Distribution characteristics of microplastics in the soil of mangrove restoration wetland and the effects of microplastics on soil characteristics

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Abstract
The dense vegetation in the wetland could effectively retain microplastic particles, and the distribution of microplastics varied significantly under different planting densities. In addition, microplastics in the soil environment can affect soil properties to a certain extent, which in turn can affect soil functions and biodiversity. In this study, we investigated the distribution of soil microplastics in a mangrove restoration wetland under different planting densities and their effects on wetland soil properties. The results indicated that the average abundance of soil microplastics was 2177.5 n/500 g, of which 70.9% exhibited a diameter ranging from 0.038–0.05 mm, while the remaining soil microplastics accounted for less than 20% of all microplastics, indicating that smaller-diameter microplastics were more likely to accumulate in wetland soil. The microplastic abundance could be ranked based on the planting density as follows: 0.5 × 0.5 m > 1.0 × 0.5 m > 1.0 × 1.0 m > control area. Raman spectroscopy revealed that the predominant microplastic categories in this region included polyethylene terephthalate (PET, 52%), polyethylene (PE, 24%) and polypropylene (PP, 15%). Scanning electron microscopy (SEM) images revealed fractures and tears on the surface of microplastics. EDS energy spectra indicated a large amount of metal elements on the surface of microplastics. Due to the adsorptive features of PET, this substance could influence the soil particle size distribution and thus the soil structure. All physicochemical factors, except for the soil pH, were significantly affected by PET. In addition, the CV analysis results indicated that soils in vegetated areas are more susceptible to PET than are soils in bare ground areas, leading to greater variation in their properties.

Keywords Microplastics · Wetland soil · Polyethylene terephthalate · Soil physicochemical properties · Abundance

Introduction
Wetlands are uniquely located between terrestrial and aquatic ecosystems, creating favourable conditions for the enrichment and transport of microplastics, and wetlands have become important hubs for microplastic transport in global ecosystems (Liu et al., 2019a). Microplastics in wetland environments are mainly derived from nearby human activities. Garcés-Ordoñez et al. (Garces-Ordonez et al., 2019) sampled microplastics in mangrove wetland soils near and away from population centres in the Colombian Caribbean and found that the abundance of microplastics increased the closer the population centres were. In addition, microplastics originating from human activities can enter the mangrove wetland environment and accumulate in wetland soils through effluent discharge (Naji et al., 2019; Wang et al., 2020). Edo et al. (Edo et al., 2020) found, by comparing the abundance of plastic particles in several natural and lagoonal wetlands, that the amount of plastic entering the lagoonal wetlands was 40 times higher than that entering the natural wetlands.

In coastal wetland environments, external forces such as mechanical erosion and chemical and biological processes...
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Materials and methods

Soil and microplastics sample

In September 2020, topsoil samples were collected in restored and control areas of the mangrove restoration wetland in the Jinjiang Estuary under different planting densities of Kandelia candel - Rhizophora stylosa - Bruguiera gymnorrhiza. The collected soil samples were sealed and labeled in self-sealing bags and transported to the laboratory for natural air drying on the same day. A portion of the air-dried soil was analysed to determine physical and chemical factors, and the results are listed in Table S1.

Based on the study of soil microplastics in this mangrove wetland in 2017 (Deng et al., 2020), polyethylene terephthalate (PET) was found to be the most abundant in this area. For this reason, PET was selected as microplastic material (Fig. S1a; analytical grade powder, Youngling-TECH Company, Beijing, China) in this study on the effect of microplastics on soil properties. The particle size distribution of PET powder was examined using a Malvern particle sizer, and the results demonstrated that the particle size of this material was <700 μm, with 8.3% of the microplastic PET particles exhibiting a particle size ranging from 700–355 μm, 31.71% exhibiting a particle size ranging from 355–158 μm, 43.01% exhibiting a particle size ranging from 158–50 μm, and 16.96% exhibiting a particle size <50 μm (Fig. S1b). In addition, microplastic powder SEM images were obtained with a field emission full scanning electron microscope (Nippon Co., Ltd., Japan) (Fig. S1c, d), which indicated that the microplastic particles contained a rough and porous surface.

Experimental methods

Flotation of soil microplastics under different planting densities

Microplastics were extracted from soil samples through density separation in a saturated NaCl solution (Amrutha and Warrier, 2020). An unground soil sample of 50 g was accurately weighed, 250 mL of saturated NaCl solution was added, and the mixture was stirred for 10 min with a magnetic stirrer. The supernatant, including microplastics, was passed through a grading sieve system consisting of four stacked stainless-steel sieves with different pore sizes (all 10 cm in diameter, with pore sizes of 2, 1, 0.5 and 0.038 mm). The process was repeated five times to ensure adequate microplastic extraction from the soil samples. The intercepted material was rinsed from the sieve into a clean beaker and left to stand for 24 h before filtering through a vacuum extraction device using a 0.45-μm glass cellulose membrane, which was then placed in a glass Petri dish. To

can affect the surface morphology of microplastics, and a rough or cracked texture is typical evidence of microplastic degradation and can reflect the degree of ageing and surface adsorption of microplastics (Veerasingam et al., 2016; Zhou et al., 2018). This change also results in microplastics as carriers of organic and inorganic pollutants (Kingiopoulou et al., 2022). Studies have demonstrated that PE can adsorb p-chlorobenzene and flurolin and other organic pollutants (or organic pollutants) in aqueous solutions, which confirms the adsorption and application of microplastics (Tubic et al., 2021). Concerning elemental attachment on the surface of microplastics, Mehdinia et al. (Mehdinia et al., 2020) identified and analysed microplastics in coastal sediments of the southern Caspian Sea in northern Iran, and examined elemental attachment on the surface of microplastics via SEM/EDS, indicating that microplastics can adsorb metal elements.

It has been reported that microplastics can be combined with soil to form a variety of agglomerates: shredded plastics can form loose agglomerates, and plastic fibres can form denser agglomerates (Rillig et al., 2017; Wong et al., 2020). In addition, microplastics can affect the soil-water cycle and thus can alter the water infiltration capacity of soil, thereby affecting soil water evaporation and contaminant migration along with fractures into deeper soils (Boots et al., 2019). Soil enzymes are closely related to a variety of soil biochemical processes and play an important role in regulating soil nutrient cycling. However, microplastics also significantly affect soil enzyme activity (Hu et al., 2020). Liu et al. (Liu et al., 2017) reported that high concentrations of polypropylene microplastics (~28%, w/w) greatly increased the nutrient (C, N, and P) content in soluble organic matter (DOM). Qi et al. (Qi et al., 2020) found significant differences in the effects of different microplastic treatments on the soil pH, conductivity and the carbon to nitrogen ratio.

Although hydrological conditions in wetland environments can allow plastics or microplastics migration, vegetations in wetlands play an important role in intercepting plastic or microplastic migration (Martin et al., 2019). However, whether different planting densities in mangrove wetlands significantly affect the distribution of microplastics in wetland soil sediments deserves further exploration. In this study, we investigated the distribution characteristics of soil microplastics under different vegetation planting densities in a mangrove restoration wetland in the Jinjiang Estuary, Fujian Province, and clarified the effects of microplastics (polyethylene terephthalate, PET) on soil properties in a mangrove restoration wetland, intending to provide a scientific basis and reference for research on the environmental behaviour and effects of microplastics in estuarine restoration wetlands.

Materials and methods

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prevent contamination from external sources, the separation, extraction, and observation processes of microplastics were conducted in a clean and airtight environment, and the Petri dishes containing filter membranes were covered with tin foil.

Identification and analysis of soil microplastics

The above-mentioned separation steps allowed the microplastics in the soil samples to be classified into three classes according to the diameter: 1–2 mm, 0.5–1 mm, and 0.038–0.5 mm (Peng et al., 2017). Microplastics remaining on the membrane were observed and photographed with an optical electron microscope (Olympus SZ61, Japan). Starting from the top and moving in a zigzag pattern from left to right, small squares were observed one at a time to identify microplastics, which were visually identified and classified by the shape and colour according to the same rules as those used by Han et al. (Han et al., 2020). The microplastics could be classified into three shapes, namely, fibrous, spherical, and fragmented microplastics, and seven colours, i.e., clear, blue, black, white, green, red, and orange particles. In the blue category, dark blue, light blue, and green particles were included. In the red category, purple and pink particles were included, but the white category did not include transparent microplastics. A Raman spectrometer (Renishaw inVia, UK), which does not require sample pretreatment, was chosen in this study to identify the various microplastic polymer types. A laser device equipped with a 785-nm laser was chosen for the experiments, scanning the spectral range from 100–3200 cm\(^{-1}\) (Frere et al., 2016). Each sample was scanned three times under the same conditions, with each scanning position randomly switched to ensure the accuracy of the results. In addition, the morphology of typical microplastics was studied with a scanning electron microscope (Hitachi S4800, Japan), and the elemental composition of the microplastic surface was determined with an energy spectrometer (X-Act, Oxford) (Wang et al., 2017).

Experimental design of effects of microplastics on soil properties

Referring to the soil culture method of Liu et al. (Liu et al., 2019b), a portion of the pre-treated wetland soil was incubated in appropriately sized beakers (using cling film with eyelets for closure). In this study, the amount of microplastics applied (w/w, 7% (M1) and 28% (M2)) was determined according to Lwanga (Lwanga et al., 2016). Our experiment included six treatments: (1) \(^6\)CK: bare ground soil only; (2) \(^6\)M1: 7% microplastics + bare ground soil; (3) \(^6\)M2: 28% microplastics + bare ground soil; (4) \(^6\)CK: vegetated area soil only; (5) \(^6\)M1: 7% microplastics + vegetated area soil; (6) \(^6\)M2: 28% microplastics + vegetated area soil. The soil was slightly compacted with a small manual soil compactor to ensure even compaction among all samples. The soil moisture content was maintained at 50% throughout the experiment, and each soil sample was incubated in an artificial climate chamber (incubation conditions: light-to-dark ratio: 14 h:10 h; humidity: 80%). As the soil was air dried, a pre-treatment period of 1 week was required to re-establish microbial metabolism. One sub-soil sample was obtained from each pot on days 1, 3, 7, 14, and 30 after the end of the pre-treatment period, with a final collection of 90 sub-samples (6 treatments x 3 replicates x 5 sampling points), which were labelled and preserved. Some of the fresh soil samples treated with PET were used to determine soil ammonium and nitrate nitrogen. The remaining soil samples were dried naturally, ground in a mortar, sieved through 20- and 100-mesh sieves, and then stored in self-sealing bags for the determination of soil physical and chemical properties.

Determination of soil properties of microplastics

The above unground soil was digested in H\(_2\)O\(_2\) and HCl for more than 12 h, the supernatant was removed with a siphon, and the precipitate was then dispersed with sodium hexametaphosphate. Finally, the soil particle size was determined with a Malvern laser particle size analyser. According to the American soil particle size classification standard, the soil particle size could be divided into three grades: clay (<2 μm), silt (2–20 μm), and sand (>20 μm). A pH meter (Sartorius, PB-10, Germany) was used to determine the soil pH value. The total phosphorus content in the sampled soil was determined. After the soil was calcined at 450 °C in a muffle furnace, the soil was moved to a centrifuge tube, and 3.5 mol L\(^{-1}\) HCl was added, followed by shaking for 16 h. The supernatant was obtained via centrifugation, and the total phosphorus content in the extract was analysed via molybdenum blue colorimetry. The soil alkaline phosphatase activity was determined through disodium benzene phosphate colorimetry. After adding 0.5 mL toluene to a certain amount of soil for 15 min and then adding a boric acid solution of 6.75 g L\(^{-1}\) disodium benzene phosphate (pH = 10), the mixture was shaken well and placed in an incubator at 37 °C. After a certain period, a boric acid solution of pH = 9.0, containing 2.5% potassium ferricyanide, and a 0.5% 4-aminoantipyrine solution were used. The absorbance was measured at a wavelength of 570 nm. Soil total nitrogen was determined with the potassium persulfate (K\(_2\)S\(_2\)O\(_8\)) oxidation method. An oxidant (0.074 mol L\(^{-1}\) K\(_2\)S\(_2\)O\(_8\) + 0.24 mol L\(^{-1}\) NaOH) was added to the soil and sterilized after 2 h of oscillation. The supernatant was centrifuged, and the total nitrogen content was determined via UV spectrophotometry.
determination methods for ammonium nitrogen and nitrate nitrogen in the soil samples were extraction-indophenol blue colorimetry and ultraviolet spectrophotometry, respectively. Determination of soil ammonium nitrogen: a 2 mol·L$^{-1}$ KCl extract was mixed with a fresh soil sample, and the mixture was shaken on a shaker and filtered. KCl extract was added to the filtrate, and 2.5 mL of a phenol solution and 2.5 mL of a sodium hypochlorite alkaline solution were added in turn, and the extract was shaken well. After 1 h at room temperature, 1 mL of a masking agent was added, and the absorbance was finally determined at 625 nm. Determination of soil nitrate nitrogen: a 0.01 mol·L$^{-1}$ calcium chloride extract was mixed with fresh soil, the mixture was shaken and filtered, and the filtrate was acidified by adding 1.00 mL of a 1:9 sulfuric acid solution. Finally, absorbance values (A210 and A275) were measured at 210 and 275 nm, respectively, in a 1-cm optical diameter quartz cuvette.

Data analysis

One-way analysis of variance (ANOVA) was used to analyse the variability in the abundance of soil microplastics under the different planting densities and the differences in the soil particle size, pH, alkaline phosphatase, total phosphorus, total nitrogen, ammonium nitrogen, and nitrate nitrogen under the different microplastic addition treatments ($p < 0.05$). Pearson correlation analysis was employed to analyse the correlation between each soil property. In addition, all experimental data processing and statistical analysis operations were completed in Excel 2019 and SPSS 25.0, respectively, and graphs were generated with Origin Pro 2020b, PowerPoint 2019, and Word 2019.

Results

Morphology and abundance characteristics of soil microplastics at different planting densities in mangrove restoration wetlands in the Jinjiang River estuary

Morphological characteristics of soil microplastics in mangrove restored wetlands

Micrographs of the different types of microplastic samples are shown in Fig. S2. The microplastics shown in (a) are white and green lumpy microplastics, irregularly shaped and of a certain thickness, and harder than the film. The microplastics shown in (b) and (c) are blue and red filamentous fibres, respectively, with elongated circumferences, which are easy to identify. The microplastics shown in (d) and (e) are transparent and red films, respectively, which are irregular in shape, exhibit a small thickness but a large area and are easily deformed or bent with tweezers. The microplastics shown in (f) include transparent fibres and spherical granules, of which the granules are regular in shape, round, small in diameter, with a glossy surface and are easy to identify. Based on microscopy observations, this study was conducted to count the abundance of microplastics and to determine the percentage by the shape and colour. Due to the nature of the microplastic appearance (Fig. 1A), strip and granular microplastics were the most represented in the study area, at 45% and 40.4%, respectively, followed by fragmented microplastics at 13.7% and films and agglomerated fibres at less than 1%. Based on the colour of the detected microplastics (Fig. 1B), transparent and white microplastics were the most abundant, at 47% and 38.1%, respectively. Microplastics of all other colours accounted for less than 10% of the total microplastics, in the order of black (6%), blue (4.5%), red 2.1%, yellow (1.3%) and green (1.1%) microplastics.

Identification of microplastics by Raman spectroscopy

In this study, a total of 100 soil microplastic samples were identified via Raman spectroscopy. Through comparative analysis with a standard atlas or spectra reported in the literature, the occurrence of microplastics was finally
determined, as well as the combination of their polymer types. Among the 100 microplastic samples identified and analysed via Raman spectroscopy, the three types of plastics with the highest proportion were (a) polyethylene terephthalate (PET; identified based on the peaks at 1612, 1726 and 3082 cm\(^{-1}\)), accounting for 52% of the total microplastics, (b) polyethylene (PE; identified based on the two peaks at 2864 and 2881 cm\(^{-1}\)), accounting for 24% of the total microplastics, and (c) polypropylene (PP; identified based on the peaks at 2839, 2881, 2904 and 2960 cm\(^{-1}\)), accounting for 15% of the total microplastics. Raman spectra for these three microplastics are shown in Fig. S3.

Surface microstructure and elemental composition of microplastics

Scanning electron microscopy (SEM) was applied to observe the microplastics isolated from the sampled wetland soil, and typical surface characteristics of some of the samples are shown in Fig. 2. SEM images of microplastic
films are shown at different magnifications in (a, b). It can be observed that the surface of these films is relatively flat, but there are obvious holes and folds. SEM images of transparent fibres are shown at different magnifications in (c, d). The fibres are elongated, and their surfaces are rough with certain substances adhered. SEM images of white granular microplastics are shown at different magnifications in (g, h). Compared to the observed films and fibres, the granular microplastics exhibit a smooth surface with obvious protrusions and substances adhered to the surface at the locations of these protrusions. SEM images of the bulk microplastics are shown at different magnifications in (j, k). The surface of the bulk microplastics is much rougher than that of the films, exhibiting obvious cracks, fractures, and voids, again with material adhered to the surface.

Abundance distribution characteristics of soil microplastics at different planting densities in mangrove restored wetlands

The distribution of the soil microplastic abundance under the different planting densities in the mangrove restoration wetland is shown in Fig. 3. The soil microplastic abundance in the entire study area ranged from 563.3–6196.7 n/500 g (dry soil) with a mean abundance of 2177.5 n/500 g (dry soil), with the maximum value occurring in the restored area under a planting density of 0.5 × 0.5. In the present study, the use of graded sieve separation allowed direct classification of the microplastics into three types based on the diameter (Fig. 3B). On the one hand, analysis of variance (Fig. 3A) clearly indicated that the abundance of microplastics with diameters of 0.038–0.5 mm significantly differed ($p < 0.05$). Based on the abundance, the microplastics in the recovery zone under the different planting densities could be ranked as 0.5 × 0.5 m > 1.0 × 0.5 m > 1.0 × 1.0 m > control zone. On the other hand (Fig. 3B), most of the microplastics in the restoration zone varied between 0.038 and 0.5 mm in diameter, which accounted for 70.9% of the total microplastics, and approximately 15.5% of the microplastics varied between 0.5 and 1 mm in diameter, while approximately 13.6% of the microplastics ranged from 1–2 mm in diameter, indicating that smaller-diameter microplastics were more likely to accumulate in wetland soil.

Effects of microplastics (PET) on soil physical and chemical properties

Effect of PET on the particle size of wetland soil

Figure 4 shows changes in the proportion of soil clay, silt, and sand particles in the bare soil and vegetated areas of the mangrove restoration wetland after the addition of PET. Table S2 provides statistical analysis results of the changes in soil clay, silt, and sand particles. In regard to soil clay particles (Fig. 4A, B), the proportion of clay particles in bare soil treated with $^a$M1 (7% PET) ranged from 5.34–6.39%, with a low fluctuation overall, but a slowly increasing trend from the 14th day of culture. However, the percentage of clay in bare soil treated with $^b$M2 (28% PET added) ranged from 4.55–6.77%, with fluctuations overall, indicating a trend of increasing, then decreasing and then slowly increasing. The clay content in bare soil treated with $^a$M1 and $^b$M2 was significantly higher than that in $^c$CK soil (0% PET) on the 7th and 14th days ($p < 0.05$; Table S2). In terms of mean value analysis, the clay content in bare land

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**Fig. 3** Distribution characteristics of soil microplastics under different planting densities in mangrove restoration wetlands. A Difference analysis of soil microplastics abundance distribution (different lowercase letters (a/b/c) indicated significant difference in soil microplastics distribution under different planting densities, $p < 0.05$); B Proportional distribution of soil microplastics with different diameters.
under the \(^a\)M1 and \(^a\)M2 treatments was higher than that in the CK group, and the mean value indicated the order of \(^a\)M1 (5.67%) > \(^a\)M2 (5.42%) > \(^a\)CK (5.35%). However, the proportion of soil clay in the vegetation area after PET treatment was the opposite to that in the bare soil area. From the 3rd day of cultivation, the proportion of soil clay in the vegetation area after the \(^b\)M1 and \(^b\)M2 treatments indicated a significant downward trend. On the seventh day of cultivation, the proportion of soil clay increased, but the proportion remained below that in \(^b\)CK soil. In addition, the average proportion of clay particles in the vegetation area under the \(^b\)M1 and \(^b\)M2 and \(^b\)CK treatments indicated the order of \(^b\)CK (6.07%) > \(^b\)M2 (5.72) > \(^b\)M1 (5.47). Combined with coefficient of variation (CV) analysis, the results

Fig. 4 Changes in the proportions of clay, silt and sand after the addition of PET. A, C, E Bare soil and B, D, F vegetation soil
revealed the order of CV_{bM1} (19.19%) > CV_{aM1} (7.68%), but CV_{bM2} (7.48%) < CV_{aM2} (14.31%).

In terms of soil silt particles (Fig. 4C, D), the fluctuation in the bare and vegetation soil samples after PET treatment was low, and the variation range was 64.37–77.96% and 65.05–82.07%, respectively. The difference analysis results revealed that the percentage of silt in bare soil treated with aM1 on the 7th and 30th days significantly differed from that in CK soil, and the percentage of silt in bare soil treated with aM2 on the 1st, 7th and 30th days significantly differed from that in CK soil ($p < 0.05$; Table S2). However, after PET treatment, the distribution of silt particles in the vegetation area significantly differed from that in CK group only on the 14th day under the bM1 treatment ($p < 0.05$). Combined with CV analysis, the CV values for silt in the bare and vegetation soil samples after PET addition were less than 10%, indicating that silt could still maintain a relatively stable distribution state after PET addition.

Regarding soil sand particles (Fig. 4E, F), the change in the bare and vegetation soil samples after PET treatment notably fluctuated. The sand proportion in bare soil under the aM1 and aM2 treatments ranged from 19.45–27.40% and 24.76–33.14%, respectively, while that in vegetation soil under the bM1 and bM2 treatments ranged from 12.43–33.01% and 12.48–23.94%, respectively. The results of the different analysis methods indicated that there was no significant difference in the percentage of sand grains on the 30th and 7th days under the aM1 and aM2 treatments in bare soil from CK levels ($p < 0.05$), but there was a significant difference in the percentage of sand grains on the 1st, 7th, 14th and 30th days under the bM1 and bM2 treatments in the vegetation area from bCK levels ($p < 0.05$). Combined with mean value analysis, it was found that the proportion of sand particles in the bare and vegetation soil samples increased to varying degrees after PET treatment (Table S2). In addition, combined with CV analysis, it was determined that the CV values for the sand proportion in the bare and vegetation soil samples increased after PET treatment (>10%). Among the various treatments, the CV values for bare soil treated with aM1 and aM2 were less than 17%, and the specific order was CV_{aM1} (16.73%) > CV_{aM2} (11.84%). The CV values for vegetation soil treated with bM1 and bM2 were greater than 20%, and the specific order was CV_{bM1} (42.49%) > CV_{bM2} (23.46%).

In summary, the proportion of clay particles in the sampled soil after PET treatment decreased, while the proportion of silt particles remained stable, but the proportion of sand particles increased. In addition, overall, the CV values for the soil in the vegetation zone after the addition of PET were higher than those for the soil in the bare zone, indicating that the distribution of the proportion of each grain size in the soil of the vegetation zone was more unstable and more notably fluctuated after PET treatment than the distribution in the soil of the bare zone, which could further indicate, to a certain extent, that the grain size distribution in the soil of the vegetation zone was more susceptible to the influence of PET.

**Effect of PET on the pH of wetland soil**

Figure 5 shows changes in the soil pH after adding PET to the bare and vegetation soil samples obtained from the mangrove restoration wetland, and Table S3 provides statistical analysis results of the soil pH. The pH values of the bare soil treated with aM1 and aM2 ranged from 7.83–7.93 and 7.77–7.90, respectively, with a narrow range (Fig. 5A). Combined with difference analysis, it was found that there existed no significant difference in the pH of bare soil under the aM1 and aM2 treatments from CK levels ($p > 0.05$; Table S3). The CV analysis results also revealed that the CV values of all treatments were low (all <1.3%) (Table S3). However, it should be noted that from the 14th day under the aM1 and aM2 treatments, the pH of bare soil was significantly lower than that of CK soil (Fig. 5A). Regarding the soil in the vegetation area, the pH values after treatment with bM1 and bM2 ranged from 7.03–7.44 and 6.94–7.45, respectively, with notable fluctuations, and the 14th day after treatment was the turning point, indicating a trend of first rising and then declining (Fig. 5B). In addition, there were significant differences in the pH of
vegetation soil on the 3rd day after the M1 treatment, the 3rd day after the M2 treatment, and the 7th day after the M2 treatment (p < 0.05; Table S3). Combined with CV analysis, it was determined that the CV value for vegetation soil after PET treatment was higher than that for bare soil (all > 2%). In terms of mean change (Table S3), the pH value of the bare and vegetation soil samples under the M1 and M2 treatments slightly differed from CK levels, but the overall trend was still downwards, indicating that PET increased the soil acidity to a certain extent, which should be studied further.

**Effects of PET on total nitrogen, ammonium nitrogen and nitrate nitrogen in wetland soil**

Figure 6 shows changes in the soil total nitrogen (TN), ammonium nitrogen (NH$_4^+$-N), and nitrate-nitrogen (NO$_3^-$-N) after adding PET culture. A, C, E Bare soil and B, D, F vegetation soil.

**Fig. 6** Change characteristics of total nitrogen, ammonium nitrogen and nitrate nitrogen in wetland soil after adding PET culture. A, C, E Bare soil and B, D, F vegetation soil.

CK: 0% PET; M1: 7% PET; M2: 28% PET
N) contents in the bare soil and vegetation areas of the mangrove restoration wetland after PET treatment. Tables S4–6 provide statistical analysis results of the soil total nitrogen (TN), ammonium nitrogen (NH$_4^+$-N), and nitrate-nitrogen (NO$_3^-$-N) contents. The TN content in bare soil treated with $^a$M1 and $^a$M2 ranged from 140.43–450.43 mg/kg and 66.03–462.83 mg/kg, respectively, with a wide range (Table S4). After the $^a$M1 treatment, TN first increased and then stabilized, while after the $^a$M2 treatment, TN first increased and then decreased (Fig. 6A). The difference analysis results indicated that the TN content in bare soil treated with $^a$M1 and $^a$M2 on the first and 30th days was significantly higher than that in CK soil ($p < 0.05$), and the mean value demonstrated that the TN content in bare soil treated with $^a$M2 was 21.07% lower than that in CK soil. The change trend of soil TN in the vegetation area after PET treatment differed from that of soil TN in the bare soil area. The soil TN content in the vegetation area under the $^b$M1 and $^b$M2 treatments ranged from 103.23–779.02 mg/kg and 301.63–729.42 mg/kg, respectively, with a wide variation range. The soil TN content in the vegetation area under both treatments exhibited a trend of first decreasing and then increasing and then slowly increasing (Fig. 6B). However, compared to $^b$CK, the average TN content in vegetation soil under the $^b$M1 and $^b$M2 treatments was still 14.31% lower than that under the CK treatment (Table S4). The CV analysis results indicated that the CV values of TN in the bare and vegetation soil samples after PET treatment increased, especially in bare soil (>50%) and in vegetation soil under the M1 treatment (>50%) (Table S4).

Regarding soil ammonium nitrogen (NH$_4^+$-N), the change trend in the bare soil and vegetation areas under the PET treatment was obvious (Fig. 6C, D). The content of NH$_4^+$-N in bare soil under the $^a$M1 treatment ranged from 20.06–30.65 mg/kg, which exhibited a wide variation range and an overall declining trend, while the content of NH$_4^+$-N in the soil treated with $^a$M2 ranged from 18.73–19.84 mg/kg, which did not significantly change (Fig. 6C). In addition, the NH$_4^+$-N content in bare soil was significantly higher than that in CK soil on the first, 7th, and 14th days after treatment with $^a$M2 ($p < 0.05$; Table S5). The content of NH$_4^+$-N in the vegetation area after the $^b$M1 and $^b$M2 treatments was high, ranging from 1.84–20.10 mg/kg and 4.93–24.45 mg/kg, respectively, exhibiting fluctuations (Fig. 6D). The NH$_4^+$-N content in the soil of the vegetation area was significantly reduced by 67.93% and 79.84% below that in the $^b$CK group on the 30th day after the $^b$M1 and $^b$M2 treatments, respectively ($p < 0.05$; Table S5). According to CV analysis, it was found that the CV value of NH$_4^+$-N in bare soil was lower after PET treatment, especially under the $^a$M2 treatment (2.46%), while that in vegetation area soil treated with PET was generally higher than 30%. Regarding nitrate-nitrogen (NO$_3^-$-N), the content of NO$_3^-$-N in bare soil treated with $^a$M1 ranged from 1.44–19.29 mg/kg, while that in soil treated with $^a$M2 ranged from 17.19–35.23 mg/kg, with a wide variation range, and the overall trend was a decreasing trend (Fig. 6E). The mean NO$_3^-$-N content in bare soil treated with $^a$M2 was significantly higher than that in CK soil, but there was no significant difference between $^a$M1 and $^a$CK. The NO$_3^-$-N content in the soil of the vegetation area after the $^b$M1 and $^b$M2 treatments was significantly higher than that under the CK treatment ($p < 0.05$; Table S6), and there occurred a significant upward trend (Fig. 6F), which was contrary to the experimental results of soil ammonium nitrogen (NH$_4^+$-N). Combined with CV analysis, the CV value of NO$_3^-$-N was greater than 25% under the various treatments.

**Effects of pet on total phosphorus and phosphatase in wetland soil**

Figure 7 shows the change in the total phosphorus (TP) content in the bare soil and vegetation areas of the mangrove restoration wetland after PET treatment. Table S7 provides statistical analysis results of the total soil phosphorus (TP) content. In this study, the content of TP in bare soil generally ranged from 352.27–479.43 mg/kg, the change range was narrow, and the change curve of TP fluctuated slightly (Fig. 7A). In addition, the TP content in bare soil treated with $^a$M1 and $^a$M2 was significantly lower...
than that in CK soil. After the \(^aM2\) treatment, except for a content of 402.31 mg/kg in the samples on the 7th day, the TP content in the other samples was lower than 400 mg/kg, and the mean TP content in the samples treated with \(^aM2\) was 13.36\% lower than that in the CK group (\(p < 0.05\); Table S7). The total phosphorus content in the vegetation area ranged from 699.28–1229.24 mg/kg, the change range was wide, and the change curve of the total phosphorus (TP) content fluctuated greatly. The total phosphorus (TP) content in soil treated with \(^bM1\) and \(^bM2\) first increased, then decreased and increased again on the 14th day (Fig. 7B). In addition, the total phosphorus (TP) content in soil treated with \(^bM1\) and \(^bM2\) in the vegetation area was significantly lower than that in \(^bCK\) soil (\(p < 0.05\); Table S7), especially in the \(^bM2\) treatment group, and the mean TP content was 18.73\% lower than that in the \(^bCK\) group. According to CV analysis, the CV values of soil phosphorus (TP) in bare soil treated with \(^aM1\) and \(^aM2\) reached 8.06\% and 5.45\%, respectively, while the CV values of TP in the vegetation soil samples under the \(^bM1\) and \(^bM2\) treatments were higher than 13\%, 14.02\%, and 13.26\%, respectively. It was found that the distribution of phosphorus in mangrove restoration wet soil could result in an uneven soil phosphorus distribution after PET treatment.

Figure 8 shows the characteristics of soil phosphatase changes in the bare and vegetation soil samples obtained from the mangrove restoration wetland after PET addition, and Table S8 provides the statistical analysis results of soil phosphatase. The phosphatase content in the bare soil samples in this study was generally low, at 0.32–0.68 mg/(g·h), with a narrow range of variation and slight fluctuations in the phosphatase change curve (Fig. 8A). Furthermore, based on analysis of variance, the phosphatase content in the bare soil samples treated with \(^aM1\) only significantly differed from that in the \(^aCK\) samples on days 14 and 30 of incubation, whereas the phosphatase content in the \(^aM2\)-treated soils was significantly different from that in the \(^aCK\) samples on days 3, 7 and 14 of incubation, which was significantly lower. Moreover, the phosphatase content was significantly higher than that in the \(^aCK\) samples on days 1 and 30 (\(p < 0.05\); Table S8). The overall higher phosphatase levels ranging from 1.42–2.01 mg/(g·h) in the vegetation soil samples indicated a wide range of variability and fluctuating phosphatase profiles (Fig. 8B). According to analysis of variance, the phosphatase content in the vegetation soil samples treated with \(^bM1\) significantly differed from that in the CK samples except on day 1, and the phosphatase content in the samples treated with \(^bM2\) significantly differed from that in the CK samples on days 3, 14, and 30 (\(p < 0.05\); Table S8). Moreover, the phosphatase levels were all lower under the \(^bM2\) treatment than those under the CK treatment. Through mean analysis, the phosphatase content in the vegetated area soils was 10.82\% and 19.07\% lower under the \(^bM1\) and \(^bM2\) treatments, respectively, than that in the \(^bCK\) group samples. In addition, CV analysis revealed that the CV values of soil phosphatase in bare ground soils treated with \(^aM1\) and \(^aM2\) were 15.14\% and 23.69\%, respectively, while the CV values of soil phosphatase in vegetated area soils treated with \(^bM1\) and \(^bM2\) were 9.62\% and 9.04\%, respectively.

**Correlation analysis**

Figure 9 shows a thermodynamic diagram of the correlation coefficient matrix between soil physical and chemical factors, reflecting the correlation between the characteristics of wetland soil after PET addition. Clay was significantly positively correlated with silt and TN (\(p < 0.01\)) and negatively correlated with sand (\(p < 0.05\)), but there occurred no significant correlation between clay and the other physical and chemical factors. There existed no significant correlation between silt and \(\text{NH}_4^+\)-N but an extremely significant correlation occurred between silt and the other physical and chemical factors (\(P < 0.01\)). Among the various fractions, silt attained an extremely significant positive correlation with TN, \(\text{NO}_3^-\)-N, TP, and phosphatase (\(p < 0.01\)) and an extremely significant negative correlation with sand and pH (\(p < 0.01\)). Sand also attained no significant correlation with \(\text{NH}_4^+\)-N but achieved a significant correlation with the other physical and chemical factors, such as the pH.
Correlation coefficient matrix heat map between soil properties (Fig. 9). Significant correlation, *p* < 0.05; extremely significant correlation, **p** < 0.01.

Morphological characteristics and abundance distribution of soil microplastics under different planting densities in mangrove restoration wetland of Jinjiang Estuary

Source apportionment and surface characteristics of soil microplastics

Based on the complexity of migration, transportation, and transformation processes of microplastics in the environment, it is relatively difficult to determine the exact source of microplastics, but the source could be inferred from the appearance and physical and chemical properties of microplastics (Nor and Obbard, 2014; Peng et al., 2017). In this study, Raman spectroscopy was used to identify soil microplastics to distinguish between microplastics and polymers (Dong et al., 2020). In this study, polypropylene (PP), polyethylene (PE) and polyethylene terephthalate (PET, polyester resin) were the main soil microplastic polymers (Fig. S3). On the one hand, Jinjiang is China’s shoe capital and a textile and clothing production base. A large amount of wastewater containing clothing fibres stemming from industrial production and daily life laundry activities may flow into Jinjiang through the drainage system and into the East China Sea through the Jinjiang Estuary (Deng et al., 2020). Some fibrous microplastics exhibited obvious filamentous characteristics under the microscope (Fig. S2), and the polymers mainly included PE and PP, which fully verifies that one of the sources of fibrous microplastics comprises clothing fibres. On the other hand, PET, PE, and PP were the most abundant microplastic polymers in this study, and these microplastic polymers are widely used in food containers, bottles, plastic bags, general packaging, clothing, fishing nets, carpets, etc (Tan et al., 2019). However, these plastic products can gather in large quantities near the estuarine wetland and become the main source of microplastics.

Scanning electron microscopy (SEM) images revealed that fractures, tears, pits, flakes, and other attachments were common degradation features (Fig. 2), demonstrating that surface soil microplastics in mangrove wetlands of the Jinjiang Estuary had undergone varying degrees of mechanical erosion and chemical weathering, mainly due to wave action and mechanical abrasion such as sanding (Wang et al., 2017). In addition, partial degradation, such as fractures, tears, and pits, could lead to an increase in the surface area-to-volume ratio of microplastics, providing more space for microorganism colonization, and the increased surface area accelerates the degradation process (Chamas et al., 2020). The energy spectra indicated that the elemental composition of the surface of the different microplastic samples greatly varied, with the highest proportion of O atoms, but metal elements such as Pb, Fe, Mn, K, and Na were also detected on the surface of microplastics at a high proportion, indicating that microplastics in water in wetland soils could act as carriers of heavy metal ions (Ashton et al., 2010). Moreover, where the surface of microplastics contained depressions, cracks, or filaments, the composition of surface elements was highly diverse, and heavy metal elements were more easily detected, indicating that plasticageing could lead to increased absorption of heavy metals (Ashton et al., 2010).

Characteristics of microplastic abundance distribution in wetland soils and its influencing factors

The average abundance of microplastics in wetland soil interstitial water in the study area ranged from 563.3–6196.7 n/500 g, which was significantly higher than the average abundance of microplastics in this area in 2017 (490–1170 n/500 g) (Deng, 2019). It should be that the proximity of the sampling area to mariculture areas and the relatively intensive production activities are important
causes of microplastic accumulation in the region. It has been pointed out that exogenous inputs from human activities exhibit a very significant correlation with the distribution of microplastic contaminants, and the fewer human activities there are, the lower the level of microplastic pollution (Nor and Obbard, 2014). In addition, as Jinjiang, Fujian Province, is the footwear capital and a textile and garment production base in China, a large number of microplastics may remain in industrial production and residential sewage flows, while the present study area is located at the mouth of the Jinjiang River and East China Sea. Microplastics carried through the drainage system may be partially intercepted in mangrove wetland soil, resulting in a high abundance of soil microplastics within the area (Ben-David et al., 2021). In addition, the abundance of soil microplastics in the mangrove restoration wetland study area varied significantly among the different planting densities ($p < 0.05$, Fig. 3A) and was the highest in the 0.5 × 0.5 m area. On the one hand, dense vegetation in wetlands could effectively trap microplastics (Sutton et al., 2016), and although plastics or microplastics may be transferred from surface water into seawater in mangrove ecosystems, mangrove vegetation could effectively inhibit this transfer process (Li et al., 2019), thus allowing microplastics to be retained in the wetland soil environment. On the other hand, microplastics could also adhere to mangrove branches or be trapped by other trailing plants in wetlands and enter the soil through rainfall, wind, and other forces (Li et al., 2018).

Influence of microplastics (PET) on soil physicochemical properties

Effect of PET on the particle size of wetland soils

The distribution of the various soil particle sizes (clay, silt, and sand) greatly influenced soil properties, especially the soil texture (Zhang et al., 2016). There were significant differences in the distribution of each soil particle size (clay, silt, and sand) after the addition of exogenous microplastic PET to the wetland bare ground and vegetated area soil samples. The proportion of clay particles in the wetland soil samples decreased after PET addition, while the proportion of silt particles remained unchanged, but the proportion of sand particles increased. In other words, the proportion of clay particles decreased, and the proportion of sand particles increased, which is consistent with the significant negative correlation between clay and sand particles in this study ($p < 0.05$, Fig. 9). The reason for this phenomenon is that the small size and stickiness of organic matter particles among the soil particles may cause adhesion upon contact with PET, thus forming large soil particles (Jeong, 2014), while the surface of PET contains many voids (Fig. S1f), which could provide space for small soil particles to enter. This could lead to a decrease in the proportion of clay particles and an increase in the proportion of sand particles in the treated soil (Wang et al., 2019). In addition, it has been suggested that soils with a low clay content and a high sand content are not conducive to plant growth, as these conditions could reduce soil water and nutrient retention and fertility on the one hand and lower soil respiration on the other hand (Garcia-Meza et al., 2004; Hall et al., 2010; Nguyen and Marschner, 2014). In this study, the proportion of clay particles in the bare ground soil samples did not exhibit a similar trend to that of the proportion of clay particles in the vegetated area soil samples, but there occurred a significant increase in the proportion of sand particles in the bare ground soil samples. On the one hand, adhesion among the bare ground soil samples was usually lower than that among the vegetated soil samples (Boer and Puigdefabregas, 2010), which does not facilitate the adhesion of small- and medium-sized soil particles to form large-sized soil particles, so the proportion of clay particles in the bare ground soil samples did not significantly decrease after PET treatment. However, adhesion to PET could also facilitate the formation of large-sized soil particles such as sand particles. The coefficients of variation (Table S2) indicated whether the distribution of clay, silt, and sand particles was uniform between the bare ground and vegetated soil samples after PET treatment (Altunkaynak and Gamgam, 2019). First, the CV values of silt grains were <10% for both types of soils samples after PET treatment, indicating that the distribution of silt grains remained relatively stable after PET addition. In contrast, the CV values of both clay and sand grains were higher, especially the CV values of sand grains, at >10%, indicating that the distribution of clay and sand grains did not remain constant after PET addition. In addition, the CV values for the vegetated soil samples were higher than those for the bare ground soil samples after PET treatment, indicating that the distribution of each particle size in the vegetated soil samples was more unstable and more notably fluctuated than that of each particle size in the bare ground soil samples after PET treatment, which also indicates, to a certain extent, that the particle size distribution in the vegetated soil samples was more susceptible to the influence of PET.

Effect of PET on pH of wetland soil

The soil pH is an intensity value of the soil acidity, representing the active acidity of soil, which directly affects the occurrence state, transformation, and effectiveness of soil nutrients, which in turn affects plant growth and development (Hong et al., 2018). In this study, the soil pH of the wetland bare ground soil and vegetated area soil samples after PET addition exhibited little change over CK levels

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but still revealed an overall decreasing trend, indicating that PET increases the soil acidity to a certain extent. Bandow et al. (Bandow et al., 2017) also reported that the presence of microplastics (e.g., HDPE) can reduce the soil pH, and they found that HDPE particles reduced the soil pH after 6 and 12 weeks of exposure to light and thermal oxidation conditions. Similar to HDPE particles, PET could alter cation exchange in soil, which in turn could lead to changes in the soil pH. Janczak et al. (Janczak et al., 2018) found that the presence of plastic lowered the soil pH and increased the number of redox sites in soil after 6 months of incubation, and there is some similarity to this study, but the mechanism of change should be further investigated. However, changes in the soil pH can directly interfere with the diversity and composition of soil microorganisms, thus affecting both the soil microbial community and inter- 
rooted microbial community (Boots et al., 2019). Recent studies have demonstrated that the addition of 28% w/w polypropylene microplastics to soil altered the microbial activity and nutrient content in dissolved organic matter as measured via the hydrolysed fluorescein diacetate method over levels when no microplastics were added (Liu et al., 2017). Similarly, Machado et al. (Machado et al., 2018) reported that the soil microbial activity responds to the addition of microplastics, depending on the addition amount. Microbial activity plays an important role in the decomposition of soil organic matter and the cycling of major nutrients, which are essential for plant growth and root development, and the transformation of functional processes, including microbially driven processes, could impact this process (Yan et al., 2017).

Effects of PET on soil nitrogen

Of all essential plant nutrients, nitrogen is the most important factor limiting plant growth and yield, and the ability of the soil nitrogen supply can be elucidated by analysing the total soil nitrogen content and morphology (Qi et al., 2021). In addition, soil effective nitrogen exists in the form of nitrogen that can be directly absorbed by plant roots, while ammonium and nitrate nitrogen can be directly absorbed by plant roots and are often collectively referred to as effective nitrogen (Mengel, 1991). In this study, the trends of the total N content in the wetland bare ground soil samples after PET addition significantly differed from those of the total N content in the vegetated area soils samples, and both types of soil samples exhibited a significant decrease in the total N content from CK group levels (Fig. 6A, B). This is similar to the study by Wang et al. (Wang et al., 2016), where the presence of microplastics could significantly reduce the soil N content. In addition, combined with analysis of soil ammonium nitrogen (NH₄⁺-N) and nitrate-nitrogen (NO₃⁻-N), it was found that the overall change in the NH₄⁺-N content in the wetland soil samples after PET treatment revealed a decreasing trend, especially the NH₄⁺-N content in the vegetated area soil samples, which significantly decreased by 67.93% and 79.84% after the 7% and 28% PET treatments, respectively, from CK levels (p < 0.05; Table S7). This result is similar to the findings of Chen et al. (Chen et al., 2020), who indicated that the concentration of NH₄⁺-N in soil was significantly reduced by 44.3% and 21.3% from CK levels between days 12 and 15 under the action of microplastics. It has also been noted that the NH₄⁺-N concentration in soils containing HDPE was half that in sediments without microplastics (Green et al., 2016). Regarding NO₃⁻-N, all wetland soil samples treated with PET indicated a significant increase in the NO₃⁻-N content over CK group levels (p < 0.05; Table S6). On the one hand, PET can chelate NH₄⁺-N via carbonyl (=O) and hydroxyl groups (-OH) on the surface (Yuan et al., 2020). On the other hand, we found that the NO₃⁻-N concentration exhibited an increasing trend under the M1 and M2 treatments (Fig. 6F), which indicates that PET could accelerate the soil nitrification process and thus reduce the content of NH₄⁺-N, which in turn could affect the transformation of N in the treated soil. MPs have been reported to alter the soil biophysical environment and increase the soil porosity, which may increase the air flux in soil and explain the promotion of ammonia oxidation to provide sufficient dissolved oxygen (Machado et al., 2018; Rubol et al., 2013). Furthermore, it has been suggested that the presence of microplastics can alter the availability of substrates for soil denitrification and anaerobic ammonia oxidation processes, which could significantly affect soil N conversion (Huang et al., 2021).

Effects of PET on phosphorus and phosphatase in wetland soil

Although the total phosphorus content in soil does not directly reflect the soil phosphorus supply capacity, if the total phosphorus content in soil is very low (<0.4 g/kg), the phosphorus supply may be inadequate (Chi et al., 2011). In this study, the total phosphorus (TP) content in the bare ground soil samples treated with aM1 and aM2 was significantly reduced from 4CK levels (p < 0.05; Table S7), especially the TP content in the bare ground soil samples treated with aM2 (dropped below 400 mg/kg), which could result in an inadequate soil phosphorus supply (Fig. 7A). Although the soil in the vegetated area contained a higher overall total phosphorus content (>690 mg/kg), the addition of PET also significantly reduced the total soil phosphorus content from 5CK levels (p < 0.05; Table S7), indicating that the presence of PET could result in an inadequate soil phosphorus supply. The reason for this phenomenon is the sorption of phosphate by soil due to microplastics (Huang
et al., 2019). Soil phosphatase, an enzyme that catalyses the mineralization of soil organic phosphorus, directly impacts the decomposition and conversion of soil organic phosphorus and its biological effectiveness (Yan et al., 2014). Typically, soil enzymes are derived from soil microorganisms, animal excretions and plant roots, and debris decomposition, with soil microorganisms as the main source of soil enzyme formation and accumulation (Weintraub and Schimel, 2005). Enzymes and microorganisms combine to transform organic matter in soil and play an important role in the biogeochemical cycling of chemical elements and pollutant degradation (Sarand et al., 1998), but the soil enzyme activity is susceptible to physical, chemical, and biological factors as well as exogenous contamination (Fernandez et al., 2004). In this study, the phosphatase activity in both the bare ground and vegetated area soil samples after PET addition significantly differed from that in the CK soil samples (p < 0.05; Table S8), especially in the vegetated area soil samples, where the soil enzyme activity was significantly lower after the bM1 and bM2 treatment than that after the bCK treatment (p < 0.05; Table S8). Yi et al. (Yi et al., 2021) also found that microplastics reduced soil urease, dehydrogenase, and phosphatase activities. The decrease in phosphatase activity observed in this study could be mainly attributed to the high PET content (7% and 28%, w/w); however, this finding is in contrast to the inhibition of FDase activity caused by PVC or PE, where the addition of MPs to soil enhanced urease and phosphatase activity levels (Machado et al., 2019), a difference that may be due to soil characteristics and MP type (Fei et al., 2020). In addition, microplastics have been suggested to alter soil properties by increasing the soil permeability and reducing heat uptake, which could affect the soil enzyme activity (Carson, 2013).

Conclusion

The distribution of microplastics in mangrove restoration wetlands varied significantly under different planting densities. In this study, the microplastic abundance could be ranked based on the planting density as follows: 0.5 × 0.5 m > 1.0 × 0.5 m > 1.0 × 1.0 m > control area. The smaller the particle size of microplastic particles, the more likely they are to accumulate in wetland soil, which in turn could exert a notable negative impact on the soil environment. In addition, Polyethylene terephthalate (PET), polyethylene (PE), and polypropylene (PP) were the three most dominant microplastic types in the sampled soils in the study area, indicating that textile and clothing production, clothing laundering, and mariculture activities and waste plastic products originating from daily human life are the main sources of microplastics in the area. In addition, microplastics can constitute important source, transport and exposure pathways for heavy metals. In a study on the properties of PET in soil, it was found that PET can influence the distribution of the soil particle size due to its adsorptive characteristics. All physicochemical factors, except the soil pH, were significantly changed by PET. On the one hand, PET can significantly reduce the content of nitrogen and phosphorus and the activity of soil phosphatase in wetland soils. On the other hand, PET can accelerate the nitrification process in soils, thus reducing the content of ammonium nitrogen and affecting nitrogen transformation in soils. The CV analysis results reflect the fact that soils in vegetated areas are more susceptible to PET than are bare ground soils, leading to greater changes in each characteristic.

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Compliance with ethical standards

Conflict of interest The authors declare no competing interests.

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