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REVIEW ARTICLE

Microplastic pollution on the soil and its consequences on the nitrogen cycle: a review

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Abstract
Microplastics (MPs) correspond to plastics between 0.1 µm and 5 mm in diameter, and these can be intentionally manufactured to be microscopic or generated from the fragmentation of larger plastics. Currently, MP contamination is a complicated subject due to its accumulation in the environment. They are a novel surface and a source of nutrients in soils because MPs can serve as a substrate for the colonization of microorganisms. Its presence in soil triggers physical (stability of aggregates, soil bulk density, and water dynamics), chemical (nutrients availability, organic matter, and pH), and biological changes (microbial activity and soil fauna). All these changes alter organic matter degradation and biogeochemical cycles such as the nitrogen (N) cycle, which is a key predictor of ecological stability and management in the terrestrial ecosystem. This review aims to explore how MPs affect the N cycle in the soil, the techniques to detect it in soil, and their effects on the physico-chemical and biological parameters, emphasizing the impact on the main bacterial groups, genes, and enzymes associated with the different stages of the N cycle.

Keywords Plastisphere · Soil microbiome · Enzyme activities · Biogeochemical cycles · Microplastic identification · Soil fauna · Microbial nitrogen genes

Introduction

Plastics have a range of unique properties and have numerous applications; they can be used at an extensive range of temperatures, they are corrosion resistant, very strong, and though. Furthermore, their low cost, diversity, and utility make them suitable for various applications (Table 1) (Andrady and Neal 2009; PlasticsEurope 2021). Nowadays, we are in the plastic age; the current global usage of plastic is enormous and has been increasing in recent years, reaching 368 million tons in 2019 (Thompson et al. 2009; PlasticsEurope 2021). Plastics represent 10% of waste generated around the world, while some plastic wastes are recycled, and the majority end up in the environment like landfills and agriculture fields (Barnes et al. 2009; Wang et al. 2019). According to Horton et al. (2017), between 473,000 and 910,000 tons of plastic waste are released and retained annually in continental environments of the European Union. These quantities correspond to between 4 and 23 times the estimated amount that is released in the oceans.

Plastic pollution is considered to be a major factor responsible for the global decline in biodiversity. This is a threat to the soil system's functioning and has been documented in ecosystems worldwide (Barnes et al. 2009; Qi et al. 2020). Therefore, the abundance and persistence of plastics and microplastics (MPs) is a severe environmental risk (Scheurer and Bigalke 2018; Steffen et al. 2015).

The term "MPs" was coined by Thomson et al. in 2004, to refer to microscopic-sized plastics and have often been defined as particles between 5 mm and 100 nm in diameter. MPs are classified according to their origin; in this way, we can distinguish primary and secondary MPs. Primary MPs are those intentionally manufactured microscopic and can

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Table 1 Types of plastics commonly used worldwide (Rocha-Santos and Duarte 2017; PlasticsEurope 2021). *No information

Plastic type	Application	Relative density	Demand (%)
Polypropylene (PP)	Packaging, bottle caps, ropes, carpets, laboratory equipment, drinking straws	0.83–0.85	19.4
Low-density polyethylene (LDPE)	Packaging, general purpose containers, shower curtains, floor tiles.	0.91–0.93	17.4
High-density polyethylene (HDPE)	Milk containers, detergent bottles, tubing	0.94	12.4
Polyvinyl chloride (PVC)	Pipes, window frames, flooring, shower curtains	1.38	10
Polyurethane (PUR)	Building insulation, pillows, and mattresses, insulating foams for fridges	0.871–1.42	7.9
Polyethylene terephthalate (PET)	Soft drink bottles, food packaging, thermal insulation, blister packs	1.38	7.9
Polystyrene (PS)	Packaging foam, disposable cups, food containers, CDs, building materials	1.05	6.2
High impact polystyrene (HIPS)	Electronics, cups in vending machines, refrigerator liners	1.08	*
Polyamides (PA—nylon)	Textiles, toothbrush bristles, fishing lines, automotive	1.13–1.35	*
Acrylonitrile butadiene styrene (ABS)	Musical instruments, printers, drainage pipes, protective equipment	1.06–1.08	*
Polycarbonate (PC)	C.D.s, DVDs, construction materials, electronics, lenses	1.20–1.22	*
Polyester (PES)	Textiles	1.4	*

be found in personal care products like toothpaste, cosmetics, and cleaning products. On the other hand, secondary MPs are originated from the fragmentation of larger plastic products, such as plastic mulch films and household garbage (Duis and Coors 2016; Qi et al. 2020; Rocha-Santos and Duarte 2017; Wang et al. 2019). MPs contain mixtures of chemical additives, fillers, residual monomers, catalysts, and non-intentionally added substances (NIAS). Also, they act as a vector for pathogens and absorb contaminants such as polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), dichlorodiphenyltrichloroethane (DDT), hexachlorocyclohexane (HCH), pharmaceuticals, pesticides, perfluoroalkyl substances (PFAS), and heavy metals. Furthermore, they accumulate in the food web by direct uptake from the soil or by consumption of contaminated soil biota (Besseling et al. 2017; Fendall and Sewell 2009; Hodson et al. 2017; Huerta Lwanga et al. 2017; Rochman et al. 2013; Wang et al. 2019).

MP pollution is listed as one of the top environmental problems by the United Nations Environment Programme (UNEP) and have gained attention due to their adverse effects on the soil, soil biota, and ecosystems in general. These effects are produced due to their small size and ubiquity (Rocha-Santos and Duarte 2017; Scheurer and Bigalke 2018; UNEP 2014). There are numerous sources of MP entry to soils (Fig. 1) and have been detected in industrial areas, agricultural soils, greenhouses, home gardens, coastal soils, and alluvial plains with a wide range of concentrations, which are well summarized in the study of Xu et al. (2020).

It has been estimated that up to 430,000 and 300,000 tons of MPs enter each year to agricultural land in Europe and North America, respectively (Nizzetto et al. 2016). Moreover, China has reported between 50 and 260 kg ha^{-1} of plastic in farmland soils after 30 years of extensive use of agricultural plastic films (Liu et al. 2014). In Australia, concentrations as high as 7% of MPs have been reported in highly contaminated topsoils (Fuller and Gautam 2016).

Determination of MPs in soils

MP pollution has been documented in various environments, and their determination is highly challenging. Therefore, is essential to choose correct methodologies in the stages of sampling, processing, detection, and quantification of MPs (Fig. 2) (Möller et al. 2020; Zhang 2007). The soil is a heterogeneous matrix comprised of minerals with a range of particle sizes, distributions, and organic matter at varying stages of decomposition. Also, the distribution and quantity of MPs can vary considerably. Therefore, the first stage in determining MPs in soils is the sampling, which must be representative and always avoid adding MPs from the sampling or transport materials (IAEA 2004; Möller et al. 2020; Yang et al. 2021).

Sample processing

To date, there is no consensus methodology for soil processing; the analytical methods for MPs research vary among research groups. First, the sample must be dry, and the purpose is to analyze a known quantity of mass to normalize by MP abundance (g, mg, or particles) per kilogram of dry soil (Möller et al. 2020; Yang et al. 2021). Then, the soil must be sieved (5 mm), in order to separate stones, roots, or other more prominent elements. Is recommended to disrupt the soil aggregates and pass them through the sieve to recover MPs from the soil aggregate fractions (Möller et al. 2020; Yang et al. 2021; Zhang and Liu 2018).

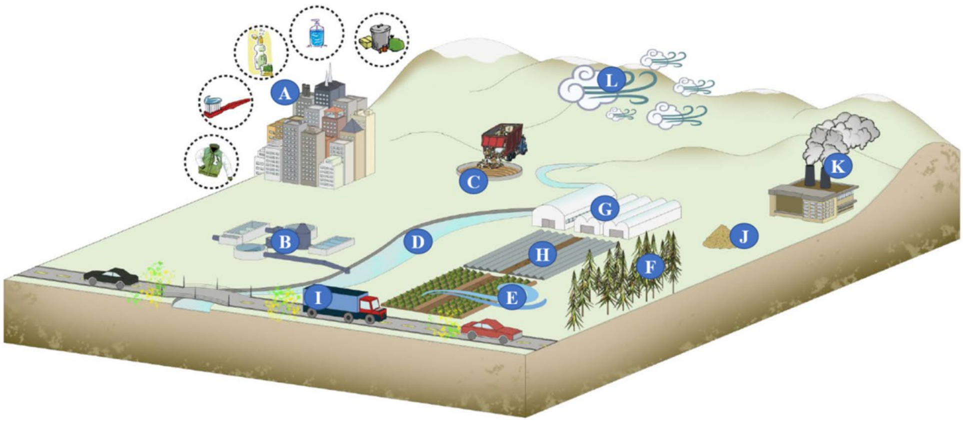
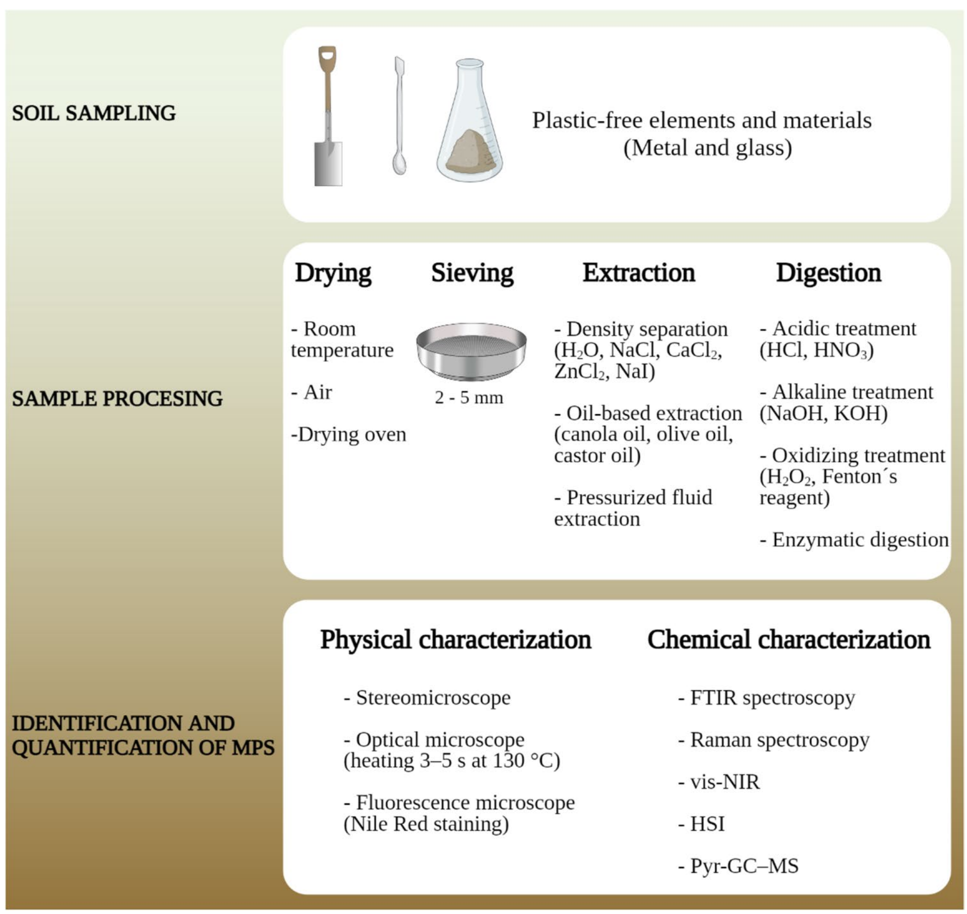


Fig. 1 Main sources of dispersion and entry of MPs to soils. A Use and disposal of household products that have primary and secondary MPs. B Wastewater treatment plants produce sewage sludge which contains MPs, due to their high nutrient load they are used to fertilize agricultural and forestry fields. C Household garbage is disposed of in landfills, where plastics are fragmented forming MPs. D MPs in watercourses reach the adjacent soils by irrigation. E Soil erosion by wind disperses the MPs in the environment. F Forest's foliage traps MPs present in the air, with the fall of leaves or the rain they are deposited on soils. G Plastics fragmentation of greenhouses generates MPs. H Plastic fragmentation of mulch film generates MPs. I Car tires release MPs due to physical wear of the tires, as well as the wear of the brakes. J Organic amendments such as compost, made with household or municipal waste will contribute MPs to the soils. K Industrial zones are highly contaminated with MPs. L Air masses transport MPs, which pollute soils by atmospheric deposition

Fig. 2 Methodologies used in the main stages of MP determination in soils



After sieving, the MPs must be isolated from the soil matrix, and several methods exist (Fig. 2). Density fractionation methods are widely used to extract MPs from complex matrices such as soil and compost. This technique uses solutions with a similar density to plastics, and plastic particles have a lower density than sediments and soil (typically 2.65–2.7 g cm^{-3}) (Li et al. 2020a; Rocha-Santos and Duarte 2017). The best results have been using saturated NaI solution (density 1.8 g cm^{-3}), an expensive reagent. However, it can reach a density that allows the vast majority of MPs to be separated without damaging them (Scheurer and Bigalke 2018; Yang et al. 2021; Zhang and Liu 2018)

Oil-based extraction techniques take advantage of most plastics' lipophilic properties. This methodology consists of mixing the sample with water and some oil, then MPs are separated from the matrix by shaking. Due to its oleophilic surface, MPs remain in the oil, from where they can be filtered and extracted (Crichton et al. 2017; Mani et al. 2019; Möller et al. 2020; Scopetani et al. 2020). The oil-based extracting technique has several advantages; spiked polymers are not chemically altered during treatment, and it requires minimal reagents and essential laboratory equipment. However, filters and MPs need to be carefully rinsed with hexane to remove oil traces and the interaction is not strong enough to extract fluorinated plastics like polytetrafluoroethene (PTFE) from solid samples (Crichton et al. 2017; Mani et al. 2019; Möller et al. 2020; Scopetani et al. 2020).

Fuller and Gautam (2016) developed an extraction method based on pressurized fluid extraction (PFE); this technique uses solvents at subcritical temperature and pressure conditions as an alternative. This method is fully automatized and fast, as it does not require sample purification. However, it is a destructive method, it does not allow to extract MPs larger than 30 µm, and only enable mass-quantitative analysis, not providing information on number, size, and shape of the polymer particles (Bläsing and Amelung 2018; Fuller and Gautam 2016; Möller et al. 2020).

To guarantee a reliable identification and quantification of MPs in soil, they must be purified from any biogenic material present (alive and non-living). Soil organic matter (SOM) should be removed because it interferes with some MP identification techniques such as Fourier transform infrared (FTIR) and Raman spectroscopy (Bläsing and Amelung, 2018). Various methodologies have been developed (Hurley et al. 2018; Scheurer and Bigalke 2018; Yang et al. 2021) (Fig. 2). Hurley et al. (2018) obtained the best results with Fenton's reagent, which is an oxidation reagent that uses H_2O_2 in the presence of a catalyst (Fe^{2+}). This method is performed at room temperature, it is low cost, fast, and effectively destroys highly chlorinated aromatic or inorganic compounds, typically recalcitrant in H_2O_2. Recently, Mbachu et al. (2021) developed a simple protocol for soil samples based on the application of cellulase, hemicellulase, lipase, and protease enzymes that digest the natural components of lignocellulosic biomass. This method proved to be effective reducing approximately 90% of organic matter. However, the authors used plant materials to simulate organic matter; nevertheless, it will be relevant to study this method in different soil samples.

Identification and quantification of MPs

There are several methods to determine MPs in soil samples. From simple visual sorting (MPs are identified by color, shape, or surface texture), to more complex techniques where MPs are determined by their chemical composition (Li et al. 2020a; Zhang et al. 2018). All these techniques are summarized with their advantages and limitations in Table 2. Nowadays, FTIR spectroscopy techniques are the most popular to identify and quantify MPs and are a promising tool for automated MP analysis. This technique provides information regarding MP abundance, shape, size, and precise identification of polymer types by recording the spectral chemical fingerprint of samples and comparing them with spectral databases. It has been used to detect MPs down to 5–10 µm (Chen et al. 2020a; Li et al. 2020a; Möller et al. 2020; Yang et al. 2021). Another common technique is Raman spectroscopy; this technique identifies substances with aromatic bonds, where FTIR has weaker intensity. Therefore, the combination of FTIR and Raman spectroscopy would be optimal for complete and reliable chemical characterization of MPs (Chen et al. 2020a; Käppler et al. 2016).

Another promising tool for MP determination in soil samples is visible-near-infrared spectroscopy (vis-NIR). This technique, which allows to be implemented in portable devices, measures the reflectance spectrum of a sample, that can be used to the identify its chemical composition (Corradini et al. 2019b).

Hyperspectral imaging (HSI) has also been used in soils by Shan et al. in 2018. In this technique, the spectrum, which can be obtained in the vis-NIR or middle infrared (MIR) region, is recorded in each pixel of an image, giving spatial context to chemical information (Möller et al. 2020).

On the other hand, Watteau et al. (2018) used pyrolysis-gas chromatography/mass spectrometry (Pyr-GC–MS); to determine MPs in soil amended with municipal solid waste composts. This technique decomposes the sample in an inert gas at high temperature, then separates using gas chromatography, and finally analyzes by mass spectrometry the composition of the MPs (Junhao et al. 2021).

Accumulation of MPs in agricultural environment

Soils are essential components of terrestrial ecosystems and have intense pressure due to MP contamination. Rillig in 2012 was the first to expose this problem; he documented

Table 2 Most common techniques used to MPs determination in soils

Method	Advantages	Limitations	References
Visual sorting	Fast and inexpensive to implement for soil samples. Can identify, record the physical characteristics and abundance of MPs. Can be optimized using the heating method (130°C for 3–5 s) to differentiate MPs from other particles. Can be optimized using Nile Red to stain the MPs and visualize them with fluorescence microscopy.	Cannot determine the composition of the MPs and is less suitable for particles with a diameter smaller than 50 μm. Visual identification under a light microscope is extremely prone to bias, with error rates ranging from 20 to 70%, because particles of natural origin could look like as synthetic fragments. Not all plastic particles surpass the brightness threshold.	(Erni-Cassola et al. 2017; Piehl et al. 2018; Zhang et al. 2018; Crew et al. 2020; Möller et al. 2020; Vermeiren et al. 2020; Yang et al. 2021)
Fourier transform infrared spectroscopy (FTIR)	It allows identification of polymer types, their abundance, shape, and size. Non-destructive technique. Micro-FTIR spectroscopy allows the identification of particles ranges from 10 to 500 μm.	Cannot analyze samples wet samples. Irregularly shaped MPs will generate light scattering which cause non linearities in the spectrum and differences in the effective path length (can be avoided using ATR).	(Bläsing and Amelung, 2018; Chen et al. 2020a; Möller et al. 2020; Yang et al. 2021)
Near infrared spectroscopy (vis-NIR)	No pre-treatment required; fast and cheap sample analysis.	It is mostly used for pollution hotspots due to its low accuracy (10 g kg^{-1}) and detection limit of around 15 g kg^{-1}. This can be avoided using chemometric analysis, although it requires prior calibration with a reference method. Water interference Black particles often result in unidentifiable spectra, due to the high absorption of infrared radiation.	(Becker et al. 2017; Corradini et al. 2019b; Li et al. 2020a)
Raman spectroscopy	It allows identification of polymer types, their abundance, shape, and size. It allows MP identification down to a pixel resolution of 500 nm and could be improved up to 100 nm with silver colloid for surface-enhanced Raman spectroscopy (SERS). It can analyze wet samples and simultaneously identify fillers or pigments.	Background fluorescence of biological, organic, and inorganic (e.g., clay minerals) contaminations in the polymers may strongly interfere with the real spectra, making them unidentifiable. Poor signal to noise ratio (can be avoided using SERS)	(Käppler et al. 2016; Lv et al. 2020; Möller et al. 2020; Yang et al. 2021)
Hyper spectral imaging (HSI)	No pre-treatment required, have the potential for fast and automated identification analysis in soil samples. Nondestructive identification method, giving information on the spatial position, size, and chemical composition.	Method limited by the particle size. Is only capable of scanning MPs (0.5–5 mm) and can only be applied to the soil surface, whereas polymer particles situated in deeper soil strata are neglected. Highly expensive	(Shan et al. 2018; Möller et al. 2020; Yang et al. 2021)

Table 2 (continued)

Method	Advantages	Limitations	References
Pyrolysis coupled with gas chromatography–mass spectrometry (Pyr-GC–MS.)	Easy to analyze sample with organic plastics additives in one run without the use of solvents and hence background contamination can be avoided.	Per run only one particle with a certain weight can be assessed and its database is available only for selected polymers. The method requires sample preparation and choice of pyrolysis type, so it is difficult for researchers to obtain similar results. No amount or shape of MPs can be identified	(Watteau et al. 2018; Baruah et al. 2021; Junhao et al. 2021)
Time-of-flight secondary ion mass spectrometry (ToF-SIMS)	It can provide information on particle sizes and their distribution.	MP analysis is susceptible to interference from natural organic matter present in the soil.	(Du et al. 2020)

that there is a large accumulation of MPs in the environment mainly due to factors such as its durability and the existing technological limitation to discard or recycle the plastic produced (Barnes et al. 2009). MP's presence in THE soil trigger changes in physical and chemical parameters, which can alter the degradation of organic matter (Liu et al. 2017). Qi et al. (2020) incubated soil with low-density polyethylene (LDPE) MPs for 4 months. They observed an electroconductivity increase, which is relevant because along with pH, and it affects the mobility of nutrients and heavy metal absorption by plants (Marschner and Rengel 2012; Zeng et al. 2011). It has also been shown that MPs can adsorb heavy metals on their surface. Soil incubation experiments with high-density polyethylene (HDPE) MPs demonstrated adsorption of zinc and an increased soil desorption capacity of cadmium (Cd). This suggests that MPs can increase the percentage of exchangeable Cd (Hodson et al. 2017; Wang et al. 2020a).

MP's presence also produces alterations of soil physical parameters; in agricultural soils, it has been shown that 72% of the MPs were associated with soil aggregates (Zhang and Liu 2018). de Souza-Machado et al. (2018) and de Souza-Machado et al. (2019) demonstrated that polyamide (PA), polyester (PES) fibers, and PA microspheres decrease the water-stable aggregates, unlike the HDPE, polyethylene terephthalate (PET), polypropylene (PP), and polystyrene (PS) fragments that did not show statistically significant results. This indicates that the shape of the microplastic, especially the microfibers, is an essential factor influencing the soil aggregates and would decrease the soil's structural stability (Zhang and Liu 2018). In addition, PA, PS, and HDPE increased the evapotranspiration in the soil. Evapotranspiration is relevant for numerous processes like microbial activity, precipitation, and the associated latent heat flux that helps to control surface temperatures (de Souza Machado et al. 2019; Jung et al. 2010).

Plastics are often less dense than many minerals present in soil; therefore, there is an bulk density parameter decreased by the addition of HDPE, PET, PP, and PS MPs at a concentration of 2% w/w (de Souza-Machado et al. 2018, 2019). However, experiments with lower concentrations of PS microfibers (0.3%) did not alter soil bulk density significantly (Zhang et al. 2019c). Due to MP pollution, a decrease in bulk density alters the soil pore structure, which may reduce penetration resistance for plant roots, enhance soil aeration, and influence water transport, increasing evaporation rate. In addition, physical and chemical parameters affect soil water dynamics, decomposition of organic matter, and biogeochemical cycles (de Souza Machado et al. 2019).

Regarding agricultural soil's contamination, the most important MP entry-ways are sludge from sewage treatment and the use of plastic covers. There are also other ways of contamination, such as the use of organic amendments, compost, irrigation, flooding, fragmentation of plastic waste, and atmospheric deposition (Bläsing and Amelung 2018; Ng et al. 2018; Nizzetto et al. 2016; Xu et al. 2020).

Sewage sludge

Agricultural soils are one of the main reservoirs of MPs, and the application of sludge from water treatment plants corresponds to the highest entry of MPs (Nizzetto et al. 2016). Sewage sludge is widely used as fertilizer because its richness in organic and inorganic plant nutrients is economically advantageous to increase yields in agricultural applications (Bläsing and Amelung 2018; Nizzetto et al. 2016). Nizzetto et al. (2016) estimated that through direct application of sewage sludge, between 125 and 850 tons MPs per million inhabitants are added annually to European agricultural soils. In addition, MPs accumulate in soils with successive sludge applications over time, thus increasing their concentration. Moreover, fibers have been found in agricultural soils where sewage sludge was applied 15 years ago, and these fibers were still maintaining their original properties (Corradini et al. 2019a; Zhang and Liu 2018; Zubris and Richards 2005).

Comparing MP concentrations of sewage sludge from different countries, Chile has an average of 34,000 MP particles kg^{-1} (Corradini et al. 2019a), Spain has an average of 50,000 MP particles kg^{-1} (Van den Berg et al. 2020), and Canada has up to 11,469 MPs kg^{-1} (Crossman et al. 2020). MP polymer differs too. For example, Ren et al. (2020) found that 41% of MP particles in sewage sludge from Yangling in China were PVC. In contrast, Crossman et al. (2020) reported mainly PS (44%) in sewage sludge from Ontario, Canada. The differences in concentration and resin are because the regions have different dietary habits, human activities, industrial manufactures, and different wastewater treatment processes (Zhou et al. 2020b).

Fragmentation plastic covers or mulch film

Plastic films are covering around 128,652 km^2 of agricultural land worldwide. The 80% of the mulched surface is found in China with estimated applications of around 700,000 t $year^{-1}$, where the growth rate is approximately 25% per year (Espi et al. 2006; Zhang et al. 2019a). Plastic mulch films are widely used in intensive production systems because they contribute to modify soil temperatures, improve the water content reducing evapotranspiration, increase rooting, control weeds, and significantly increase the productivity of crops. However, plastic polymers efficiently accumulate other harmful pollutants from the surrounding environment during its use, including several persistent, bioaccumulative, and toxic substances like PCBs, dioxins, DDTs, and PAHs (Nizzetto et al. 2016). For example, Ramos et al. (2015) evidenced a concentration of deltamethrin in mulch film (584–2284 μg pesticide g^{-1} plastic) higher than the concentration in soil (13–32 μg pesticide g^{-1} soil). Furthermore, there was a recalcintrant effect on the degradation of deltamethrin adsorbed in PE film. This aspect could be very concerning because PE and PP MPs have been found in agricultural soil where plastic mulch was applied for at least 20 years and with unknow consequences for soil biota and/or biodiversity (Piehl et al. 2018).

Recent studies in soils with plastic covers and mulch films have shown up to 18,760 MPs per kilogram of soil. Moreover, soils with mulch film have more than twice MPs compared to non-mulch since the remaining plastic decomposes into smaller pieces under the action of various physical, chemical, and biological factors (Zhang and Liu 2018; Huang et al. 2020; Zhou et al. 2020a).

Atmospheric deposition

The third most important entryway of MPs to soils is through atmospheric deposition. Atmospheric deposition is understood as the flux of substances from the atmosphere onto the earth's surface. Due to their small size and relatively low density (compared to other natural sediments), MPs are easily transported by air masses. Moreover, MPs can be transported to remote locations as has happened to MPs found in the Alps, the Pyrenees, and even the Arctic (Allen et al. 2019; Bergmann et al. 2019; Dris et al. 2016; Evangeliou et al. 2020; Klein and Fisher 2019). Currently, studies of the presence of MPs in atmospheric deposition have focused mainly on urban centers due to the possible impact on human health (Liu et al. 2019). In monitoring carried out throughout the year in Creteil (France), the atmospheric deposition of MPs ranged from 2 to 355 particles m^{-2} per day, indicating a high annual variability (Dris et al. 2016). In the case of the Hamburg metropolitan area (Germany), an average abundance of 275 particles m^{-2} per day has been reported, similar to the Chinese city of Dongguan, where up to 313 MPs particles m^{-2} per day were found (Cai et al. 2017; Klein and Fisher 2019).

An essential source of MPs into the atmosphere is road traffic. Cars release MPs due to physical wear of tires, as well as the wear of brakes (Kole et al. 2017). Dowarah et al. in 2020 studied the abundance of MPs in road dust in 16 sites in India, finding 227 particles per 100 g of dust, where most were fibers (92%). An alarming fact about this situation is there is a correlation between changes in the dominant wind direction and the number of MPs measured during the same period (Klein and Fisher 2019). It has also been shown that the wind can erode the soil, such as uncovered agricultural soil, and drag MPs that can be re-suspended to the atmospheric load and be transported to remote sites (Rezaei et al. 2019).

Impact of MPs on soil fauna

Soil fauna is the total population of endopedonic (living inside the soil) and amphihabitant animals (living for a time in the soil and then outside) (Bunnenberg and Taeschner 2000). Studies show that MPs and soil contaminated with MPs negatively affect soil fauna, and the magnitude of its impact depends on several factors such as the species, concentration, size, and polymers present (Huerta Lwanga et al. 2016; Pflugmacher et al. 2020). Additionally, MPs can indirectly affect soil fauna by changing the soil's physicochemical parameters (Kim et al. 2020). Decomposition of SOM is performed in 90% by microorganisms such as bacteria and fungi. Also, decomposition is facilitated by ants, termites, earthworms, and others, which create channels, pores, aggregates, and mounds that influence the gases and water transport (Brussaard 1997; García-Palacios et al. 2013).

According to the body width, soil fauna is classified into three categories. The macrofauna (fauna of size >2 mm in diameter) are recognized as litter transformers by converting organic matter into organic structures (fecal pellets) (Xu et al. 2020). Selonen et al. (2020) studied the effect of soil contaminated with 0.02 to 1.5% w/w of PS microfibers on *Porcellio scaber* and observed that contaminated soil decreases feeding activity and allocates energy resources from proteins and lipids to carbohydrates, suggesting a potential depletion in energy reserves. Prendergast-Miller et al. (2019) also evidenced the effects of PS fibers in *Lombricus terrestris*. Treatments of 1% w/w showed a 1.5-fold lower cast production and a change in stress biomarker genes responses (24.3-fold increase metallothionein expression and a 9.9-fold decline in heat shock protein-70 expression). On the other hand, the significantly higher concentration of LDPE MPs (<150 um, 28% w/w) increased mortality and decreased the growth rate of *L. terrestris* (Huerta Lwanga et al. 2016). Otherwise, Song et al. (2019) demonstrated that PET microfibers can be ingested and depurated throughout the digestive system of terrestrial snails *Achatina fulica*. This behavior caused effects like villi damage, decreased food intake, excretion rate, glutathione peroxidase content, and total antioxidant capacity (T-AOC).

In the case of mesofauna (fauna with a size between 100 μm and 2 mm in diameter), studies have focused on the species *Enchytraeus crypticus* and *Folsomia candida*. Pflugmacher et al. (2020) showed that an increase in the concentration of HDPE MPs of 0 to 8% w/w in soil resulted in an increased *E. crypticus* mortality from 2 to 14%, respectively. Furthermore, when enchytraeids are exposed to soils with different concentrations of MPs, they preferred an environment with lower MP dose or an MP-free environment. MP particles used in this study (4 mm) were too large to be consumed by the oligochaete. It probably changed the soil structure, which resulted in unfavorable conditions for the Enchytraeids (Pflugmacher et al. 2020). Similar behaviors were evidenced in *F. candida*; springtails exhibited avoidance behaviors at 0.5 and 1% of PE MPs (w/w), and the avoidance rate was 59 and 69%, respectively. Other effects in springtails (1% MPs in soil w/w) were a decrease in the reproduction rate (70.2%) and an increase in mortality (26%) compared to the control group (Ju et al. 2019).

Lin et al. (2020) studied a high-dose of MP addition (15 g m^{-2}), finding a decrease of abundance of oribatid mites, dipteran larvae, lepidopteran larvae, and hymenoptera ants. However, Barreto et al. (2020) found no effects on the abundance and species richness of the groups Oribatida, Prostigmata, Astigmata, Mesostigmata, and Collembola in a loamy sand soil with addition of PE and PP MPs (0.4% w/w). These different results can be explain by the use of different MPs and concentrations in both studies.

Regarding microfauna (fauna of size <200 μm in diameter), in vitro experiments using *Caenorhabditis elegans* nematode, show that effects depend on MPs' concentration, size, polymer content, and additives. MPs triggered a decrease in offspring and survival rates and also produce more oxidative stress, intestinal damage, and shorter defecation intervals than the control group (Lei et al. 2018; Schöpfer et al. 2020). Recently Kim et al. (2020) studied the effect of soils contaminated with PS nanospheres or microspheres on *C. elegans*, finding that offspring number significantly decreased at concentrations of 10 mg kg^{-1} of soil, and nematodes were more sensitive to MPs (530 nm) than nanoplastics (42 nm). Moreover, a principal component analysis showed that soil composition and properties like bulk density, cation exchange capacity, clay, and sand content significantly affect the toxicity induced by these 530-nm-sized PS particles (Kim et al. 2020).

Gut microbiota present in soil fauna (springtail *F. candida* and oligochaete *E. crypticus)* has also been studied. Insects exposed to soils with MPs had a different structure of gut microbial community than insects in soils without MPs; gut microbes play a vital role in host reproduction, nutrient supply, and immunity (Ju et al. 2019; Zhu et al. 2018). Exposure to HDPE MPs increased the relative abundance of *Bradyrhizobiaceae*, *Ensifer*, and S*tenotrophomonas*, all associated with N fixation (Ju et al. 2019). It is estimated that biological fixation contributes globally with 180 million metric tons of ammonia per year, and these fixation processes are performed by a great variety of bacteria that have nitrogenases (Tilak et al. 2005).

Effect of MPs on the soil microbiota and nitrogen cycle

MPs are a novel surface and serve as a substrate for microorganism colonization; this ecosystem which in marine environments was called "Plastisphere" (Zettler et al. 2013) is also present in soils. Recently, next-generation sequencing (NGS) analysis of MP surface from soils evidenced different microbial communities, with lower richness and evenness in MPs compared to microbial community of soil (Huang et al. 2019; Yi et al. 2020). Also, there was differences in the microbiome on PET and LDPE, suggesting that chemical properties of MPs play an important role directing the evolution of the soil microbiome (Huang et al. 2019; Ng et al. 2021; Wang et al. 2021).

MPs in soil serve as a "special microbial accumulator" as well, enriching the bacterial groups involved in their own biodegradation (Zhang et al. 2019b). An example of this colonization is the phylum Actinobacteria, which is the most sensitive to MP addition, because it decreases in the soil, but is enriched on the surfaces of PE (Huang et al. 2019; Yi et al. 2020; Wang et al. 2020b). *Actinomycetes* produce extracellular polymers such as dextran, glycogen, levan, and *N*-acetylglucosamine-rich slime polysaccharides, facilitating their attachment to plastic surfaces for subsequent microbial action (Amobonye et al. 2020).

N dynamics in the biosphere include biological processes such as, N fixation, mineralization, nitrification, denitrification, and anaerobic oxidation of ammonium. Its incorporation is essential for soil fertility and, therefore, for plant productivity. Microbial communities play a significant role in these processes, and when soil microbial ecology is disturbed, biological processes such as nutrient cycling will be affected (Cerón and Aristizábal 2012; Rong et al. 2021). Several studies have reported that MP addition to the soil could have an impact on the N cycle at different levels; altering the microbiota and the abundance of genes, and therefore, the enzymes that catalyze the different stages of the N cycle (Fei et al. 2020; Huang et al. 2019; Qi et al. 2020; Rong et al. 2021; Wang et al. 2020b).

There is consensus that phyla *Acidobacteria*, *Bacteroidetes*, *Gemmatimonadetes*, and *Proteobacteria* are significantly more abundant in soils with the addition of PE and PP, and the composition of microbial communities plays a fundamental role in SOM decomposition (Fei et al. 2020; Huang et al. 2019; Rong et al. 2021; Yi et al. 2020; Wang et al. 2020b). In *Proteobacteria*, there are the families *Burkholderiaceae* (which is documented as a N-fixing bacteria), *Pseudomonaceae* (with the ability to promote both nitrification and denitrification), and *Xanthobacteraceae* (with fixing-N capacity). These three families increased their abundance in loamy and sandy soils with the addition of LDPE (1 and 5% w/w), PVC (1 and 5% w/w), and PP (2% w/w) (Fei et al. 2020; Yi et al. 2020; Wiegel 2006). Furthermore, in the study of Qian et al. (2018), the MP addition produced an increase in nitrite-oxidizing bacteria belonging to the Phylum *Nitrospirae*, which also participate in soil nitrification in agricultural ecosystems. On the other hand, the phylum Acidobacteria decreased with the addition of LDPE MPs at both 1% and 5%. Some members of this group have been reported as nitrate reducers (Kielak et al. 2016; Qian et al. 2018). Regarding silty loam soil, exposure of LDPE MPs of 150–200 μm (2 and 7% w/w) affected the soil bacterial diversity and structure. It triggered a shift in the abundance of some bacterial genera involved in soil N-cycling processing. Bacterial diversity significantly increased with 2% of LDPE MPs amendment at day 7 and significantly decreased in soils with 7% w/w of LDPE MP amendment at day 60 (Rong et al. 2021). Besides, a high concentration of LDPE MPs (7% w/w) altered the structure of nitrogen-cycling bacterial community. This increased the proportions of *Mycobacterium*, *Gordonia*, and *Rhodococcus*, but decreased the proportion of *Azoarcus* compared to control (Rong et al. 2021). In general, MPs alter microbial communities of the soil and these changes mainly depend on the polymer's shape, quantity, and composition (de Souza Machado et al. 2018; Xu et al. 2020).

Other ubiquitous microorganisms in which the influence of MPs has also been studied are arbuscular mycorrhizal fungi (AMF). Studies showed that MPs alter symbiosis with roots. MPs of PES and PP increased the root colonization ~8 and ~1.4 times, respectively, but PET reduced root colonization ~50% (De Souza Machado et al. 2019). In addition, next-generation sequencing (NGS) analysis evidenced that depending on the type and dose, MPs also alter the structure and diversity of the AMF community (Wang et al. 2020a). MPs in soil have the potential to alter the role of AMF in the nitrogen cycle, like improving soil structure and nitrogen retention (the global AM fungal N pool may be at least 70% of that in the root pool) (Hodge and Fitter 2010). This role is connected to key ecosystem services important for soil and, eventually, human health (Leifheit et al. 2021). In conclusion, the addition of MPs affects different stages of the nitrogen cycle, as seen in Fig. 3.

Effect of MPs on genes related to the nitrogen cycle

The alteration of bacterial communities due to MP addition changes the abundance of bacterial genes related to the nitrogen cycle. To date, there are few studies on the effect on these genes. Qian et al. in 2018 studied the use of plastic film and its effect on soil communities involved in the nitrogen cycle. They found that the abundance of the *nifH* gene increased by approximately 48%; this gene is used as

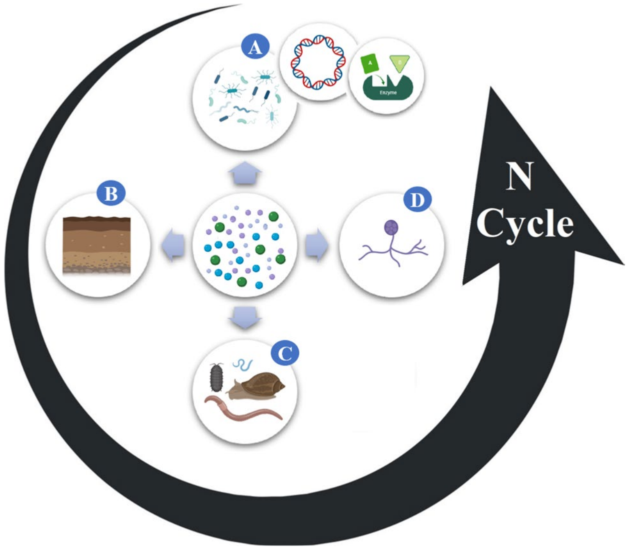


Fig. 3 Factors affected by MPs that alter the soil nitrogen cycle. A Bacterial communities, genes, and enzymes. B Soil physiochemical parameters (soil aggregate fractions, evapotranspiration, soil bulk density, electroconductivity). C Soil fauna (macrofauna, mesofauna, and microfauna). D Arbuscular mycorrhizal fungi

a marker in the nitrogen fixation stage. On the other hand, the abundance of *amoA* (marker gene related to the nitrification stage) decreased by 9.8%. Regarding the *nosZ* and *nirS* genes, these genes increased 80 and 83%, respectively, but the abundance of *nirK* decreased 37% (Qian et al. 2018). The abundance of the *nirS* and *nosZ* genes was positively correlated with the activity of nitrate reductase. However, it showed no correlation with *nirK* gene abundances, indicating that the marker genes of the denitrification stage *nirS*-type and *nosZ*-type contribute more to nitrate reduction and are more active. This suggests that functional communities involved in denitrification respond differently to soils covered with plastic (Iqbal et al. 2020; Qian et al. 2018).

Nitrogen fixation

The study of the MP effects on the abundance of nitrogen cycle marker genes is a recent topic in soils as in other ecosystems as sediments and freshwater systems. However, to date, it has been shown that soils with LDPE MPs at 0.5% (w/w) did not produce significant effects on the abundance of marker genes of the nitrogen fixation stage, such as the *nifD*, *nifH*, and *nifK* genes (Feng et al. 2022). But high doses of LDPE (7% w/w) promoted the abundance of *nifH* gene (Rong et al. 2021). This results can be associated with the increase of certain genera related to nitrogen fixation, such as the genus *Burkholderiaceae* that significantly increased after MP addition (LDPE 1% and 5% w/w and PVC 5% w/w) (Fei et al. 2020). Furthermore, mass balance calculation of total nitrogen at the beginning and at the end of a microcosm experient with freshwater suggested a possible N input caused by biological nitrogen fixation produced by biofilms on PP MPs (Chen et al. 2020b).

Nitrification

Regarding the nitrification stage, studies with soil are based on the abundance of *amoA* gene, which codes for the ammonia monooxygenase enzyme that oxidizes ammonia (NH_3^+) to hydroxylamine (NH_2OH) (Seeley et al. 2020). Rong et al. (2021) showed that addition of LDPE MPs (2% w/w) promoted the bacterial *amoA* gene abundance on day 15, but not the following days. These results showed a positive correlation with nitrifying bacteria *Nitrosopira* ($r = 0.662$, $p =$ 0.007). Moreover, the addition of high-dosage LDPE MPs (7% w/w) promoted the bacterial and archaeal *amoA* genes abundance on day 60. However, the addition of LDPE MPs (2% and 7% w/w) also produced a decrease in the *amoA* gene abundance of archaeas on day 15. This suggests that LDPE MPs can occasionally inhibit the abundance of ammonia-oxidizing archaea (AOA)-*amoA* gene (Rong et al. 2021). The *amoA* gene abundance has also been studied in other environments such as sediments and freshwater. In sediment studies with MPs, it has been shown that abundance of ammonia-oxidizing bacteria (AOB)-*amoA* gene increased from day 7 to day 16, suggesting enhanced nitrification potential with time (Seeley et al. 2020). Furthermore in freshwater systems, the addition of MPs with biofilms further increased nitrification ability in the system (Chen et al. 2020b).

Denitrification

Regarding the denitrification stage, the addition of high doses of LDPE MPs to the soil (7% w/w) promoted the abundance of *nirK* gene on day 90 and *nirS* genes on days 7 and 15. However, the stimulative effects on *nirS* gene were temporary and decreases by day 90. The results about *nirK* gene abundance are positively correlated with the abundances of denitrifying bacterias *Pseudomonas*, *Stenotrophomonas*, *Brachybacterium*, and *Achromobacter* ($r > 0.5$, $p =$ 0.5) (Rong et al. 2021). On the other hand, Ren et al. (2020) studied the effect of the MP addition to a fertilized soil on the emission of greenhouse gases, concluding that LDPE MPs (5% w/w) decreased the emission of N_2O by changing the abundances of microbes related to N_2O emissions. The impact of MPs on the nitrogen cycle has also been studied in sediments, where microcosm experiment with sediment and PVC MPs exhibited a decrease of relative abundance of *nirS* gene and a low potential rate of denitrification too (Seeley et al. 2020). However, in experiments adding MPs to activate sludge and MPs with biofilms at a freshwater systems, the denitrification has been promoted (Chen et al. 2020b; Li et al. 2020b).

Studies conclude that addition of MPs to the soil produces effects on the nitrogen cycle and additional studies are required to measure the real impact on the different stages of the nitrogen cycle.

Effect of MPs on soil enzymatic activity

Bacterial communities are the main enzyme producers in soils, and MPs alter the bacterial structure and affect the soil enzymatic activity (de Souza Machado et al. 2018; Xu et al. 2020; Zhang and Liu 2018). Urease catalyzes the conversion of urea to ammonium that will be oxidize in the nitrification process. The effects on this enzyme depend mainly on the MPs used, concentration, and experiment extension time. In the study of Yi et al. (2020), a higher urease activity was observed in soils treated with MPs of LDPE and PP at 2% (w/w) on day 14, but this activity decreased by 31% on day 29 compared to the control. However, in a different study, urease activity was stimulated in soil with LDPE MPs during the 90 days of the experiment, although this effect was probably due to the lower concentration of MPs that was used (0.0076% w/w) (Huang et al. 2019). Alterations in the community structure, gene expression, and synthesized enzymes result in variations in the nitrogen content in soil. For example, in the study of Liu et al. (2017), when MPs of PVC at 7 and 28% (w/w) were added, the total dissolved nitrogen (TDN) and dissolved organic nitrogen (DON) content increased significantly after days 7 and 14, respectively. However, between days 7 and 30, the addition of MPs did not produce significant changes in NO_3^- and NH_4^+ compared to control. On the other hand, Yan et al. (2020) showed that paddy soil with concentrations of 1% of PVC had a 13% lower NO_3^- content than soil without MPs. These contradictory results are likely because different concentrations of PVC or soils were used, and therefore, there were different physical, chemical, and biological characteristics. It has also been shown that MPs alter the nitrogen cycle directly too, by enriching the soil with nitrogen, particularly when PA MPs are added because their composition is rich in nitrogen (de Souza Machado et al. 2019).

Conclusión, challenges, and future directions

It is important to know the MP load that the soils contain; for this purpose, a representative sampling and suitable processing of soils must be performed. Then, a combination of FTIR and Raman spectroscopy would be optimal for complete and reliable chemical characterization of MPs. MPs are classified as emerging pollutants and like any pollutant, and it alters the ecosystem it enters. It has been shown that the addition of MPs to soils alters biogeochemical cycles, such as the nitrogen cycle, and does it directly by adding MPs that have nitrogen in their chemical structure. However, these alterations can be indirectly too, i.e., by modifying the microbiota/enzymes that catalyze reactions in the different stages of the nitrogen cycle in the soil, by changing the soil fauna that is responsible for facilitating the decomposition of organic matter, and/or by altering the physicochemical parameters of the soil such as evapotranspiration, electrical conductivity, and/or the proportion of microaggregates.

The global effects of MPs on the nitrogen cycle are still unknown, since the studies to date have been performed under soil plant (leguminous) in laboratory conditions. Also, field experiments to study the changes in the nitrogenous species should be performed for better understanding of the N-biological stages and processes afected. This is necessary since the consumption of plastic is increasing, and with it, the accumulation of MPs, as their natural degradation, is limited. These analyses would show strong evidence that could be used to conduct appropriate agronomic practices and public policies to reduce the consumption and disposal of plastics to mitigate their effects. Additionally, understanding the impact of MPs on the nitrogen cycle is important because this cycle is a key predictor of ecological stability and management in the terrestrial ecosystem.

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Declarations

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References

Allen S, Allen D, Phoenix VR, Le Roux G et al (2019) Atmospheric transport and deposition of microplastics in a remote mountain catchment. Nat Geosci 12:339–344. https://doi.org/10.1038/s41561-019-0335-5

Amobonye A, Bhagwat P, Singh S, Pillai S (2020) Plastic biodegradation: frontline microbes and their enzymes. Sci Total Environ 143536https://doi.org/10.1016/j.scitotenv.2020.143536

Andrady AL, Neal MA (2009) Applications and societal benefits of plastics. Philos Trans R Soc B Biol Sci 364:1977–1984. https://doi.org/10.1098/rstb.2008.0304

Barnes DK, Galgani F, Thompson RC, Barlaz M (2009) Accumulation and fragmentation of plastic debris in global environments. Philos Trans R Soc B Biol Sci 364:1985–1998. https://doi.org/10.1098/rstb.2008.0205

Barreto C, Rillig M, Lindo Z (2020) Addition of polyester in soil affects litter decomposition rates but not microarthropod communities. Soil Org 92:109–119. https://doi.org/10.25674/so92iss2pp109

Baruah A, Sharma A, Sharma S, Nagraik R (2021) An insight into different microplastic detection methods. Int J Environ Sci Technol 1-10https://doi.org/10.1007/s13762-021-03384-1

Becker W, Sachsenheimer K, Klemenz M (2017) Detection of black plastics in the middle infrared spectrum (MIR) using photon up-conversion technique for polymer recycling purposes. Polymers 9:435. https://doi.org/10.3390/polym9090435

Bergmann M, Mützel S, Primpke S, Tekman MB et al (2019) White and wonderful? Microplastics prevail in snow from the Alps to the Arctic Sci Adv. 5:eaax1157. https://doi.org/10.1126/sciadv.aax1157

Bläsing M, Amelung W (2018) Plastics in soil: analytical methods and possible sources. Sci Total Environ 612:422–435. https://doi.org/10.1016/j.scitotenv.2017.08.086

Besseling E, Quik JT, Sun M, Koelmans AA (2017) Fate of nano-and microplastic in freshwater systems: a modeling study. Environ Pollut 220:540–548. https://doi.org/10.1016/j.envpol.2016.10.001

Brandon J, Goldstein M, Ohman MD (2016) Long-term aging and degradation of microplastic particles: comparing in situ oceanic and experimental weathering patterns. Mar Pollut Bull 110:299–308. https://doi.org/10.1016/j.marpolbul.2016.06.048

Brussaard L (1997) Biodiversity and ecosystem functioning in soil. Ambio 26(8):563–570

Bunnenberg C, Taeschner M (2000) Soil fauna transport versus radionuclide migration. Radiat Prot Dosim 92:35–38. https://doi.org/10.1093/oxfordjournals.rpd.a033280

Cai L, Wang J, Peng J, Tan Z, Zhan Z, Tan X, Chen Q (2017) Characteristic of microplastics in the atmospheric fallout from Dongguan city, China: preliminary research and first evidence. Environ Sci Pollut Res 24:24928–24935. https://doi.org/10.1007/s11356-017-0116-x

Cerón LE, Aristizábal FA (2012) Nitrogen and phosphorus cycles dynamics in Soils. Rev Colomb Biotecnol 14:285–295

Chen Y, Wen D, Pei J, Fei Y, Ouyang D, Zhang H, Luo Y (2020) Identification and quantification of microplastics using Fourier transform infrared spectroscopy: current status and future prospects. Curr Opin Environ Sci Health 18:14–19. https://doi.org/10.1016/j.coesh.2020.05.004

Chen X, Chen X, Zhao Y, Zhou H et al (2020) Effects of microplastic biofilms on nutrient cycling in simulated freshwater systems. Sci Total Environ 719:137276. https://doi.org/10.1016/j.scitotenv.2020.137276

Corradini F, Meza P, Eguiluz R, Casado F, Huerta-Lwanga E, Geissen V (2019) Evidence of microplastic accumulation in agricultural soils from sewage sludge disposal. Sci Total Environ 671:411–420. https://doi.org/10.1016/j.scitotenv.2019.03.368

Corradini F, Bartholomeus H, Lwanga EH, Gertsen H, Geissen V (2019) Predicting soil microplastic concentration using vis-NIR spectroscopy. Sci Total Environ 650:922–932. https://doi.org/10.1016/j.scitotenv.2018.09.101

Crew A, Gregory-Eaves I, Ricciardi A (2020) Distribution, abundance, and diversity of microplastics in the upper St. Lawrence River. Environ Pollut 260:113994. https://doi.org/10.1016/j.envpol.2020.113994

Crichton EM, Noël M, Gies EA, Ross PS (2017) A novel, density-independent and FTIR-compatible approach for the rapid extraction of microplastics from aquatic sediments. Anal. Methods 9:1419–1428. https://doi.org/10.1039/C6AY02733D

Crossman J, Hurley RR, Futter M, Nizzetto L (2020) Transfer and transport of microplastics from biosolids to agricultural soils and the wider environment. Sci Total Environ. 724:138334. https://doi.org/10.1016/j.scitotenv.2020.138334

de Souza Machado AA, Lau CW, Till J, Kloas W et al (2018) Impacts of microplastics on the soil biophysical environment. Environ Sci Technol 52:9656–9665. https://doi.org/10.1021/acs.est.8b02212

de Souza Machado AA, Lau CW, Kloas W, Bergmann J, Bachelier JB, Faltin E, Becker R, Gorlich AS, Rillig MC (2019) Microplastics can change soil properties and affect plant performance. Environ Sci Technol 53:6044–6052. https://doi.org/10.1021/acs.est.9b01339

Dris R, Gasperi J, Saad M, Mirande C, Tassin B (2016) Synthetic fibers in atmospheric fallout: a source of microplastics in the environment? Mar Pollut Bull 104:290–293. https://doi.org/10.1016/j.marpolbul.2016.01.006

Du C, Wu J, Gong J, Liang H, Li Z (2020) ToF-SIMS characterization of microplastics in soils. Surf Interface Anal 52:293–300. https://doi.org/10.1002/sia.6742

Duis K, Coors A (2016) Microplastics in the aquatic and terrestrial environment: sources (with a specific focus on personal care products), fate and effects. Environ Sci Eur 28:2. https://doi.org/10.1186/s12302-015-0069-y

Erni-Cassola G, Gibson MI, Thompson RC, Christie-Oleza JA (2017) Lost, but found with Nile red: a novel method for detecting and quantifying small microplastics (1 mm to 20 µm) in environmental samples. Environ Sci Technol 51:13641–13648. https://doi.org/10.1021/acs.est.7b04512

Espi E, Salmeron A, Fontecha A, García Y, Real AI (2006) Plastic films for agricultural applications. J Plast Film Sheeting 22:85–102. https://doi.org/10.1177/8756087906064220

Evangeliou N, Grythe H, Klimont Z, Heyes C et al (2020) Atmospheric transport is a major pathway of microplastics to remote regions. Nat Commun 11:1–11. https://doi.org/10.1038/s41467-020-17201-9

Feng Y, Han L, Li D, Sun M et al (2022) Presence of microplastics alone and co-existence with hydrochar unexpectedly mitigate ammonia volatilization from rice paddy soil and affect structure of soil microbiome. J Hazard Mater. 422:126831. https://doi.org/10.1016/j.jhazmat.2021.126831

Fei Y, Huang S, Zhang H, Tong Y et al (2020) Response of soil enzyme activities and bacterial communities to the accumulation of microplastics in an acid cropped soil. Sci Total Environ 707:135634. https://doi.org/10.1016/j.scitotenv.2019.135634

Fendall LS, Sewell MA (2009) Contributing to marine pollution by washing your face: microplastics in facial cleansers. Mar Pollut Bull 58:1225–1228. https://doi.org/10.1016/j.marpolbul.2009.04.025

Fuller S, Gautam A (2016) A procedure for measuring microplastics using pressurized fluid extraction. Environ Sci Technol 50:5774–5780. https://doi.org/10.1021/acs.est.6b00816

Gao Y, Xie Y, Jiang H, Wu B, Niu J (2014) Soil water status and root distribution across the rooting zone in maize with plastic film mulching. Field Crops Res 156:40–47. https://doi.org/10.1016/j.fcr.2013.10.016

García-Palacios P, Maestre FT, Kattge J, Wall DH (2013) Climate and litter quality differently modulate the effects of soil fauna on litter decomposition across biomes. Ecol Lett 16:1045–1053. https://doi.org/10.1111/ele.12137

Hodge A, Fitter AH (2010) Substantial nitrogen acquisition by arbuscular mycorrhizal fungi from organic material has implications for N cycling. Proc Natl Acad Sci 107:13754–13759. https://doi.org/10.1073/pnas.1005874107

Hodson ME, Duffus-Hodson CA, Clark A, Prendergast-Miller MT, Thorpe KL (2017) Plastic bag derived-microplastics as a vector for metal exposure in terrestrial invertebrates. Environ Sci Technol 51:4714–4721. https://doi.org/10.1021/acs.est.7b00635

Horton AA, Walton A, Spurgeon DJ, Lahive E, Svendsen C (2017) Microplastics in freshwater and terrestrial environments: evaluating the current understanding to identify the knowledge gaps and future research priorities. Sci Total Environ 586:127–141. https://doi.org/10.1016/j.scitotenv.2017.01.190

Huang Y, Zhao Y, Wang J, Zhang M, Jia W, Qin X (2019) LDPE microplastic films alter microbial community composition and enzymatic activities in soil. Environ Pollut 254:112983. https://doi.org/10.1016/j.envpol.2019.112983

Huang Y, Liu Q, Jia W, Yan C, Wang J (2020) Agricultural plastic mulching as a source of microplastics in the terrestrial environment. Environ Pollut 260:114096. https://doi.org/10.1016/j.envpol.2020.114096

Huerta Lwanga E, Gertsen H, Gooren H, Peters P, Salánki T, Van Der Ploeg M, Besseling E, Koelmans AA, Geissen V (2016) Microplastics in the terrestrial ecosystem: implications for Lumbricus terrestris (Oligochaeta, Lumbricidae). Environ Sci Technol 50:2685–2691. https://doi.org/10.1021/acs.est.5b05478

Huerta Lwanga E, Mendoza Vega J, Ku Quej V, de los Angeles Chi J et al (2017) Field evidence for transfer of plastic debris along a terrestrial food chain. Sci Rep 7:1–7. https://doi.org/10.1038/s41598-017-14588-2

Hurley RR, Lusher AL, Olsen M, Nizzetto L (2018) Validation of a method for extracting microplastics from complex, organic-rich, environmental matrices. Environ Sci Technol 52:7409–7417. https://doi.org/10.1021/acs.est.8b01517

IAEA (International Atomic Energy Agency) (2004) Soil sampling for environmental contaminants. IAEA-TECDOC-1415, IAEA, Vienna

Iqbal S, Xu J, Allen SD, Khan S, Nadir S, Arif MS, Yasmeen T (2020) Unraveling consequences of soil micro-and nano-plastic pollution for soil-plant system with implications for nitrogen (N) cycling and soil microbial activity. Chemosphere 127578https://doi.org/10.1016/j.chemosphere.2020.127578

Ju H, Zhu D, Qiao M (2019) Effects of polyethylene microplastics on the gut microbial community, reproduction and avoidance behaviors of the soil springtail, Folsomia candida. Environ Pollut 247:890–897. https://doi.org/10.1016/j.envpol.2019.01.097

Jung M, Reichstein M, Ciais P, Seneviratne SI et al (2010) Recent decline in the global land evapotranspiration trend due to limited moisture supply. Nature 467:951–954. https://doi.org/10.1038/nature09396

Junhao C, Xining Z, Xiaodong G, Li Z, Qi H, Siddique KH (2021) Extraction and identification methods of microplastics and nanoplastics in agricultural soil: a review. J Environ Manage 294:112997. https://doi.org/10.1016/j.jenvman.2021.112997

Käppler A, Fischer D, Oberbeckmann S, Schernewski et al (2016) Analysis of environmental microplastics by vibrational microspectroscopy: FTIR, Raman or both? Anal Bioanal Chem 408:8377–8391. https://doi.org/10.1007/s00216-016-9956-3

Kielak AM, Barreto CC, Kowalchuk GA, van Veen JA, Kuramae EE (2016) The ecology of Acidobacteria: moving beyond genes and genomes. Front Microbiol 7:744. https://doi.org/10.3389/fmicb.2016.00744

Kim SW, Kim D, Jeong SW, An YJ (2020) Size-dependent effects of polystyrene plastic particles on the nematode Caenorhabditis elegans as related to soil physicochemical properties. Environ Pollut 258:113740. https://doi.org/10.1016/j.envpol.2019.113740

Klein M, Fischer EK (2019) Microplastic abundance in atmospheric deposition within the Metropolitan area of Hamburg, Germany. Sci Total Environ 685:96–103. https://doi.org/10.1016/j.scitotenv.2019.05.405

Kole P, Löhr AJ, Van Belleghem F, Ragas A (2017) Wear and tear of tyres: a stealthy source of microplastics in the environment. Int J Environ Res Public Health 14:1–4. https://doi.org/10.3390/ijerph14101265

Lei L, Wu S, Lu S, Liu M et al (2018) Microplastic particles cause intestinal damage and other adverse effects in zebrafish Danio rerio and nematode *Caenorhabditis elegans*. Sci Total Environ 619:1–8. https://doi.org/10.1016/j.scitotenv.2017.11.103

Leifheit EF, Lehmann A, Rillig MC (2021) Potential effects of microplastic on arbuscular mycorrhizal fungi. Front Plant Sci 12https://doi.org/10.3389/fpls.2021.626709

Li J, Song Y, Cai Y (2020) Focus topics on microplastics in soil: analytical methods, occurrence, transport, and ecological risks. Environ Pollut 257:113570. https://doi.org/10.1016/j.envpol.2019.113570

Li L, Song K, Yeerken S, Geng S et al (2020) Effect evaluation of microplastics on activated sludge nitrification and denitrification. Sci Total Environ 707:135953. https://doi.org/10.1016/j.scitotenv.2019.135953

Lin D, Yang G, Dou P, Qian, et al (2020) Microplastics negatively affect soil fauna but stimulate microbial activity: insights from a field-based microplastic addition experiment. Proc R Soc B 287:2020. https://doi.org/10.1098/rspb.2020.1268

Liu EK, He WQ, Yane CR (2014) 'White revolution' to 'white pollution'—agricultural plastic film mulch in China. Environ Res Lett 9:091001. https://doi.org/10.1088/1748-9326/9/9/091001

Liu H, Yang X, Liu G et al (2017) Response of soil dissolved organic matter to microplastic addition in Chinese loess soil. Chemosphere 185:907–917. https://doi.org/10.1016/j.chemosphere.2017.07.064

Liu K, Wang X, Wei N, Song Z, Li D (2019) Accurate quantification and transport estimation of suspended atmospheric microplastics in megacities: implications for human health. Environ Int 132:105127. https://doi.org/10.1016/j.envint.2019.105127

Lv L, He L, Jiang S, Chen J et al (2020) In situ surface-enhanced Raman spectroscopy for detecting microplastics and nanoplastics in aquatic environments. Sci Total Environ 728:138449. https://doi.org/10.1016/j.scitotenv.2020.138449

Marschner P, Rengel Z (2012) Chapter 12 - Nutrient Availability in Soils. In: Marschner P (Ed) Marschner's Mineral Nutrition of Higher Plants, 3nd edn. Academic Press, San Diego, pp 315–330. https://doi.org/10.1016/B978-0-12-384905-2.00012-1

Mani T, Frehland S, Kalberer A, Burkhardt-Holm P (2019) Using castor oil to separate microplastics from four different environmental matrices. Anal Methods 11:1788–1794. https://doi.org/10.1039/C8AY02559B

Mbachu O, Jenkins G, Pratt C, Kaparaju P (2021) Enzymatic purification of microplastics in soil. MethodsX 8:101254. https://doi.org/10.1016/j.mex.2021.101254

Möller JN, Löder MG, Laforsch C (2020) Finding microplastics in soils: a review of analytical methods. Environ Sci Technol 54:2078–2090. https://doi.org/10.1021/acs.est.9b04618

Ng EL, Lwanga EH, Eldridge SM, Johnston P et al (2018) An overview of microplastic and nanoplastic pollution in agroecosystems. Sci Total Environ 627:1377–1388. https://doi.org/10.1016/j.scitotenv.2018.01.341

Ng EL, Lin SY, Dungan AM, Colwell JM, Ede S et al (2021) Microplastic pollution alters forest soil microbiome. J Hazard Mater 409:124606. https://doi.org/10.1016/j.jhazmat.2020.124606

Nizzetto L, Futter M, Langaas S (2016) Are agricultural soils dumps for microplastics of urban origin? Environ Sci Technol 50:10777–10779. https://doi.org/10.1021/acs.est.6b04140

Prendergast-Miller MT, Katsiamides A, Abbass M, Sturzenbaum SR, Thorpe KL, Hodson ME (2019) Polyester-derived microfibre impacts on the soil-dwelling earthworm Lumbricus terrestris. Environ Pollut 251:453–459. https://doi.org/10.1016/j.envpol.2019.05.037

Pflugmacher S, Huttunen JH, Wolff MAV, Penttinen OP et al (2020) Enchytraeus crypticus avoid soil spiked with microplastic. Toxics 8:10. https://doi.org/10.3390/toxics8010010

Piehl S, Leibner A, Löder MG, Dris R, Bogner C, Laforsch C (2018) Identification and quantification of macro-and microplastics on an agricultural farmland. Sci Rep 8:1–9. https://doi.org/10.1038/s41598-018-36172-y

PlasticsEurope (2021) Plásticos – Situación en 2020 Un análisis de los datos sobre producción, demanda y residuos de plásticos en Europa. https://www.plasticseurope.org/es/resources/publications/4803-plasticos-situacion-en-2020 (accessed 06 july 2021)

Qi Y, Ossowicki A, Yang X, Lwanga EH et al (2020) Effects of plastic mulch film residues on wheat rhizosphere and soil properties. J Hazard Mater 387:121711. https://doi.org/10.1016/j.jhazmat.2019.121711

Qian H, Zhang M, Liu G, Lu T et al (2018) Effects of soil residual plastic film on soil microbial community structure and fertility. Water Air Soil Pollut 229:261. https://doi.org/10.1007/s11270-018-3916-9

Ramos L, Berenstein G, Hughes EA, Zalts A, Montserrat JM (2015) Polyethylene film incorporation into the horticultural soil of small periurban production units in Argentina. Sci Total Environ 523:74–81. https://doi.org/10.1016/j.scitotenv.2015.03.142

Ren X, Sun Y, Wang Z, Barceló D, Wang Q, Zhang Z, Zhang Y (2020) Abundance and characteristics of microplastic in sewage sludge: a case study of Yangling, Shaanxi province, China. Case Stud Chem Environ Eng 2:100050. https://doi.org/10.1016/j.cscee.2020.100050

Rezaei M, Riksen MJ, Sirjani E, Sameni A, Geissen V (2019) Wind erosion as a driver for transport of light density microplastics. Sci Total Environ 669:273–281. https://doi.org/10.1016/j.scitotenv.2019.02.382

Rillig MC (2012) Microplastic in terrestrial ecosystems and the soil? Environ Sci Technol 46:6453–6454. https://doi.org/10.1021/es302011r

Rillig MC, Ziersch L, Hempel S (2017) Microplastic transport in soil by earthworms. Sci Rep 7:1–6. https://doi.org/10.1038/s41598-017-01594-7

Rillig MC, de Souza Machado AA, Lehmann A, Klümper U (2019) Evolutionary implications of microplastics for soil biota. Environ Chem 16:3–7. https://doi.org/10.1071/EN18118

Rocha-Santos TAP, Duarte AC (2017) Characterization and analysis of Microplastics. Elsevier, Oxford

Rochman CM, Hoh E, Kurobe T, Teh SJ (2013) Ingested plastic transfers hazardous chemicals to fish and induces hepatic stress. Sci Rep 3:1–7. https://doi.org/10.1038/srep03263

Rong L, Zhao L, Zhao L, Cheng Z, Yao Y, Yuan C, Wang L, Sun H (2021) LDPE microplastics affect soil microbial communities and nitrogen cycling. Sci Total Environ 773:145640. https://doi.org/10.1016/j.scitotenv.2021.145640

Scheurer M, Bigalke M (2018) Microplastics in Swiss floodplain soils. Environ Sci Technol 52:3591–3598. https://doi.org/10.1021/acs.est.7b06003

Schöpfer L, Menzel R, Schnepf U, Ruess L et al (2020) Microplastics effects on reproduction and body length of the soil-dwelling nematode Caenorhabditis elegans. Front Environ Sci 8:41. https://doi.org/10.3389/fenvs.2020.00041

Scopetani C, Chelazzi D, Mikola J, Leiniö V et al (2020) Olive oil-based method for the extraction, quantification and identification of microplastics in soil and compost samples. Sci Total Environ 733:139338. https://doi.org/10.1016/j.scitotenv.2020.139338

Seeley ME, Song B, Passie R, Hale RC (2020) Microplastics affect sedimentary microbial communities and nitrogen cycling. Nat Commun 11:1–10. https://doi.org/10.1038/s41467-020-16235-3

Selonen S, Dolar A, Kokalj AJ, Skalar T et al (2020) Exploring the impacts of plastics in soil–the effects of polyester textile fibers on soil invertebrates. Sci Total Environ 700:134451. https://doi.org/10.1016/j.scitotenv.2019.134451

Shan J, Zhao J, Liu L, Zhang Y, Wang X, Wu F (2018) A novel way to rapidly monitor microplastics in soil by hyperspectral imaging technology and chemometrics. Environ Pollut 238:121–129. https://doi.org/10.1016/j.envpol.2018.03.026

Song Y, Cao C, Qiu R, Hu J et al (2019) Uptake and adverse effects of polyethylene terephthalate microplastics fibers on terrestrial snails (Achatina fulica) after soil exposure. Environ Pollut 250:447–455. https://doi.org/10.1016/j.envpol.2019.04.066

Steffen W, Richardson K, Rockström J, Cornell SE et al (2015) Planetary boundaries: guiding human development on a changing planet. Science 347:6223. https://doi.org/10.1126/science.1259855

Steinmetz Z, Wollmann C, Schaefer M, Buchmann C et al (2016) Plastic mulching in agriculture. Trading short-term agronomic benefits for long-term soil degradation? Sci Total Environ 550:690–705. https://doi.org/10.1016/j.scitotenv.2016.01.153

Thompson RC, Swan SH, Moore CJ, Vom Saal FS (2009) Our plastic age. Philos Trans R Soc Lond B Biol Sci 364:1526. https://doi.org/10.1098/rstb.2009.0054

Tilak KVBR, Ranganayaki N, Pal KK, De R et al. (2005) Diversity of plant growth and soil health supporting bacteria. Curr Sci 89(1):136–150. http://www.jstor.org/stable/24110439.

UNEP (2014) ANNUAL REPORT 2014 https://www.unep.org/resources/annual-report/unep-2014-annual-report (accessed 06 july 2021).

van den Berg P, Huerta-Lwanga E, Corradini F, Geissen V (2020) Sewage sludge application as a vehicle for microplastics in eastern Spanish agricultural soils. Environ Pollut 261:114198. https://doi.org/10.1016/j.envpol.2020.114198

Vermeiren P, Muñoz C, Ikejima K (2020) Microplastic identification and quantification from organic rich sediments: a validated laboratory protocol. Environ Pollut 262:114298. https://doi.org/10.1016/j.envpol.2020.114298

Wang J, Liu X, Li Y, Powell T et al (2019) Microplastics as contaminants in the soil environment: a mini-review. Sci Total Environ 691:848–857. https://doi.org/10.1016/j.scitotenv.2019.07.209

Wang F, Zhang X, Zhang S, Zhang S, Sun Y (2020a) Interactions of microplastics and cadmium on plant growth and arbuscular mycorrhizal fungal communities in an agricultural soil. Chemosphere 126791https://doi.org/10.1016/j.chemosphere.2020.126791

Wang J, Huang M, Wang Q, Sun Y, Zhao Y, Huang Y (2020) LDPE microplastics significantly alter the temporal turnover of soil microbial communities. Sci Total Environ 726:138682. https://doi.org/10.1016/j.scitotenv.2020.138682

Wang J, Peng C, Li H, Zhang P, Liu X (2021) The impact of microplastic-microbe interactions on animal health and biogeochemical cycles: a mini-review. Sci Total Environ 145697https://doi.org/10.1016/j.scitotenv.2021.145697

Watteau F, Dignac MF, Bouchard A, Revallier A, Houot S (2018) Microplastic detection in soil amended with municipal solid waste composts as revealed by transmission electronic microscopy and pyrolysis/GC/MS. Front Sustain Food Syst 2:81. https://doi.org/10.3389/fsufs.2018.00081

Wiegel J (2006) The genus Xanthobacter. The prokaryotes 5:290-314. Third Edition. Springer, New York. https://doi.org/10.1007/0-387-30745-1

Xu B, Liu F, Cryder Z, Huang D et al (2020) Microplastics in the soil environment: occurrence, risks, interactions and fate–a review. Crit Rev Environ Sci Technol 50:2175–2222. https://doi.org/10.1080/10643389.2019.1694822
Yan Y, Chen Z, Zhu F, Zhu C et al. (2020) Effect of polyvinyl chloride microplastics on bacterial community and nutrient status in two agricultural soils. Bull Environ Contam Toxicol 1–8https://doi.org/10.1007/s00128-020-02963-1
Yang L, Zhang Y, Kang S, Wang Z, Wu C (2021) Microplastics in soil: a review on methods, occurrence, sources, and potential risk. Sci Total Environ 146546https://doi.org/10.1016/j.scitotenv.2021.146546
Yi M, Zhou S, Zhang L, Ding S (2020) The effects of three different microplastics on enzyme activities and microbial communities in soil. Water Environ Res 93:24–32. https://doi.org/10.1002/wer.1327
Zeng F, Ali S, Zhang H, Ouyang Y, Qiu B, Wu F, Zhang G (2011) The influence of pH and organic matter content in paddy soil on heavy metal availability and their uptake by rice plants. Environ Pollut 159:84–91. https://doi.org/10.1016/j.envpol.2010.09.019
Zettler ER, Mincer TJ, Amaral-Zettler LA (2013) Life in the “plastisphere”: microbial communities on plastic marine debris. Environ Sci Technol 47:7137–7146. https://doi.org/10.1021/es401288x
Zhang C (2007) Fundamentals of environmental sampling and analysis. John Wiley & Sons, Hoboken, New Jersey, U.S.A.
Zhang GS, Liu YF (2018) The distribution of microplastics in soil aggregate fractions in southwestern China. Sci Total Environ 642:12–20. https://doi.org/10.1016/j.scitotenv.2018.06.004
Zhang S, Yang X, Gertsen H, Peters P, Salánki T, Geissen V (2018) A simple method for the extraction and identification of light density microplastics from soil. Sci Total Environ 616:1056–1065. https://doi.org/10.1016/j.scitotenv.2017.10.213
Zhang XY, Yao SY, Tian YQ, Li JS (2019a) Comparison of plastic film, biodegradable paper and bio-based film mulching for summer tomato production: soil properties, plant growth, fruit yield and fruit quality. Sci Hortic 249:38e48. https://doi.org/10.1016/j.scienta.2019.01.037
Zhang M, Zhao Y, Qin X, Jia W, Chai L, Huang M, Huang Y (2019) Microplastics from mulching film is a distinct habitat for bacteria in farmland soil. Sci Total Environ 688:470–478. https://doi.org/10.1016/j.scitotenv.2019.06.108
Zhang GS, Zhang FX, Li XT (2019) Effects of polyester microfibers on soil physical properties: perception from a field and a pot experiment. Sci Total Environ 670:1–7. https://doi.org/10.1016/j.scitotenv.2019.03.149
Zhou B, Wang J, Zhang H, Shi H et al (2020) Microplastics in agricultural soils on the coastal plain of Hangzhou Bay, east China: multiple sources other than plastic mulching film. J Hazard Mater 388:121814. https://doi.org/10.1016/j.jhazmat.2019.121814
Zhou G, Wang Q, Zhang J, Li Q et al (2020) Distribution and characteristics of microplastics in urban waters of seven cities in the Tuojiang River basin, China. Environ Pollut 189:109893. https://doi.org/10.1016/j.envres.2020.109893
Zhu D, Chen QL, An XL, Yang XR et al (2018) Exposure of soil collembolans to microplastics perturbs their gut microbiota and alters their isotopic composition. Soil Biol Biochem 116:302–310. https://doi.org/10.1016/j.soilbio.2017.10.027
Zubris KAV, Richards BK (2005) Synthetic fibers as an indicator of land application of sludge. Environ Pollut 138:201–211. https://doi.org/10.1016/j.envpol.2005.04.013