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Contrasting effects of food waste and its biochar on soil properties and lettuce growth in a microplastic-contaminated soil

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Abstract

The incorporation of organic amendments, such as food waste (FW) and biochar, into soil is an established agronomic practice known for enhancing soil fertility and improving overall soil health. However, the individual and combined effects of FW and biochar on soil properties in microplastic (MP)-contaminated soil–plant systems remain poorly understood. To address this knowledge gap, we conducted a field experiment to investigate the individual and combined effects of polystyrene MPs, FW, and FW-derived biochar on soil properties and lettuce growth. Soil chemical properties were unaffected by the addition of MPs. However, the application of FW and biochar increased the soil pH, with the highest pH (8.2) observed in the combined treatment of biochar and MPs. Despite the presence of MPs, FW application resulted in notable increases in soil electrical conductivity (EC; 2.04 dS m^{-1}), available nitrogen (NO_3^-–N: 325.5 mg kg^{-1}, NH_4^+–N: 105.2 mg kg^{-1}), available phosphorus (88.4 mg kg^{-1}), and total exchangeable cations (18.6 $cmol_{(+)}$ kg^{-1}). However, these values decreased after lettuce cultivation. In soil cultivated with lettuce, the coexistence of MPs and biochar reduced soil Fluorescein diacetate hydrolase enzyme activity by 46.2% and urease activity by 94.0%. FW addition doubled acid phosphatase activity, whereas FW and its coexistence with MPs decreased alpha diversity. The relative abundance of Actinobacteria decreased with MP application, whereas that of Acidobacteria and Actinobacteria decreased with FW treatment. Gemmatimonadetes and Nitrospirae decreased in soil treated with FW and biochar. The highest relative abundances of Firmicutes and Proteobacteria were observed in the FW-added soils, and Planctomycetes were the highest in the biochar-added soils. FW application negatively affected lettuce growth. Overall, the coexistence of MPs with FW or biochar had limited effects on soil properties and lettuce growth, with FW and biochar serving as the primary factors in modifying soil–plant systems. Future studies should investigate the effects of different MPs and their interactions with organic soil amendments on soil properties and crop growth under different management practices.

Keywords Biochar interactions, Environmental implications, Sustainability, SDG 15 life on land, Sustainable waste management

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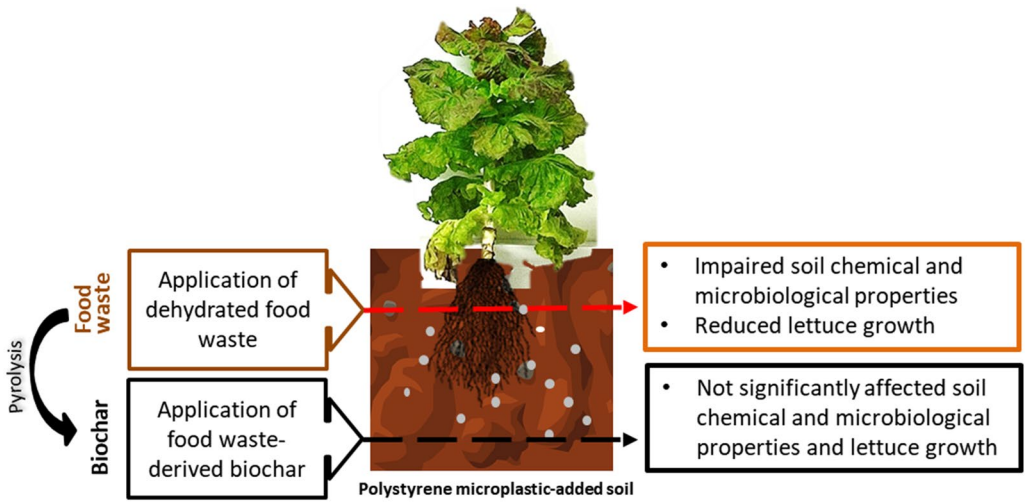
Graphical Abstract



Introduction

Global plastic pollution has surged significantly owing to rising consumption and insufficient plastic waste management, which is exacerbated by the limited biodegradability of plastic [1]. In 2015, out of the 6300 Mt of plastic waste produced, only approximately 9% was recycled, while around 79% was either deposited in landfills or accumulated in the natural environment [2]. The agricultural sector contributes significantly to plastic pollution in the soil, primarily through the use of plastic mulch films, weed-barrier sheets, tunnels, greenhouse plastic films, and plastic nets [3–5]. These agricultural plastics undergo various physicochemical changes owing to weathering, ultraviolet radiation, and hydrolysis, resulting in the formation of microplastics (MPs) with particle sizes smaller than 5 mm. During agricultural practices such as tillage, MPs can migrate downward and accumulate along the soil profile. Yang et al. [6] estimated an annual accumulation of 18.1 million MP items per hectare in the plowed layer. The presence of MPs has been found to alter the physical, chemical, and microbial properties of soil, thereby exerting negative effects on plant growth and microorganisms. Studies have shown that exposure to polyethylene (PE) MPs inhibits the growth of lettuce and wheat by disrupting antioxidant enzymes and photosynthetic systems [7, 8]. Moreover, plants can take up micro-(nano-)plastics through their roots and transport them to their shoots, leading to contamination of the food chain and potential impacts on human health [9].

In addition to plastic waste, the generation of FW (food waste) is a global issue that significantly contributes to environmental degradation. FW generation contributes to a significant environmental footprint associated with food production, highlighting the pressing need for sustainable practices that minimize waste and conserve resources [10–12]. According to the United Nations, approximately one billion tons of food are wasted globally each year [13], and the Food and Agriculture Organization predicts that FW generation will reach 3.62 Gt 2030 [14]. Generally, FW is either disposed of in landfills or incinerated for energy generation [15]. FW in landfills poses various environmental hazards, including the emission of CH_4, NH_3, and volatile fatty acids, as well as high chemical oxygen demand [16]. Leachates released during FW degradation have the potential to contaminate the surrounding surface water and groundwater [17]. Furthermore, the incineration of FW produces volatile organic compounds and particulate matter, which pose risks to human and animal health. Therefore, the valorization of FW into value-added products such as compost, biofertilizers, and biochar has been recognized as a sustainable solution [18]. The application of FW-derived products to soil has been shown to enhance soil fertility, mitigate organic and inorganic soil contaminants, and improve crop productivity [19, 20]. For instance, O'Connor et al. [21] revealed that the applying dehydrated FW enhanced the availability of macronutrients (N, P, and K) to plants and stimulated microbial activity in various soil types. However, they cautioned that repeated or high-dose applications could potentially

lead to increased soil salinity, phytotoxin accumulation, anoxic conditions, and water repellence [21]. Moreover, several studies have demonstrated that FW-derived biochar acts as a fertilizer and soil amendment, exerting beneficial effects on soil physicochemical and biological properties, including increased water-holding capacity, enhanced microbiological activity, and elevated levels of carbon (C), N, P, and K [22–24].

The occurrence of MPs and the application of FW-derived soil amendments have become common in terrestrial ecosystems, leading to an increased prevalence of their coexistence. However, limited research has been conducted on the effect of this coexistence on ecosystem functions. The effects of the coexistence of MPs and biochar on soil greenhouse gas emissions have been studied. For instance, Li et al. [25] found that the coexistence of PE MP and straw biochar increased N_2O emissions by 37.5% and decreased CH_4 emissions by 35.8%. In a study on NH_3 volatilization from rice paddy soils, Fang et al. [26] observed that the co-effects of PE and polyacrylonitrile MPs with straw-derived hydrochar increased NH_3 volatilization by 37.8–46.2% compared with MP alone, indicating that the coexistence of MPs and hydrochar can weaken the mitigation effects of MP on NH_3 volatilization. Furthermore, they found that the NH_4^+ sorption capacity increased by 17% in the presence of PE MP and straw biochar and by 7.1% in the presence of PE MP and manure biochar in an aqueous solution [26]. This increased sorption was likely due to deprotonation of the functional groups and a decrease in dissolved organic C with a small molecular size [27]. The expandable and flexible nature and low-density of plastic polymers typically facilitate interactions with biochar [28]. Similar to exogenous organic inputs, biochar can adsorb or agglomerate into MP [29, 30]. MP can be trapped inside biochar pores, [31] and the surface functional groups of the biochar may provide binding sites for PE-like MPs [27]. In addition, MP spheres can become stuck, trapped, or entangled in the biochar because of the large internal pore space of the biochar and the immobilization of MP [32]. Moreover, biochar accelerates the weathering of PE MP in paddy soils through physical friction [33]. These key findings provide evidence for different mechanisms of MP immobilization by biochar in soil.

To date, no prior research has explored the potential interactions and consequences of MP and their coexistence with FW and FW-derived biochar on soil chemical properties, microbial activity, and crop growth. We hypothesized that the presence of biochar in MP-contaminated soils, compared with dehydrated FW, can sustain soil fertility while mitigating the potential negative impacts induced by MP on crop growth. To test this hypothesis, we selected polystyrene (PS) MP, which is prevalent in agricultural soils [34], in conjunction with dehydrated FW and FW-derived biochar. We investigated their individual and combined effects on soil chemical properties and microbial attributes, including soil enzyme activity, bacterial diversity, and community composition, as well as on the growth of lettuce. This study is the first of its kind, specifically focusing on elucidating the effects of the coexistence of PS MPs and FW-derived soil amendments on soil properties and crop growth.

Materials and methods

Microplastics, food waste, and biochar

PS, one of the most prevalent types of plastics in the soil environment, was selected as the MP used in this study. The PS particles used were purchased from SINWON Industrial Co. Ltd. (Korea) and had a white color with a particle size ranging from 95 to 300 μm (Additional file 1: Table S1). The FW was obtained from an FW treatment facility (Gimpo Urban Management Corporation, Gimpo-Si, Korea) and consisted of a mixture of plant- and animal-derived FW from households and restaurants. The FW was ground and dehydrated, and the dried FW was then placed in a pyrolysis reactor (HM Corp., Hwaseong-si, Korea) in batches of 10 kg, where it was pyrolyzed at 500 °C with a retention time of 20 min. After pyrolysis, the FW-derived biochar was cooled to room temperature (15–20 °C) and sieved through a 2 mm stainless steel sieve. To reduce the chlorine content, the produced biochar was demineralized for 30 min by mixing it with deionized water at a ratio of 1:10 (w:v) to reduce the chlorine content [35].The demineralized biochar and brine were separated using a 1.2 μm Whatman GF/C filter (Buckinghamshire, UK). The biochar was dried in an oven at 70 °C for 24 h and stored until further analysis. The basic characterization methods and properties of the dehydrated FW and FW-derived biochar are presented in the Additional file 1.

Field experiment

The field experiment was conducted in Banghak-dong, Seoul, South Korea (37°39′34.20″N, 127°01′18.84″E) from March to September 2021. The study area falls within the humid continental climate zone (Köppen climate classification; Dwa), which is characterized by a mean annual precipitation of 1233 mm and a mean annual temperature of 11.3 °C. For the PS MP application, a concentration of 1% (w w^{-1}), which is considered environmentally relevant for soils with high anthropogenic influence, was used [36]. FW and biochar were added at a rate of 5% (w w^{-1}), equivalent to 85 t ha^{-1} [11]. Thus, the six treatment combinations were as follows: (1) control with no amendment (control), (2) 1% MP (MP-1), (3) 5% FW (FW-5), (4) 5% biochar (BC-5), (5) 1%

MP + 5% FW (MP + FW), and (6) 1% MP + 5% biochar (MP + BC). Three replicates were established for each treatment, resulting in 18 experimental units arranged in a completely randomized block design. The field site was divided into three blocks with six raised beds within each block. Each raised bed measured 0.65 m × 0.45 m and was separated from neighboring beds by wooden frames (20 cm high) inserted approximately 5 cm into the soil (Additional file 1: Fig. S1). Initially, the MPs, FW, and biochar were applied to the soil according to their respective treatments and incorporated to a depth of approximately 20 cm using a hoe/iron-toothed rake for thorough mixing. To allow for interactions between the soil microbiome and MPs, FW, and biochar, the plots were incubated (pre-incubation) for five weeks after treatment application. After the pre-incubation period, one-week-old lettuce (*Lactuca sativa*) seedlings were transplanted into each plot, with four seedlings per pot. The field was managed using conventional farming practices, including weeding and watering.

Soil sampling and analysis

Soil samples were collected from each plot at the end of the pre-incubation and cultivation stages. Three replicates were randomly selected from a depth of 0–20 cm depth and were thoroughly mixed to create one composite sample per plot. Visible roots, stones, and organic residue were manually removed. Fresh sub-soil samples were sieved (< 2 mm) and stored at − 4 °C for microbial analyses. The remaining soil was air-dried, sieved (< 2 mm), and stored in airtight bags for chemical analysis.

Soil pH was measured in soil–water (1:5 w:v) suspensions using a pH meter (pH meter (Orion Star A211, Thermo Scientific, USA). The soil solution was then centrifuged and filtered using a Whatman 42 filter paper, and the electrical conductivity (EC) was determined using an EC meter (Orion Star A211, Thermo Scientific, USA) [11]. To determine the available N content in the soil, NO_3^-–N and NH_4^+–N were extracted with 1 M KCl at a 1:4 (w:v) ratio and analyzed by steam distillation [37]. The total C (TC) and total N (TN) contents of the soils were determined using a CHN elemental analyzer (Elementar Analysensysteme GmbH, Germany, Vario-Micro Cube model). The available soil P was determined using a 0.5 M $NaHCO_3$ extraction solution (pH 8.5) at a 1:20 (w:v) soil-solution ratio, following the Olsen method described by Olsen et al. [38] The available P in the extractant was quantified using a UV–VIS spectrophotometer (Thermo Fisher Scientific Solutions, LLC) at a wavelength of 880 nm. Exchangeable cations (Ca^{2+}, Mg^{2+}, K^+, and Na^+) were extracted using 1 M ammonium acetate (NH_4OAc, pH 7) at a soil: solution ratio of 1:10 (w:v) [11]. The concentrations of exchangeable cations in the soil extracts were determined by inductively coupled plasma–optical emission spectrometry (ICP–OES; Agilent 700 series, Japan). The sum of Ca^{2+}, Mg^{2+}, K^+, and Na^+ was recorded as the total exchangeable cations (TEC) [11].

Soil enzyme activity

Fluorescein diacetate hydrolase (FDAase), urease, and acid phosphatase activities were analyzed. For FDAase activity measurement, one gram of fresh soil was added to 4 mL of 60 mM sodium phosphate buffer (pH 7.6) supplemented with 400 µL FDA (10 µg mL^{-1}). The soil-buffer mixture was incubated in a rotary shaker at 24 °C and 160 rpm for 60 min. FDAase hydrolysis was terminated by adding acetone (final concentration 50% v v^{-1}). The soil was separated by centrifugation at 6000 rpm for 5 min and filtered through a Whatman 2 filter paper [39]. The optical density of the filtrate was measured at 490 nm by using a UV/VIS spectrophotometer (Infinite M200 PRO; TECAN, Austria).

The soil urease activity was measured using the method described by Kandeler et al. [40] One gram of each soil sample was mixed with 0.5 mL of 0.72 M urea solution and borate buffer (pH 10), and the mixtures were incubated at 37 °C for 2 h. After adding 10 mL of 1.00 N potassium chloride and 0.01 N hydrogen chloride solutions, the final solution was shaken for 30 min at 160 rpm and then filtered using Whatman 2 filter papers. The filtrates were mixed with 1 mL of sodium salicylate solution and 0.4 mL of 0.1% sodium dichloroisocyanurate. The resulting mixtures were incubated for 30 min at 25 °C, and the urease activity was determined by measuring the optical density at 690 nm using a UV/VIS spectrophotometer (Infinite M200 PRO, TECAN, Austria), with ammonium chloride used as the standard.

For soil acid phosphatase analysis, the methodology described by Tabatabai et al. [41] was used. One gram of soil was mixed with 4 mL of modified universal buffer (pH 6.5), 0.25 mL of toluene, and p-nitrophenyl phosphate solution (1 mL). The soil solution was then incubated at 37 °C for 1 h. After incubation, 1 mL of 0.5 M calcium chloride and 0.5 M sodium hydroxide was added, and the samples were filtered through a folded filter paper (Whatman No.1). Phosphatase activity was measured (as the optical density) by spectrophotometer at 420 nm using a UV/VIS spectrophotometer (Infinite M200 PRO, TECAN, Austria), with p-nitrophenol used as the standard for soil acid phosphatase.

Soil microbial DNA extraction and sequencing

Genomic DNA was extracted from the soil (0.5 g) using the FastDNA® Spin Kit (MP Biomedicals, USA) to analyze the microbial composition of the soil. The purity and concentration of genomic DNA were measured using

a NanoDrop 2000 spectrophotometer (Thermo Fisher Scientific, USA), and soil DNA was pyrosequenced at Macrogen Inc. (Seoul, South Korea). The bacterial 16S gene from the extracted DNA was amplified in the V3–V4 region using universal primers 341F and 805R, following amplification, initial denaturation, 25 annealing cycles, and final extension. The products were normalized and pooled using PicoGreen, and the library size was verified using a TapeStation DNA ScreenTape D1000 (Agilent, Germany). The products were sequenced on a MiSeq platform (Illumina, San Diego, CA, USA), following established methods [42–44]. Alpha diversity indices, including Chao1, Shannon, Gini-Simpson, and Good's coverage, were calculated based on operational taxonomic units using MOTHUR 40, and then statistically separated using the Statistical Analysis System ver. 9.4 (SAS, Cary, NC, USA).

Plant sampling and analysis

At harvest (five weeks after planting), the height of the lettuce plants was measured from the soil surface to the tip of the longest leaf, and the number of leaves per plant was determined. After destructive harvesting and thorough cleaning with distilled water, the fresh biomass of the aboveground (stems and leaves) and belowground (roots) plant parts was quantified [45]. Plant dry biomass was quantified after oven drying the samples at 70 °C for constant weight.

Quality control and statistical analysis

All measurements were performed in triplicate, and the values are reported as the mean ± standard error. Blanks and analytical-grade reagents were used for soil analysis. One-way analysis of variance (ANOVA) was performed using Statistical Analysis System ver. 9.3 (SAS, Cary, USA). Tukey's honest significant difference (HSD) test was conducted to determine the significant differences between the various treatments at a significance level of 0.05.

Results and discussion

Effects of microplastic, food waste, and biochar addition on soil chemical properties

The addition of MP did not affect the soil pH in either the incubated or cultivated soils (Fig. 1a). The application of FW increased the soil pH to a near-neutral level (FW-5 = 7.14) in the incubated soils, but the coexistence of MP and FW (MP + FW = 6.86) did not have a significant impact on soil pH compared with FW-5. The pH of FW was acidic (4.87), however, during the incubation, the degradation of organic matter in FW likely led to an increase in soil pH. The soil pH significantly increased ($p < 0.05$) in the BC-5 and MP + BC treatments in both incubated and lettuce-cultivated soils compared to the control and MP-1 treatments. Previous studies have also reported that the addition of 1% (w w^{-1}) low-density polyethylene (LDPE) and 0.1% (w w^{-1}) polylactic acid MP to the soil did not have an impact on soil pH [46, 47]. However, Qi et al. [48] found that soil pH increased two months after adding 1% (w w^{-1}) LDPE MP to the soil. The exact mechanism by which MP influences soil pH remains unclear [49], but it has been suggested that the shape, size, and residence time of MP in soils may contribute to soil pH fluctuations [50]. In general, MP foams and fragments significantly affected the soil pH, which was attributed to improved aeration and porosity. Their introduction, coupled with chemical leaching, influenced soil biota, leading to alterations in pH. Microplastic films also increased pH by modifying the nitrogen-fixing bacterial diversity and elevating the NH_4^+ content. Variations in shape or additives may explain the differences in the impact of pH. Soil pH is influenced by factors such as soil organic matter, acid buffering, and cation retention. The type of soil and the presence of plants can influence the pH impact of MPs, which is potentially mitigated by plants compared with bare soils [50]. During FW decomposition, hemicellulose is initially hydrolyzed by microbes, producing organic acids that lower the pH. As decomposition progresses, organic acids volatilize, increasing the pH. Ammonia from N-containing organic matter can also increase the pH. The presence of cellulose and lignocellulose in FW indirectly contributes to the pH increase by inhibiting soil acidification [51]. After lettuce cultivation, the pH values significantly decreased ($p < 0.05$) in both the FW-5 and MP + FW treatments. This decrease could be attributed to the leaching of excess basic cations, nutrient uptake by plants, and organic matter decomposition. These processes result in the loss of alkaline cations and the release of organic acids, gradually lowering the soil pH [52]. Furthermore, the degradation of fatty acids present in the FW could potentially contribute to the decrease in soil pH [21] The FW-derived biochar had an alkaline pH (9.1), and base ions (including Ca^{2+}, Mg^{2+}, K^+, and Na^+) could remain in the residual ash as oxides or carbonates. These ions can be released into the soil solution and exchanged with acidic ions, thereby increasing the soil pH [53]. Similarly, Xu et al. [54] reported that the addition of kitchen waste-derived biochar led to an increase in soil pH, reaching 7.4 compared to the initial pH of 6.3, with the elevated pH attributed to the release of Na-like cations.

Similar to soil pH, the addition of MP did not affect soil EC in either the incubated or cultivated soils (Fig. 1b). The application of FW increased soil EC from 0.18 dS m^{-1} (control) to 1.74 dS m^{-1} (FW-5) and 2.04 dS m^{-1} (MP + FW) in the incubated soils, with

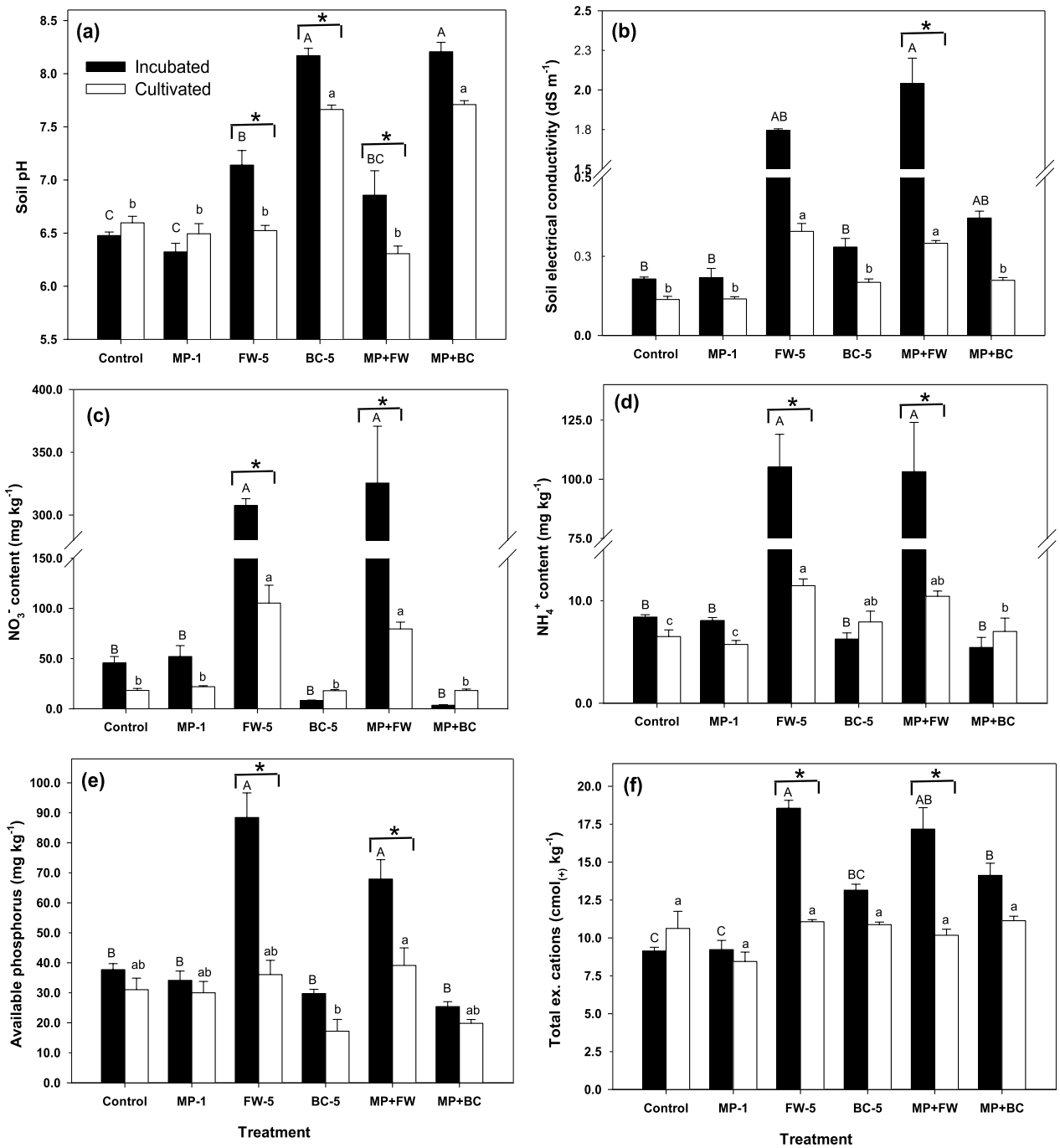


Fig. 1 Changes in soil properties in incubated and lettuce-cultivated soils. **a** pH; **b** electrical conductivity (dS m^{-1}); **c** NO_3^-–N (mg kg^{-1}); **d** NH_4^+–N (mg kg^{-1}); **e** available phosphorus (mg kg^{-1}); and **f** total exchangeable cations ($cmol_{(+)}$ kg^{-1}). Error bars denote ± standard error (n = 3). For different parameters, letters "A, B, and C" and "a, b" represent significant differences among soil treatments in incubated and lettuce-cultivated soils, respectively. According to the Tukey test, having the same letters on two bars implies that soil properties are not different at $p < 0.05$. Asterisks (*) represent a significant difference ($p < 0.05$) between incubated and lettuce-cultivated soils

a significant increase ($P < 0.05$) observed only in the MP + FW treatment. Similarly, in cultivated soils, FW significantly increased ($p < 0.05$) soil EC in both the FW and MP + FW treatments, with the highest value observed as 0.33 dS m^{-1} in FW-5. However, the coexistence of MP and FW (MP + FW) did not significantly impact soil EC compared to FW-5. The application of biochar and its coexistence with MP did not have a significant impact on

soil EC in either incubated or cultivated soils. This indicated that soil-soluble salts were not affected by the presence of MP in the soils. Given that our study employed pristine PS-MP, which was not aged and had an incubation period of only six months, it is plausible to hypothesize that this duration of exposure might not have been adequate to induce discernible changes in EC. It is conceivable that higher concentrations of MP or an extended exposure period may be necessary to observe measurable effects on soil EC. This consideration aligns with the understanding that the dynamics of MP-soil interactions can be influenced by factors such as the concentration, duration, and inherent properties of the MP themselves. Further investigation under various experimental conditions may provide valuable insights into the temporal aspects of MP-induced changes in soils. The high EC in the FW-added soils was likely due to the high inherent EC of FW (5.75 dS m^{-1}), which was relatively high owing to the salt content and protein-rich substances [18]. Consequently, the application of such materials to the soil can result in the accumulation of salts and an increase in soil EC [20]. Overall, the increase in soil EC due to FW treatments did not exceed the salinity threshold for vegetable crops (1–2.5 dS m^{-1}) [55]. Lee et al. [56] observed that the addition of FW-derived compost increased the soil EC to 3.37 dS m^{-1}, while the control soil had an EC of 0.34 dS m^{-1}; however, this increase in EC did not have an adverse effect on lettuce growth. Compared to FW (5.75 dS m^{-1}), biochar had a lower EC (2.39 dS m^{-1}), probably due to the demineralization process during the biochar production [35]. Speratti et al. [57] observed an increase in EC to > 2 dS m^{-1} following the application of biochar derived from local agricultural waste, including swine manure and cotton. However, these changes in soil EC are mainly influenced by the physicochemical properties of biochar and can vary depending on the type of feedstock and the production method used [58].

The application of MP (MP-1) did not have significant impacts on soil NO_3^--N, NH_4^+-N, available P, and TEC contents compared to the controls in both incubated and cultivated soils (Fig. 1c–f). The application of MP, biochar, and their coexistence (MP + BC) did not have significant impacts on soil NO_3^-–N contents in both incubated and cultivated soils (Fig. 1c). FW-5 and MP + FW showed increased NO_3^--N content compared to the control in the incubated soil. However, NO_3^--N contents in incubated soils significantly decreased ($p < 0.05$) by 65.8% and 75.6% in FW-5 and MP + FW respectively, after lettuce cultivation. A similar trend was observed for NH_4^+-N, where FW-5 and MP + FW exhibited the highest NH_4^+-N content in incubated soils and the values significantly decreased ($p < 0.05$) by 89.1% and 89.9% in FW-5 and MP + FW, respectively, after lettuce cultivation (Fig. 1d). The NH_4^+-N content increased in BC-5 and MP + BC compared to both the control and MP-1 in cultivated soils. However, the coexistence of MP with either FW or biochar did not show any significant effect on soil available N compared to the sole application of FW or biochar in both soils. The highest available P contents were observed in the FW-5 (88.4 mg kg^{-1}) and MP + FW (67.9 mg kg^{-1}) treatments in incubated soils and the values significantly reduced ($p < 0.05$) up to 36.1 mg kg^{-1} (FW-5) and 39.1 mg kg^{-1} (MP + FW) after lettuce cultivation (Fig. 1e). The application of MP, biochar, and their combination (MP + BC) did not have a significant impact on soil available P in either the incubated or cultivated soils (Fig. 1e). In addition, the highest TEC was observed in FW-5 (18.6 $cmol_{(+)}$ kg^{-1}) and MP + FW (17.2 $cmol_{(+)}$ kg^{-1}) in incubated soils, and these values significantly reduced ($p < 0.05$) by 40.3% (FW-5) and 40.7% (MP + FW) after lettuce cultivation (Fig. 1f). The soil TC content increased in all MP, FW, and biochar treatments compared to the controls in both the incubated and cultivated soils (Table 1). Among them, the coexistence of MP and biochar (MP + BC) showed the highest increments, with TC increasing 5.2 times in the incubated soil and 4 times in the cultivated soil compared to the controls. The addition of MP did not have a significant impact on the

Table 1 Changes in soil total carbon and total nitrogen in incubated and lettuce-cultivated soils

Treatment	Total C (%)		Total N (%)	
	Incubated	Cultivated	Incubated	Cultivated
Control	2.67 ± 0.2 (d)	2.85 ± 0.1 (c)	0.25 ± 0.0 (d)	0.26 ± 0.0 (c)
1% Microplastics (MP-1)	5.70 ± 0.7 (c)	5.38 ± 0.2 (cd)	0.26 ± 0.0 (d)	0.24 ± 0.0 (c)
5% Food waste (FW-5)	4.88 ± 0.0 (cd)	3.87 ± 0.0 (de)	0.65 ± 0.0 (bc)	0.44 ± 0.0 (b)
5% Biochar (BC-5)	9.62 ± 1.0 (b)	8.80 ± 0.8 (b)	0.89 ± 0.1 (ab)	0.82 ± 0.1 (a)
1% MP + 5% FW (MP + FW)	7.41 ± 0.6 (bc)	6.16 ± 0.2 (c)	0.60 ± 0.1 (c)	0.41 ± 0.0 (bc)
1% MP + 5% Biochar (MP + BC)	13.79 ± 0.6 (a)	11.43 ± 0.3 (a)	1.05 ± 0.1 (a)	0.82 ± 0.0 (a)

Letters "a, b, c, d and e" represent significant differences between the different soil treatments. Data are presented as mean ± standard error (n = 3)

TN content in soils (Table 1). However, the application of FW, biochar, and their co-application with MPs significantly increased TN in the incubated soil ($p < 0.05$), and a similar trend was observed in the cultivated soils.

When reviewing existing research on the impact of MPs on soil properties, it becomes apparent that a consensus is lacking. The diversity in the results and findings is primarily due to variations in the experimental design and conditions. Some researchers have employed pristine plastics with different shapes, colors, and quantities, whereas others have focused on plastic-based products such as mulch films and packaging materials made from various plastics. Furthermore, the duration of these studies ranged from months to years, and different types of soils with varying textures and chemical and physical properties were used. Despite the increasing prevalence of MP contamination in soils and the urgent need for soil remediation, the wide array of variables and approaches poses a challenge for comparing and drawing definitive conclusions regarding the impact of MP on soil properties.

Yi et al. [59] reported that PE MPs had no impact on NH_4^+–N and NO_3^-–N contents in paddy soil. Meng et al. [60] also observed that soil NH_4^+–N and NO_3^-–N contents did not change upon the addition of LDPE MPs at 0.5–2.5%, (w w^{-1}) application rates. On the other hand, some studies have shown that the addition of MPs such as PP and PE increased the NH_4^+-N content while decreasing the NO_3^--N content in the soil, regardless of the addition levels, which could be attributed to the inhibition as well as activation of microbial activities under different types of MPs [61]. Feng et al. [62] observed no alteration in available P with the application of 2% (w w^{-1}) PE and PS MPs, whereas Yan et al. [63] reported a significant increase in available P following the addition of 0.1% and 1% high-density polyethylene plasticized MPs. Furthermore, the authors found that adding 1% unplasticized polyvinyl chloride (PVC) MPs significantly reduced available P content from 38.4 to 26.9 mg kg^{-1} [63] Thus, it can be postulated that the effect of MPs on soil P availability varies based on the type of MPs and application dose. Palansooriya et al. [64] observed that the application of 7% (w w^{-1}) LDPE MPs to soils did not affect the TEC concentration in soils. Yi et al. [59] found that the application of PE MPs decreased available K content by 8% in common rice-cultivated soil, but increased it by 11% in hybrid rice-cultivated soil, suggesting that the effect of MPs on the soil largely depends on the plant species.

The increased contents of available N, P, and TEC in the FW-added treatments could be due to the high levels of NPK and other micronutrients in the FW [65]. FW-related waste is mostly lignocellulosic with high cellulose and lignin contents [66]. During decomposition, soluble nutrients are released into the soil, resulting in increased soil nutrients. In particular, the relatively high amount of P in FW can readily mineralize in soils [67, 68]. The significant decrease in available N, P, and TEC contents in FW-5 and MP + FW in cultivated soil implies that NH_4^+–N, NO_3^-–N, P, Ca^{2+}, Mg^{2+}, K^+, and Na^+ were lost from the soil through leaching, volatilization, or plant uptake [69, 70]. Thus, there is a risk of applying FW to the soil, as it causes surface and groundwater pollution through leaching, and air pollution through NH_3 volatilization and NOx fluxes [71]. Therefore, converting FW into biochar and pretreating and diluting FW before soil application are recommended to avoid unintentional consequences on soil and plant growth [20]. Generally, biochar is rich in nutrients and is used for soil amendments [72]. However, the effects of biochar on available soil N, P, and TEC were not significant in our study, except for some treatments. Only the cultivated soil treated with biochar (BC-5 and MP + BC) showed a significant increase ($p < 0.05$) in NH_4^+–N compared with the control and MP-1. Similarly, a significant increase ($p < 0.05$) in TEC was observed only in the soil incubated with the MP + BC treatment compared to the control and MP-1 treatments. These different effects can be attributed to changes in soil chemical properties, microbial activities, and the gradual release of nutrients over time [73]. In our study, the sole application of MP, as well as its coexistence with FW or biochar, did not significantly affect the soil's chemical properties. However, these effects were primarily driven by the FW and biochar, with the FW exerting a dominant influence. Further investigation is essential to gain a comprehensive understanding of the short- and long-term effects of MPs and their coexistence with other organic amendments on soil properties. The higher TC content in the MP-, FW-, and biochar-treated soils was due to the high C content in the MPs [74], FW, and biochar [75] (Additional file 1: Table S2). MP contains less N; [76] hence, its impact on soil TN was not notable. Chen et al. [61] also observed that the application of PVC, polypropylene (PP), PE, PS, polyethylene terephthalate MPs at 0.25%, 2%, and 7% (w w^{-1}) concentrations had no significant impact on soil TN. Compared to MP, FW, and biochar had higher TN contents; thus, their application increased soil TN in FW- and biochar-treated soils than MP MP-treated soils.

Effects of microplastic, food waste, and biochar addition on soil enzyme activity

The biological and biochemical properties of the soil play a vital role in soil pollution and indicate the influence of pollutants on soil systems [77]. FDAase activity in soil is a useful indicator of short-term changes in

soil quality and represents the overall metabolic activity of soil microorganisms [78]. Urease activity in the soil is closely related to the N cycle, promoting the hydrolysis of N-containing organic matter [79]. Acid phosphatase activity in soils is strongly linked to P cycling and is often used as an indicator of changes in soil fertility under contaminant stress [80, 81]. In our study, compared with the control, FDAase activity significantly decreased ($p < 0.05$) in MP-1 by 35.6% (Fig. 2a). Although the application of biochar did not show a significant impact on the FDAase activity, the coexistence of MP and biochar (MP + BC) significantly decreased ($p < 0.05$) FDAase activity by 46.2%. In contrast, the application of FW significantly increased ($p < 0.05$) FDAase activity by 43.3% in FW-5. Similar to FDAase activity, urease activity significantly decreased ($p < 0.05$) in MP-1 by 88.7%, and the application of biochar did not show a significant impact, but the coexistence of MP and biochar (MP + BC) significantly decreased ($p < 0.05$) urease activity by 94.0% (Fig. 2b). However, the application of FW and its coexistence with MPs did not significantly affect soil urease activity. Only FW-5 showed a significant increase ($p < 0.05$) in acid phosphatase activity, nearly doubling that of the control (Fig. 2c). Previous studies have observed that the addition of PVC and PE MPs to soils decreased FDAase activity, possibly because of the toxicity of MPs to microorganisms, inhibiting their activity [82]. The effects on FDAase activity following the addition of MPs to soils can vary depending on the MP type, soil properties, and diversity of bacteria present [78]. Yang et al. [83] have shown that urease activity significantly decreased (by 43–80%) in soils with PP MPs and PE MPs over time. Yu et al. [84] reported that applying PE MPs to soils decreased urease activity by 16.9–40.8% after 180 d of incubation. These findings indicate that the presence of MP in the soil has short- and long-term adverse effects on urease activity. MPs, due to their large surface area and strong adsorption capacity, MPs can absorb substrates such as soil organic matter and inhibit soil enzyme activity. Moreover, soil microorganisms are unable to utilize MPs, and

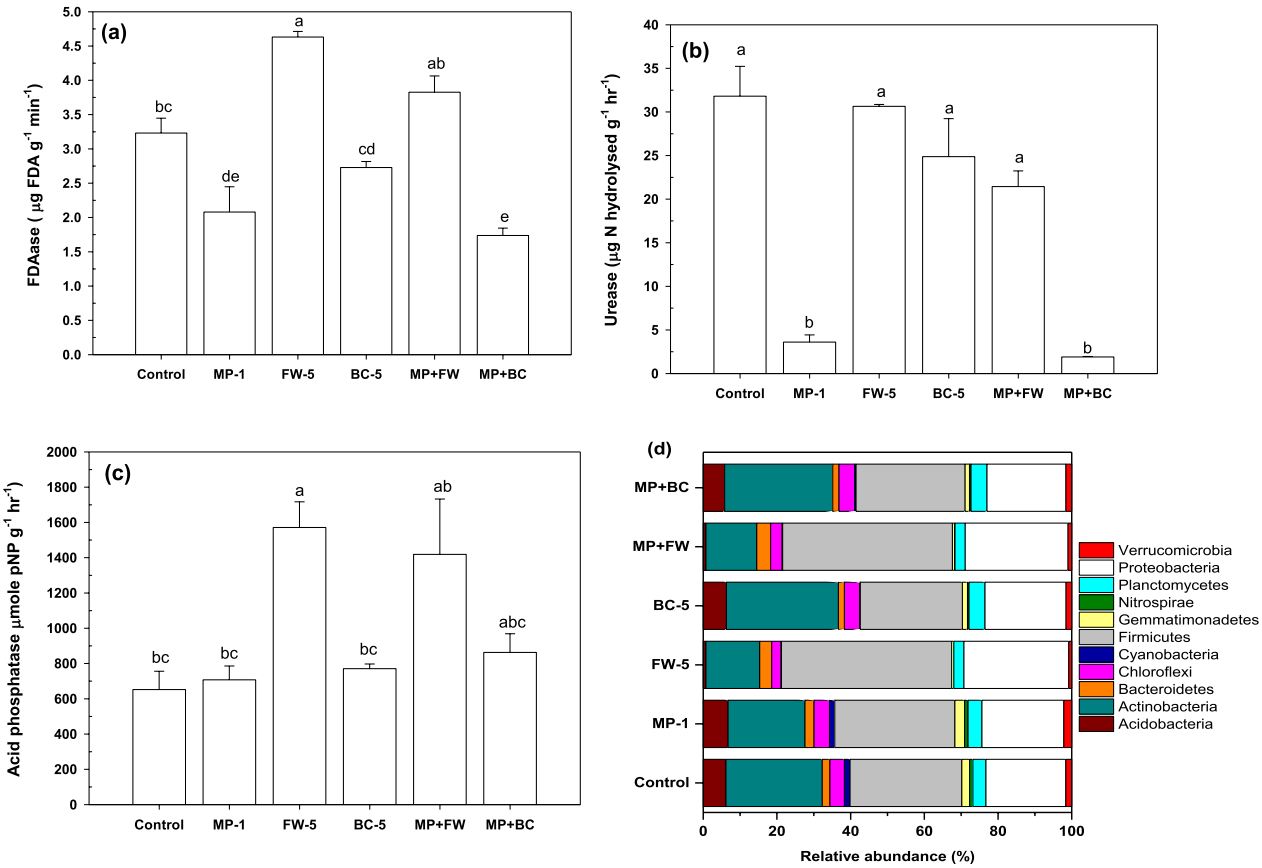


Fig. 2 Changes in soil enzyme activity and bacterial abundance in lettuce-cultivated soil. **a** FDAase; **b** urease; **c** acid phosphatase activity; and **d** relative abundance of bacteria at major phyla levels under different treatments. Error bars denote ± standard error (n = 3). For different parameters, letters "a," "b," "c," "d," and "e" represent significant differences among soil treatments. According to Tukey's test, the same letters on each bar imply that soil enzyme activity is not different at a significance level of $p < 0.05$

these MPs can potentially compete for physicochemical niches, leading to reduced microbial activity and the inhibition of soil enzyme activities. Thus, the presence of MPs in the soil can disrupt the soil structure, consequently damaging the physicochemical niches necessary for microorganism growth and function [84].

According to Chintala et al. [85], the application of biochar derived from corn stover, switchgrass, and ponderosa wood residues decreases FDAase activity by 23%, 28%, and 21%, respectively. However, the authors noted that compared to biochar, the incorporation of raw biomass (without pyrolysis) increased FDAase activity by 1.5 and 2 times at application rates of 10 and 50 g kg^{-1}, respectively. These findings suggest that increased microbial metabolic activity in soils is closely associated with the presence of readily available C sources that are more abundant in raw biomass, such as FW, than in biochar. These readily available C sources initially act as the main C sources for microorganisms [85], stimulating heterotrophic microbial activity, and thereby increasing enzyme activity in the soil. FW contains C and N, which can be easily used as energy and nutrient sources by soil microorganisms, resulting in increased soil microbial populations and high enzyme activity. According to Dick and Tabatabai [86], the initial acid phosphatase concentration depends on the quantity of microbial biomass in the substrate and its subsequent activities. In line with our study, Lee et al. [56] also observed increased acid phosphatase activity following the addition of FW compost (289–355 μg p-nitrophenol g^{-1} soil h^{-1} at 2 weeks) to the soil when compared to control soils (26–69 μg p-nitrophenol g^{-1} soil h^{-1}). This increase in phosphatase activity can be attributed to the hydrolysis of organically bound phosphate into free ions that are available for plant uptake [56].

Unlike FW, recalcitrant biochar contains minor quantities of dissolved organic C that can support microbial activity [87]. Chintala et al. [85] showed that biochar can act as a quorum quencher in soil by binding to quorum-sensing molecules, thereby reducing their availability to soil microorganisms. This may be another reason for the lower enzymatic activity observed in soils treated with biochar than in those treated with FW. Luo et al. [88] reported that the application of peanut shell biochar significantly decreased urease activity by 13.0–49.8% in an incubation experiment. Similarly, Wu et al. [89] found that urease activity was significantly reduced with increasing rates of biochar application, indicating an inhibitory effect of biochar on urease activity. Several factors may contribute to this phenomenon, including the adsorption of enzyme molecules and/or substrates by biochar, which can affect their apparent affinity for substrates or block reaction sites [90]. Additionally, biochar can influence enzyme activity through changes in soil physicochemical properties (e.g., soil pH). Moreover, biochar may release small molecules that act as inhibitors of specific enzymes [91].

Interestingly, the coexistence of MPs and biochar played a significant role in decreasing the soil FDAase and urease activities, possibly because of the combined negative effects of MPs and biochar. Soil pH is a crucial factor regulating bacterial abundance and diversity. The increased soil pH resulting from biochar application (Fig. 1a) likely decreased the abundance and diversity of certain microorganisms. Additionally, the presence of MPs exerts pressure on soil microorganisms, further contributing to the collective negative impact on microbial activity and subsequently reducing FDAase and urease activities.

Effects of microplastic, food waste, and biochar addition on soil bacterial community and composition

Alpha diversity was used to assess the complexity of bacterial species diversity in the soil using four indices: Chao1, Shannon, Gini-Simpson, and Good's coverage. The application of MPs, biochar, and their combination did not have a significant impact on alpha diversity (Additional file 1: Table S3). These findings align with those of previous studies by Ma et al. [92] and Rong et al. [93], where the addition of PE and LDPE MPs, respectively, did not affect the alpha diversity of soil bacteria. This implies that MPs derived from petroleum hydrocarbons have minimal short-term influence on soil bacterial alpha diversity, possibly because of their slow degradation process, which can span decades and result in minimal changes to soil bacterial communities. Many studies have reported an increase in the alpha diversity of soil bacteria following biochar application [94]. However, other studies have indicated that biochar addition can lead to a decrease in alpha diversity owing to biofilm formation on the biochar surface, which restricts the homogeneous diffusion process [95]. Han et al. [95] also found that the coexistence of polyethylene terephthalate MPs and biochar did not affect soil bacterial diversity. The limited impact of MPs, biochar, and their coexistence on alpha diversity may be explained by the inherent resilience of soils to various disturbances, including extreme or combined disruptions, which enables their ability to withstand and recover [93]. Nevertheless, the application of FW and its coexistence with MPs significantly reduced ($p < 0.05$) the Chao 1 index by 47.5% (FW-5) and 49.5% (MP + FW), as well as the Shannon index by 10.8% (FW-5) and 10.8% (MP + FW). These results suggest that the application of FW and its coexistence with MPs can negatively affect soil microorganisms, reducing bacterial richness (Chao1) and diversity (Shannon index).

This could be due to increased soil salinity and excessive nutrient accumulation in the soil resulting from the addition of FW (Fig. 1). Meng et al. [96] reported that the application of FW biogas slurry disrupted the stability and abundance of Ketobacter, which was attributed to the development of soil salinity [96].

To gain further insights into the effects of the treatments on the soil bacterial community, changes in the bacterial community composition at the phylum level were studied (Fig. 2d). The dominant phyla observed across all treatments were Acidobacteria, Actinobacteria, Bacteroidetes, Chloroflexi, Cyanobacteria, Firmicutes, Gemmatimonadetes, Nitrospirae, Planctomycetes, Proteobacteria, and Verrucomicrobia (Fig. 2d). Among them, only the relative abundance of Actinobacteria showed a significant decrease ($p < 0.05$) with the application of MP (MP-1 = 19.72%) compared with the control (24.73%). These results indicate that the impact of MPs on certain soil bacteria was minimal, suggesting the potential resilience of such taxa to MPs. Previous studies also reported a decrease in the abundance of Actinobacteria upon the addition of PE, PS, and PVC MPs to the soil [82, 97]. This can be attributed to the alteration of the bacterial community structure, which is potentially driven by the enrichment of bacterial groups associated with MP biodegradation [82]. In addition, the impact of MPs on the microbial community can be influenced by various factors, such as soil properties, type of MPs, their concentration, and particle size [98, 99].

The application of FW and its coexistence with MPs significantly decreased ($p < 0.05$) the relative abundances of Acidobacteria, Actinobacteria, Gemmatimonadetes, and Nitrospirae compared with the other treatments, with the most pronounced decrease observed in Acidobacteria. In contrast, compared to the control (28.66%), the relative abundance of Firmicutes significantly increased ($p < 0.05$) in FW-5 (44.79%) and MP + FW (44.69%). Similarly, the relative abundance of Proteobacteria significantly increased ($p < 0.05$) in FW-5 (27.60%) and MP + FW (27.07%) compared to the control (20.49%). Based on 16S rRNA data from different soils, Fierer et al. [100] and Kielak et al. [101] also observed a high abundance of Acidobacteria in soils with low C mineralization rates. FW and animal manure contain a high percentage of dissolved organic C in the form of amino acids and carbohydrates [102], which can potentially reduce the proliferation of Acidobacteria. Kalam et al. [103] reported that most Acidobacteria species favor acidic conditions (3.0–6.5 pH) for their proliferation. The addition of FW to the soil increased the soil pH from 6.47 to 7.14, which might have inhibited its growth and reduced its abundance. Meng et al. [96] found a significant increase in the relative abundance of Firmicutes with the addition of FW slurry, particularly with increasing soil salinity. Pan et al. [104] also reported that Firmicutes dominated FW and anaerobic digestate samples, comprising a substantial proportion (33–83%) of the total microbial community. Firmicutes are known for their ability to efficiently degrade complex organic matter into short-chain fatty acids, even under extreme conditions such as low pH. Their tolerance to varying salinity gradients and pH levels ranging from 7.5 to 9.0 allows them to thrive under such extreme conditions [96]. Proteobacteria are commonly found in various soil ecosystems, including rhizosphere, saline, and semi-arid soils. The relative abundance of Proteobacteria increased with the increasing availability of organic C [105, 106]. In the present study, regardless of MP contamination, the high amount of labile C in soils with added FW contributed to the increased abundance of Proteobacteria. Mickan et al. [107] also observed a higher relative abundance of Proteobacteria in soils amended with FW digestates.

The relative abundance of Chloroflexi was highest in MP + BC (4.09%), whereas Planctomycetes showed the highest relative abundance in BC-5 (4.11%) and MP + BC (4.12%). In contrast, Gemmatimonadetes and Nitrospirae exhibited lower relative abundances in the BC-5 and MP + BC treatments than in the control, but their relative abundances were higher compared to the FW-added treatments. Ran et al. [108] found that the co-application of PP MPs and biochar increased the relative abundance of Chloroflexi compared to the treatment with MPs alone. Chloroflexi and Planctomycetes are commonly associated with plant rhizospheres and play vital roles in maintaining the balance between root nutrient absorption and the microenvironment [108, 109]. Therefore, they suggested that biochar amendment could promote equilibrium between root nutrient absorption and the bacterial community microenvironment in soil contaminated with MPs. Zhang et al. [110] observed a decrease in the relative abundances of Gemmatimonadetes and Nitrospirae under different biochar application rates. The increased nutrient and soil moisture content potentially limit the growth of certain bacteria because they prefer dry and low-nutrient conditions in soils [111]. There was no significant difference in the soil bacterial abundance between the FW/biochar treatments and their co-application with MP. This indicated that the impact on bacterial abundance was primarily driven by either FW or biochar, and the coexistence of MPs with either FW or biochar had a limited effect on soil bacterial abundance.

Effect of microplastic, food waste, and biochar addition on lettuce growth

Overall, the application of MPs, biochar, and the co-application of MPs and biochar did not significantly

affect lettuce growth (Fig. 3). Considering all growth parameters, the application of FW negatively affected lettuce growth (Fig. 3). Specifically, FW-5 showed significant reductions ($p < 0.05$) in the number of leaves (36.9%), shoot dry weight (56.7%), root fresh weight (51.8%), and root dry weight (64.5%) compared with the control. The retarded growth of lettuce in soils treated with FW may be attributed to alterations in soil properties and microbial communities that adversely affect plant growth. O'Connor et al. [21] showed that high application rates of dehydrated FW as a fertilizer can decrease plant biomass due to phytotoxins, the potential development of anoxic conditions, and high salinity levels. Lee et al. [112] observed a decrease in rice and pepper yields as FW compost application rates increased from 0 to 60 t ha^{-1}. By contrast, Lee et al. [56] reported a significant increase in the fresh weight of lettuce after the application of FW. These differential effects could be due to compositional differences in the FW used in these studies. However, despite the negative effects of FW, the application of FW-derived biochar was able to maintain the growth of lettuce, in a similar way to the control soils, even in the presence of MPs. This suggests that converting FW into biochar is a promising approach to mitigate the detrimental effects of FW and improve plant growth, rather than the direct application of dehydrated FW to soils as an amendment. Consistent with our findings, Ran et al. [108] reported that biochar application to MP-contaminated soil increased the number of beneficial bacteria, particularly enhancing the N and P metabolism cycles in the soil and plants, thereby effectively promoting the growth of pepper plants in MP-contaminated soil. Moreover, Liu et al. [113] observed that the coexistence of plastic film mulch and biochar improved soil C sequestration, reduced greenhouse gas emissions into the atmosphere, and increased maize yield in rainfed

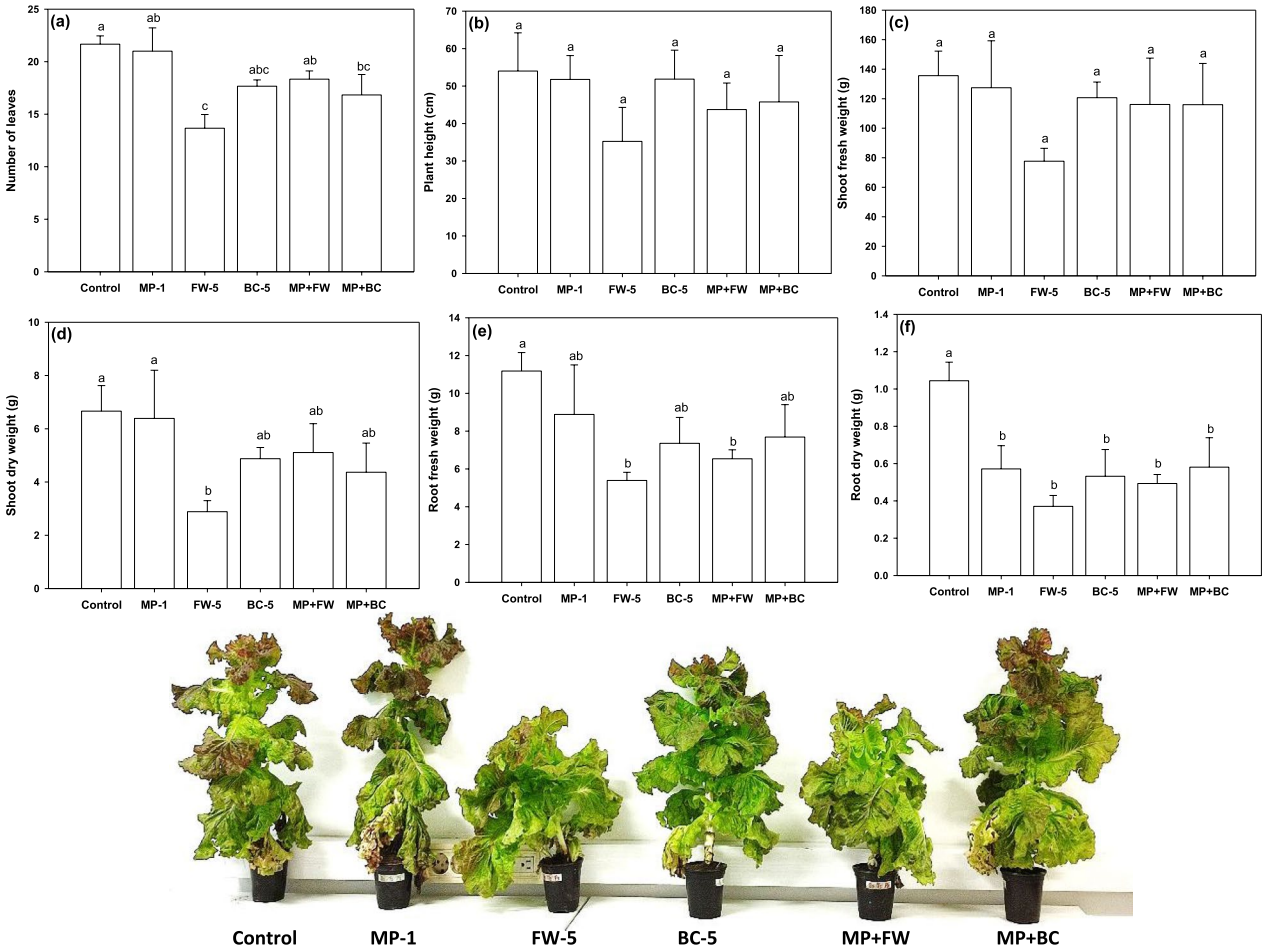


Fig. 3 Changes in **a** number of leaves; **b** plant height; **c** shoot fresh weight; **d** shoot dry weight; **e** root fresh weight; and **f** root dry weight of lettuce plants under different soil treatments. Error bars denote ± standard error (n = 3). For different parameters, letters "a," "b," and "c" represent significant differences among different soil treatments. According to Tukey's test, the same letters on two bars imply that the plant growth parameters for these treatments are not significantly different at $p < 0.05$

farmlands. Therefore, utilizing biochar instead of FW in soils contaminated with MPs offers a potentially sustainable waste management approach in agroenvironments. In addition, we emphasize that the observed variations in plant growth are not solely attributable to the presence of MPs, FW, or biochar, or their coexistence. These variations may arise from their presence and the alterations in soil properties following their introduction. Although the current study did not directly focus on microbial diversity, the observed variations in plant growth may have been influenced by microbial activity in the soils [50, 95]. Furthermore, soil physical properties, such as bulk density and porosity, change due to the presence of MP, FW, and biochar in soils. Alterations in the soil properties have the potential to affect crop growth. This underscores the need for future studies to delve deeper into the soil–plant system and thoroughly examine how the presence of different amendments in MP-contaminated soils could affect soil properties and, in turn, influence crop growth.

Abbreviations

C	Carbon
EC	Electrical conductivity
FDAase	Fluorescein diacetate hydrolase
FW	Food waste
ICP-OES	Inductively coupled plasma-optical emission spectrometry
LDPE	Low-density polyethylene
MP	Microplastic
N	Nitrogen
P	Phosphorous
PP	Polypropylene
PS	Polystyrene
PVC	Polyvinyl chloride
K	Potassium
SEM	Scanning electron microscope
TEC	Total exchangeable cations
TC	Total carbon
TN	Total nitrogen
XRD	X-ray powder diffraction
XPS	X-ray photoelectron spectroscopy

Supplementary Information

The online version contains supplementary material available at https://doi.org/10.1186/s13765-023-00851-w.

Additional file 1: Table S1. Physical and chemical properties of polystyrene powder. **Table S2.** Basic properties of food waste and biochar. **Table S3.** Alpha diversity of bacteria community in lettuce cultivated soil. **Figure S1.** Field experiment setup; **a** soil bed preparation and pre-incubation with food waste and biochar and **b** lettuce plant growing. **Figure S2.** Scanning electron microscopy with energy-dispersive X-ray spectroscopy (SEM/EDX) graph of **a** food waste and **b** biochar. **Figure S3.** X-ray photoelectron spectroscopy (XPS) spectrum of **a**, **b** C1s **c** O1s and **d** N1s for food waste and biochar.

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Author contributions

All authors discussed the results and contributed to the preparation of the manuscript. KNP and PAW analyzed the data and wrote the manuscript with regular discussions with YSO. Substantial contributions were made via discussion with YJ, MKS, YC and GH. SXC and YSO reviewed and edited the manuscript before submission.

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Availability of data and materials

All data generated or analysed during this study are included in this published article and its additional files.

Declarations

Competing interests

The authors declare that they have no competing interests.

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