Pedosphere **36**(2): 387–401, 2026
doi: 10.1016/j.pedsph.2025.09.014
ISSN 1002-0160/CN 32-1315/P

Published by Elsevier B.V. and Science Press

PEDOSPHERE

www.elsevier.com/locate/pedosphere

Amelioration of coastal saline soils of Chinese river deltas *via* soil amendments: A meta-analysis

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(Received February 8, 2025; revised August 24, 2025; accepted September 10, 2025)

ABSTRACT

The amelioration of coastal saline soils has the potential to deliver large areas of farmlands to meet the increasing global food demand. Although numerous studies have been conducted in recent years, a systematic evaluation of the efficacy of various amendments for saline soil amelioration remains absent. Hence, a meta-analysis was performed to compare the effects of various amendments on soil physicochemical properties and crop yield, based on data extracted from 97 peer-reviewed publications. *Our results indicate* that the application of organic amendments (*i.e.*, biochar, compost, crop residues, and organic wastes) reduces soil salinity and bulk density, while simultaneously increasing the soil macroaggregate ratio and crop yield. Due to the purposive selection of raw materials, organic amendments generally exhibit positive effects on replenishing soil carbon, nitrogen, and phosphorus pools, which alleviates nutrient deficiencies in coastal saline soils. However, gypsum is unsuitable for ameliorating coastal saline soils, since there are no significant improvements observed in soil salinity and nutrient contents. *Our meta-regression results* indicate that the effects of amendments are either positively correlated or show no relationship with application amounts. Overall, *this meta-analysis reveals* that the application of organic amendments is an effective way for ameliorating coastal salt-affected soils. However, optimal application rates should be assessed to mitigate potential environmental risks in future studies.

Key Words: amendment efficacy, biochar, compost, crop residue, crop yield, organic amendment, organic waste, salt-affected soil, soil nutrient, soil organic carbon, soil salinity

Citation: Tao J, Liu X, Spielvogel S. 2026. Amelioration of coastal saline soils of Chinese river deltas *via* soil amendments: A meta-analysis. *Pedosphere*. **36**(2): 387–401.

INTRODUCTION

Soil salinization is one of the most serious processes of global soil degradation, which threatens both the quantity and quality of farmlands and endangers food security. The reclamation of saline soils is a potential strategy to meet the growing food demand, as these soils can be transformed into productive farmlands with proper management practices (Garcia-Franco *et al*., 2021; Mukhopadhyay *et al*., 2021). More than 424 million hectares of topsoil (0–30 cm) are subjected to soil salinization (FAO, 2021). Approximately one million hectares of coastal soils in China are suffering from soil salinization, primarily distributed in the Yellow River Delta (YRD) and the Yangtze River Estuary (YRE) (Zhao *et al*., 2013; Xie *et al*., 2021). These two regions are among the most dynamic and densely populated coastal areas globally (Huang and Lu, 2015; Hu *et al*., 2019; Liu *et al*., 2021). However, nearly 70% of the total area of the YRD is threatened by soil salinization (Fan *et al*., 2012), while the salinized area in the YRE accounts for approximately 25% of the mudflat area in China. The primary cause of soil salinization in coastal areas is seawater intrusion caused by rising sea levels, and the extent of salinization continues to expand at an unprecedented annual rate (Fan *et al*., 2012; Zhao *et al*., 2020). Given the extensive distribution of saline soils in these two regions, their reclamation has the potential to supply a considerable area of arable land.

Compared to engineering approaches and plant-based technologies, the application of soil amendments is more economical and convenient (Fall *et al*., 2018; Gunarathne *et al*., 2020; Pedrero *et al*., 2020) and can alleviate soil nutrient deficiency (Montiel-Rozas *et al*., 2018). The diversity of soil amendments originates from the wide choice of raw materials. Most raw materials for soil amendment are byproducts or waste materials derived from other industries. Flue gas desulfurization gypsum, a byproduct of power generation plants, is widely used to ameliorate saline-sodic soils (Luo *et al*., 2018; Wang and Yang, 2018; Zhang *et al*., 2020). Some studies reported that gypsum application decreased soil salinity by improving soil structure (Chi *et al*., 2012; Morsy *et al*., 2022), whereas others observed that adding

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gypsum increased soil salinity by introducing calcium sulfate (Zia *et al*., 2006; Rasouli *et al*., 2013).

Agricultural wastes and livestock manure are utilized as organic fertilizers to enhance soil fertility, either directly or following appropriate treatments (*e.g*., pyrolysis and composting) (Rubio *et al*., 2013; Yang *et al*., 2018; Chen *et al*., 2020). Crop residue return enhances soil organic carbon (SOC) stock in saline soils (Setia *et al*., 2013; Zhang W W *et al*., 2019), although this effect may be influenced by soil salinity level (He *et al*., 2022). Crop residue return generally inhibits soil salt accumulation by promoting soil aggregate formation (Choudhury *et al*., 2014). However, Wang *et al*. (2015) reported that straw return increased soil salinity, as certain cations, *e.g*., sodium (Na^+), were released into the soil solution during straw decomposition. In addition, straw amendment is generally believed to increase soil nutrient contents. However, Yang *et al*. (2020) indicated that soil available phosphorus (AP) content decreased following straw incorporation.

Biochar and compost have been frequently reported to alleviate the constraints of saline soils. Some studies observed that biochar addition reduced soil salt content more efficiently than compost (Ghazi, 2018; Wang *et al*., 2022), whilst Foronda (2022) reported an opposite result. In addition, excessive application of biochar or compost promotes SOC sequestration but may concurrently increase soil salinity (Miller *et al*., 2017; Gondek *et al*., 2020; Liu *et al*., 2022). However, the relationship between the ameliorative effects of amendments and their application amounts remains unclear. Nowadays, the production of domestic and industrial wastes is mounting due to rapid urbanization and population growth (Srivastava *et al*., 2015; Cao *et al*., 2019; Meena *et al*., 2019). Some of these waste products may be reused as soil amendments due to their high nutrient contents. However, potential safety concerns, such as heavy metal pollution and elevated soil salinity, must be carefully considered (Hargreaves *et al*., 2008; Meena *et al*., 2019; Jat Baloch *et al*., 2023).

Therefore, the effects of soil amendments on salt-affected soil reclamation remain ambiguous and sometimes controversial, despite extensive research having been conducted. Recent studies have highlighted the importance of amendment applications for saline soil amelioration. However, most studies have included both inland and coastal soils, despite notable differences in their genesis and salt composition. To ensure the sustainable development of coastal saline soils, a comprehensive evaluation of the efficacy of soil amendments is essential. Therefore, we conducted this meta-analysis to synthesize and summarize the results of existing publications. Here, two classical river deltas (YRD and YRE) in China were selected to minimize the potential influence of soil heterogeneity. The objectives of this meta-analysis are to: i) compare the effects of different amendments on soil structure, salinity, pH, and nutrient contents and crop yield; ii) explore key factors that may influence the performance of amendments; and iii) evaluate the feasibility of applying soil amendments in agricultural practices.

DETAILED DESCRIPTION OF YRD AND YRE

The YRD and YRE are essential ecological transitions between the continent and the sea in China. Situated on the southern coast of Bohai Bay and the western coast of Laizhou Bay (37°22′–38°04′ N, 118°14′–119°05′ E), the YRD covers an area of 5 171 km^2 (Yu *et al*., 2024). It belongs to the warm temperate continental monsoon climate zone, and the average annual temperature, precipitation, and potential evaporation are 11.5–12.9 °C, 549.3 mm, and 2 049.4 mm, respectively. The ratio of average evaporation to average precipitation is 3.73. The typical soil types in this area are Fluvisol and Solonchak, and the winter wheat-summer maize rotation system is the dominant planting system.

The YRE is situated at the confluence of the Yangtze River and the Pacific Ocean (27°02′–35°08′ N, 114°54′–123°10′ E). This area belongs to the subtropical monsoon climate zone, and the average annual temperature, precipitation, and potential evaporation are 16 °C, 1 144 mm, and 1 364 mm, respectively (Gao *et al*., 2020). The evaporation to precipitation ratio is 1.19. Rice is a typical crop in the area. Herein, we included studies that were conducted in the northern part of the YRE and the adjacent northern part of Jiangsu Province. This is because the main soil type of this area is Fluvisol which is comparable to the soils in YRD. Soil textures of the two regions cover a broad range from loamy sand to clay.

DATA COLLECTION AND ANALYSES

A systematic literature search for peer-reviewed publications was carried out up to December 2023 using the following databases: Web of Science Core Collection and China National Knowledge Infrastructure. The exact search string utilized is listed in Table SI (see Supplementary Material for Table SI). The EndNote 20 software was used to screen publications (Fig. S1, see Supplementary Material for Fig. S1). After removing reviews and duplicates, the remaining articles were further selected according to the following criteria to minimize publication bias and reduce heterogeneity: i) only field and pot experiments were included, while incubation and column experiments were excluded; ii) field experiments must be carried out in the YRD or YRE, and soil samples for pot experiments must be taken from these two regions; iii) amended and unamended treatments must be conducted under the same environmental conditions; and iv) at least one pair (treatment and control) of soil properties or crop yield must be reported in each study. All the data used in this meta-analysis were either obtained directly from the tables or extracted from the figures of the respective publication. Standard error was transformed to standard deviation (SD) by multiplying the respective square root of the sample size. Missing SDs were estimated by multiplying the average coefficient of variations calculated from the observations with SDs. Each variable was unified into its most commonly-used unit. Soil salinity measured by different methods was transformed to electrical conductivity (EC) (Table SII, see Supplementary Material for Table SII).

A total of 97 publications were included in the meta-analysis (Tables SIII and SIV, see Supplementary Material for Tables SIII and SIV), covering 11 variables: *soil salinity* represented by EC, pH, bulk density (BD), macroaggregate ($>$ 250 μm) ratio (MAR), SOC, microbial biomass carbon (MBC), available nitrogen (AN), total nitrogen (TN), AP, total phosphorus (TP), and crop yield. Different types and application rates of soil amendments were recorded as independent observations. Similarly, results for the same amendment from different sites (soils) within one study were recorded separately. The same results from one experiment were included only once, even if they were reported in different publications, while different variables from the same experiment were merged into one observation when reported in different publications. Moreover, only data of topsoil (0–20 cm) from the final sampling time were collected to ensure data comparability.

To minimize heterogeneity, soil amendments were categorized into the following subgroups: biochar, biochar plus organic materials (B-O), compost, crop residues, gypsum, gypsum plus organic materials (G-O), and organic wastes. Biochar was produced from plant-derived materials (*e.g.*, straw and grass). Pyrolysis temperature, pyrolysis time, and biochar EC and pH were summarized in Table SV (see Supplementary Material for Table SV). The salt content and pH of biochar used in YRE were higher than those used in YRD. The direct application of straw from plants such as cotton, rice, reed, wheat, maize, barley, and grass was defined as crop residue return. The average carbon (C)/nitrogen (N) and C/phosphorus (P) ratios of the various types of straw were 58 and 201, respectively (Table SVI, see Supplementary Material for Table SVI). Organic wastes referred to municipal sewage sludge and the compost out of it, which were rich in organic matter and nutrients. Compost covered various types of composted organic materials except for organic wastes, including farmyard manure, commercial organic fertilizer, vermicompost, and microbial organic fertilizers. The average C/N and C/P ratios of compost were higher than those of organic wastes. For the B-O subgroup, compost or living microorganisms were added and mixed with biochar. The organic materials used in G-O were either straw or compost.

Effect size was expressed as the natural logarithm of the response ratio (lnRR), which was calculated using Eq. 1, and its variance, Var(lnRR), was calculated using Eq. 2.

$$\mathrm{lnRR} = \ln\left(\frac{x_\mathrm{e}}{x_\mathrm{c}}\right) \tag{1}$$

$$\mathrm{Var(lnRR)} = \frac{\mathrm{SD}_\mathrm{e}^2}{n_\mathrm{e}x_\mathrm{e}^2} + \frac{\mathrm{SD}_\mathrm{c}^2}{n_\mathrm{c}x_\mathrm{c}^2} \tag{2}$$

where x_e and x_c are the means of the amended treatment and the control, respectively, $\mathrm{SD_e}$ and $\mathrm{SD_c}$ are the SDs of the amended treatment and the control, respectively, and n_e and n_c are the sample sizes of the amended treatment and the control, respectively. Extreme values can substantially affect the result of a meta-analysis, as the analysis is pooled by the weighted mean of all effect sizes, potentially influencing the validity and robustness of the conclusion (Viechtbauer and Cheung, 2010; Zhang *et al*., 2022).

To minimize the effect of extreme values, outliers were detected and removed using the Z-score method with a threshold of 2 (Pearson *et al*., 2014; Sehovic *et al*., 2022), and Z score was calculated using Eq. 3.

$$Z \text{ score} = \frac{|\mathrm{lnRR} - \mu|}{\sigma} \tag{3}$$

where μ and σ are the mean and SD of all the effect sizes from one variable, respectively. The mean effect sizes following outlier removal were not significantly different from those of the original data. A random effects model was applied to pool the mean effect size of each subgroup using the restricted maximum likelihood method, where the inverse variance method was used for weighting (Viechtbauer, 2005). The 95% confidence interval (CI) of the mean effect size was estimated using the Knapp and Hartung method (Knapp and Hartung, 2003). The percent of change (PC, %) in each variable was calculated using Eq. 4.

$$\mathrm{PC} = (\mathrm{e}^{\mathrm{lnRR}} - 1) \times 100 \tag{4}$$

The effects of different amendments were deemed significant when the 95% CIs did not overlap with 0. As for yield, MAR, and soil elemental pools, the effects were deemed significantly positive (negative) if their 95% CIs were completely higher (lower) than 0. However, the effects of amendments on soil salinity, pH, and BD were positive only if their 95% CIs were completely below 0. Egger's regression test for funnel plot asymmetry and the fail-safe number analysis (file drawer analysis) were conducted to examine potential publication bias. The result of a meta-analysis was deemed robust when the fail-safe number was greater than $5n + 10$ (where n is the number of observations). In addition, the relationships between effect sizes (lnRR) and soil amendment application amounts (or soil salinity) were examined using meta-regression analysis with a linear regression approach (if $n \geqslant 10$). The meta-analysis and meta-regression analysis were performed using the meta (version 6.5-0) and metafor (version 4.4-0) packages in R (version 4.3.0).

To enhance the robustness of this meta-analysis, the following criteria were developed to assemble and compare the overall effects of different amendments on soil properties and crop yield based on the sample size and the 95% CI of the effect sizes. The first criterion was that the effect of an amendment was defined and marked as "not clear" when the number of observations was less than five, or the number of studies was less than three, due to the "small sample effect". The second criterion was that when the mean effect size was positive (negative), but the 95% CI overlapped with 0, we assumed that the effect might exist and marked it as "potentially positive (negative)". The third criterion was that only when both the mean effect size and the 95% CI were above (below) 0, we assumed that the effect was significant, and marked it as "probably positive (negative)". In terms of soil salinity, pH, and BD, the second and third criteria were applied reversely since decreases in these variables had positive effects.

EFFECT OF AMENDMENTS ON SOIL PHYSICAL PROPERTIES AND CROP YIELD

The mean effect sizes of various amendments on soil salinity were generally negative (ranging from −0.22 to −0.08), except for gypsum which caused an overall increase in soil salinity by 7.6% compared to the unamended control (Fig. 1). The application of organic wastes, compost, and crop residues significantly reduced soil salinity by 20.1%, 18.2%, and 10.8%, respectively, since their 95% CIs were completely below 0. In addition, the application of G-O likely reduced soil salinity by 10.4%. However, this effect was not significant because the upper bound of the 95% CI was slightly higher than 0 (7.34×10^{-4}). Statistically, biochar application exhibited a significant desalinization effect, although the upper bound of the 95% CI was very close to 0 (-2.77×10^{-3}). Meta-regression results revealed that the effect sizes of biochar, compost, and crop residues were insensitive to initial soil salinity level (Fig. 2). In contrast, the effect size of organic wastes was negatively correlated with the initial soil salinity level. Meta-regression analysis indicated that the desalinization effect of organic wastes was significant only when the initial soil salinity exceeded a threshold of 1.96 dS m^{-1}.

All amendments had negative mean effect sizes on soil pH, but the reductions in soil pH were less than 5% (Fig. 1). Application of organic wastes led to the highest reduction of soil pH by 4.4%, which was two-fold larger than those caused by compost and crop residues. The effect of gypsum application on soil pH was not stable, although its mean effect size was almost equal to that of organic wastes. In addition, biochar addition had the lowest mean effect size (−0.003), and the upper bound of its 95% CI even surpassed 0, meaning that soil pH responded weakly to biochar addition.

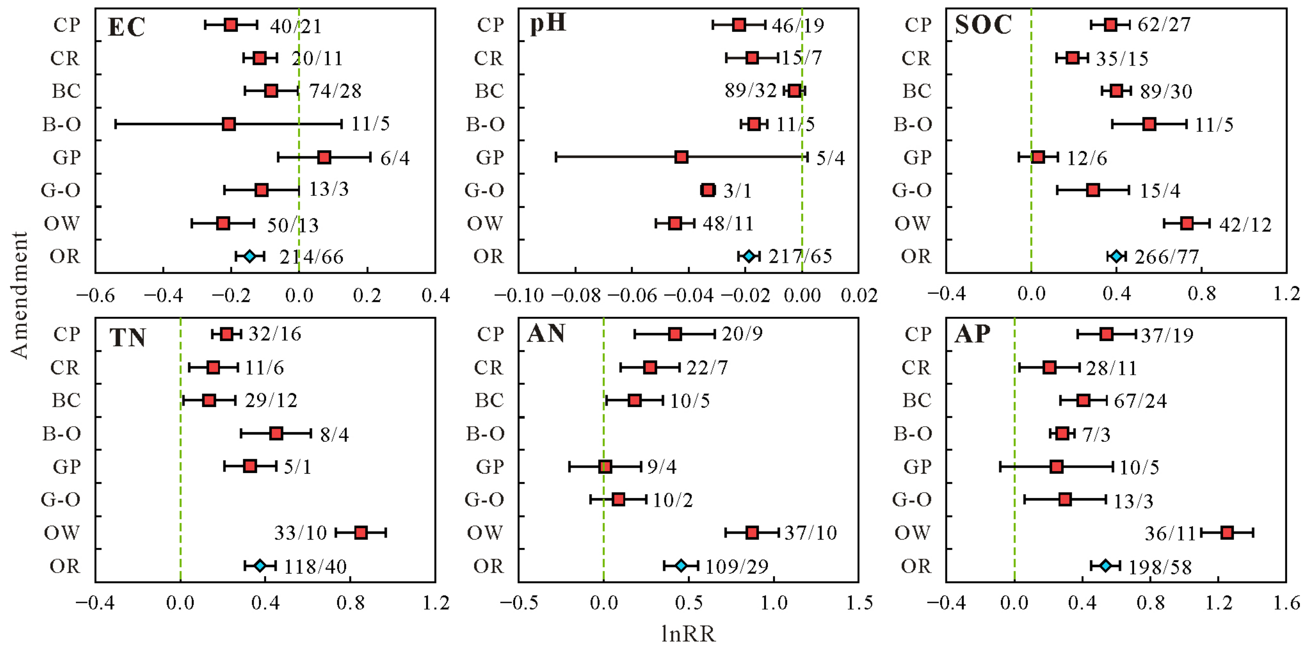

Fig. 1 Effect sizes, expressed as the natural logarithm of the response ratio (lnRR), of different amendments on soil salinity (represented by electrical conductivity (EC)), pH, organic C (SOC), total N (TN), available N (AN), and available P (AP) in the Yellow River Delta and the Yangtze River Estuary of China. Error bars represent 95% confidence intervals. The numbers beside error bars are the numbers of observations/numbers of publications. CP = compost; CR = crop residues; BC = biochar; B-O = biochar plus organic materials; GP = gypsum; G-O = gypsum plus organic materials; OW = organic wastes; OR = overall.

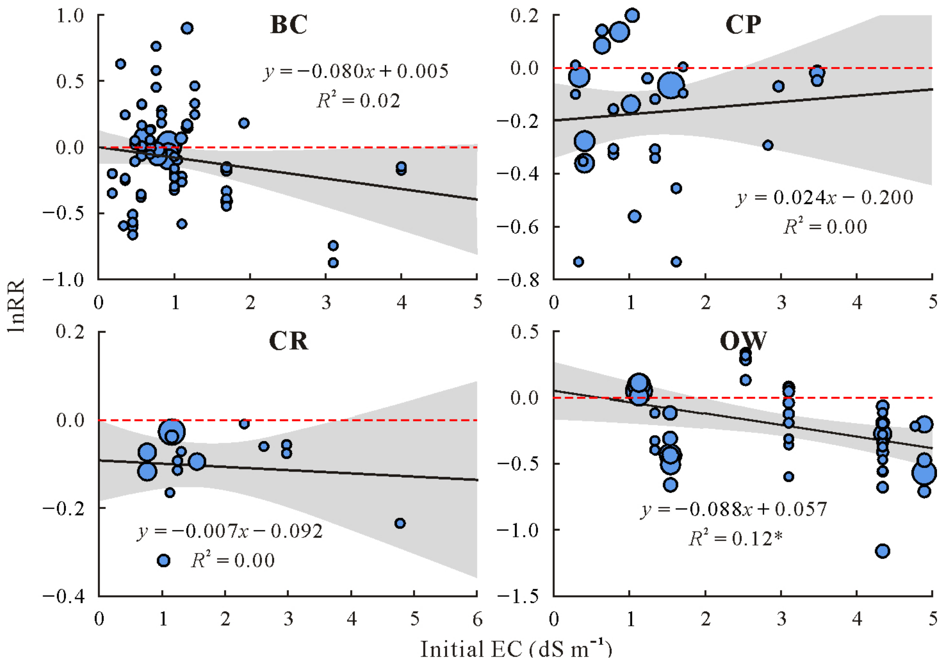

Fig. 2 Relationships between the effect sizes, expressed as the natural logarithm of the response ratio (lnRR), of different amendments on soil salinity and initial soil salinity levels, represented by electrical conductivity (EC) values, in the Yellow River Delta and the Yangtze River Estuary of China. Circle size denotes the inverse variance of effect size. Shaded area indicates the 95% confidence interval of the regression line. Asterisk * indicates significance at $P <$ 0.05. BC = biochar; CP = compost; CR = crop residues; OW = organic wastes.

Our results confirmed that the application of crop residues, biochar, compost, and organic wastes significantly increased soil MAR, decreased soil BD, and enhanced crop yield (Table SVII, see Supplementary Material for Table SVII). The improvements of these variables followed the same trend: organic wastes > biochar > compost > crop residues, except that biochar application led to the lowest mean reduction in soil BD (by 4.8%). In addition, crop yield increased by 125.1% following the application of organic wastes, which was 4.2, 6.5, and 14.3 times greater than those caused by biochar, compost, and crop residue application, respectively. However, the effect of organic wastes on soil BD was not significantly different from those of the other amendments.

EFFECTS OF AMENDMENTS ON SOIL C, N, AND P POOLS

The SOC pool increased considerably after the application of organic amendments (Fig. 1). The addition of organic wastes led to the highest overall increase in SOC by 107.8%, which was 2.2, 2.4, and 5.1 times greater than those caused by biochar, compost, and crop residues, respectively. The application of G-O increased SOC by 33.5%, while the sole application of gypsum had no impact on SOC. Additionally, the application of compost, biochar, and crop residues increased MBC by 42.1%, 44.0%, and 31.4%, respectively (Table SVII), although no significant difference was found among these amendments.

All amendments significantly promoted the accumulation of TN, with the addition of organic wastes resulting in the highest increase of 134.1% (Fig. 1). The increases in TN induced by biochar, crop residues, and compost were 89.1%, 87.4%, and 81.8%, respectively, significantly lower than that induced by organic wastes. Similarly, applying organic wastes induced the highest increase in AN by 139.3%, which was 7.0, 4.5, and 2.7 times higher than those induced by biochar, crop residues, and compost, respectively.

The effects of all amendments on soil AP were generally positive and significant except for gypsum (Fig. 1). The highest increase in AP was observed with the application im organic wastes (249.3%), followed by compost (71.9%) and biochar (50.1%). The application of B-O or G-O did not promote soil AP accumulation compared to the single application of biochar or gypsum. Furthermore, applying organic wastes and compost significantly increased soil TP by 77.2% and 54.3%, respectively, while biochar application and crop residue return did not demonstrate a clear effect on soil TP (Table SVII).

EFFECTS OF AMENDMENTS INFLUENCED BY CUMULATIVE APPLICATION AMOUNTS

The cumulative application amounts of soil amendments differed significantly across studies (Fig. 3, Table SVIII, see Supplementary Material for Table SVIII). However, the limited sample sizes only allowed us to examine the potential interactions between the effects of biochar, crop residues, compost, and organic wastes on soil properties and crop

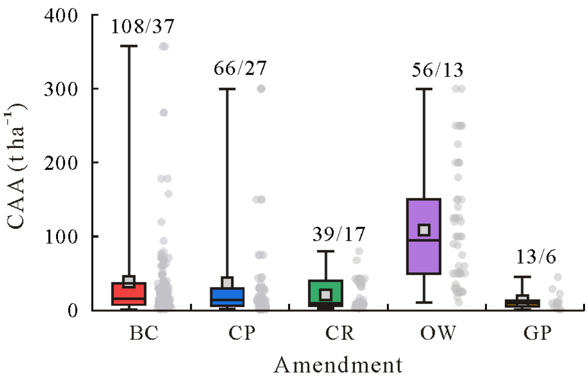

Fig. 3 Boxplots with individual data points of cumulative application amounts (CAAs) of different amendments of biochar (BC), compost (CP), crop residues (CR), organic wastes (OW), and gypsum (GP) in the Yellow River Delta and the Yangtze River Estuary of China. Boxes show 25–75 percentiles, whiskers show 10–90 percentiles, horizontal lines within boxes are medians, and open squares are means. The numbers above the upper whiskers are the numbers of observations/numbers of publications.

yield and the respective cumulative application amounts of these amendments. The effects of organic wastes on soil properties and crop yield increased with increasing cumulative application amounts (Table SIX, see Supplementary Material for Table SIX). Similarly, the effects of compost on soil salinity, pH, TN, AN, AP, and MAR were more pronounced under higher cumulative application amounts, whereas its effects on SOC, TP, BD, and crop yield were insensitive to cumulative application amounts (Fig. 4). In contrast, our results showed that the effect sizes of crop residues and biochar were generally insensitive to cumulative application amount variations except for the effect of biochar on SOC. The replenishing effect was positively correlated with biochar cumulative application amounts. Additionally, the cumulative application amounts of organic wastes were considerably higher than those of other amendments. This is because the lower quartile of organic waste addition was greater than the upper quartile of the cumulative application amounts of biochar, compost, and crop residues (Fig. 3, Table SVIII).

EFFECTS OF BIOCHAR AND COMPOST IN YRD AND YRE

The effects of biochar application on soil properties were similar in YRD and YRE (Fig. 5a). The increase in SOC in YRE was generally higher than that in YRD, although no significant difference was found between the two regions. However, the promotion of crop yield in YRE was significantly lower by 60.2% than that in YRD. In addition, soil TN and AN were insensitive to biochar addition in both regions, although they were significantly increased when the dataset was not subdivided into regions (Fig. 1). Likewise, biochar application had no significant effect on soil salinity in both regions, since the upper bounds of the 95% CIs in YRE (0.005) and YRD (0.026) were both larger than 0.

Compost application was generally more beneficial in YRE than in YRD, since the cumulative application amount of compost was also higher in YRE (Table I). Specifically, the increases in SOC, TN, and AP in YRE were 2.0, 2.3, and 3.2 times higher than those in YRD (Fig. 5b). Moreover, compost application had no significant effects on soil pH, TP, and crop yield in YRD, whereas it significantly reduced soil pH by 3.6% and increased soil TP and crop yield by 79.2% and 34.5%, respectively, in YRE.

PUBLICATION BIAS AND OVERALL EVALUATION

The results of Egger's regression test suggested no significant potential publication bias for soil salinity, pH, SOC, MBC, AN, AP, and BD. The potential publication biases for TN, TP, MAR, and crop yield would not affect the results of the meta-analysis, since the (Rosenthal's) fail-safe numbers were considerably larger than $5n + 10$ (Table SX, see Supplementary Material for Table SX). Generally, the application of organic amendments (*i.e.*, biochar, compost, crop residues, and organic wastes) was beneficial for ameliorating coastal saline soils. Compost was the only amendment that significantly improved all variables evaluated in this study (Fig. 6). In addition, organic wastes exhibited prevailing positive effects on saline soil reclamation except for MBC, which was not reported in the original publications.

PROPER UTILIZATION OF AMENDMENTS IN COASTAL SALINE SOILS

Soil desalinization occurs either by enhancing leaching through soil structure improvement or by deactivating excessive soluble ions. The primary soluble ions in coastal areas are Na^+ and Cl^- (Liu *et al.*, 2023), which cannot be removed or deactivated *via* chemical reactions (*e.g.*, deposition) due to their high chemical stability and solubility. The most feasible ways for desalinizing coastal soils are leaching and adsorption (Chaganti *et al.*, 2015). However, adsorption only temporarily reduces soil soluble Na^+ and Cl^- concentrations, because it does not remove these ions from the soil profile. Deficiencies in organic matter and nutrients are the second major constraint on the productivity of saline soils (Liu *et al.*, 2015; Zhang *et al.*, 2015). Most soil amendments replenish soil elemental pools because they are derived from organic materials (Rubio *et al.*, 2013; Elgharably and Ito, 2014; Chahal *et al.*, 2017; Yang *et al.*, 2018). Amendment application has the potential to simultaneously address the two primary limitations of coastal saline soils. However, soil amendments may have adverse effects on soil salinity, because the raw materials may contain certain amounts of salts. Moreover, soil nutrient contents, especially available nutrients, may decrease due to stoichiometric imbalances induced by these amendments. Amendments modulate nutrient availability and metal dynamics by stimulating soil

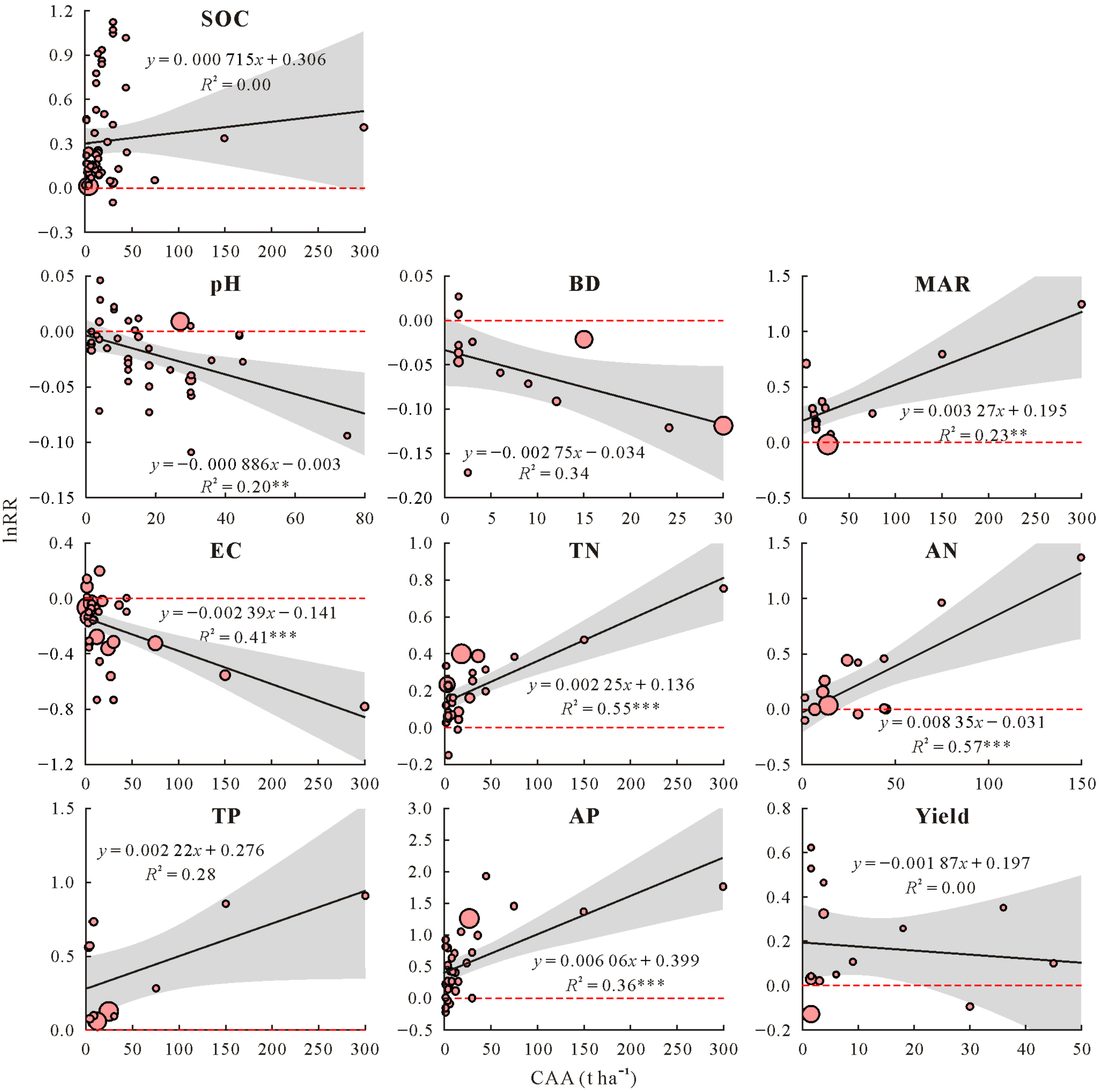

Fig. 4 Relationships between the effect sizes, expressed as the natural logarithm of the response ratio (lnRR), of compost on soil organic C (SOC), electrical conductivity (EC), representing soil salinity, pH, bulk density (BD), macroaggregate ratio (MAR), total N (TN), available N (AN), total P (TP), and available P (AP) and crop yield and its cumulative application amounts (CAAs) in the Yellow River Delta and the Yangtze River Estuary of China. Circle size denotes the inverse variance of effect size. Shaded area indicates the 95% confidence interval of the regression line. Asterisks ** and *** indicate significances at $P < 0.01$ and $P < 0.001$, respectively.

microbial enzymatic activities, particularly dehydrogenase activity, which in turn drive nutrient cycling and enhance soil biochemical functioning (Klik *et al*., 2025). Therefore, the effects of amendments may be overstated if only one or two variables are considered. Herein, we synthesized the effects of various amendments on soil properties and crop yield through a meta-analysis, offering a comprehensive evaluation of the commonly used amendments in coastal saline soils.

Biochar desalination mechanisms and its nutrient effects

It is reported that biochar application can improve soil structure, thereby enhancing salt leaching (Amini *et al*., 2016; Saifullah *et al*., 2018; Fouladidorhani *et al*., 2020). Our results confirm that biochar application indeed increases MAR and decreases soil BD (Table SVII). Enhanced soil porosity provides the prerequisite for salt leaching, but the efficiency of salt leaching depends largely on the quantity and

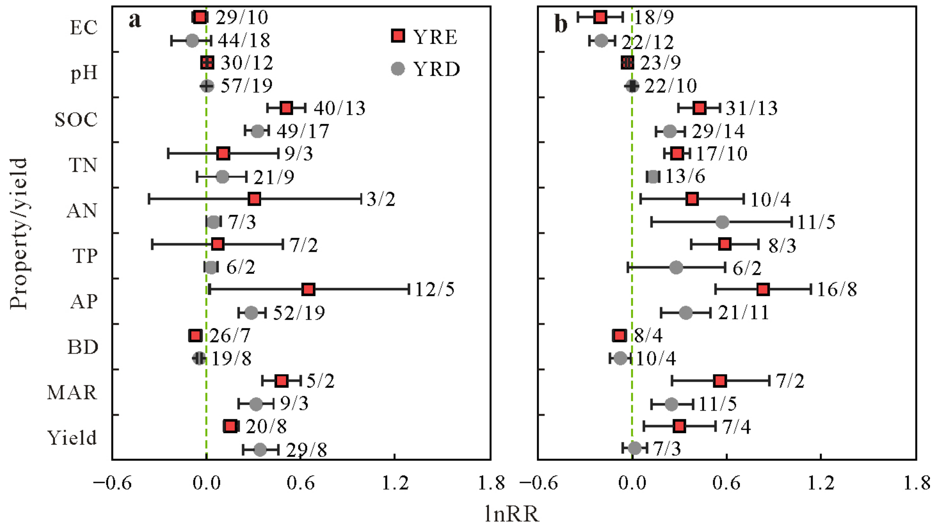

Fig. 5 Effect sizes, expressed as the natural logarithm of the response ratio (lnRR), of biochar (a) and compost (b) on soil organic C (SOC), electrical conductivity (EC), representing soil salinity, pH, bulk density (BD), macroaggregate ratio (MAR), total N (TN), available N (AN), total P (TP), and available P (AP) and crop yield in the Yellow River Delta (YRD) and the Yangtze River Estuary (YRE) of China. Error bars represent 95% confidence intervals. The numbers beside error bars are the numbers of observations/numbers of publications.

TABLE I

Descriptive statistics[a)] of cumulative application amounts (CAAs) of biochar and compost in the Yellow River Delta (YRD) and the Yangtze River Estuary (YRE) of China

Amendment	Min	Q1	Median	Mean	Q3	Max
	t ha^{-1}					
	YRD					
Biochar	1.3	6.0	12.0	21.7	28.2	94.4
Compost	1.5	4.0	10.3	12.4	14.2	44.0
	YRE					
Biochar	1.5	15.0	27.0	63.0	64.0	357.5
Compost	1.5	12.1	30.0	58.6	60.0	300.0

[a)] Min = minimum; Q1 = first quartile, the value below which 25% of the CAAs fall; Q3 = third quartile, the value below which 75% of the CAAs fall; Max = maximum.

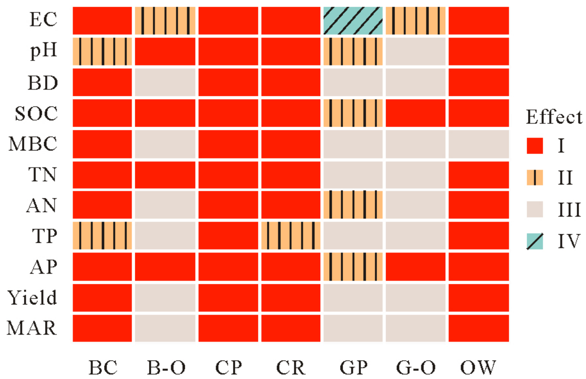

Fig. 6 Effects of different amendments on soil organic C (SOC), electrical conductivity (EC), representing soil salinity, pH, bulk density (BD), macroaggregate ratio (MAR), total N (TN), available N (AN), total P (TP), and available P (AP) and crop yield in the Yellow River Delta and the Yangtze River Estuary of China. BC = biochar; B-O = biochar plus organic materials; CP = compost; CR = crop residues; GP = gypsum; G-O = gypsum plus organic materials; OW = organic wastes; I = probably positive; II = potentially positive; III = not clear; IV = potentially negative.

quality of irrigation water. In addition, studies have shown that the porous structure of biochar and the functional groups on its surface can adsorb soluble ions in the soil (Ali *et al.*, 2017; Elshaikh *et al.*, 2018; She *et al.*, 2018; Yu *et al.*, 2019). Long-term field application of biochar enhances its specific surface area, pore volume, and abundance of oxygen-containing functional groups (Yao *et al.*, 2010; Dong *et al.*, 2017). This finding indicates its potential role in mitigating soil salinity over extended periods. High-temperature pyrolysis biochar exhibits enhanced cation exchange capacity and specific surface area (Zhang J H *et al.*, 2019), thereby effectively mitigating soil salinity. However, the adsorbed ions may be released back into the soil over time, as the sorption capacity of biochar may decrease or even disappear due to biochar aging and weathering (Ren *et al.*, 2018; Saifullah *et al.*, 2018). Moreover, biochar itself contains a certain amount of salts (Table SV), depending mainly on the feedstock type and pyrolysis conditions (Ippolito *et al.*, 2020). The biochar included in this study is made from plant materials, which may contain more soluble salts than those made from manures or biosolids (Ippolito *et al.*, 2020). These soluble salts are released into the soil with water flow within 1–3 weeks following application (Joseph *et al.*, 2021). Therefore, the effectiveness of biochar application in reducing soil salinity primarily depends on the balance between salt input from biochar and salt output from the soil. This is strongly affected by environmental conditions and biochar properties. Some studies have reported that biochar application may salinize the soil (Li *et al.*, 2018; Palansooriya *et al.*, 2019; Santos *et al.*, 2022), while other studies have drawn opposite conclusions (Zong *et al.*, 2023; Rassaei, 2024; Wang *et al.*, 2024). Here, biochar application has an overall desalinization effect

(Fig. 1), indicating that it can potentially reduce soil salinity in coastal areas. Furthermore, the positive and negative results of biochar are almost evenly distributed on both sides of the y-axis when the initial soil salinity levels are below 2.00 dS m^{-1} (Fig. 2). This suggests that the desalinization effect of biochar may be affected by the initial soil salinity, although our meta-regression analysis shows a non-linear relationship between initial soil salinity and the desalinization effect of biochar. A recent study indicates that the desalinization effect of biochar is strongly influenced by initial soil salinity and becomes significant only in severely saline soils (Wang *et al*., 2024). In natural conditions, the saline soils in YRE are more prone to desalinization than those in YRD, mainly due to abundant precipitation and a low evaporation/precipitation ratio. However, the effects of biochar on soil salinity are not significantly different between the two regions (Fig. 5a). This may be because the application amount and salt content of biochar in YRE are much higher compared to those in YRD, which potentially introduces more salts into the soil (Tables I and SV).

Biochar application directly introduces a considerable amount of C into the soil. Therefore, increases in SOC are positively correlated with biochar application amounts. This corresponds to the finding that the mean replenishing effect of biochar on SOC is more pronounced in YRE than in YRD, due to the higher application amount of biochar in YRE (Table I). A considerable amount of the freshly applied biochar-C is rarely available, considering the highly condensed structure and long turnover time of biochar in soil (Kuzyakov *et al*., 2014). However, biochar may contribute to long-term SOC sequestration by altering soil microbial communities and increasing recalcitrant organic C fractions (Liu *et al*., 2016; Zheng *et al*., 2022). Long-term SOC sequestration is primarily driven by biochar-C stability, with biochar produced at high temperatures or derived from plants exhibiting particularly high resistance to degradation (Singh *et al*., 2012). Biochar application generally increases soil TN and AN pools (Fig. 1), but its replenishing effects become insignificant in both regions (Fig. 5a). This suggests that the overall positive effects of biochar on soil N pools may be overestimated. Therefore, it is more robust to conclude that biochar application can potentially increase soil N pools in coastal areas. Actually, the AN content in biochar is very low because N mainly exists in the form of C-N heterocycles within biochar (Lehmann *et al*., 2006; Chen *et al*., 2019), and the N content decreases with increasing pyrolysis temperature (de Oliveira Paiva *et al*., 2024). Therefore, the positive effects of biochar on soil N pools may be attributed to the enhanced ammonium adsorption and mitigated N leaching (Zheng *et al*., 2013), and these positive effects tend to intensify over time. In addition, studies have shown that biochar application retains more AP in the soil by altering the P sorption-desorption equilibrium (Gao *et al*., 2019; Tesfaye *et al*., 2021). These results indicate that biochar application enhances the retention of available nutrients in soil rather than directly providing nutrients, which increases nutrient use efficiency and crop yield. Therefore, the effects of biochar on soil nutrient pools and crop yield show no significant linear relationships with cumulative application amounts (Table SIX). The cumulative application amount of biochar is generally higher in YRE, yet the increase in crop yield in YRE is lower compared to YRD (Table I and Fig. 5a). This is probably due to the stronger mean desalinization effect of biochar in YRD (Fig. 5a). Collectively, these results suggest that higher application amounts of biochar may not yield more benefits. Furthermore, the production cost of biochar is estimated to range from US \$233 to US \$1 847 t^{-1} (Nematian *et al*., 2021; Patel and Panwar, 2024). The benefit-cost ratio decreases when the application amount of biochar exceeds a threshold of 8 t ha^{-1} (Patel and Panwar, 2024). In addition, the massive energy consumption during pyrolysis may generate more environmental problems (Brassard *et al*., 2018). Given these technical obstacles and drawbacks, it may not be worthwhile to turn agricultural wastes into biochar for ameliorating saline soils at present.

Other amendments

The primary differences among organic amendments lie in the source of raw materials and the associated pretreatments (*e.g*., pyrolysis and composting). Biochar is not a typical organic fertilizer, as it cannot directly provide large amounts of nutrients to the soil (Garbowski *et al*., 2023). In contrast, composting accelerates the natural biodegradation of organic materials. This process produces substantial amounts of readily available nutrients and effectively destroys pathogens and weed seeds in the raw materials (Partanen *et al*., 2010; Onwosi *et al*., 2017; Soobhany *et al*., 2017). The average production costs are estimated to be US \$52 t^{-1} organic wastes (Song and Lee, 2010) and US \$72 t^{-1} compost (Dong *et al*., 2020). These estimated costs are considerably lower than those associated with biochar production. Our results confirm that applying compost and organic wastes induces higher mean increases in soil N and P pools compared to biochar (Fig. 1, Table SVII). Moreover, the application of compost and organic wastes generally has stronger effects in reducing soil salinity and BD than biochar. This is probably because compost and organic wastes provide more organic substances that bind soil particles (Sleutel *et al*., 2006). In contrast, biochar reduces soil BD through its porous internal structure (Agegnehu *et al*., 2017).

Agricultural waste and manure are traditional composting materials and usually co-composted with other additives, such as earthworms and microorganisms. Attempts have been made to use municipal sewage sludge as a raw material for composting (Wei *et al*., 2000; Roca-Pérez *et al*., 2009; Onwosi *et al*., 2017; Sayara *et al*., 2020). Here, municipal sewage sludge and its compost are subgrouped independently as organic wastes to distinguish them from normal compost, given their potential to contain hazardous materials. The positive effects of organic wastes are more pronounced than those of compost, especially with respect to SOC and TN (Fig. 1). This is probably due to the significantly higher application amounts of organic wastes relative to compost (Fig. 3). Diacono and Montemurro (2010) reported that long-term applications of compost and organic wastes promoted SOC sequestration, with farmyard manure being the most effective. In addition, our meta-regression analysis confirms that the ameliorating effects of organic wastes become stronger with higher application amounts (Table SIX). However, the long-term excessive application of organic wastes inevitably introduces more soluble salts and pollutants (*e.g*., heavy metals) into the soil, which may increase the risks of soil salinization and pollution (Hargreaves *et al*., 2008; Meena *et al*., 2019; Ofori *et al*., 2021). These risks are strongly influenced by amendment-induced changes in cation exchange capacity and metal sorption processes. The average contents of heavy metals in organic wastes are in the sequence of zinc $>$ copper $>$ chromium $>$ manganese $>$ nickel $>$ lead $>$ cadmium (Table SXI, see Supplementary Material for Table SXI). For safe reutilization of organic wastes, their heavy metal contents must comply with the agricultural use limits established by the Food and Agriculture Organization and World Health Organization guidelines (FAO, 1992; Chang *et al*., 1995). Domestic sludge is preferred over industrial sludge because the latter typically contains higher contents of heavy metals (Rizzardini and Goi, 2014; Islam *et al*., 2017). Our results show that the effects of organic wastes on soil salinity are negatively correlated with initial soil salinity levels (Fig. 2). This suggests that substantial applications of organic wastes may disrupt the salt balance in slightly and moderately saline soils, resulting in salt accumulation in the topsoil (Perez-Espinosa *et al*., 2000; Wong *et al*., 2001; Zoghlami *et al*., 2016). Moreover, the overuse of organic wastes may cause eutrophication (Liu, 2016; Carlos *et al*., 2017).

Although compost generally contains fewer hazardous materials than sewage sludge (Yu *et al*., 2022; Garbowski *et al*., 2023), environmental concerns are still relevant. The application of livestock and poultry manures in agriculture may still pose risks to the environment due to the potential presence of heavy metals and pharmaceuticals (Qi *et al*., 2023). Manures from cows, pigs, and chickens are frequently seen in our dataset, as these animals are widely bred in China. However, most of the publications in our dataset do not provide information on the contaminant contents in the compost. Some studies have reported that cow manure generally contains lower levels of nutrients and heavy metals than pig and chicken manures (Pagliari and Laboski, 2012; Wang *et al*., 2013; Yang and Han, 2013; Chadwick *et al*., 2015; Qi *et al*., 2023). For example, Moreno-Caselles *et al*. (2002) indicated that the average N contents in cow, pig, and chicken manures were 18.6, 21.7, and 31.4 g kg^{-1}, respectively, while the average P contents were 3.1, 14.4, and 13.2 g kg^{-1}, respectively. Similarly, Qi *et al*. (2023) reported that the average cadmium contents in cow, pig, and chicken manures were 0.84, 2.97, and 1.34 mg kg^{-1}, respectively, while the average arsenic contents were 2.09, 16.47, and 6.88 mg kg^{-1}, respectively. These findings indicate that the total addition of heavy metals from various animal manures may be similar under equivalent nutrient application levels. Considering the heterogeneity of animal manure across regions and countries, determining the optimal application amount of compost is crucial for agricultural practices and should be a key focus in future studies.

Direct return of crop residues is an economical method for recycling agricultural wastes, as it does not involve any pretreatment or transportation. Crop residues interrupt the continuity of soil capillaries and block the upward movement of salts from deep soil layers, thereby preventing salt accumulation in the topsoil (Zhao *et al*., 2016). In addition to their role as a physical barrier, crop residues serve as additional sources of C input that may supply a certain amount of nutrients. However, the mean increases in soil nutrient pools following crop residue return are lower than those caused by compost and organic wastes (Fig. 1). This is probably due to the low nutrient content and high C/nutrient ratio of crop residues (Table SXII, see Supplementary Material for Table SXII) (Ding *et al*., 2014; Wang *et al*., 2019; Siedt *et al*., 2021). In addition, the effects of crop residues on soil properties are highly similar to those of biochar, except that biochar significantly increases crop yield compared to crop residues (Table SVII). This is probably because biochar can retain greater amounts of available nutrients compared to crop residues. Nevertheless, crop residue return may pass down soil-borne pests and diseases to subsequent crops, which may influence crop yield and quality (Cherubin *et al*., 2018; Shan *et al*., 2021).

Gypsum is the most commonly-used inorganic amendment for reclaiming sodic soils characterized by high pH and excessive concentrations of Na^+, carbonate (CO_3^{2-}), and bicarbonate (HCO_3^-). Singh *et al*. (2016) demonstrated through a three-year field experiment that applying gypsum to a highly sodic soil significantly decreased soil pH from 10.5 to 8.9 and reduced the exchangeable Na percentage from 89.0 to 28.5. Chi *et al*. (2012) indicated that applying a high amount of gypsum to an inland saline-sodic soil in China reduced soil pH from 10.5 to 8.0 and decreased soil salinity from 26.0 to 2.7 dS m^{-1}. Gypsum applications reduce soil alkalinity primarily through the replacement of Na^+ by calcium (Ca^{2+}) released from gypsum (Li and Xu, 2013; Zhang *et al*., 2021). In addition, HCO_3^- and CO_3^{2-} react with Ca^{2+}, leading to the precipitation of calcium carbonate (Wang *et al*., 2017). However, gypsum applications may even cause soil secondary salinization, particularly in some regions with poor drainage systems (Zhao *et al*., 2020; Xia *et al*., 2024). A meta-analysis conducted by Wang *et al*. (2021) revealed that gypsum applications in both saline (EC > 4 dS m^{-1}, pH < 8.6) and weakly sodic (pH of 8.6–9.0) soils significantly increased soil salinity. Nan *et al*. (2016) also reported that applying gypsum to a coastal saline soil (EC $= 5.93$ dS m^{-1}, pH $= 7.43$) led to an increase in soil salinity. These results agree with our finding (Fig. 1) that gypsum applications may, in turn, increase soil salinity levels, probably because gypsum is inherently a salt (calcium sulfate). Therefore, the usage of gypsum may not be necessary in coastal areas, since these soils rarely exhibit extremely high soil pH levels (Wu *et al*., 2019).

Combined application of soil amendments

The combined application of different amendments has been insufficiently studied in previous publications, probably because most amendments are organic and share similar roles in the soil. The combined application of organic materials and biochar may compensate for the deficiencies of available nutrients in biochar (Lentz *et al*., 2014). This study demonstrates that B-O significantly increases soil TN compared to the sole application of biochar (Fig. 1). However, the combination of biochar with organic materials exhibits no significant synergistic effects on SOC relative to biochar alone. This may be because the stable C in biochar mainly contributes to SOC sequestration, whereas the labile C from organic materials stimulates microbial decomposition of SOC (Bass *et al*., 2016). In addition, B-O exhibits no significant effect on soil desalinization. These results suggest that the combined application of biochar and organic materials may have antagonistic effects on saline soil amelioration. Moreover, the high price of biochar may restrain its widespread application in agricultural practice. Compared to the co-application of biochar and organic materials, combining compost with crop residues is a more realistic and economical approach, since our results show that the effects of crop residues and biochar are highly similar.

CONCLUSIONS

This meta-analysis offers a comprehensive evaluation of the effects of various amendments on soil physicochemical properties and crop yield in two coastal areas of China. It reveals that gypsum application averagely increases soil salinity by 7.6% without affecting SOC. Hence, gypsum is not recommended for ameliorating coastal saline soils. Organic amendments (*i.e*., biochar, compost, crop residues, and organic wastes) generally exert positive effects on SOC and nutrient pools, increasing crop yield, improving soil structure, and mitigating soil salinity in coastal saline areas. However, improper application of organic wastes may increase soil salinity when the initial soil salinity (EC) is below 1.96 dS m^{-1}. Therefore, amendment application amounts should be optimized based on local soil conditions to mitigate the risk of secondary salinization. Overall, this meta-analysis provides a basis for ameliorating coastal saline soils *via* amendment application. However, several potential limitations exist in this meta-analysis. Firstly, the data are confined to the 0–20 cm soil depth, which may limit the applicability of the results to deeper soil horizons. Secondly, the impacts of amendment applications on soil pollution (*e.g*., heavy metal pollution) and microbial activities are not analyzed owing to limited data availability. Finally, the majority of studies included in this meta-analysis span only one to three years, whereas long-term stability (*e.g*., SOC sequestration and salt rebound) is crucial for achieving sustainable reclamation. Therefore, further studies are needed to address these knowledge gaps.

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.

ACKNOWLEDGEMENT

We are grateful to acknowledge the financial support from the China Scholarship Council (CSC) to Jianyu Tao.

SUPPLEMENTARY MATERIAL

Supplementary material for this article can be found in the online version.

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