %), which can be calculated using the following formulae:

$$MWD = \sum_{i=1}^{n} \overline{x}_i W_i \tag{6}$$

$$GMD = \exp\left(\sum_{i=1}^{n} W_i ln\overline{x}_i\right) \tag{7}$$

$$WR_{0.25} = \left[1 - \frac{M_{x<0.25}}{M_T}\right] \times 100 \tag{8}$$

where $\overline{x}_i$ is the mean diameter of the *i*th size class (mm); W_i is the proportion of aggregates in size class *I*; $M_{x<0.25}$ is the mass of the aggregates size smaller than 0.25 mm (g); M_T is the total mass of the aggregates (g).

Soil chemical properties Five topsoil cores were collected from each plot and mixed as a composite sample for determination of soil chemical properties and microbial community analysis. The fresh soil samples were screened with a 2 mm sieve to remove residual roots and stones. Each sample was then divided into four parts: one subsample was oven-dried to measure soil water content, one subsample was air-dried for the determination of soil EC, pH and TOC, one subsample was stored at 4°C for soil available nitrogen (AN) analysis, and the other subsample was stored at -80°C soil bacterial community analysis. Soil EC and pH were measured by a conductivity meter (SG3-ELK742, Mettler-Toledo International Inc., Switzerland) and a pH meter (FE28, Mettler-Toledo International Inc., Switzerland), respectively (1:5 soil:water ratio). The AN (10 g fresh soil sample was extracted with 1 mol L^{-1} KCl at a ratio of 1:5 (w/v)) was determined using a continuous flow analytical system (Auto Analyzer 3, 224 Bran + Luebbe, SEAL Analytical GmbH, Germany). The TOC was determined using the potassium dichromate oxidation-oil bath heating method (Bao 2000).

2.5. DNA extraction, PCR amplification and high throughput sequencing

Bacterial genomic DNA was extracted from 0.50 g of soil sample using the Fast DNA SPIN extraction kits (MP Biomedicals, Santa Ana, CA, USA) according to the manufacturer's protocols. The DNA concentration and purity were measured in 1% agarose gel and with a NanoDrop 2000 UV-vis spectrophotometer (Thermo Scientific, Wilmington, USA). The bacterial 16S rRNA gene at V3-V4 regions was amplified using the forward primer 338F (5′-ACTCCTACGGGAGGCAGCA-3′) and the reverse primer 806R (5′-GGACTACHVGGGTWTCTAAT-3′), respectively. The PCR product was sequenced through an Illumina Miseq platform, which was completed by Personalbio Biotechnology Co., Ltd., Shanghai, China. Raw sequence data analyses were conducted using open-source software of Quantitative Insights into Microbial Ecology (QIIME2).

2.6. Soil quality index

As an effective parameter to evaluate soil quality, soil quality index (SQI) is widely used for evaluating soil quality of various types of ecosystems (Shao *et al*. 2020; Chen *et al*. 2024; Teng *et al*. 2024). The first step was to determine the total data set (TDS), including soil physical, chemical and biological indicators in this study. This step involved selecting representative soil indicators from the TDS to construct a minimum data set (MDS). Principal component analysis (PCA) was used to reduce the dimensionality of these indicators after determining the redundancy between the indicators by Pearson correlation analysis. In particular, the Kaiser-Meyer-Olkin (KMO) test (>0.50) and Bartlett's spherical test (P<0.05) were conducted before analysis to ensure that the data was suitable for PCA.

Only the principal components (PCs) with eigenvalues ≥1 that explained at least 5% of the data variation were chosen for the MDS (Shao *et al*. 2020). Indicators with factor loading of PC≥0.50 were grouped together, and any indicator that exhibited a loading greater than 0.5 across multiple PCs was assigned to the group with the higher loading. The indicators whose Norm value fell within 10% of the maximum Norm value in each group was preselected for MDS. If there was a strong correlation between any two preselected indicators (r>0.5), the indicator with the highest Norm value was selected for the MDS. If there was not a strong correlation between any two preselected indicators (r≤0.5), then all indicators were selected for the MDS.

The Norm value can be calculated using the following equation:

$$N_{ik}=\sqrt{\sum_{i=1}^{k}u_{ik}^{2}\lambda_{k}} \tag{9}$$

where N_{ik} is comprehensive loading of *i*th soil indicator on the first *k* PCs with eigenvalues≥1; u_{ik} is the loading of the *i*th indicator on the *k*th PC; λ_k is the eigenvalue of the *k*th PC.

In this study, a standardized scoring function method was applied to transform and normalize the selected MDS indicators within the range of 0.0 to 1.0 (Andrews *et al*. 2002). The MDS metrics were standardized using two scoring functions, "more is better" and "less is better", which can be expressed by the following equations (10)-(11):

$$\mu_i=\frac{x_i}{\max(x_i)} \tag{10}$$

$$\mu_i=\frac{\min(x_i)}{x_i} \tag{11}$$

where μ_i is the standardized score of indicators; $\max(x_i)$ and $\min(x_i)$ are the maximum and minimum values of indicators in different treatments, respectively.

Finally, the SQI was calculated based on the sum of the standardized indicators and their corresponding weights in each treatment (Armenise *et al*. 2013):

$$SQI=\sum_{i=1}^{n}u_i w_i \tag{12}$$

where w_i is the weighting of the indicators, which were calculated as the ratio of each indicator's communality to the sum of communalities of all indicators:

$$w_i=\frac{C_i}{\sum_{i=1}^{n}C_i} \tag{13}$$

where C_i is the communality value of each indicator.

2.7. Statistical analysis

One-way ANOVAs and Duncan's multiple comparison tests were used to assess statistically significant differences among different treatments for soil properties and SQI. Three-way ANOVAs were used to analyze the effects of biochar application, irrigation and water salinity, as well as their interactions on soil properties and SQI. Pearson correlation analysis was conducted to test the relationships between soil bacterial phylum and soil physicochemical properties, and the relationships among soil indicators of the TDS. Linear regression analysis was conducted to determine the relationship between soil TP or AC and soil EC, as well as the relationship between MDS-SQI and TDS-SQI. Nonmetric Multidimensional scaling (NMDS) based on Unweighted-unifrac and PERMANOVA analysis was used for testing for significant differences in bacterial community structure. PCA was performed to select the most representative soil indicators for soil quality assessment. The partial least squares path model (PLS-PM) was used to assess the direct and indirect effects of soil properties on bacterial composition and soil quality.

3. Results

3.1. Effects of biochar application on soil properties under different water-salt conditions

Soil hydraulic and chemical parameters Biochar application had diverse effects on soil physicochemical properties under different water-salt conditions (Table 2; Appendix B). Under the same irrigation level without biochar application (B0), brackish water irrigation (S1) significantly increased soil BD compared to fresh water irrigation (S0). Biochar application (B1) significantly reduced soil BD by 8.21-19.85% in all water-salt treatments, but the reduction was greater under full irrigation (W1). Brackish water irrigation significantly reduced soil TP, AC and K_{ns}, but had no significant impact on FC or AWC. Biochar application significantly increased all soil porosity indicators, and enhanced K_{ns} by 17.65-133.33%. The SWC under the brackish water (S1) and full irrigation (W1) treatments was higher than that under fresh water (S0) and deficit irrigation (W2) treatments, respectively. The higher SWC caused by biochar treatment (B1) was only observed under W1S1 treatment. There was a higher EC under S1 compared with S0, and biochar application reduced soil salinity by 32.43% under W1S1 treatment. Overall, salt treatment significantly reduced soil pH, while water and biochar treatments had no significant impact on soil pH. Biochar application was the most significant factor affecting soil TOC, with biochar application causing increases ranging from

196.40-227.90%. Deficit irrigation, brackish water irrigation, and biochar application all increased soil AN to some extent compared with their respective control treatments. Furthermore, regression analysis revealed that both soil TP and AC decreased significantly with increase in soil EC under the no biochar (B0) treatment (Fig. 2). The decrease slowed and the correlation became non-significant after biochar application, suggesting that biochar application could mitigate the damaging effects of soil salinity on soil TP and AC.

The distribution and stable indices of soil water-stable aggregates Compared to fresh water irrigation, brackish water irrigation significantly reduced the proportion of large aggregates (>2 mm) by 40.83% on average, but had no significant impact on the proportion of small aggregates (0.25-2.00 mm) under both W1 and W2 irrigation (Fig. 3). Brackish water irrigation also increased the proportion of microaggregates (0.053-0.25 mm) as well as silt and clay fractions (<0.053 mm) under full irrigation. Compared to W1S0 treatment, the proportion of large aggregates was lower, and microaggregates was higher, than under the W2S0 treatment, respectively. Biochar application did not significantly affect the proportions of large aggregates and microaggregates, but it exhibited significant differences in the effect on the proportion of small aggregates as well as silt and clay fractions between W1 and W2. For instance, biochar increased the proportion of small aggregates as well as silt and clay under W1, but did not significantly affect them under W2, except for increasing the proportion of silt and clay fractions in the W2S0 treatment (Fig. 3-A). Compared to the control treatment (W1S0), both water and salt treatments reduced soil aggregate stability indices to varying degrees. Salt treatment was the most significant factor affecting soil aggregate stability indices, and there was an interactive effect between the salt and water treatments on these indices. Brackish water irrigation significantly reduced MWD, GWD, and $WR_{0.25}$ under W1 by 35.00%, 32.49%, and 19.61%, respectively, while its effect on these indices did not reach statistical significance under W2 (Fig. 3-B–D). The ability of biochar to improve soil aggregate stability depended on irrigation level, that is, biochar improved MWD, GWD and $WR_{0.25}$ under W1 by 9.74%, 26.04%, and 13.91%, respectively, while biochar had no significant effect on these indicators under W2.

Soil bacterial richness and diversity indices The Chao1 and Observed species indices under deficit irrigation were significantly lower ($P<0.05$) by 10.97% and 10.81% than that under full irrigation on average (Table 3). Although brackish water irrigation reduced Chao1 and Observed species by an average of 6.87% and 6.17%, respectively, it did not reach statistical significance, compared to fresh water irrigation. Biochar application did not significantly affect the richness and diversity indices of soil bacterial communities.

Soil quality index The results of KMO and Bartlett's sphericity tests indicated that soil indicators were suitable for PCA after excluding highly redundant and low-quality data sets (FC and K_{ns}) in this study (Appendix D). The TP, GWD, SWC, Observed species and Simpson indices were selected as the representative indicators for evaluating soil quality index (SQI) through the MDS method (Appendices E-G). The regression analysis between SQI-MDS and SQI-TDS values indicated the MDS method could effectively reflect the overall soil quality status (Fig. 4-A). Water-salt, biochar application and the two-way interaction between water and salt, and between water and biochar application were all found to significantly affect SQI (Fig. 4-B). Under the S0B0, S0B1 and S1B1 treatments, the SQI under deficit irrigation was 3.79, 9.60, and 5.32% lower than that under full irrigation, respectively. In contrast, the SQI was similar between the two irrigation rates under the S1B0 treatment. Brackish water irrigation significantly reduced the average SQI by 6.49% under W1, while it did not significantly affect SQI under W2, compared to fresh water irrigation. Biochar application significantly improved the SQI of the W1S0 and W1S1 treatments by 6.13% and 10.40%, respectively, while it had no significant impact on SQI under deficit irrigation. Compared to full irrigation with fresh water, deficit irrigation significantly reduced the average maize yield over the two years, while full irrigation with brackish water slightly reduced maize yield. Biochar application increased maize yield to some extent under W1S0, W1S1, and W2S0 treatments. Overall, there was a significant and positive correlation between the soil quality index and the average maize yield over the two years ($P<0.01$) (Appendix A).

3.2. Effects of biochar application on soil bacterial composition and soil structure under different water-salt conditions

The relative abundances of bacterial phyla under different treatments are shown in Fig. 5. The relative abundance of the top ten bacterial phyla, namely Actinobacteria, Proteobacteria, Chloroflexi, Acidobacteria, Gemmatimonadetes, Bacteroidetes, Firmicutes, Rokubacteria, Patescibacteria, and Nitrospirae, collectively accounted for 97.6-98.5% of the total bacterial phyla (Fig. 5-A). Overall, the findings from correlation analysis and three-way ANOVA both revealed that only a minority of bacterial phyla were influenced by different treatments and soil environments (Fig. 5-B; Appendix H). For instance, deficit irrigation led to an increase in the relative abundance of Actinobacteria compared to full irrigation, excepting for there was a similar value between W2S1B0 and W1S1B0. This increase was evident in the negative correlation observed between its relative abundance and SWC. On average, the relative abundance of Proteobacteria under B1 was

4.75% higher than that under B0, and its relative abundance under S1 was 4.14% lower than that under S0. Consequently, the relative abundance of Proteobacteria showed a negative correlation with EC and a positive correlation with TOC. Brackish water irrigation significantly increased the relative abundance of Chloroflexi by 13.21% compared to fresh water irrigation, thus its relative abundance increased with increase in soil EC, and both irrigation amount and biochar treatment had no significant impact on its relative abundance. Deficit irrigation reduced the relative abundance of Acidobacteria except in the S0B0 treatment, and its relative abundance showed a negative correlation with soil AN. Biochar application significantly increased the relative abundance of Patescibacteria by 52.90% on average, resulting in a positive correlation between its relative abundance and soil TOC. Deficit irrigation decreased the relative abundance of Nitrospirae by 17.14% on average, and its relative abundance exhibited a positive correlation with SWC and a negative correlation with soil AN.

The NMDS analysis demonstrated that the bacterial community structure under different treatments was separated to different degrees on the two principal coordinate axes ($P<0.001$) (Fig. 6-A). The four treatments under full irrigation were distributed in the four quadrants of the NMDS coordinates, while the four treatments under deficit irrigation were distributed in the second and fourth quadrants. The four treatments with biochar application were distributed in the positive direction of the NMDS2 axis, while the four treatments without biochar application were distributed in the negative direction of the same axis. Overall, the PERMANOVA analysis suggested that there were significant differences in the bacterial community structure between W1 and W2 ($P<0.01$), as well as between B0 and B1 ($P<0.001$). However, there were no significant differences in the bacterial community structure between S0 and S1 (Fig. 6-B–D).

3.3. The key soil properties regulating soil bacterial community structure and soil quality under water-salt-biochar treatment

The PLS-PM was performed to test how soil physicochemical properties affected the soil bacterial community, and their joint impact on soil quality under combined water, salt and biochar treatments (Fig. 7). The fitting results of the PLS-PM illustrated that the model showed a good fitting effect of the data (SRMR=0.056, dULS=0.067, dG=0.161, NFI=0.841). In general, the changes in soil physical properties (TP and GWD) were the most significant factors, followed by soil microbial property (Observed species) in affecting SQI, which explained a large proportion of the variation in SQI. However, no direct or indirect influence of soil chemical property (pH) or soil bacterial structure on SQI was found. Moreover, the PLS-PM revealed that soil pH significantly influenced the soil bacterial richness index, but did not exert a notable impact on soil bacterial structure. Conversely, soil physical properties had a pronounced effect on soil bacterial structure, but showed no significant influence on soil bacterial richness index. Consequently, biochar application primarily aimed to enhance soil structure in terms of soil physical properties, thereby improving the SQI under full irrigation.

4. Discussion

4.1. Effects of water, salt and biochar treatments on soil properties

This study demonstrated that deficit and brackish water irrigation affected soil physical and chemical properties. As expected, SWC and TOC were significantly lower under deficit irrigation than under full irrigation, while soil EC and pH did not differ significantly between the deficit and full irrigation treatments (Table 2). Williams and de Vries (2020) also found that SWC stress caused by deficit irrigation could lead to a decrease in soil TOC, mainly due to the reduction in crop root exudates and residues, while Erdem *et al.* (2001) and Hirich *et al.* (2012) found that deficit irrigation had no significant impact on soil EC and pH in the short term. However, AN was higher under deficit irrigation than full irrigation (Table 2), which is mainly attributed to the limitation of crop nitrogen absorption under drought stress, resulting in a large amount of soil residual AN (Gonzalez-Dugo *et al*. 2010; Jehan *et al*. 2022).

Brackish water irrigation significantly increased soil EC, SWC and AN, but reduced soil pH and TOC (Table 2). Salt accumulation in the soil root zone is a direct result of using brackish water irrigation (Alomar and Jena 2024), and this accumulation subsequently decreases the water absorption rate of the root system and the repulsive forces between soil particles, thereby increasing the SWC (Wang *et al*. 2019; Wu *et al*. 2024a). Previous studies have found that brackish water irrigation inhibited crop growth, leading to a reduction in the absorption of soil AN and the C input to soil by crops, which in turn caused a decrease in soil TOC (Rietz and Haynes 2003; Wilson *et al*. 2018; Haj-Amor *et al*. 2022) and an increase in AN content (Guo *et al.* 2023). The accumulation of strongly acidic ions (i.e., SO_4^{2-} and Cl^-) from brackish water irrigation might be the major reason for decreasing soil pH (Fan *et al*. 2009; Du *et al*. 2023).

Biochar application significantly increased soil TOC and AN, and slightly enhanced SWC (Table 2). Biochar has rich and stable carbon, and can promote the C input of crop root systems and the formation of aggregates that protect organic carbon from decomposition, thereby increasing soil TOC (Sohi *et al.* 2010; Wang *et al*. 2016; Duan *et al*. 2021; Omar *et al*. 2023). Feng *et al*. (2021) found that stover biochar application significantly increased soil AN content in a corn farmland, which they ascribed to the decomposition of readily oxidized carbon in biochar (Phillips *et al*. 2022) and the reduction of leaching of available nitrogen (Liao *et al*. 2021). Furthermore, the hydrophilic functional groups on the biochar surface and in the porosities of biochar's graphite flakes can improve the capacity for soil water retention, ultimately increasing SWC (Uzoma *et al*. 2011; Yao *et al*. 2017). We found that biochar application significantly decreased soil EC and increased pH only under full irrigation with brackish water (Table 2). The soil with biochar addition has abundant pores and greater hydraulic conductivity that might promote salt leaching under adequate irrigation, leading to a decrease in soil EC (Githinji 2014; Chaganti *et al*. 2015). As described above, the reduction in soil acidic ions might increase soil pH under biochar application. Therefore, biochar has the potential to improve soil chemical properties under special water and salt conditions.

Soil aggregates and porosity were not significantly affected by different irrigation water amounts, but brackish water irrigation significantly disrupted soil structure (Table 2; Fig. 3). Brackish water irrigation increases the soil exchangeable Na^+ content, which raises the electrokinetic potential and causes swelling and dispersion of soil aggregates (Li *et al*. 2019; Wu *et al*. 2020). Similarly, the present study found that brackish water irrigation significantly reduced the percentage of soil large aggregates (>2 mm) and aggregate stability indicators (e.g., MWD, GMD, and $WR_{0.25}$), especially under full irrigation. The destruction of soil aggregate structure could cause soil compaction (increased BD), poorer aeration (decreased AC), and a decrease in soil TP, which is in line with the findings of previous studies (Wolkowski and Lowery 2008; Shah *et al*. 2017). Numerous studies have shown that soil structure is a key factor affecting saturated hydraulic conductivity (Singh *et al*. 2022; Chang 2023; Mozaffari *et al*. 2024). Brackish water irrigation has adverse effects on both soil aggregates and pore structure, and significantly reduces K_{ns} (Qadir *et al*. 2007; Ji *et al*. 2014).

In this study, under full irrigation biochar application significantly improved the soil water-stable aggregate structure, primarily by increasing the percentage of 0.25-2.00 mm aggregates (Fig. 2). This improvement occurred because soil particles form aggregates primarily through the cohesive cementation of inorganic compounds and organic polymers (Lal 2000). Biochar can adsorb nutrients and humus providing essential organic cementing substances for the formation of soil aggregates (Duan *et al*. 2021). The oxidized carboxyl groups and minerals on the surface of biochar enhance the connectivity between different components of soil (Zheng *et al*. 2018), ultimately promoting the formation of stable aggregates. Furthermore, a good soil water environment is conducive to microbial activities, enabling their metabolites to better function as a binder between biochar and soil particles. This might explain why biochar better promoted the formation of 0.25-2.00 mm aggregates under full irrigation than under deficit irrigation (Zhu *et al*. 2019). In addition, we found that biochar application significantly improved soil porosity structure and K_{ns} under all water-salt conditions (Table 2). Previous researchers have found that the lightweight texture and well-developed porous structure of biochar reconstructs the soil pore structure, and can significantly increase soil total porosity (TP) and different sizes of porosity (Mukherjee and Lal 2013; Blanco-Canqui 2017; Alghamdi 2018), thereby markedly enhancing soil hydraulic conductivity (Oguntunde *et al*. 2008; Asai *et al*. 2009).

Soil biological properties are key indicators reflecting the quality status of soil (Gholamhosseinian *et al*. 2022). Previous studies have shown that the loss of microbial diversity could negatively affect nutrient cycling and soil function, ultimately reducing soil quality (Sharma *et al*. 2011; Philippot *et al*. 2013). Therefore, studying the changes in soil microbial diversity is essential for evaluating soil health. In the present study, the richness indices of soil bacterial communities were significantly lower under deficit irrigation than under full irrigation (Table 3). Soil water stress alters the niche environments that bacteria have adapted to, impacting nutrient transport and substrate availability. This in turn, leads to a decrease in the number of bacterial communities that are unable to adapt to the new conditions (Han *et al*. 2007; Bastida *et al*. 2017), ultimately resulting in a reduction in soil bacterial richness (Peralta *et al*. 2014; Ren *et al*. 2018).

Previous studies have shown that soil microbial community richness and diversity exhibited varying responses to different irrigation water qualities (Chen *et al*. 2017; Zhao *et al*. 2020). Although some studies have suggested that saline water irrigation could reduce (Min *et al*. 2016; Chen *et al*. 2019) or increase soil microbial diversity (Chen *et al*. 2017), we found that two years of brackish water irrigation did not significantly alter soil bacterial community richness and diversity, which is consistent with the findings of other researchers (Hu *et al*. 2020; Sun *et al*. 2022; Ding *et al*. 2023). The impact of brackish water irrigation on the soil bacterial community depends on the ion composition and concentration of the water, as well as the original soil conditions and climatic environment.

Most studies have shown that biochar has great potential for increasing soil microbial richness and diversity (Chen *et al*. 2015; Palansooriya *et al*. 2019). However, we found that biochar application had no significant effect on the richness and diversity of the soil bacterial community across all water-salt treatments after two years (Table 3). Similarly, Liu H X *et al*. (2019) found that soil bacterial diversity did not change significantly 4 years after biochar application in corn farmland. Initially, biochar application may significantly increase soil bacterial richness and diversity in the short-term, however, over time, soil bacteria adapt to the biochar-rich environment (Nguyen *et al*. 2018), and the low content of readily decomposable carbon that is easily utilized by bacteria ultimately fails to stimulate an increase in bacterial richness and diversity in the long term (Jiang *et al*. 2016).

4.2. Effects of water, salt and biochar treatments on soil bacterial composition and structure

This study utilized high-throughput sequencing analysis to identify the soil bacteria. Actinobacteria, Proteobacteria, Chloroflexi, Acidobacteria, and Gemmatimonadetes were the dominant bacterial phyla in the study soil (Fig. 5), which is consistent with previous studies (Sun *et al*. 2022; Ding *et al*. 2023; Guo *et al*. 2023). Overall, deficit irrigation significantly increased the relative abundance of Actinobacteria, but decreased the relative abundance of Acidobacteria and Nitrospirae compared to full irrigation (Fig. 5-A). Actinobacteria have strong drought tolerance, and they are more active in a low-water environment with good air permeability and sufficient oxygen (Barnard *et al*. 2013; Li *et al*. 2021). In contrast, Acidobacteria prefer moist conditions (Zavaleta *et al*. 2003). Barnard *et al*. (2013) and Rodriguez-Ramos *et al*. (2022) both found that the abundance of Actinobacteria was significantly higher under drought conditions, whereas Acidobacteria showed the opposite trend. Previous studies have found that water stress decreased the relative abundance of the Nitrospirae (Santos-Medellín *et al*. 2017; Wang *et al*. 2020), which is in agreement with our results. Furthermore, the present study found that brackish water irrigation decreased the relative abundance of Proteobacteria, but increased the relative abundance of Chloroflexi compared to fresh water irrigation (Fig. 5-A). Proteobacteria are susceptible to environmental disturbances (Schimel *et al*. 2007), and their activity and reproductive ability are highly dependent on soil organic carbon content (Mukhopadhya *et al*. 2012), which may explain the lower abundance of Proteobacteria under the low input of rhizospheric carbon under brackish water irrigation. Similarly, Guo *et al*. (2019) found that brackish water irrigation significantly reduced the relative abundance of Proteobacteria by 6.57% compared to that under fresh water irrigation. The increased relative abundance of Chloroflexi under brackish water irrigation may be associated with their spore-forming ability and Gram-positive cell walls, making them highly resistant to salt stress (Schimel *et al*. 2007) and thus are usually found in high-salt environments (Valenzuela-Encinas *et al*. 2009; Hu *et al*. 2020). Furthermore, we found that biochar application significantly increased the relative abundances of Proteobacteria and Patescibacteria (Fig. 5-A). Yin *et al*. (2021) reported that the relative abundance of Proteobacteria was increased by 13.0% under corn stover biochar treatment in comparison with a no-biochar treatment. This may be because Proteobacteria are nutrient-rich bacteria, and biochar improves soil nutrient status (Ding *et al*. 2016), creating a favorable environment for Proteobacteria. Patescibacteria are a diverse group of bacteria constituting most of the microbial dark matter (Wang Y X *et al*. 2023). Shi *et al*. (2023) found that biochar application significantly increased the relative abundance of Patescibacteria. This may be because the porous structure of biochar provided shelter and increased soil nutrient retention (e.g., TOC, AN), thereby favoring Patescibacteria survival and reproduction.

NMDS and PERMANOVA analyses revealed that water and biochar treatments significantly altered the bacterial community structure, while brackish water treatment did not significantly change it (Fig. 6). The soil bacterial community structure is influenced by the abundance of the various bacterial community phyla and their interactions, which are easily changed by the soil microenvironment. As mentioned above, the significant changes in SWC, TOC, and AN caused by deficit irrigation were the main factors that altered the structure of the soil bacterial communities. Previous studies have found that water stress also causes microbial cells to lyse and release intracellular enzymes, leading to alteration of the microbial community structure (Gleeson *et al*. 2008; Nguyen *et al*. 2018).

The altered soil structure and nutrient content caused by biochar can significantly affect the relative abundance of soil bacteria and alter the competition or cooperation relationships within the bacterial community. Most researchers believe that biochar may alter bacterial community composition by improving soil physicochemical properties, and further affect the community structure (Stark *et al*. 2012; Zhang *et al*. 2018; Zhou *et al*. 2019b; Woolet and Whitman 2020). Although in our study two years of brackish water irrigation changed the relative abundance of some dominant bacteria, it did not significantly alter the community structure of bacteria overall. This result is consistent with the finding of Mark Ibekwe *et al*. (2017), who reported that brackish water irrigation (0.85-15.0 dS m^{-1}) did not significantly affect soil bacterial community structure in their study. In summary, we have observed that water, salt, and biochar exert various effects on the composition and structure of soil bacterial communities, and in turn, the reconfiguration of bacterial communities also has potential impacts on soil nutrient cycles and ecological functions in the future.

4.3. Mechanisms of water, salt and biochar treatments in changing soil quality

In the study we selected TP, GWD, SWC, Observed species, and Simpson indexes from sixteen soil indicators through the MDS method to evaluate soil quality. Furthermore, the PLS-PM was used to analyze the mechanisms that biochar altered soil quality under various water-salt treatments. Compared to full irrigation with fresh water (W1S0), deficit and brackish water irrigation significantly reduced SQI through decreasing some soil indicators, which had a potential adverse impact on maize yield (Fig. 4; Appendix A). Previous studies have also shown that deficit or brackish water irrigation may reduce some soil physical and chemical properties (Worku and Bedadi 2016; Zhang *et al*. 2019; Deng *et al*. 2021), and soil microbial richness and diversity (Litchfield *et al*. 2009; Prudent *et al*. 2020), ultimately reducing the soil quality composite index (Xian *et al*. 2019; Saha *et al*. 2020; Lin *et al*. 2023). Interestingly, we found that biochar application under full irrigation with both fresh and brackish water enhanced soil quality only by improving soil aggregate and pore structures (Figs. 4 and 7). Wang *et al*. (2003) and Nimmo (2004) believe that good soil pore structure allows the ability to absorb and release water, transport nutrients, and support microbial communities, thus favoring soil quality. Soil aggregate structure resists mechanical damage from erosion, and affects soil structure and water conductivity, which is a key indicator for assessing soil quality (Shukla *et al*. 2006; Delelegn *et al*. 2017). Previous studies have shown that biochar could enhance soil quality by improving soil aggregate and pore structures (Głąb *et al*. 2016; Zhang *et al*. 2021; Hafeez *et al*. 2022). However, in the present study biochar did not effectively improve soil aggregate structure under 50% of full irrigation, thus failing to significantly enhance soil quality under this treatment. It was evident that excessively dry soil conditions hindered the effectiveness of biochar in improving soil aggregate structure, so future research needs to explore more suitable irrigation levels for biochar application to improve soil quality while achieving high water use efficiency. Although PLS-PM indicated that soil bacterial community richness was also a crucial factor affecting soil quality, biochar failed to effectively increase bacterial community richness. Therefore, the use of a biochar-based synthetic microbial community (Yu *et al*. 2024) or biochar-based fertilizers (Zhou *et al*. 2019a; Qasim *et al*. 2024) to improve microbial communities and enhance soil quality may be more effective than applying biochar alone. Based on these views, we aim to maximize soil quality and crop yield through more effective water, salt, and biochar management strategies in the future.

Previous studies have indicated that microbial community structure is widely recognized as an effective indicator of soil quality (Zornoza *et al*. 2009; Khan *et al*. 2010; García-Orenes *et al*. 2013). However, our results showed that the changes in microbial community structure did not directly affect soil quality (Fig. 7). In a review, Philippot *et al*. (2023) found that there was a mutual feedback relationship between soil microbes and soil properties including soil structure. It is possible that individual microbial communities improved soil quality by influencing soil structure, while they did not necessarily dominate the changes in the overall community structure. The beneficial phyla of Proteobacteria and Bacteroidetes were positively correlated with soil pore and aggregate structures, respectively, which could evidence this point (Fig. 5-B). Some previous studies have also reported that the changes in these specific bacterial phyla (e.g., Proteobacteria and Bacteroidetes) could directly affect soil quality (Shi *et al*. 2021; Kruczyńska *et al*. 2023). Thus, biochar may improve soil quality by influencing specific bacterial communities, providing a reference for the future targeted modification of soil bacterial communities (Ruan *et al*. 2024).

5. Conclusion

In this study, we assessed soil quality indices by observing soil physicochemical and microbiological properties after two years of biochar application under different water-salt conditions in the arid region of Northwest China. The results indicated that deficit and brackish water irrigation both degraded soil quality. Comparative analysis combined with PLS-PM indicated that biochar improved soil quality by enhancing soil pore and aggregate structure under full irrigation with fresh and brackish water. However, biochar only improved soil pore structure but failed to significantly enhance soil aggregate structure and microbial diversity, resulting in no significant improvement in soil quality under deficit irrigation. Both water and biochar treatments significantly altered the soil bacterial community structure, but brackish water treatment had no significant impact on it. Moreover, there was no direct influence of bacterial community structure on soil quality, however applying biochar could significantly enhance soil quality under more suitable soil water and salt regimes. These findings provide valuable insights into the applicability and mechanisms of biochar for improving soil quality under different water-salt conditions, as well as guidance for future directions in soil improvement in arid regions.

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Declaration of competing interests

The authors declare that they have no conflict of interest.

Appendices associated with this paper are available at http://www.ChinaAgriSci.com/

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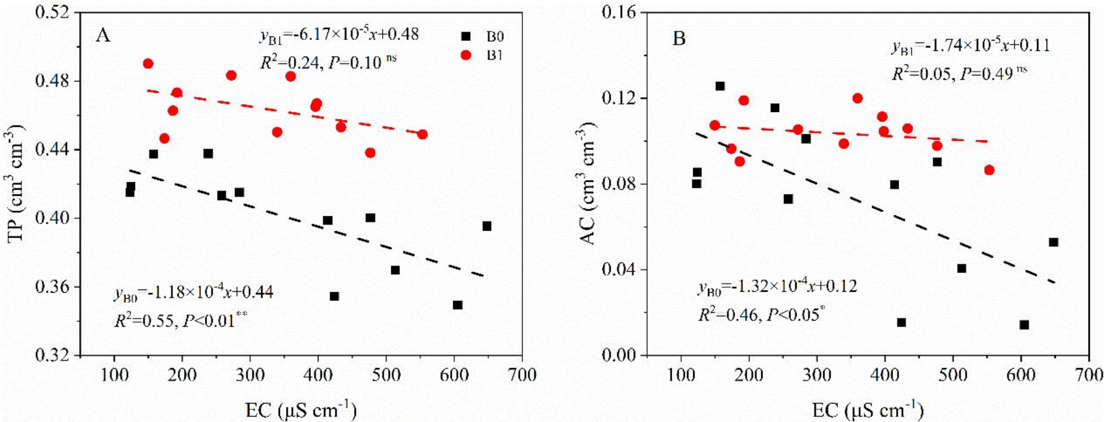

Fig. 2 Regression analysis between soil total porosity (TP) and electrical conductivity (EC) (A), as well as between air capacity (AC) and electrical conductivity (B) for biochar application (B1, red circles) and without biochar application (B0, black squares) treatments. ns, non-significant; *, $P<0.05$; **, $P<0.01$.

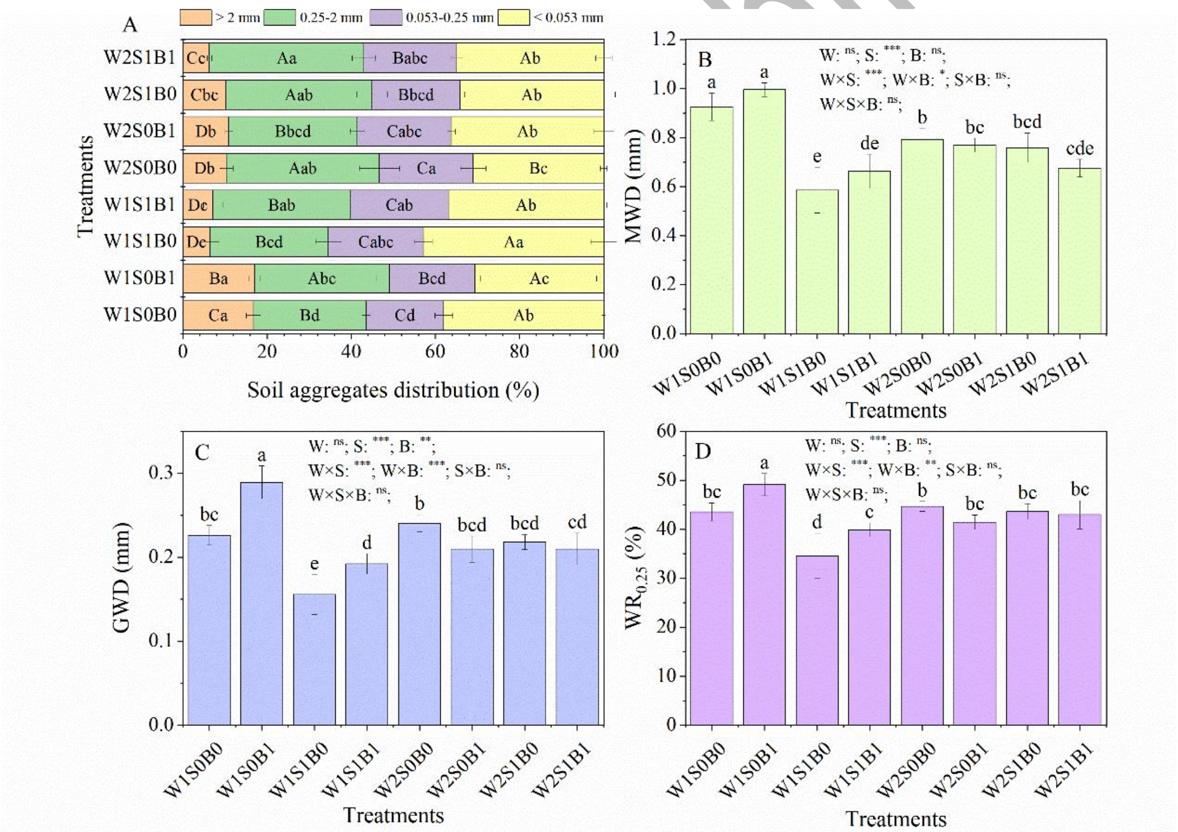

Fig. 3 Effects of biochar application on soil water-stable aggregate distribution (A) and aggregate stability indexes (MWD, B; GWD, C; $WR_{0.25}$, D) under different water-salt conditions. W1, full irrigation; W2, deficit irrigation; S0, fresh water; S1, brackish water; B0, no biochar application; B1, biochar application; MWD and GWD are the mean weight diameter and geometric mean diameter of soil water-stable aggregates, respectively; $WR_{0.25}$ is the proportion of water-stable aggregates with a diameter greater than 0.25 mm; ns,

non-significant; *, $P<0.05$; **, $P<0.01$; ***, $P<0.001$. Values are mean±SD (n=3), and bars with different capital letters indicate significant differences among aggregate size group, and bars with different lowercase letters indicate significant differences among treatments ($P<0.05$).

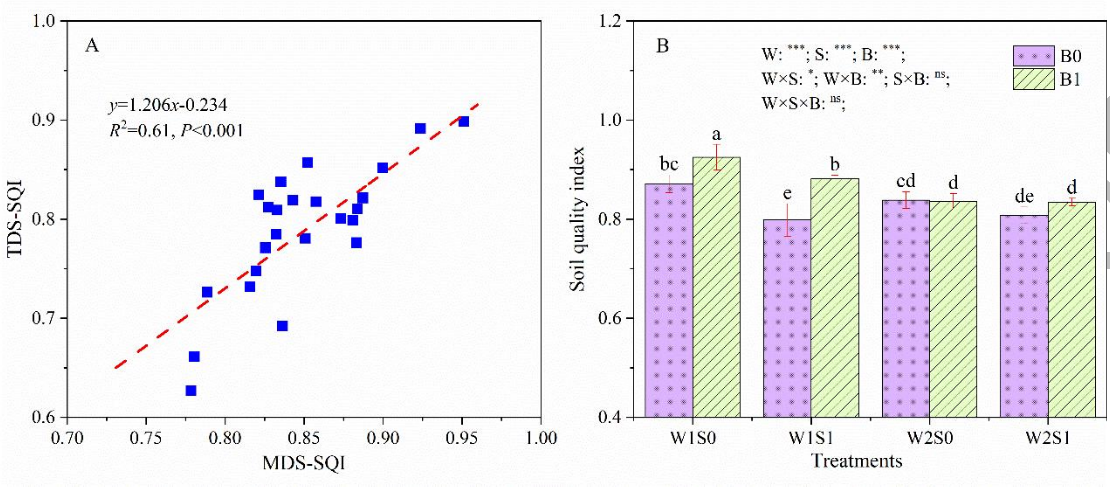

Fig. 4 Regression analysis for soil quality index between Minimum Data Set (MDS) and Total Data Set (TDS) (A), and the comparison of soil quality index calculated by Minimum Data Set under different treatments (B). W1, full irrigation; W2, deficit irrigation; S0, fresh water; S1, brackish water; B0, no biochar application; B1, biochar application; ns, non-significant; *, $P<0.05$; **, $P<0.01$; ***, $P<0.001$. In (B), values are mean±SD (n=3), and bars with different lowercase letters indicate significant differences among treatments ($P<0.05$).

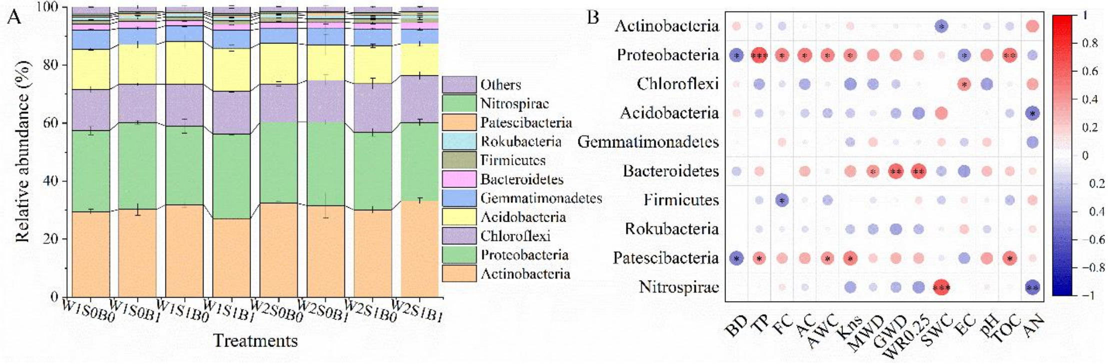

Fig. 5 Relative abundance of the top 10 bacterial phyla under different treatments (A), and Pearson correlation analysis between the relative abundance of the top 10 bacterial phyla and soil physicochemical properties (B). W1, full irrigation; W2, deficit irrigation; S0, fresh water; S1, brackish water; B0, no biochar application; B1, biochar application. BD, bulk density; TP, total porosity; FC, field capacity; AC, air capacity; AWC, available water content; K_{ns}, near saturated hydraulic conductivity; SWC, soil water content;

EC, electrical conductivity; TOC, total organic carbon; AN, available nitrogen; MWD, mean weight diameter; GWD, geometric mean diameter; $WR_{0.25}$, the proportion of water-stable aggregates with a diameter greater than 0.25 mm. *, $P<0.05$; **, $P<0.01$; ***, $P<0.001$.

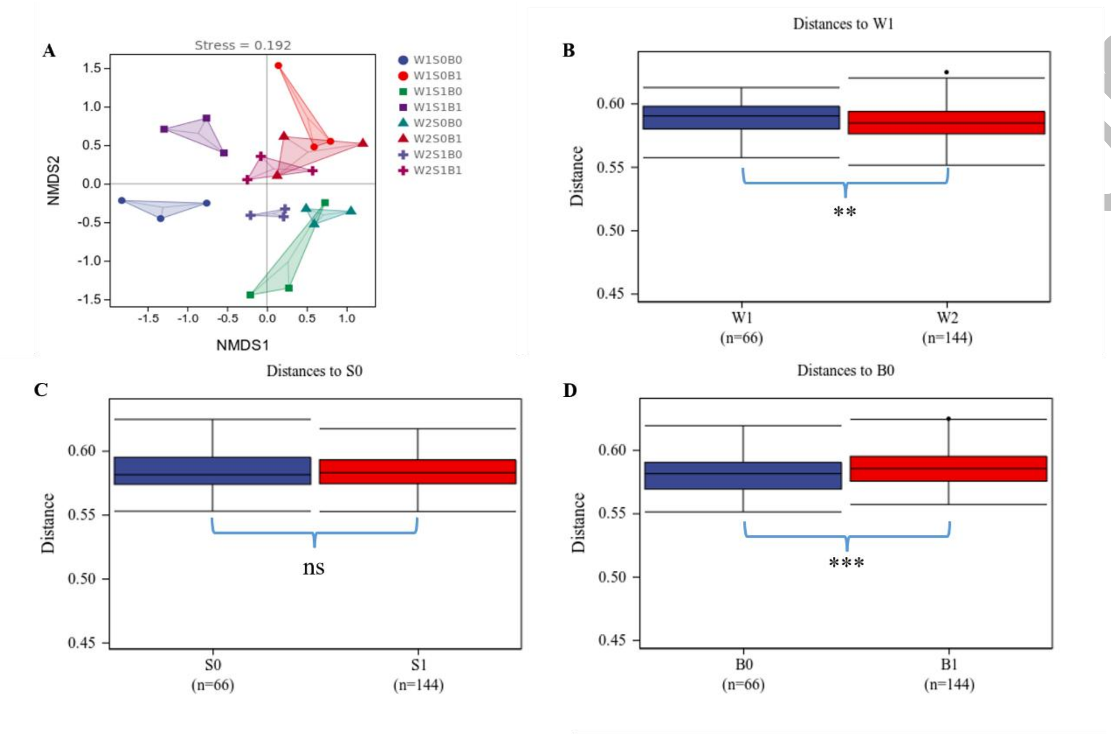

Fig. 6 Non-metric multidimensional scaling (NMDS) analysis for the soil bacterial community structure under different treatments (A). The PERMANOVA analysis was used to test for the significance of differences in bacterial community structure between full (W1) and deficit irrigation (W2) (B), between fresh water (S0) and brackish water irrigation (S1) (C), and between without biochar application (B0) and biochar application (B1) (D). ns, non-significant; **, $P<0.01$; ***, $P<0.001$.

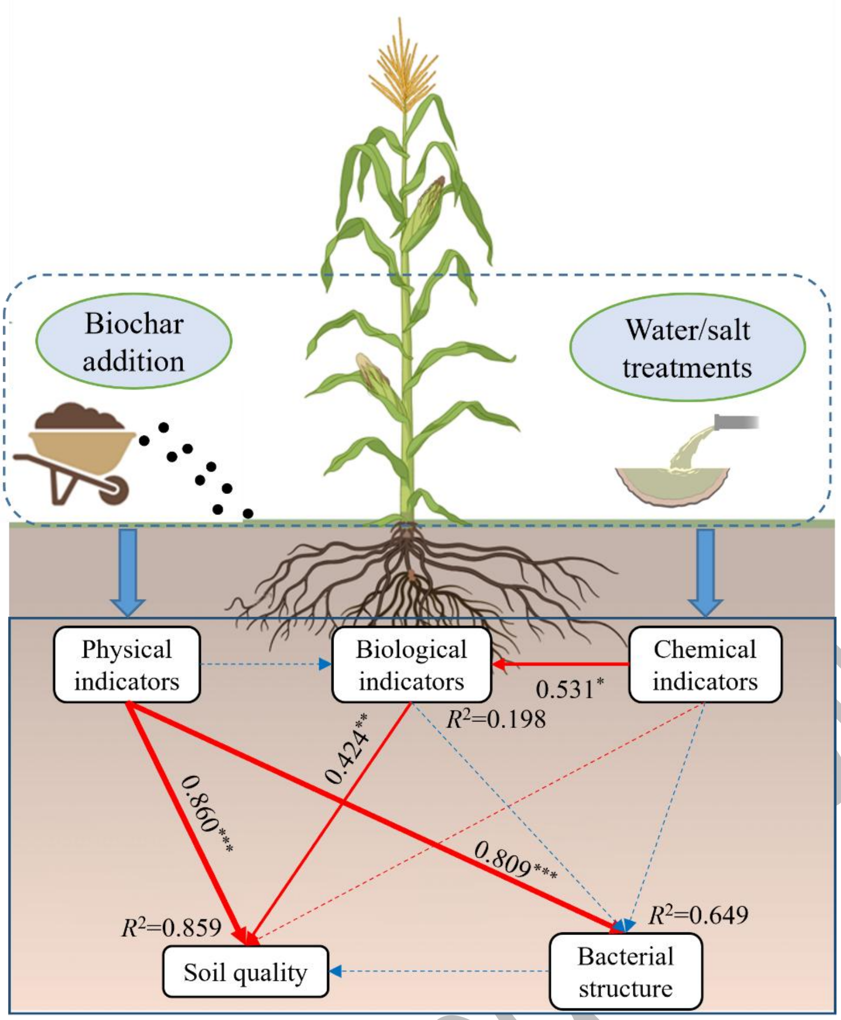

Fig. 7 Partial least squares path models (PLS-PM) describing direct and indirect effects of soil properties on bacterial composition and soil quality under combined water-salt-biochar treatments system. Red and blue solid lines indicate significant positive and negative coefficients, respectively, and arrow thickness indicates the magnitude of the standardized path coefficient. Red and blue dashed lines indicate insignificant positive and negative coefficients, respectively. The number near the solid arrows indicates the path coefficients and the significance level. R^2 value indicates the proportion of variance explained for each endogenous variable. The first axis of NMDS was used to represent microbial community composition. *, $P<0.05$; **, $P<0.01$; ***, $P<0.001$.

Table 1 Irrigation water , water salinity, and biochar application treatments in 2020 and 2021, respectively.

Treatment[1)]	2020			2021		
	Irrigation water (mm)	Water salinity ($g\ L^{-1}$)	Biochar application ($t\ ha^{-1}$)	Irrigation water (mm)	Water salinity ($g\ L^{-1}$)	Biochar application ($t\ ha^{-1}$)
W1S0B0	421	0.71	0	532	0.71	0
W1S0B1	421	0.71	60	532	0.71	0
W1S1B0	421	4.00	0	532	4.00	0
W1S1B1	421	4.00	60	532	4.00	0
W2S0B0	230.5	0.71	0	286	0.71	0
W2S0B1	230.5	0.71	60	286	0.71	0
W2S1B0	230.5	4.00	0	286	4.00	0
W2S1B1	230.5	4.00	60	286	4.00	0

[1)] W1, full irrigation; W2, deficit irrigation; S0, fresh water; S1, brackish water; B0, no biochar application; B1, biochar application.

Table 2 The effects of irrigation, water salinity and biochar application treatments on soil physical and chemical quality parameters[1)]

Treatment[2)]	BD (g cm^{-3})	TP (cm^3 cm^{-3})	FC (cm^3 cm^{-3})	AC (cm^3 cm^{-3})	AWC (cm^3 cm^{-3})	K_{ns} (cm d^{-1})	SWC (cm^3 cm^{-3})	EC (µS cm^{-1})	pH	TOC (g kg^{-1})	AN (mg kg^{-1})
W1S0B0	1.47 b	0.415 de	0.336 d	0.079 bc	0.175 b	178.56 bc	0.184 c	168.6 d	8.75 a	7.31 d	5.06 e
W1S0B1	1.26 de	0.475 a	0.370 a	0.106 ab	0.212 a	354.24 a	0.184 c	176.2 d	8.70 ab	21.67 a	6.26 e
W1S1B0	1.53 a	0.366 f	0.339 cd	0.027 d	0.171 b	69.12 c	0.205 b	559.1 a	8.55 d	5.91 e	6.54 de
W1S1B1	1.24 e	0.461 ab	0.356 abc	0.105 ab	0.209 a	112.32 bc	0.220 a	377.8 bc	8.68 abc	19.38 b	8.53 d
W2S0B0	1.38 c	0.430 cd	0.316 e	0.114 a	0.178 b	195.84 bc	0.153 e	226.9 d	8.68 abc	5.67 e	14.96 c
W2S0B1	1.27 de	0.471 ab	0.364 ab	0.107 ab	0.224 a	230.40 b	0.157 e	268.5 cd	8.68 abc	18.03 c	17.51 b
W2S1B0	1.44 b	0.389 ef	0.319 e	0.070 c	0.169 b	95.04 bc	0.170 d	468.2 ab	8.59 cd	5.99 e	16.14 bc
W2S1B1	1.30 d	0.447 bc	0.350 bcd	0.097 abc	0.209 a	221.76 b	0.178 cd	487.7 ab	8.63 bcd	17.77 c	20.00 a

[1)] BD, bulk density; TP, total porosity; FC, field capacity; AC, air capacity; AWC, available water content; K_{ns}, near saturated hydraulic conductivity; SWC, soil water content; EC, electrical conductivity; TOC, total organic carbon; AN, available nitrogen.

[2)] W1, full irrigation; W2, deficit irrigation; S0, fresh water; S1, brackish water; B0, no biochar application; B1, biochar application.

Values within a column without a letter in common are significantly different by Duncans multiple range test ($P<0.05$).

Table 3 Effects of biochar addition, irrigation and salinity on soil bacterial richness and diversity indices.

Treatment[1)]	Richness indices		Diversity indices	
	Chao1	Observed species	Shannon	Simpson
W1S0B0	6,608 a	5,805 a	11.05 a	0.9989 a
W1S0B1	6,072 ab	5,353 ab	10.96 a	0.9980 a
W1S1B0	5,841 ab	5,176 ab	10.86 a	0.9986 a
W1S1B1	5,917 ab	5,268 ab	11.00 a	0.9990 a
W2S0B0	5,941 ab	5,162 ab	10.83 a	0.9987 a
W2S0B1	5,808 ab	5,050 ab	10.81 a	0.9986 a
W2S1B0	5,388 b	4,812 b	10.84 a	0.9987 a
W2S1B1	5,412 b	4,796 b	10.78 a	0.9986 a

[1)] W1, full irrigation; W2, deficit irrigation; S0, fresh water; S1, brackish water; B0, no biochar application; B1, biochar application.

Values within a column without a letter in common are significantly different by Duncans multiple range test (*P*<0.05).