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Manuscript title: The risk of nano- and micro-plastics contamination on the geoenvironment: an ecotoxicological perspective

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Abstract

Plastic pollution in the terrestrial environment is emerging as another significant manmade threat to ecosystem function and health. Plastic contamination can range from the macro-to-nano scale, and environmental impacts are evident at each level. Although significant knowledge gaps remain regarding the interactions between the natural environment and nano- and micro-plastics (NMPs), there is an increasing body of evidence concerning detrimental effects on a wide range of taxa. The surface properties of NMPs lead to the adsorption of heavy metals, endocrine-disrupting chemicals, antibiotics and other persistent organic pollutants, which, therefore, can result in their co-migration in the terrestrial environment. Although airborne and dietary transmission routes of NMPs have been observed, their effects to human health are still not fully understood, which is of concern to the scientific community. This state-of-the-art review paper firstly examines available evidence for, and knowledge of, various sources of NMP contamination to the terrestrial environment. Attention then focuses on (i) the biological processes from source to soils and plants, (ii) potential impacts of NMPs on soil and subsurface ecosystems, (iii) trophic interactions and function, and (iv) implications for environmental and human health. The paper concludes by identifying knowledge gaps and presents recommendations on prioritised research needs.

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Introduction

All life on Earth, from microorganisms to people, is becoming increasingly exposed to plastic pollution. Society's addiction to plastics, from uses in clothing to food packaging, is driving a spiral of destruction of the planetary ecosystem on a scale comparable to climate change, whilst simultaneously being intricately linked with it. The impact of plastic pollution in oceans, on coastlines, soils and freshwater environments, and ultimately in food, water and air, has become a hot topic worldwide, and increasingly for public-health specialists. Whilst the focus so far has predominantly been on the levels, types and impacts of plastics in marine environments, it is estimated that the levels of plastics entering terrestrial ecosystems are between 4 and 23 times greater than those entering the oceans (Horton *et al*., 2017). The severity has heightened with the emergence of the COVID-19 pandemic, which has increased single-use plastic usage globally for personal protection and hygiene, and intensified the indiscriminate disposal of plastic wastes (Vaverková *et al*., 2021).

Although no internationally agreed definition of their size range exists (Hartmann *et al*., 2019), microplastics (MPs) are generally defined as plastic debris with particle sizes ranging between 100 nm and 5 mm (Galgani *et al*., 2013; Thompson *et al*., 2004), whereas plastic particles < 100 nm are referred to as nanoplastics (NPs) (Alimi *et al*., 2018; Jahnke *et al*., 2017). While there remain significant knowledge gaps around the interactions between these plastics and the natural environment, and their impacts on ecosystem function and services, it is agreed that nano- and micro-plastics (NMPs) have detrimental effects on a broad range of species at different trophic levels, in both aquatic and terrestrial environments, including those

doi: 10.1680/jenge.22.00053

providing essential ecosystem services, such as earthworms (Browne *et al*., 2013; Lwanga *et al*., 2016). These effects, similar to those evident in marine ecosystems, may be either direct or indirect, and are dependent on different characteristics, such as particle-size and surface properties, of the plastic particles themselves (de Souza Machado *et al*., 2018). For example, the surface properties of NMPs (e.g. hydrophobicity, high specific-surface area and surface morphology) lead to adsorption of heavy metals, endocrine-disrupting chemicals, antibiotics and other persistent organic pollutants (Su *et al.*, 2021), and their co-migration in the terrestrial environment (Qi *et al*., 2020a). Migration of NMPs through the soil can occur via numerous mechanisms, including by the ingestion and egestion of soil organisms (e.g. earthworms), translocation by water through burrows and tubes, and tillage or soil wetting/drying cycles. Although understanding of the affects to human health is still in its infancy, the recent confirmation of plastics in human blood (Leslie *et al*., 2022) is of significant concern, and it is important that routes of transfer are identified to mitigate exposure. For example, NMPs ingested by soil organisms (such as earthworms) can enter the human food chain via predators, such as commercial poultry (viz., chicken) (Wang *et al*., 2020). Trophic transfer within the human food chain is also suggested to occur through the consumption of plants, with recent studies suggesting the transfer of NMPs from the soil into the edible parts of plants (viz., roots, leaves and fruit) (Karami *et al.*, 2017; Oliveri Conti *et al.*, 2020). A secondary route of exposure of NMPs from soil to humans through the consumption of plants may occur due to aeolian NMPs becoming deposited on leaves (Liu *et al*., 2020), or via consumption of livestock that have fed on aeolian-NMP-contaminated leaves.

Using the '*Scopus search engine*', the Research article, Review, Conference paper and Book-chapter type papers published in the last 10 years were screened using five different keyword combinations: (i) "MPs in oceans"; (ii) "MPs in marine"; (iii) "MPs in seawater"; (iv) "MPs in terrestrial"; (v) "MPs ecotoxicological effects". When plotted (Figure 1), the synthesis of data shows the dramatic increase in the numbers of publications for each category produced on yearly basis. Also, it can be realised that, compared to the other categories, the research works on MPs considering the marine environment are much greater, implying the current focus of researchers on MPs studies. Therefore, MPs in terrestrial areas also need to be explored, to get a better perspective of their presence in all environments. Furthermore, from the total of 32 publications available for the keywords "MPs ecotoxicological in terrestrial", only 13 are review type papers, with very few of them concentrated on the ecotoxicological effects of MNPs on the terrestrial ecosystem. Hence, it necessitates a comprehensive review to critically analyse the existing literature, as done in the present article, with a leaning on environmental geotechnics aspects.

This state-of-the-art review paper begins with an appraisal of available evidence for, and knowledge of, various sources of NMP contamination to the terrestrial environment. The focus then turns to NMP transport in the terrestrial environment, its impacts on soil properties, soil biology and subsurface ecosystems, and potential implications for human health. The paper concludes by identifying knowledge gaps in the literature, and presents recommendations on prioritised research needs.

NMP sources and composition

doi: 10.1680/jenge.22.00053

The types and composition of NMPs entering the environment can vary significantly between the upstream sources from which they originate. Dominant sources of NMPs to soils include from industry, domestic households, transport, wastewater treatment plants (WWTPs), agriculture, horticulture and landfills (Figure 2). NMPs can be divided into two categories, primary and secondary. Primary NMPs are manufactured, and include, for example, microbeads for use in personal care products (Golwala *et al.* 2021). Secondary NMPs are formed from fragmentation of plastics through environmental, chemical and (or) biological processes, including, for instance, microfibres arising from synthetic textiles. The NMPs' morphotype (fibre, fragment, bead or film) plays an important role in determining their fate and transport in the geoenvironment, as well as their impact on organisms that they interact with (Ren *et al.*, 2021; Shamskhany *et al.*, 2021).

WWTP practices have a significant role in the distribution of NMPs into the environment. Dominant morphotypes of plastic found in municipal wastewater are synthetic fibres, followed by fragments and films, and in some locations microbeads, when they are still permitted for use in personal care and domestic cleaning products (Golwala *et al.*, 2021; Rasmussen *et al.*, 2021; Ruffell *et al.*, 2021). As well as being represented by different morphotypes, the MPs in wastewater consist of a broad range of polymer types; dominant ones being polyester (PEST), polyethylene (PE) and polypropylene (PP) (Ruffell *et al.*, 2021). WWTPs can achieve a high MPs' removal efficiency (80–99%), although much is physical removal, such that a large percentage of them end up in the sludge residue (Carr *et al.* 2016; Rasmussen *et al.*, 2021). In the study by Rasmussen *et al.* (2021), a mass balance was performed for a large municipal WWTP in

Sweden (receiving 201.2 kg/day of plastic waste). They concluded that most plastics were removed at the screens entering the WWTP (the 20- and 2-mm bar screens respectively removed 38.2% (equivalent to 76.8 kg/day) and 35.2% (equivalent to 70.8 kg/day) of the plastics), with 13.6% (equivalent to 27.3 kg/day) contained in the sludge residue. Only a small proportion (0.7 kg/day) was contained in the WWTP effluent. However, it should be noted that there was a large proportion missing from the mass balance (12.7% or 25.5 kg/day), attributed to sampling and analysis errors and also potential loss in the digesters and activated sludge process. The high removal by the screens emphasises the importance of adequate management of screen waste. In countries where the screened waste is incinerated, the potential for NMP release into the environment is small. However, many countries dispose of screened waste in landfills, which may have a much bigger impact on the environment through transport into groundwater. The fact that most NMPs end up in sludge or biosolids has large implications in the terrestrial environment, since biosolids are applied to land as soil conditioners and fertilisers. It has been estimated that the quantity of MPs annually applied to farmland in Europe through addition of sludge could be as high as 40,000–50,000 tonnes, and between 63,000 and 430,000 tonnes in North America (Nizzetto *et al.*, 2016). More recent studies confirmed these numbers, with up to 19,000 tonnes of MPs annually applied to agricultural soils in Australia (Ng *et al*., 2018).

There is a lack of research on the extent of NMP release through on-site wastewater treatment systems; e.g. septic tank systems and package plants. It is logical that a concentration of NMPs occurs in these systems since they are not designed to remove plastic waste. NMPs end up entering the environment through the disposal or land-application field, or through sludge

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removal. Disposal fields, normally comprising sand or fine gravel, are used to discharge effluent in a controlled manner into the sub-surface, such that any NMPs present will also be discharged into the disposal field. Effluent passes through the disposal field before entering the vadose (unsaturated) zone and then groundwater. The design of disposal fields is such that additional physical removal is achieved before reaching the groundwater. Removal largely depends on size and surface charge of the NMP particles.

More research is needed to ascertain the NMP removal rates through WWTP and on-site wastewater-treatment systems. If the NMPs are removed and retained in the sludge, it normally (re-)enters the municipal wastewater treatment system, where the potential for removal has been explained above.

Another source of NMPs is food production farms, where plastics are widely used, from netting to plastic mulch film. Although designed to withstand environmental conditions through incorporation of additives (such as ultraviolet (UV) stabilizers) in their manufacturing process, plastics degrade over time because of weathering and mechanical wear and tear, forming fragments (i.e. NMPs) that stay in the soil (Kasirajan and Ngouajio, 2012; Steinmetz *et al.*, 2016). In some locations, it is common practice that, rather than being removed after the growing season, horticultural plastics (such as mulch film) are ploughed into the soil. Other sources of plastics in soils are the application of organic and inorganic fertilisers. For instance, many pelletized synthetic fertilisers possess polymer-based microcapsules (Katsumi *et al.*, 2021). After their gradual breakdown, allowing slow release of the fertiliser, the synthetic polymer remains in the soils as NMPs. The application of natural organic fertilisers resulting from the recycling of municipal green-waste (either compost or anaerobic digestate) also represents a significant source of plastic particles to soils because of the high levels of plastics that find their way into the food waste (Golwala *et al.*, 2021; Weithmann *et al.*, 2018). Although the role of agricultural and horticultural practices on the contribution of NMPs in soil have been identified, their levels are not fully understood and further research is needed to establish associated risks.

There is evidence that landfills are defuse sources of NMPs to their immediate environment (Table 1), being identified in the surrounding air, groundwater, surface water and soil (O'Kelly *et al*., 2021). Unmanaged, old or legacy landfills do not include protective liners, covers, and leachate and gas collection systems, such that leachate can migrate into the environment. The leachate contains high levels of primary and secondary MPs due to mechanical fragmentation of larger plastic items, the presence of discarded products (including personal care and industrial cleaning products), as well as NMPs present in landfilled WWTP sludge (Guerranti *et al*., 2019; Sobhani *et al*., 2020). Dominant polymers found in leachate are PE and PP (see Table 1), accounting for almost 99.4% of NMPs present, mainly because: (i) the majority of the municipal plastic waste is derived from either PE or PP (Goli *et al*., 2020; He *et al*., 2019) due to their high usage levels; (ii) since the density of these polymers is <1.0 g/cm^3, they float and get transported by the leachates. The levels of MPs in solid refuse samples have been found to be significantly higher (62,000±23,000 MP particles/kg (Su *et al*., 2019)) than those found in sewage sludge (4,196 and 15,385 MP particles/kg (Mahon *et al*., 2017)). It should be kept in view that the physico-chemical attributes of NMPs generated in municipal solid waste (MSW) facilities would be distinctive, due to the presence of a mixture of both conventional and biodegradable polymers.

Vehicular transport also acts as a significant source of plastics to the environment (Schwarz *et al*., 2019), in the form of brake wear and tyre-wear particles (Evangeliou *et al*., 2020). The wearing process depends on the vehicle and pavement characteristics and type of tyre (Grigoratos and Martini, 2014). Kole *et al*. (2017) reported movement of break- and tyre-wear particles owing to washout and runoff to freshwater and marine ecosystems. The size of these particles can be <10 μm, and thus they can also remain airborne for longer periods (Harrison *et al*., 2012). Apart from these sources, polymerized bitumen is another supply of NMPs (Rødland *et al*., 2022).

Other potential sources of local MP contamination of soils and groundwater are the addition of synthetic polymer-based (usually PP) fibres, as strength enhancement (reinforcement) for ground improvement, and synthetic polymers, including waste-tyre-derived aggregates (TDAs) used, for instance, as partial replacements for natural soils and aggregates in the construction of road embankments and pavement subgrades, and as lightweight backfill to bridge abutments and retaining walls. As well as the MP particles derived from weathering and physical breakdown of the TDA additive *in-situ*, there is the potential toxicity of leachates containing heavy metals and other chemicals common in TDA materials, as additional sources of contamination to the terrestrial environment (Vaverková and O'Kelly, 2022). For instance, Tallec *et al*. (2022) emphasised that the use of rubber-based products (e.g. crumb rubber granulates) can induce "rubber contamination" by releasing micro-rubber and (or) constitutive compounds (added during the tyre manufacturing processes) which leach out by the action of water. From the studies of Šourková *et al*. (2021a, 2021b), it is evident that leachates from waste tyre fractions are phytotoxic to highly phytotoxic for higher plants.

Further geotechnical engineering examples include polystyrene (PS) based lightweight engineering fills (Abbasimaedeh *et al*., 2021; O'Kelly and Soltani, 2022b), or use of dredge sediments from water resources as fill materials (Monkul and Özhan, 2021). Dredged sediments may contain significant amounts of MPs owing to their free dispersion in the aquatic environment (Ji *et al*., 2021). Synthetic polymer-based fibres are also used as secondary additives to increase the ductility of stabilized soils that are mainly used as barriers in landfills or subgrade soils. Expanded polystyrene (EPS), as blocks (i.e. Geofoam), or as myriads of discrete beads mixed with soil *in-situ*, is a preferred lightweight fill for embankment construction over soft soil deposits, as retaining wall backfill and to protect culverts and buried pipelines. At end-of-life for earth structures constructed using soil–EPS beads mixtures or using myriads of discrete plastic fibres mixed randomly with soil *in-situ*, the EPS beads and plastic fibres contained therein are not alienable from the soil; hence they are not recyclable (Abbasimaedeh *et al*., 2021; O'Kelly and Soltani, 2022a, 2022b) and have potential to cause substantial local NMP contamination of soil and groundwater. Based on environmental concerns, and also due to the substantial deterioration in geomechanical behaviour/properties for increasing EPS additive content, Abbasimaedeh *et al*. (2021) went so far as to recommend that geotechnical engineering practice should discontinue the approach of adding particulate EPS beads to soil for producing uncemented lightened fills.

Note that even though NPs and small MPs are widely present in the environment, they are often not detected or accurately quantified in environmental matrices, including soils, because of current methodological and analytical limitations — identified as a major shortcoming for present research efforts (Goli *et al*., 2021; O'Kelly *et al*., 2021).

NMP transport in terrestrial environment

While NMP transport above ground is driven by wind and surface water movement, once entered in the soil matrix, the NMP horizontal and vertical migration are controlled by soil physical properties, soil biota, agronomic practises and hydrological conditions (i.e. rainfall frequency and intensity) (Rillig *et al*., 2017a). Due to the pore sizes of fine-grained soils being smaller than the size of many NMPs, a large proportion of NMPs are retained by the upper soil layer (Ren *et al.*, 2021). However, owing to the absence of UV light and low stable temperatures, once in the subsurface, the NMPs can accumulate and remain unaltered for prolonged periods (Otake *et al*., 1995). Some studies have indicated that soils are not only NMP sinks, but they can also provide a feasible transport entryway to subsurface receptors via advective or colloidal transport. For instance, O'Connor *et al*. (2019) found that MPs can undergo significant vertical migration in sandy soils, with transport distances increasing for reducing MP particle size and (or) under wetting–drying cycles. Similar transport characteristics should be expected in soils with high proportions of macro-pores (e.g. bio-pore-rich loamy soils, organic-rich soils, leptosols and vertisols), for which preferential flow and transport can occur (Bläsing and Amelung, 2018). Vertical transport to deep soils and the vadose zone is further accentuated by bioturbation, in the form of fragmentation and size-selective transport (Huerta Lwanga *et al*., 2017; Rillig *et al*., 2017b; Zhu *et al*., 2018). In

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this context, MPs are prone to degradation in terrestrial environments, which decreases their particle size, increases their specific surface area and oxygen-containing functional groups, and enhances the potential chance of attachment (i.e. via sorption, electrostatic force, etc.) of microorganisms, heavy metals and other pollutants present in soil (Golwala *et al.*, 2021).

Impacts on geoenvironment

Once NMPs arrive on soils, one must consider the role of environmental conditions effecting their fate and potential impacts (see Figure 3). For instance, weathering due to UV radiation and hydrolysis affects the physical characteristics of plastics, including polymer structure changes, alteration of surface texture and promotion of fragmentation. Further, when considering NMP impacts on the terrestrial environment, one cannot only consider the physical NMPs themselves, but must also consider their associated chemicals. That is, in the manufacturing of plastics, various chemical additives, ranging from plasticisers to UV stabilisers, are included to suit the end-product use. This section of the paper focuses on NMP impacts on the soil properties, soil biology and subsurface ecosystems.

Soil properties

The impact of NMPs on the physical structure of the soil is beginning to be realised, especially in studies from China, where, since the late 1970s, there has been an increasingly common practice of plastic film mulching (PFM) for improving cash-crop yields (Huang *et al*., 2020; Ng *et al*., 2018; Qi *et al*., 2020b). Studies have shown that PFM can have physical effects on soil ecosystems, including reduced soil porosity (Koskei *et al*., 2021), changed air circulation and altered microbial communities (Li *et al*., 2014; Muroi *et al*., 2016), increased soil water-retention capacity (de Souza Machado *et al*., 2019), and greater greenhouse gas emissions (Cuello *et al*., 2015). Additionally, Wang *et al*. (2016) found that the practice of PFM significantly reduces soil biomass carbon and nitrogen contents, soil metabolic activity, and microbial function and activity. Weathering and physical breakdown of plastic mulch film results in the formation and accumulation of NMPs within the soil matrix. The NMPs presence has also been shown to have a physical impact on soil mesofauna owing to the reduction in porosity, effectively causing their immobilisation (Kim and An, 2019). Luo *et al*. (2020) reported that physicochemical factors, especially the pH and soil organic carbon, play a role in the attachment of NMPs (0.047-µm-sized PS). Five soils were tested, and a strong positive correlation was seen with soil organic carbon and iron (II) oxide; i.e., the greater the soil organic carbon or Fe_2O_3 content, the higher sorptive capacity of NMPs to the soil. In contrast, the higher the clay content, the lower the sorption of NMPs to the soils. Luo *et al*. (2020) also reported that the sorption capacity of all the test soils decreased as pH increased. They concluded that the main attractive forces attributing to NMP sorption to soils were electrostatic interaction and hydrophobic interaction. A more recent study by Wang *et al.* (2022) agreed with these findings, and added that attachment of MPs was significantly correlated with the soils' zeta potential.

Soil biology

Plastics that enter the environment will inevitably be colonised by microbial communities. In any food web, microbes are keystone organisms, often existing in biofilms to enhance survival and for protection from predation. In forming biofilms, microbes will interact with plastic in any

doi: 10.1680/jenge.22.00053

environment, as a surface to attach to, or, as a supplementary carbon source. Nevertheless, synthetic plastics are recalcitrant, and few studies have demonstrated actual plastic degradation directly arising from microbial action, most of them demonstrating that the additives used in plastic manufacture may be degraded, rather than the plastic polymer itself. Biodegradable plastics, manufactured with aliphatic PESTs (e.g. polyhydroxyalkanoate (PHA) and polylactic acid (PLA)) have been shown to degrade, although they may require specific conditions not present in the soils. Even though there are numerous studies demonstrating microbiomes (biofilms) associated with plastics across aquatic and terrestrial environments, the impact of plastics on these communities is sparsely investigated (Lear *et al*., 2021; Ng *et al*., 2021). A recent study demonstrated that distinct communities colonise MPs in soil (Zhang *et al*., 2019b), but it did not investigate the impact of this on the soil ecosystem. Ng *et al*. (2021) investigated the impact of polyethylene terephthalate (PET) and low-density PE (LDPE) MPs by comparing to a control soil from a forest environment. Shifts were observed in microbial composition, both between plastics and relative to the control soil, indicating that the chemical composition (as different plastics age in the soil) will have an impact on the soil microbiome. In addition, the presence of LDPE MPs resulted in a 7–8 fold increase in CO_2 production compared with the control soil.

A direct impact of colonisation of MPs with biofilm communities is degradation or fragmentation/disintegration. There is demonstrated evidence (predominantly in laboratory setting) that bacterial and fungal species can degrade MPs (Gambarini *et al*., 2021; Lear *et al*., 2021; Ng *et al*., 2018; Wei and Zimmermann, 2017). Another consequence of colonisation is that soil detritivores consume microbial biofilms and, hence, may also incidentally engulf MPs (Guo *et al*., 2020). Some studies have demonstrated that detritivores, such as earthworms, consume colonised organic matter (including MPs) over fresh material (Huerta Lwanga *et al*., 2017; Rillig *et al*., 2017a, 2017b), producing detrimental effects (Lwanga *et al*., 2016). These include intestinal problems, blockages and stopping vital nutrients from being taken in by the animal, or, alternatively, can give the animal a feeling of being full, so that they do not feed and starve as a result. However, detailed studies on the impacts of NMP ingestion by soil biota are limited. A reduction in activity in detritivores will change the soil physical properties, such as nutrient addition and bioturbidation. This could reduce soil structure and bioavailability of nutrients for plant growth. More studies are needed in this area to pinpoint exact impacts across biological function and ecosystem services.

Sub-soil transport

Although soils may be predominantly a sink for plastics, it is inevitable that degradation in solids will lead to NMPs becoming a source of NMPs in groundwater environments. Physical processes, such as described in section 3, can lead to contamination of the vadose zone and ultimately the groundwater environment. Recent studies have begun to illustrate the risk to groundwater from landfill sites and agricultural soils (Lwanga *et al.*, 2022; Manikanda Bharath *et al.*, 2021; Ren *et al.*, 2021; Samandra *et al.*, 2022; Zhang *et al.*, 2022). Once in the groundwater environment, NMPs can be transported in significant quantities in alluvial aquifers (Goeppert and Goldscheider, 2021). There is a sparsity of other studies looking at the transport and fate of NMPs in groundwater (Viaroli *et al.*, 2022). The potential of NMP transport depends on the lithology and geochemical conditions, such as clay and colloidal materials present. Also, environmental factors and soil parameters (such as pH, primary cations and Fe mineral, and organic matter) influence the transport behaviour of MPs in the soil matrix (Ren *et al*., 2021). One of the key issues with the study of NMPs in groundwater is the lack of consistent methods for sampling and quantification (Viaroli *et al.*, 2022). To date, only five articles (Goeppert and Goldscheider, 2021; Johnson *et al*., 2020; Panno *et al*., 2019; Selvam *et al*., 2021; Samandra *et al*., 2022) present sufficient information on the groundwater sampled and the hydrogeological information needed for assessment (Viaroli *et al.*, 2022). The NMP concentrations varied by orders of magnitude in different aquifer materials, but, as expected, they followed a pattern of increasing concentration with large pore size or fractured systems present. Karstic systems are likely to have NMP-size distributions similar to those found in surface waters (Panno *et al.*, 2019). Alluvial aquifers have been shown to transport a wide range of NMP sizes, down to 1-μm size (Goppert and Goldscheider, 2021).

Furthermore, with very little known about the groundwater ecosystems themselves, the impact on the groundwater ecosystems present is unaccounted for, such that understanding the NMP effect on their function and its subsequent impact on the water cycle is currently unknown. The fact that globally around two billion people depend on groundwater resources makes this a critical issue to be urgently addressed.

Ecotoxicity

Studies on the impacts and ecotoxicological effects of NMPs on eukaryotic organisms is more common than for prokaryotes, with studies across springtails, earthworms, nematodes, arthropods, isopods and mites. In a recent investigation on the plastic additive Bisphenol A, Gerhardt (2019) indicated acute and chronic toxicity effects on surface and groundwater crustaceans. Compared with detritivores, higher sensitivities were evident for filter-feeding crustacean, as may be expected for a dissolved toxin. Gerhardt (2019) concluded that the groundwater crustacean isopod *Proasellus slavus* was the most sensitive to both acute and chronic exposure, such that it could be a useful indicator species. Oxidative stress, histopathological changes and reproduction impediment have been indicated in response to MP exposure in the earthworm *Eisenia andrei* (Jiang *et al.*, 2020; Kwak and An, 2021; Lackmann *et al.*, 2022). Studies on predator–prey interactions at the higher levels of the food web are sparser, with most only showing indirect evidence (Helmberger *et al*., 2020).

Until recently, flora had been largely overlooked regarding ecotoxicological effects from plastics, those exceptions concerning the terrestrial environment including the investigations by Allouzi *et al*. (2021), Mateos-Cárdenas *et al*. (2021) and Ng *et al*. (2021). For instance, Mateos-Cárdenas *et al*. (2021) showed that plants can adsorb or internalise NMPs, with the mechanisms suggested to be mostly due to electrostatic forces or entrapment in uneven surfaces on the plant. Studies have shown that NPs can also cross membrane boundaries and enter plant cells, suggesting that a toxicological effect is possible (Azeem *et al*., 2021; Luo *et al.*, 2022; Mateos-Cárdenas *et al*., 2021). It has been noted that, to date, most of the studies on plant interactions and impacts have been performed in controlled laboratory environments and on single species. There is still work that is needed to assess the impact of NMPs in the environment and at an ecosystem level. In the limited studies conducted, to date, NMP effects have been reported on plant germination, elongation growth and biomass. Studies that have investigated the toxic effects of plastic particles on plant photosynthesis have varied in their findings, with some finding no effect on photosynthetic activity in controlled experiments on single species (Dovidat *et al*., 2020; Mateos-Cárdenas *et al*., 2019), while others (Gao *et al*., 2019; Qi *et al*., 2018, 2020b) have found negative effects. This variability points to the sparsity of available data, and that further studies are required on a variety of plants and plastics, investigating pertinent environmental conditions.

NMP impacts on human health

The full extent of potential impacts, particularly on humans, of exposure to NMPs remains unknown. The current agreed routes of exposure of plastic particles for humans (see Figure 4) include (i) inhalation (Amato-Lourenço *et al*., 2020; Enyoh *et al*., 2019), synthetic fibres having been found in lung tissue (Pauly *et al*., 1998), and (ii) ingestion of water and food (both fresh and processed) (Daniel *et al*., 2020; Dessì *et al*., 2021; Kedzierski *et al*., 2020; Senathirajah *et al*., 2021). There is currently no research supporting the ability of NMPs to penetrate the surface of the skin, but it is thought that they may enter the body via sweat glands, skin wounds or hair follicles (Schneider *et al*., 2009).

Atmospheric NMPs have been demonstrated in multiple locations globally, with them found in the air above urban areas, as well as in remote locations. This supports the hypothesis that their global transportation occurs (Zhang *et al*., 2019a), although the process by which aeolian NMPs evolve from soils is not yet clear. It is assumed that NMPs behave similar to nanoparticles, microorganisms and organic material — these can be re-suspended from terrestrial surfaces and transported over long distances (Griffin, 2007; Yang *et al*., 2016). Airborne synthetic fibres have been associated with respiratory diseases in humans (Prata, 2018; Turcotte *et al*., 2013), owing to (i) difficulties associated in clearing them from the respiratory system, (ii) the potential of the plastic to interact with other organic materials, (iii) through the release of chemical contaminants associated with the fibres (Enyoh *et al*., 2019). High levels of plastic fibres present within wastewater effluent (Cao *et al*., 2020; Dyachenko *et al*., 2017) and biosolids (Koutnik *et al*., 2021) applied to land may, therefore, pose a human health risk via inhalation after resuspension into the air, as well as other morphotypes also present (Enyoh *et al*., 2019; Knobloch *et al*., 2021).

It has been estimated that children and adults may ingest as many as 100,000 MP particles each day (Mohamed Nor *et al*., 2021), with other studies estimating an average ingest rate of 0.1–5 g of MPs per week through various exposure pathways (Senathirajah *et al*., 2021). Internalisation of MPs through the dietary pathway has been confirmed by evidence of their presence within human stool and blood samples (Leslie *et al*., 2022; Schwabl *et al*., 2019). Animal studies have demonstrated the translocation of NMPs from the gut (Browne *et al*., 2008; Mattsson *et al*., 2017), although, to date, similar data has not been possible to determine this for humans.

NMPs may be introduced into food during meal preparation (Knobloch *et al*., 2021; Zhang *et al*., 2020), or they may already be present within fresh food items. In addition to the potential health risk posed by NMPs within plant tissues when consumed, the presence of plastics within the soil facilitates the uptake of heavy metals into plants, which may pose a direct human health

doi: 10.1680/jenge.22.00053

risk (Wang *et al*., 2021). However, there are huge uncertainties associated with detecting, identifying and characterising different NMPs (Goli *et al*., 2021) in food.

Although there is still a need for far more research to determine the full impacts of NMPs on human health, there is sufficient evidence available to necessitate a precautionary approach to dealing with NMPs exposure. For instance, Ragusa *et al*. (2021) found plastic particles in all three layers of human placenta; the maternal side, foetal side and the chorioamniotic membrane. A better understanding of the potential damage that NMPs cause to humans will only begin to emerge when studies unravel their complex interactions with human organs.

Direct harmful effects of NMPs may be physical (mechanical) and/or chemical (toxicological), while indirect risks are posed by the presence of microbes that may colonise the plastic surface. The potential human health risks are dependent on multiple factors, including the plastic particles' size, morphology, age, associated chemicals (inherent and acquired) and microbes, and exposure pathway. These factors may work in isolation or in synergy, with some of them determined by the source of the NMPs and the land use of the soil that they have been applied to. For instance, wastewater effluent that may be applied to land contains (i) high levels of very diverse NMPs, (ii) a complex cocktail of organic and inorganic chemicals used in domestic and industrial processes, pharmaceuticals and personal care products (e.g. agrichemicals, antibiotics, biocides, UV sunscreens (Deblonde *et al*., 2011)), and (iii) a wide range of microorganisms, including potential human pathogens (Hansen *et al*., 2018).

NMPs may pose direct physiological and microbial risks, together with a direct or indirect chemical risk. Within the soil matrix, associated chemicals, whether additives or acquired from

doi: 10.1680/jenge.22.00053

the environment, may leach from the plastics, become mobile and be taken up by plants, and therefore pose a potential health risk via ingestion (Al-Lihaibi *et al*., 2019), or their presence may facilitate the uptake of non-NMPs associated contaminants (Wang *et al*., 2021). Many of these chemicals have already been determined to pose toxicological effects in humans and animals. For instance, common additives in plastics (e.g. phthalates, Bisphenol A and Bisphenol S) are considered endocrine disrupting chemicals, having been linked with reproductive and developmental disorders, including breast cancer, blood infection, early onset of puberty and genital defects (Mishra *et al*., 2019; Ribeiro *et al*., 2019). With an increasing body of evidence showing that NMPs themselves are being taken up by plants (Bosker *et al*., 2019; Li *et al*., 2021; Liu *et al*., 2020), soils may present an exposure pathway to humans through diet. In addition to acting as a vector for associated chemicals, once inside the gastrointestinal tract, NMPs may translocate to other organs (Browne *et al*., 2013; Powell *et al*., 2010) causing physiological effects. For example, the accumulation of NMPs in the liver and kidneys causes disturbance of energy and lipid metabolism, and oxidative stress (Deng and Zhang, 2019).

Although not fully understood, initial reports are identifying the enrichment of antimicrobial resistance genes (ARGs) and potential human pathogens on NMPs (Shi *et al*., 2021) present in the leachate of landfills that accept high volumes of municipal biosolid waste. The pathogens that were found in the leachate and MP environment were *Ochrobatrum anthropi*, *Acinetobacter iwoffii*, *A. baumannii*, *Afipia broomeae*, *Pseudomonas aeruginosa*, *Escherichia coli*, *Bacillus anthracis*, *Serratia marcescens* and *Aeromonas hydrophila*. These are opportunistic human pathogens, which are linked to ARGs, and are responsible for nosocomial infection, such as bacteraemia, secondary meningitis, urinary tract infection and pneumonia. The high NMPs' load in wastewater effluent and biosolids, therefore, presents a potential high risk to microbiological human health, should they become associated with either food or water, or through direct contact of the contaminating soils, resulting in accidental ingestion. These changes to microbial communities may also cause significant changes in soil and subsoil ecosystem health.

Concluding remarks and way forward

In this paper, the authors reviewed available evidence for, and knowledge of, sources of plastic contamination to the terrestrial environment, its effects on key soil ecosystem functions and the potential for soil NMPs to enter the food chain through fauna and flora. Through the review, the following research gaps in the literature, and whose investigation will require concerted multidisciplinary effort, are identified:

- Understanding NMP interactions with plants is still lacking, especially at an ecosystem level, with very few studies reporting on multiple species and environmental impacts — those that have been done show varied responses.
- More studies are needed on the interaction of microbial biofilms in the soil environment and the potentials for pathogen survival and transport.
- Studies on the whole soil ecosystem remain somewhat lacking, particularly trophic cascade implications in terrestrial environments.

Although NMPs are widely present in the environment, they are often not detected or accurately quantified in environmental matrices (including soils) due to current methodological

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and analytical limitations. This has been identified as a major shortcoming for present research efforts in quantifying the extent and amount of NMP contamination in soils and groundwater.

There is also an urgent need to develop efficient remediation methods to improve the overall soil health. To date, efforts have concentrated on NMPs reduction at the source, which appears to be the most viable and efficient way to manage the risk, control the effects and limit further spreading of NMPs. Remediation of water and soil from NMP pollution is still at very early stages, with microbial biodegradation and bioremediation of certain plastic pollutants showing promise at the experimental stage.

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Table 1. Concentrations of plastic particles identified in different components of landfills and their surrounding ecosystems

Source/ecosystem component	Location	Size (μm)	Avg. MPs concentration	Salient observations	Reference
Leachate	Shanghai, Wuxi, Suzhou and Changzhou, China	100–5000	0.42–24.58[a]	PE and PP were dominant MPs. 99.36% of MPs were derived from fragmentation of plastic waste buried in landfills. 77.48% of MPs sized between 0.1 and 1.0 mm.	He *et al*. (2019)
	Shanghai, China	20–5000	8±3 (4–13)[a]	Dominant morphotype of MPs were fibres (60%) and film. Average MP concentration in young (<3 yr), medium (~10 yr) and old leachates (>20 yr) were 8, 10 and 4 MP particles/L, respectively. Observed 9 different polymer types present based on functional groups identified using FTIR spectroscopy.	Su *et al*. (2019)
Refuse	Shanghai, China	20–5000	62,000±23,000 (20,000–91,000)[b]	MPs abundance in young (<3 y), medium (~10 y) and old leachates (>20 y) of 83,000±10,000, 68,000±6000 and 36,000±14,000 MP particles/kg, respectively. Observed 15 different types of thermoplastic and thermoset polymers present. Most dominant shape of MPs were fibres (59.82%).	Su *et al*. (2019)
Compost	Paris, France	0.45–5000†	‡	Used pyrolysis-gas chromatography-mass spectroscopy for detection of MPs presence in soils.	Watteau *et al*. (2018)

Groundwater	Chennai, India	0.45–5000	Perungudi site: 33 (7–80)[a] Kodungaiyur site: 12 (3–23)[a]	90% of MPs were derived from buried plastics. Major polymer types in groundwater were PS and PP. Dominant colours of MPs were white (38%), black (27%), red (18%), green (8%), blue (6%).	Manikanda Bharath *et al*. (2021)
Soil	Dhaka, Bangladesh	1–2000	ⱶ	Samples were collected at two different depths: topsoil and 0–20 cm layer.	Afrin *et al*. (2020)
Lichen	Tuscany, Italy	<5000	Close, 79,000 (0–95,000)[b]; intermediate (i.e., 200 m distant), 13,000 (0–15,000)[b]; remote (i.e., 1500 m distant), 7000 (3000–9000)[b]	Influence of landfill on MP concentration in lichen was determined; MP concentration reduced with increasing distance from landfill.	Loppi *et al*. (2021)

Note: FTIR, Fourier-transform infrared; PE, polyethylene; PP, polypropylene; PS, polystyrene. [a] MP particles/L; [b] MP particles/kg dry mass of sample/matrix tested. † Lower and upper boundaries of size range decided based on pore size of filter paper and apperature size of sieve, respectively, used in the experiments. ‡ Quantification cannot be obtained since, in this method, the MPs are degraded at higher temperatures before the detection stage, using mass spectrometry. ⱶ Not mentioned in original paper, as detection technique of FTIR, using KBr pellet method, cannot be used to quantify the MPs

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Figure 1. Number of publications on various MP-related topics published in last 10 years (up to June 2022): (a) "MPs in oceans"; (b) "MPs in marine"; (c) "MPs in seawater"; (d) "MPs in terrestrial"; (e) "MPs ecotoxicological effects"

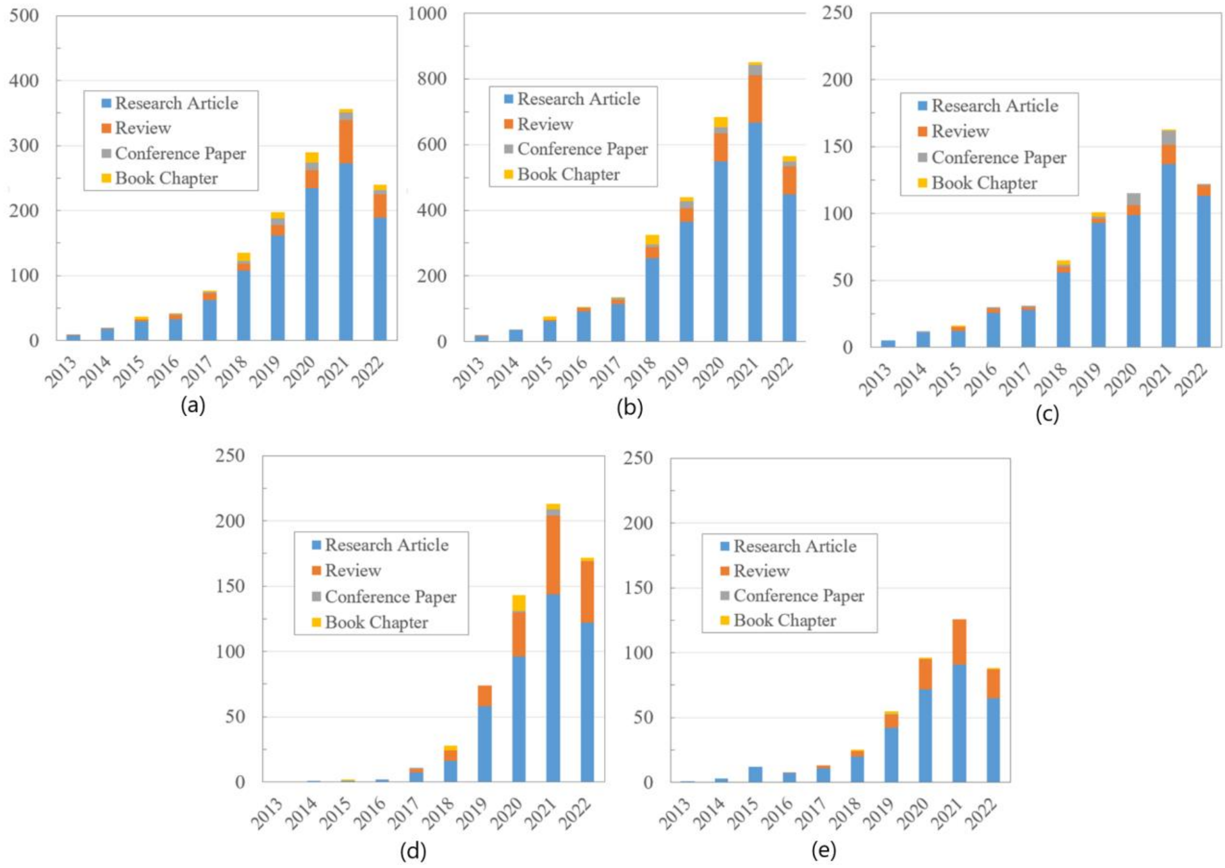

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Figure 2. NMP transport and fate in relation to soil health

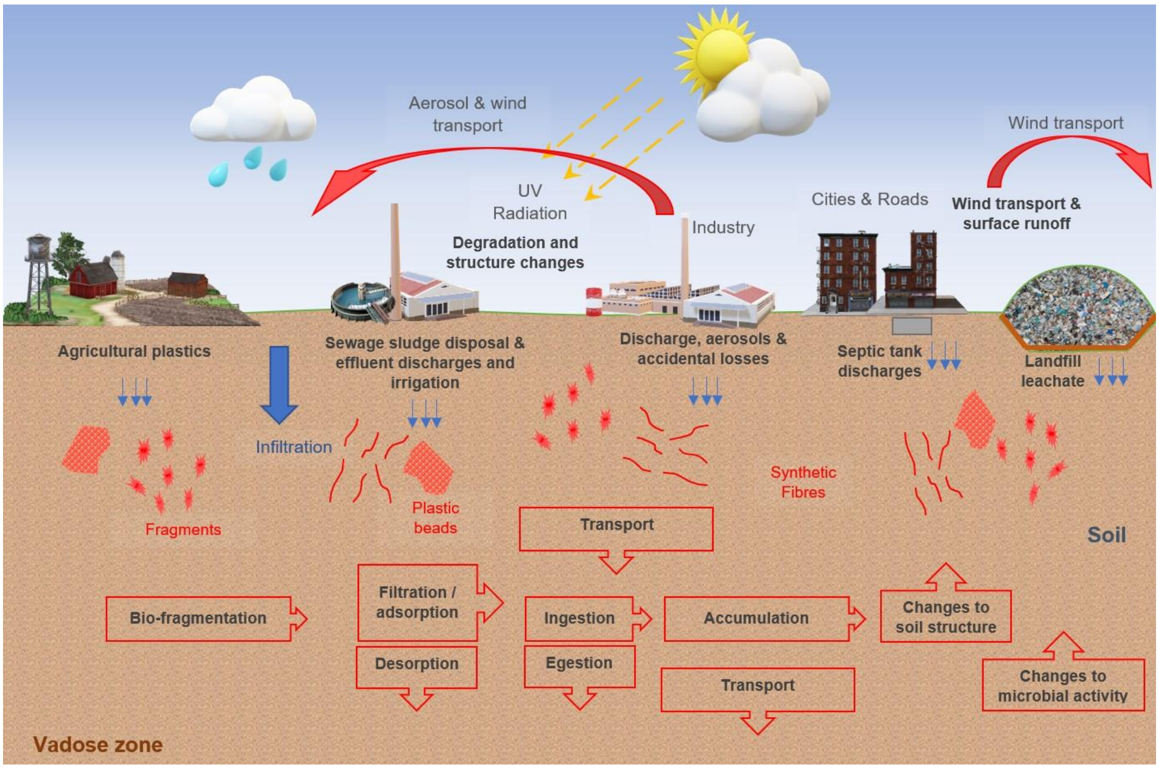

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Figure 3. Schematic representation of NMP effects and impacts on soil health

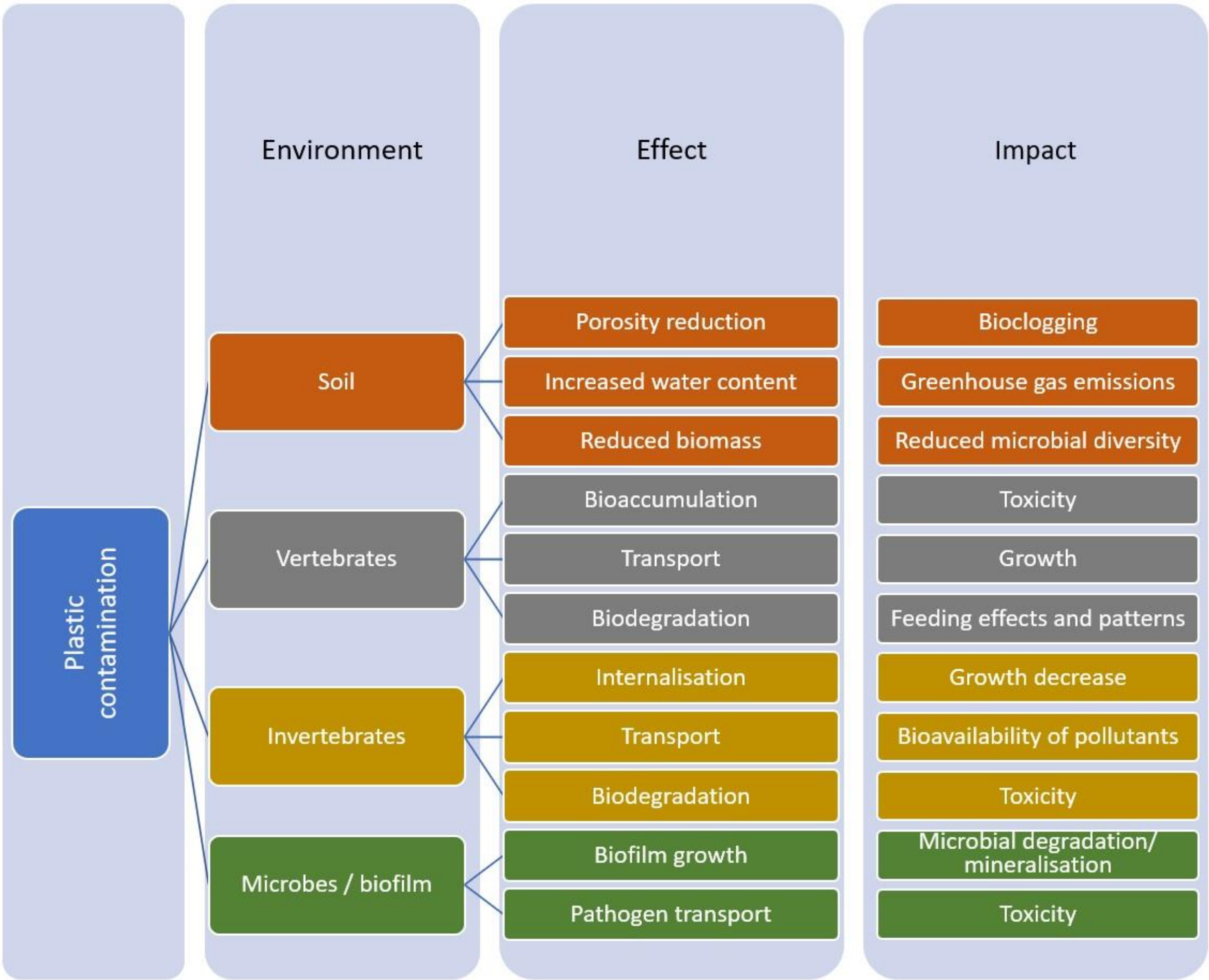

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Figure 4. NMP routes of exposure to humans: the soil–air–soil cycle