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OPINION

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Microplastics as an emerging threat to terrestrial ecosystems

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Abstract
Microplastics (plastics <5 mm, including nanoplastics which are <0.1 μm) originate from the fragmentation of large plastic litter or from direct environmental emission. Their potential impacts in terrestrial ecosystems remain largely unexplored despite numerous reported effects on marine organisms. Most plastics arriving in the oceans were produced, used, and often disposed on land. Hence, it is within terrestrial systems that microplastics might first interact with biota eliciting ecologically relevant impacts. This article introduces the pervasive microplastic contamination as a potential agent of global change in terrestrial systems, highlights the physical and chemical nature of the respective observed effects, and discusses the broad toxicity of nanoplastics derived from plastic breakdown. Making relevant links to the fate of microplastics in aquatic continental systems, we here present new insights into the mechanisms of impacts on terrestrial geochemistry, the biophysical environment, and ecotoxicology. Broad changes in continental environments are possible even in particle-rich habitats such as soils. Furthermore, there is a growing body of evidence indicating that microplastics interact with terrestrial organisms that mediate essential ecosystem services and functions, such as soil dwelling invertebrates, terrestrial fungi, and plant-pollinators. Therefore, research is needed to clarify the terrestrial fate and effects of microplastics. We suggest that due to the widespread presence, environmental persistence, and various interactions with continental biota, microplastic pollution might represent an emerging global change threat to terrestrial ecosystems.

KEYWORDS
environmental health, global change, microplastics, nanoplastics, pollution, soil geochemistry

1 | INTRODUCTION

Physical and chemical anthropogenic influences on the Earth System has achieved a level comparable to that of natural geophysical processes (Steffen, Grinevald, Crutzen, & Mcneill, 2011). Consequently, human activities are among the most significant drivers of ecosystem functions and biodiversity threats (Meybeck, 2004). A characteristic marker of human enterprise is the widespread presence of plastics (Galloway, Cole, & Lewis, 2017), a chemically diverse group of synthetic polymer-based materials used for many purposes in every-day life. All plastics are characterized by high plasticity (i.e. the capacity to change in shape in response to applied forces) at least at one point of their manufacture. More than 80% of the plastics produced are thermoplastics, which are obtained through polymerization of monomers into high-molecular-weight chains known as a thermoplastic polymer (Yang, Yaniger, Jordan, Klein, & Bittner, 2011). This polymer matrix is then subjected to subsequent modification of physical (e.g. melting, extrusion, pelletization) and chemical (mixed with additives such as antioxidants, plasticizers, clarifiers, bisphenol A-based polycarbonate, copolymer, colorants, etc.) properties (Yang et al., 2011). Therefore, plastic products embed within their physical structure a complex chemical composition. The malleability, low costs and durability of plastics have made them extremely versatile and their usage increased about 25-fold over the last 40 years

(Sutherland et al., 2010). Annual plastic production currently exceeds 380 million tons, summing up to 8300 million tons produced until 2015 (Geyer, Jambeck, & Law, 2017).

In 2014 the European demand of plastics was approximately 47.8 million tons, while only 25.8 million tons entered waste stream management (PlasticsEurope, 2015). Global plastic recovery is even lower, and it is estimated that roughly 32% of plastic waste might find its first receptacle in soils or continental aquatic ecosystems (Jambeck et al., 2015). In fact, a study considering production, usage and waste management of plastic materials estimated that approximately 6300 million tons of plastic waste have been generated, of which ~4977 million tons have accumulated in landfills and the natural environment (Geyer et al., 2017).

Environmental plastic litter can undergo aging processes (i.e. degradation and disintegration) resulting from the action of physical, chemical, and biological drivers (Barnes, Galgani, Thompson, & Barlaz, 2009; Whitacre, 2014). Recent reviews claim that most plastics, including many reported as biodegradable, are actually more prone to disintegration than degradation (Whitacre, 2014). Thus, macroscopic plastic pollution generates particles smaller than 5 mm, which are commonly referred to as microplastics (Barnes et al., 2009). Microplastics might also be released directly into the environment through the manufacture of micro-sized particles designed for diverse purposes (see section on microplastic fate). Further degradation and disintegration of microplastics generate particles with dimensions eventually smaller than 0.1 μm, which are referred to as nanoplastics ((CONTAM), 2016).

Microplastic pollution is perhaps one of the most widespread and long-lasting anthropogenic changes to the surface of our planet (Barnes et al., 2009). In fact, microplastic pollution was identified to be among the most relevant topics for biodiversity conservation at global scale (Sutherland et al., 2010). Scientific attention on this topic has tremendously increased in recent years (Figure S1), which resulted in overwhelming evidence of direct and indirect deleterious effects of microplastic pollution on the coastal and oceanic marine biota (Galloway et al., 2017; Lusher, Welden, Sobral, & Cole, 2017). Similar mechanisms are reported to impact freshwater and estuarine environments (Horton, Walton, Spurgeon, Lahive, & Svendsen, 2017). Therefore, microplastics are amongst the contaminants of emerging concern for aquatic systems (Programme, 2014, Syberg et al., 2015).

Most of the plastic litter arriving in the oceans has been produced, used, and often disposed on land (Jambeck et al., 2015; Lebreton et al., 2017; Nizzetto, Bussi, Futter, Butterfield, & Whitehead, 2016), where it undergoes environmental processes affecting its fate (Lebreton et al., 2017) and effects. Therefore, it is likely within terrestrial systems that microplastics first interact with the biota, potentially altering geochemistry and biophysical environment and causing environmental toxicity.

We here present new insights into microplastics as a global change stressor in terrestrial systems, focusing on the geochemistry, the biophysical environment, and ecotoxicology. To achieve this, we first introduce the environmental fate of microplastics in terrestrial habitats including relevant links to continental freshwater systems. Then we highlight the theoretical and observed evidence of the physical and chemical effects of micro-/nanoplastics with ecosystem and organismal functioning. Finally, we conceptualize the potentially broad-spectrum toxicity of nanoplastics on terrestrial organism and its ecological implications. We suggest that microplastic presence is ubiquitous in terrestrial environments, with several potential consequences for biodiversity as well as for human and ecosystem health. This highlights that research is urgently needed to clarify the environmental fate and effects of such small plastic particles in terrestrial systems.

2 | EMERGING CONCERNS ABOUT MICROPLASTIC POLLUTION: DIFFERENCES BETWEEN TERRESTRIAL AND AQUATIC SYSTEMS

Terrestrial systems have received far less scientific attention than their aquatic counterparts (Figure S1). Notwithstanding, microplastic contamination on land might be 4-23-fold larger than in the ocean (Horton, Walton, et al., 2017). Indeed, agricultural soils alone might store more microplastics than oceanic basins (Nizzetto, Futter, & Langaas, 2016). Microplastic threats to aquatic systems are often related to the fact that, for organisms living in a liquid environment, microplastics may represent particulate targets for ingestion (Rehse, Kloas, & Zarfl, 2016), solid surfaces for transport of contaminants (Zhan et al., 2016), or the potential of physical damage (Barnes et al., 2009). Focusing on these factors might have led to an underestimation of microplastic threats to terrestrial species, because neither particulate material nor solid surfaces are rare in continental systems.

More careful analyses of the dangers of microplastic pollution to terrestrial biodiversity are required. The abundance, composition, and physico-chemical surface properties of particulate material follow typical patterns in terrestrial and continental environments (Dubovik et al., 2002), where particles play specific roles in ecosystem functions (Mchale, Newton, & Shirtcliffe, 2005; Saxton & Rawls, 2006). Impacts on such systems cannot be ruled out since microplastics are materials of anthropogenic origin, i.e. with xenobiotic composition and structural properties that are distinguishable from natural matter (see section “Microplastic effects on continental systems”). Therefore, the protection of terrestrial biodiversity from this emerging contaminant requires that further scientific focus is devoted to understanding the effects of microplastics in continental environments.

3 | MICROPLASTIC FATE IN CONTINENTAL SYSTEMS

Diverse sources of microplastics in continental systems (i.e. terrestrial, aquatic, and semiaquatic in land environments) have been

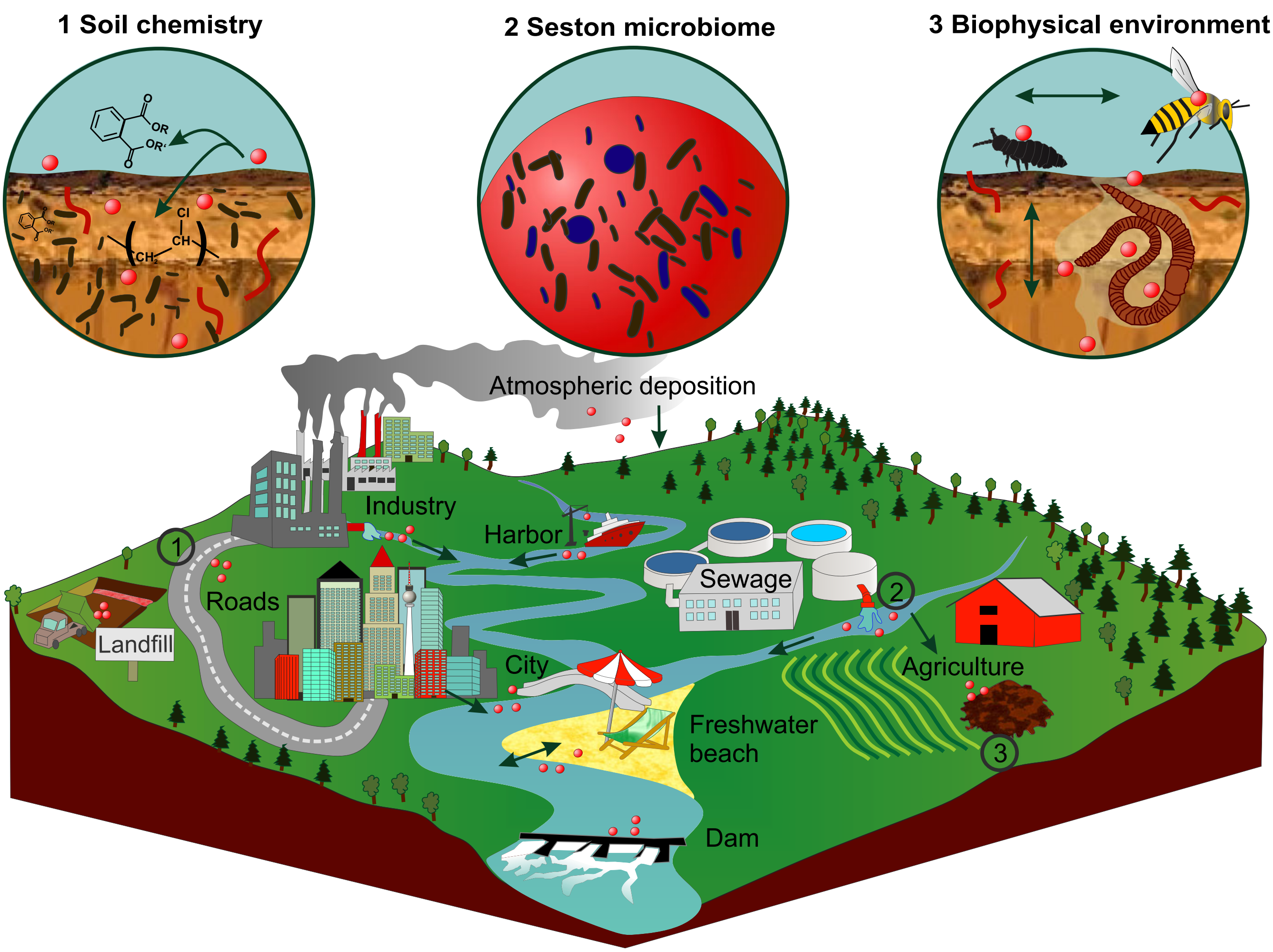

FIGURE 1 Microplastic fate in terrestrial environments and its link to freshwaters. Potential microplastic sources (nonexhaustive) are named and represented by the red-colored objects while areas of microplastic concentration are highlighted with red-filled spheres (industrial zones (Lechner & Ramler, 2015), atmosphere (Dris et al., 2016), sewage (Mahon et al., 2017; Mason et al., 2016), agricultural soils (Huerta Lwanga et al., 2017; Nizzetto, Futter, et al., 2016), freshwater beaches (Ballent et al., 2016), harbors and dams (Zhang et al., 2017), cities and roads (Horton, Svendsen, et al., 2017), and landfills (Rillig, 2012)). The three upper circular panels represent zoom on selected effects on soil chemistry (Fuller & Gautam, 2016), microbiome (Mccormick et al., 2016) and the biophysical environment (Huerta Lwanga et al., 2017; Liebezeit & Liebezeit, 2015; Maass et al., 2017; Rillig, Ziersch, et al., 2017; Zhu et al., 2018) [Colour figure can be viewed at wileyonlinelibrary.com]

reported (Figure 1). Environmental regulations allow arguably permissive levels of microplastics in effluents of industrial plants. For instance, a single production plant in the Austrian portion of the Danube River could emit 94.5 t/year of industrial microplastics considering an effluent production of 100 L/s (Lechner & Ramler, 2015). Consequently, a single production plant might contribute with more than 6% of the total estimated export of the Danube River to the Black Sea, which is about 4.2 t/day (Lechner et al., 2014). Considering the total length of the Danube and its potential for microplastic retention within its sediments, hyporheic zone (surface-groundwater mixing zone), and marginal soils, the total anthropogenic input of plastics to that river might be much higher than 4.2 t/day. Indeed, microplastic concentrations orders of magnitude higher than in marine environments are reported for rivers in industrialized areas (Basel, 2016; Mani, Hauk, Walter, & Burkhardt-Holm, 2015).

Sewage treatment plants might also be significant sources for continental systems because the untreated domestic sewage is rich in fibers from clothing and microplastic beads from personal care products, among others (Mason et al., 2016). Between 80% and 90% of the incoming microplastics are retained in the sludge (Talvitie, Mikola, Setala, Heinonen, & Koistinen, 2017). Even after treatment, sludge might contain significant amounts of microplastics with surface properties varying according to plastic type and to the sewage treatment used (Mahon et al., 2017). The resulting biosolids are often applied as fertilizer to soils (Horton, Walton, et al., 2017; Nizzetto, Futter, et al., 2016) where microplastics may remain much longer than the intended nutrients. Indeed, microplastic fibers have been reported in agricultural fields up to 15 years after sludge was amended, still maintaining their original properties (Zubris & Richards, 2005). Microplastic arriving in agricultural systems might enter the soil by diverse physical, biological, and anthropogenic mechanisms (Rillig, Ingraffia, & De Souza Machado, 2017). Between 1270 and 2130 tons of microplastics per million habitants were estimated to be annually produced in European cities, which correspond to a yearly addition via sludge of 63000-43000 tons of microplastics to European farmlands per year (Nizzetto, Futter, et al., 2016). Other hotspots of microplastic pollution have been reported in the proximity of cities (Ballent, Corcoran, Madden, Helm, & Longstaffe, 2016; Horton, Svendsen, Williams, Spurgeon, & Lahive, 2017), freshwater beaches (Horton, Walton, et al., 2017), and dams (Zhang et al., 2017).

Landfills, urban and industrial centers might contribute to the input of microplastics directly on land through the accidental loss of particles, improper handling of waste and generation of contaminated soils and aerosols (Nizzetto, Bussi, et al., 2016). The latter is of particular concern because atmospheric particles can be quickly transported over considerable distances (Dris, Gasperi, Saad, Mirande, & Tassin, 2016). Microplastic fall-out up to 355 particles m^{-2} day^{-1} was observed for the Parisian metropolitan area, which corresponded to an environmental exposure of 2–10 tons/year of fibers (Dris et al., 2016). Such contamination could be transported far away from the source in that area.

4 | THE NEED FOR ACCURATE AND PRECISE QUANTIFICATION OF MICROPLASTICS

Despite the pervasive presence of microplastics in continental systems, very few (if any) long-term or large-scale monitoring data are publicly available and most terrestrial sources of microplastics are overlooked. This is partially due to the lack of standardized methods for measuring and reporting environmental microplastics. Commonly employed methods for analyzing microplastics often involve some of the following procedures: filtering large volumes of fluids, separation from other particles via flotation, degradation of natural organic matter in samples, and visual selection under optic microscope followed by confirmation methods such as attenuated total reflection Fourier-transform infrared (FT-IR) spectroscopy, focal plane array-based transmission micro-FT-IR imaging or pyrolysis gas chromatography–mass spectrometry ((CONTAM), 2016, Filella, 2015; Mintenig, Int-Veen, Loder, Primpke, & Gerdts, 2017). Each of these steps has particular limitations for terrestrial and other continental environments.

Filtering often results in low retention and consequently underestimation of particles smaller than the mesh opening, e.g. in aquatic systems plankton nets (usually ~300 μm mesh) are used for this purpose (Filella, 2015; Hidalgo-Ruz, Gutow, Thompson, & Thiel, 2012). As a result, environmental data about the distribution of microplastics smaller than 300 μm in freshwaters are rare, and the exact relevance of their occurrence might be overlooked (Filella, 2015; Lebreton et al., 2017). Soil particles are less prone to hydrodynamic selection. Therefore, they might be characterized by a broader distribution of particle size and density. Thus, the flotation method currently used for recovering microplastics from sediments might be less efficient or more laborious for soil samples. Moreover, some of the techniques for chemical degradation of organic matter might also degrade small microplastics ((CONTAM), 2016). Visual detection is subjected to human error and has the obvious limitation of precision (Dekiff, Remy, Klasmeier, & Fries, 2014) and accuracy for small particles, i.e. the human eye equipped with optical microscopes would not resolve particles smaller than 0.42 μm (Filella, 2015). Confirmation methods are equally insensitive and present technical particle size detection limits range from 2 to 100 μm (Filella, 2015). As analytical and statistical procedures are often adapted at various levels in different studies, it is challenging to extrapolate and to compare environmental concentrations of microplastics. Also, detection of nanoplastic contamination is not possible with nearly all commonly used approaches (Bouwmeester, Hollman, & Peters, 2015). Indeed, no scientific report currently exists on nanoplastic concentrations in terrestrial ecosystems.

Another limitation is that the particle size distributions are rarely presented in either aquatic or terrestrial studies. Nevertheless, information about microplastic particle distribution is important as it conveys information about the physical nature of the system, which in turn has relevance for the inference of proximity to source, transport, and fate (Filella, 2015). The lack of such information hampers interpretation of potential fate and effects of microplastic contamination especially in particle-rich environments.

The latest promising approaches seem to address some of the issues mentioned above. Successful recovery and identification of many plastic types in soils was recently achieved by using pressurized fluid extraction (Fuller & Gautam, 2016), which provides precise quantitative information on microplastic mass in various environmental samples. Another mass-related approach (Curie-Point pyrolysis-gas chromatography-mass spectrometry combined with thermochemolysis) was recently proposed (Fischer & Scholz-Bottcher, 2017) which, together with the established FT-IR and Raman methods, could provide qualitative and quantitative information for individual polymers at the micro- and nanogram level. Guidance for the description of particle size distributions already exists (Filella, 2015). A model of the microplastic fate in the catchment of the Thames River (UK) suggested that soils might retain between 16% and 38% of these contaminants depending on soil type and management, plastic size and density, and precipitation patterns (Nizzetto, Bussi, et al., 2016). A similar study included hydrodynamic effects as well as formation of biofilms, microplastic degradation, heteroaggregation and other processes affecting the fate of microplastic beads (100 nm to 10 mm) (Besseling, Quik, Sun, & Koelmans, 2017). Results showed that retention was lowest (18–25%) for particles in the size rage of ~5 μm, which implies that microplastics at submicron size as well as larger micro- and millimeter-sized plastic are preferentially retained within a catchment (Besseling et al., 2017). Additionally, global fate of inland plastics was shown to correlate with waste management, catchment runoff, and population sizes (Lebreton et al., 2017). These model exercises provide insight into the relevance of microplastic fate within terrestrial and aquatic continental systems for a broad environmental contamination assessment. Thus, the standardization of methods for measuring environmental microplastics, reporting of particle size distributions of natural and plastic particles, and development of fate models might constitute significant advances to overcome the lack of quantification of microplastic fate in continental environments.

5 | MICROPLASTIC EFFECTS IN CONTINENTAL ECOSYSTEMS

The composition of plastics and their inherent association with human activities can result in relevant impacts on ecosystem functioning (Figure 1, Circles 1–3 therein). Within a wastewater treatment plant, microplastic surfaces might be enriched with pathogenic and opportunistic organisms (Kirstein et al., 2016). Microplastics not retained by the sewage treatment plants might enter freshwater courses (Talvitie et al., 2017) and subsequently disperse microbes within those systems. Therefore, microplastics released from sewage treatment plants to the continental waters integrate seston with a microbiome distinct (and potentially dangerous (Kirstein et al., 2016)) from the ones on natural particles. In this sense, continental microplastics might play a role as a vector for recently observed disease emergence as in the marine environment (Kirstein et al., 2016). The effects of microplastics on terrestrial microbiomes remain largely unexplored and represent a relevant area for future research.

Impacts also might occur in environments dominated by particulate material. Fuller and Gautam (Fuller & Gautam, 2016) found that topsoils near roads and industrial areas around Sydney (Australia) might contain up to ~7% of microplastics by weight. At this pollution level, the leaching of nonvolatile organochlorines (shown to be about 300-fold more than inorganic chloride in the studied area) from polyvinyl chloride (PVC) and other chlorinated microplastics caused geochemical changes in soils (Fuller & Gautam, 2016). Some authors argue that levels up to 60% of microplastics in weight of top soil of contaminated areas might be environmentally realistic (Huerta Lwanga et al., 2017). Within soils, microplastics might persist for more than 100 years due to low light and oxygen conditions (Horton, Walton, et al., 2017). Thus, microplastics might also interact with soil fauna by changing their biophysical environment, with potential consequences for their fitness and soil function (Huerta Lwanga et al., 2017; Lwanga et al., 2016; Zhu et al., 2018). For instance, springtails and earthworms have been reported to transport microplastics within soil in both horizontal and vertical directions (Maass, Daphi, Lehmann, & Rillig, 2017; Rillig, Ziersch, & Hempel, 2017). In the case of earthworms, microplastic exposure was associated with structural changes in their burrows, an endpoint that is directly linked to soil aggregation and function (Huerta Lwanga et al., 2017). For springtails, the changes in the biophysical environment affected their activity which triggered effects in their gut microbiomes (Zhu et al., 2018). Therefore, even without clear evidence of ingestion, microplastic-exposed springtails had altered gut microflora, affected isotopic signature (δ^{15}N and δ^{13}C) and displayed deleterious effects on growth and reproduction (Zhu et al., 2018).

Other terrestrial organisms might also experience changes in their biophysical environment triggered by microplastics. For instance, commercially available (industrialized and locally produced) honey might contain microplastics (Liebezeit & Liebezeit, 2013). An attempt to track the sources of this contamination suggested that microplastics might be broadly present in inflorescences of diverse species (Liebezeit & Liebezeit, 2015). Therefore, interference of microplastics with plant–pollinators interactions is likely. In this context, when 6 μm polyester beads were introduced onto transmitting tracts of styles of inflorescences of various species the particles were actively translocated by the plants to the ovary (Sanders & Lord, 1989). Thus, plastic beads of compatible pollen size might travel unidirectionally (and sometimes intercellularly) to the ovules as do pollen tubes (Sanders & Lord, 1989). The potential for deleterious environmental impacts of microplastics on important plant and pollinator ecological functions remains to be quantified, however.

6 | MECHANISMS OF POTENTIAL IMPACTS OF MICROPLASTIC ON TERRESTRIAL ORGANISMS

The non-natural and polymer-based structure of plastics together with their composition rich in xenobiotics and poorly soluble biopersistent hydrophobic particles confers to microplastic pollution the fundamental nature of "combined physical and chemical effects" (Figure 2). Previous extrapolations of some mechanisms of physical effects of microplastics from aquatic to terrestrial environments have been discussed (Horton, Walton, et al., 2017; Rillig, 2012). For instance, large plastics limit the exchange of gases and compounds that might affect environmental health (Steinmetz et al., 2016) and cause organism entanglement (Barnes et al., 2009). Smaller particles can be ingested or inhaled causing pseudosatiation and blockage of the digestive tract, or abrasion and irritation of mucosa (Barnes et al., 2009; Rehse et al., 2016). Potential chemical effects are less discussed and might have at least two components as outlined in detail below.

The first component is the leaching of plastic additives, plasticizers, and components of the polymer matrix, which occurs during use, in the environment, or within organisms ((CONTAM), 2016, Whitacre, 2014). This leaching is problematic because many of these additives such as phthalates and bisphenol A are known for their estrogenic activity and further potential endocrine disruption in vertebrates and some invertebrate species (Sohoni & Sumpter, 1998). In fact, plastic additives are now reported amongst the most commonly found anthropogenic substances in environmental samples (Whitacre, 2014). Phthalates, bisphenol, and many other plastic additives have been found at moderately high levels in potentially microplastic-rich sludge from water treatments used for agricultural purposes (Clarke & Smith, 2011). Most plastic materials leach compounds with estrogenic activity (Yang et al., 2011), which is problematic as ambient estrogenicity and demasculinizing effects in laboratory populations and in the wild have been shown to be linked (Manikkam, Tracey,

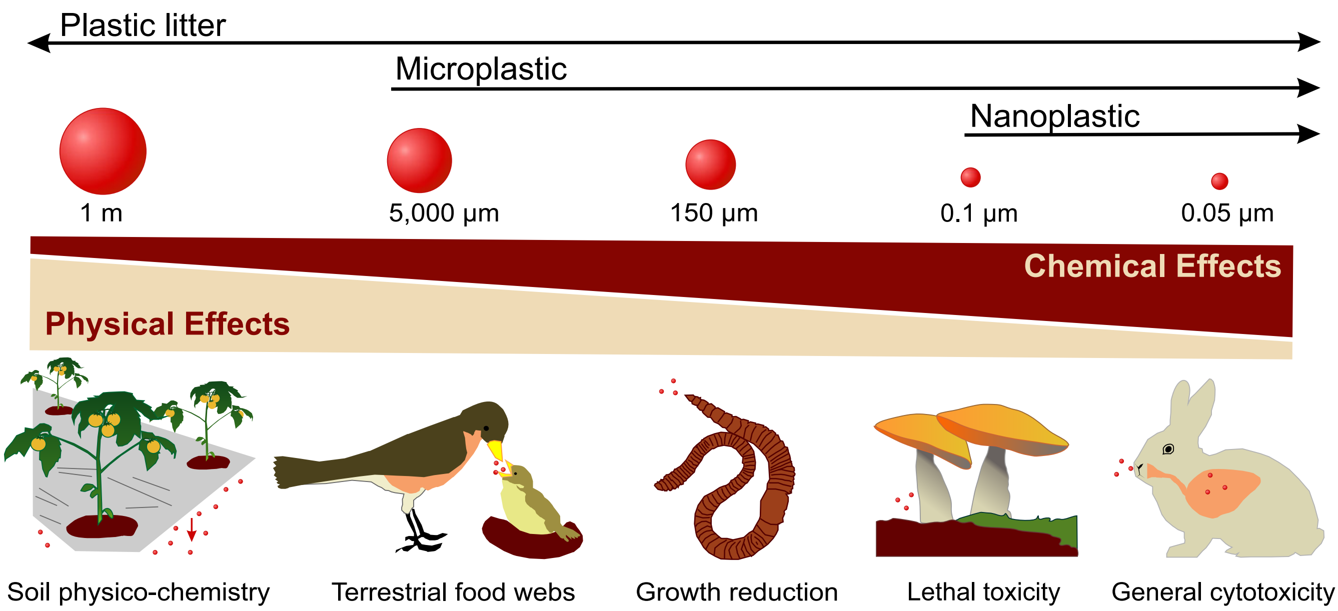

FIGURE 2 : Microplastics as trigger of combined physical or chemical-like effects. Soil biogeochemistry related to agricultural mulching (Steinmetz et al., 2016), ingestion by terrestrial and continental birds (Gil-Delgado et al., 2017; Holland et al., 2016; Zhao et al., 2016), reduction in growth of earthworms (Lwanga et al., 2016), lethal toxicity to fungi (Miyazaki et al., 2014, 2015; Nomura et al., 2016), mammal lung inflammation (Hamoir et al., 2003; Oberdorster, 2000; Schmid & Stoeger, 2016) and broad cytotoxicity (Forte et al., 2016; Kato et al., 2003) of nanoplastics [Colour figure can be viewed at wileyonlinelibrary.com]

Guerrero-Bosagna, & Skinner, 2013; Marty et al., 2017; Tamschick et al., 2016; Ziková et al., 2017). The broad use of plastics and increasing environmental concentrations of endocrine active compounds are of ecological concern: endocrine systems were reasonably well preserved during the evolution of vertebrates, and therefore endocrine disruptors might trigger wide-ranging direct consequences for animal health. It is not appropriate to assume that the many plastic components with known potential for endocrine disruption will have no ecological impacts on exposed biota. When larger plastic particles fragment into smaller pieces there is an exponential increase in the surface/volume ratio. This increases the potential for leaching estrogenically active compounds because many additives are physically, but not chemically, bound to a polymeric structure and hence can almost always leach from the polymer surface (Yang et al., 2011).

The second component of the chemical effect arises from properties of poorly soluble biopersistent small microplastics (<1 μm) that enable them to interact with biological membranes, organelles, and molecules. This can incite many effects commonly triggered by toxic chemicals such as inflammation, changes in membrane permeability, oxidative stress, among others ((Forte et al., 2016; Hamoir et al., 2003; Jeong et al., 2016; Oberdorster, 2000), also see toxicity targets of nanoplastics). The nature of physico-chemical combined effects of microplastics might be the cause of the lack of monotonicity (i.e. lack of constant or increasing effects when increasing exposure concentrations) often found in microplastic dose–response curves ((CONTAM), 2016, Mahler et al., 2012; Rehse et al., 2016). Indeed, the lack of monotonicity in acute toxicity of a particle-solute complex mixture might be associated with the nonmonotonic behavior of particles at nanoscale (Machado, Zarfl, Rehse, & Kloas, 2017). Taken altogether, future studies investigating the effects of microplastic exposure should consider the idiosyncratic interactions of plastic materials (leachable chemical components), their particle size distribution, and the chemical behavior of their surfaces.

The nature of microplastic combined effects can affect soils through physico-chemical changes on soil texture and structure, which is consequential for water cycling and ecosystem functioning in terrestrial systems and diverse plant–soil feedbacks (Bergmann et al., 2016; Zheng, Morris, Lehmann, & Rillig, 2016). In this context, microplastic-driven changes in the hydrologic properties of soils could influence soil microbial biodiversity, with potential impacts on key symbiotic associations in terrestrial ecosystems, such as mycorrhizal (Hallett et al., 2009) and N-fixing (Conrad, 1996) associations. Such potential physical impacts on soil structure and function are of particular concern for the soil microbiome because the mechanistic understanding of biodiversity loss and extinction in those ecosystems are not fully understood (Machado, Valyi, & Rillig, 2017; Veresoglou, Halley, & Rillig, 2015). Moreover, the hydrophobic surfaces of plastics and their eco-corona are known to interact with hydrophobic compounds (Barnes et al., 2009; Galloway et al., 2017; Zhan et al., 2016). Trophic effects and other ecological impacts were observed when the chemicals adsorbed on microplastics were linked to marine intra- or interspecies communication pathways (Galloway et al., 2017). In soils many hydrophobic and amphiphilic compounds also regulate species communication and ecosystem processes. For instance, hydrophobins are amphiphilic proteins ubiquitous in soils that are secreted by fungi (Rillig, 2005). These cysteine-rich polypeptides play important roles in soil hydrophobicity and soil aggregate stability, with direct potential consequences for soil erosion and biogeochemical cycles (Rillig, 2005). It was suggested that microplastics might present distinct sorption properties for soil inorganic elements (Hodson, Duffus-Hodson, Clark, Prendergast-Miller, & Thorpe, 2017), and laboratory results suggest that hydrophobins play a role in the protection against nanoplastic toxicity to filamentous fungi (Nomura et al., 2016). Relevant biogeochemical changes might arise if the hydrophobic surfaces of microplastics interact with hydrophobins or other chemical drivers of soil structure in a manner significantly different from natural soil particles. Thus, further research is required to clarify the extent to which microplastic pollution could affect soil chemistry, texture, structure, and function.

Microplastics might accumulate in terrestrial and continental food webs at levels similar to or higher than in marine counterparts, although conclusive evidence is yet to be found. Zhao et al. found microplastic present in the digestive tract of 94% of dead terrestrial birds with diverse foraging behavior in China (Zhao, Zhu, & Li, 2016). Microplastics in the guts of freshwater continental birds have also been reported (Gil-Delgado et al., 2017; Holland, Mallory, & Shutler, 2016), and microplastic from agricultural activities seem to be an important source (Gil-Delgado et al., 2017). In some cases, microplastic was considerably smaller than the usual food of those birds, which suggests microplastic ingestion to be either accidental or via trophic transfer (Zhao et al., 2016). Moreover, a first quantitative assessment of trophic transfer of microplastic found increasing microplastic concentration in soils (~0.9 particles/g), earthworm casts (~14 particles/g), and chicken feces (~129 particles/g) (Lwanga et al., 2017). The translocation of particles via the intestinal lymphatic cells is the most widely documented portal for entry of particles (0.1 μm to 150 μm, various compositions) into the body, including humans, dogs, rabbits, and small rodents (Hussain, Jaitley, & Florence, 2001). Accidental microplastic ingestion might indeed be of concern to human health. Microplastics have been reported in seafood, salt, sugar, and beers ((CONTAM), 2016, Liebezeit & Liebezeit, 2014). Some of those measurements have been criticized regarding their accuracy and possible laboratory contamination (Lachenmeier, Kocareva, Noack, & Kuballa, 2015) as there is no standardized method to assess microplastic in the food industry ((CONTAM), 2016, Filella, 2015). However, if laboratories following good analytical practice might contaminate food samples with microplastics, much higher exposures are to be expected for humans consuming food from plastic packaging in a (plastic-rich) indoor environment. In fact, microplastic bioaccumulation might be ubiquitous within terrestrial species, even within organisms that do not properly “ingest” their food. For instance, microplastics smaller than 0.5 μm accumulate in yeasts and filamentous fungi (Hamoir et al., 2003; Oberdorster, 2000; Schmid & Stoeger, 2016). This indicates potential microplastic accumulation or magnification along the soil detrital food web, which might be amongst the longest food chains on Earth.

Most large plastic particles present low lethal toxicity. Nevertheless, the exposure, intake and uptake of small microplastics might cause toxicity and act as a new long-term environmental stressor and exert selective pressure on terrestrial organisms. Sublethal negative responses such as growth reduction were observed after the exposure of earthworms to 150 μm microplastics in their food (Lwanga et al., 2016). Such effects might be partially explained by histological damage and changes in the gene expression associated with microplastic exposure (Rodriguez-Seijo et al., 2017). Moreover, microplastics could act as vector of toxic Zn to earthworms under environmental conditions due to a higher adsorption of this metal to high density polyethylene microplastic (Hodson et al., 2017). Lethal effects (100% mortality) were observed after 1 h exposure of yeast cells of *Saccharomyces cerevisiae* to polystyrene nanobeads (50 and 100 nm, 10–15 mg/L) in 5 mM NaCl culture media (Miyazaki et al., 2014). For the filamentous fungi *Aspergillus oryzae* and *Aspergillus nidulans* the nanoplastic toxicity was not uniform among species or phenotypes, which was explained by the variability in one single trait: the resistance and hydrophobicity of cell walls (Nomura et al., 2016). The response and sensitivity to polystyrene nanobeads was also compared in vertebrates, which was linked to immunological traits of the respiratory system, i.e. the quantitative and qualitative attributes of surface respiratory macrophages of the domestic duck and rabbit (Mutua, Gicheru, Makanya, & Kiama, 2011). Given the environmental persistence of microplastics and their selective toxicity to organisms there is the potential for selective pressure of species traits with consequences for phenotypic, genetic, and functional biodiversity.

7 | THE BROAD-SPECTRUM TOXICITY TARGETS OF NANOPLASTICS

It is generally accepted that the impacts of pollution on ecologically relevant endpoints (such as migratory behavior, reproduction success, and mortality) are triggered by a cascade of changes initiated at subcellular level that propagates throughout the biological hierarchy. In this context, contaminants with broader toxicity targets can affect potentially a larger number of species and ecological functions. As plastic particles fragment they gain novel physical and chemical properties that increase their potential interaction with organisms causing direct and indirect toxicity. For instance, there is a growing consensus that the Trojan effect of large microplastics transferring contaminants to aquatic organisms might be unimportant (Koelmans, Bakir, Burton, & Janssen, 2016) and, if relevant, related to indirect adsorption of contaminants to the eco-corona (Galloway et al., 2017). On the other hand, research interest is increasing in the potential of nanoplastics to deliver drugs directly to intracellular compartments ((CONTAM), 2016, Forte et al., 2016; Hamoir et al., 2003). This highlights the potential of nanoplastics as an environmental pollutant and a carrier for other contaminants (Nemmar, Hoylaerts, Hoet, Vermylen, & Nemery, 2003; Wick et al., 2010). The technological limitations to quantify environmental nanoplastics (see section on the need for quantification of microplastic abundance) might explain why most of attention on nanoplastics as pollutant has focused on laboratory evidence.

Compared to larger microplastics, nanoplastics present high surface curvature (roughly, a topographical measurement of how locally curved surfaces are (Roach, Farrar, & Perry, 2006)) and particular surface chemistry. The surface curvature determines whether molecules and membranes would perceive microplastic surfaces as planar (Roach et al., 2006), and therefore influences the loading capacity for chemicals and nanoplastic crossing of membranes. The surface chemistry affects the charge and adsorption of compounds to nanoplastics, which in turn determines toxicity mechanisms (Figure 3). Nanoplastics externally adsorbed to cells might still cause toxicity by disrupting several membrane processes essential to the intracellular homeostasis of the exposed organism. Amino terminated polystyrene beads (50 nm) could strongly adhere to cells causing high toxicity at concentrations around 10 mg/L (Miyazaki et al., 2014). Conversely, carboxyl terminated polystyrene particles (50 nm) did not enter eukaryotic cells (*S. cerevisiae*), which had little effects on the growth rate at up to 80 mg/L after 1 h exposure (Miyazaki et al., 2014). The acute toxicity of adhered positively charged nanoplastics during exposure can be attributed to the electrostatic attraction between particles and cell walls, which has the potential to strongly disrupt membrane processes (Miyazaki et al., 2014). Positively charged amine-modified (60 or 400 nm) polystyrene particles were also more effective than their negatively charged counterparts in triggering lung responses such as on bronchoalveolar lavage indices and on peripheral thrombosis in hamster and rabbits (Hamoir et al., 2003; Nemmar et al., 2003).

In addition to acute toxicity by disrupting membrane processes, nanoplastics can also be internalized by cells which might increase their cytotoxicity targets (Syberg et al., 2015). Uptake allows direct interaction between nanoplastics, genetic material, and cell organelles. Indeed, changes in gene expression, inflammatory and biochemical responses, as well as carcinogenesis have been reported after nanoplastic exposure in human and nonhuman toxicological models (Forte et al., 2016; Kato et al., 2003). As with adhesion, the uptake and toxicity mechanisms of such nanoparticles seem to depend on their specific properties (surface area, size, electric charge, and hydrophobic properties, and cell metabolism) (Miyazaki et al., 2014; Schmid & Stoeger, 2016). For instance, in the lungs of rats, lecithin-coated and uncoated polystyrene beads (240 nm) were incorporated into alveolar macrophages, while alveolar epithelial cells selectively incorporated only lecithin-coated beads (Kato et al., 2003). Some of these ingested beads were sequestered within lysosomes while others were free in the cytoplasm, potentially due to nanoplastic disruption of lysosomal membranes (Kato et al., 2003). Polystyrene nanoparticles (44 nm and 100 nm) were readily taken up by gastric adenocarcinoma cells likely via an energy dependent mechanism of internalization and a clathrin-mediated endocytosis (Forte et al., 2016). Uptake of nanoplastics by fungi seems to be less related to endocytic processes and more associated to metabolic rates (respiration), membrane charge and differences between intra- and extracellular environments, osmotic pressure, and the hydrophobicity of cell walls (Miyazaki, Kuriyama, Tokumoto, Konishi, & Nomura, 2015; Miyazaki et al.,

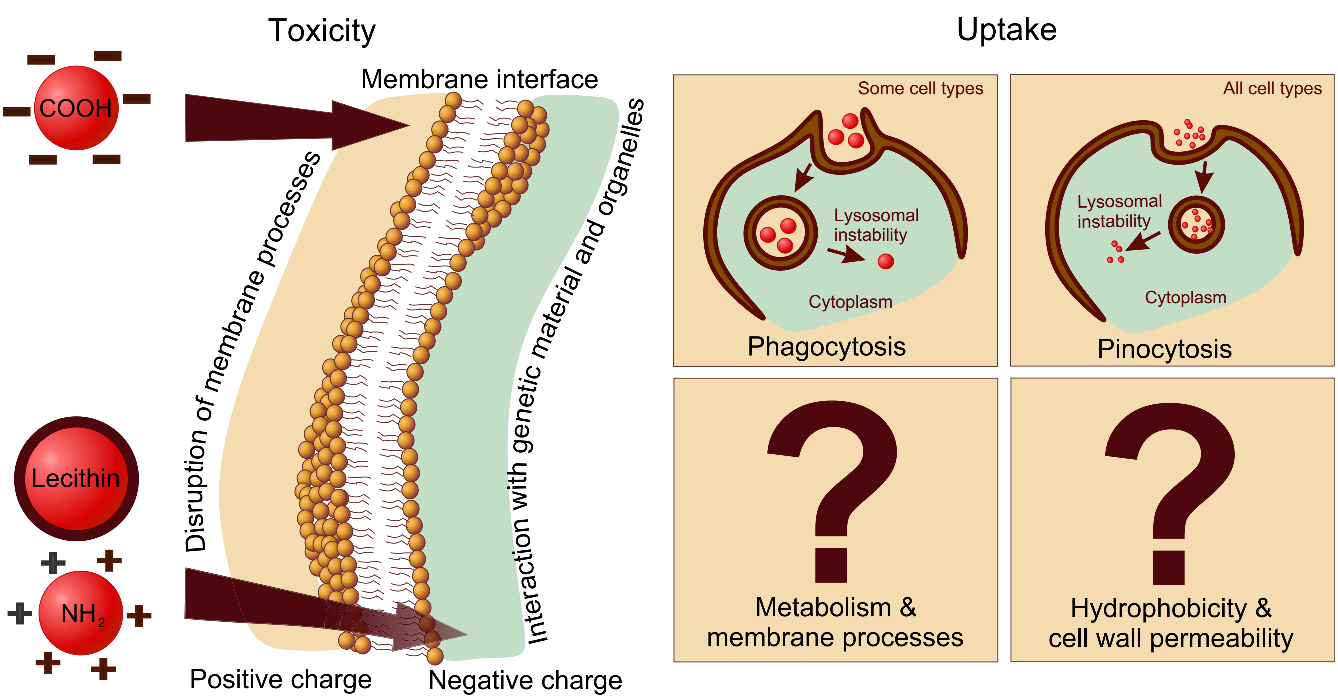

FIGURE 3 Potential toxicity and uptake mechanisms of nanoplastics. Carboxyl (COOH) and amino (NH_2) terminated or lecithin coated polystyrene beads yield nanoparticles with diverse cellular fate, which influences the toxicity mechanism. Reported processes (Kato et al., 2003; Miyazaki et al., 2014, 2015; Syberg et al., 2015) do not fully explain toxicity and uptake. Further mechanisms await discovery [Colour figure can be viewed at wileyonlinelibrary.com]

TABLE 1 Priority research for a more robust quantitative assessment of potential threats of microplastics to terrestrial ecosystems

Critical Open Question Explanation
What is the fate of microplastics in terrestrial and continental systems? Research is urgently needed to address the distribution of microplastics in terrestrial environments, to investigate the natural and anthropogenic processes affecting microplastic behavior (transport, degradation, and disintegration), and to improve methodological accuracy and precision for detection of microplastic quality (polymer matrix, additives, etc.) and quantity (particle numbers, size distributions, surface areas) for micro- and nanoparticles.
Which are the potential effects of microplastics in terrestrial ecosystems? Additional research should investigate whether the sorption capacity of microplastic surfaces, its physical structure, and leaching of components cause significant biogeochemical effects in soil ecosystems, and if so, for how long such effects are measurable. Similarly, it is important to study how microplastic contamination affects the biophysical environment in terrestrial ecosystems and its effects on the microbiota.
Are there potential impacts of microplastics on terrestrial organisms? Evidence suggests that most studied terrestrial animals can uptake microplastics while particles smaller than 1 μm are taken up by nearly all studied organisms. Therefore, it is important to investigate the potential of microplastics to physically and chemically disrupt physiologically important functions. This should also include communities from natural and urban ecosystems, including ecotoxicological models that are rarely studied such as plants, and natural microbial assemblages.
What are the uptake and toxicity mechanisms of nanoplastics in terrestrial organisms? Recent studies point to broad toxicity targets and uptake of nano-sized plastics. Thus toxicological, ecotoxicological, and toxicokinetic studies on organismal and suborganismal endpoints are required. The internalization of nanoplastics via the digestive tract and respiratory system, its transport to other parts of the body and the consequent local effects must be analyzed. Additionally, the role of nanoplastics to facilitate exposure to environmental hydrophobic pollutants remains to be clarified.

2014; Nomura et al., 2016). The uptake of nanoplastics through chitin-rich fungi cell walls highlights their capacity of crossing important impermeable barriers for many other toxics. Nanoplastics that were inhaled by rats were transported from the alveolar space by monocytes to the capillary lumen, from where the particles could be distributed to the rest of the body (Kato et al., 2003). In fact, there is substantial evidence that nanoplastics might cross highly selective membranes such as the brain-blood barrier and the human placenta ((CONTAM), 2016, Wick et al., 2010). At least in fish, the arrival of nanoplastics to the brain was associated with behavioral changes (Mattsson et al., 2017). As nanoplastics are eventually the final particle produced during the plastic degradation and disintegration, more research on its uptake and toxicity mechanisms in human and other terrestrial species is required.

8 | CONCLUSIONS

In isolation, microplastics might not be the single most toxic (lethal or sublethal) environmental contaminant. However, there are consistent past, present, and future trends of increasing a near-permanent plastic contamination of natural environments at global scale (Geyer et al., 2017). In light of the diverse and nonexhaustive possible interactions of microplastics with biotic and abiotic aspects of terrestrial ecosystem function hypothesized here, microplastics might well represent an important driver of global change across major terrestrial and continental ecosystems of the planet.

It is possible that terrestrial species are already or will be exposed to levels of plastic pollution capable of shifting baselines of physiological and ecosystem processes. Some species, in particular those with short generation times, may already be under evolutionary pressure from this new anthropogenic stressor. Therefore, microplastic and nanoplastic pollution might have potentially important, although almost completely neglected, impacts on biodiversity of continental systems. There is an urgent need to prioritize research dealing with this topic, and to provide sound information about environmental behavior, as well as ecotoxicological data about diverse microplastic contaminants in terrestrial ecosystems (Table 1). We think that scientifically sound information on the fate and effects of microplastics is essential for policy development as well as strategic management of microplastic pollution and its threats to terrestrial ecosystems.

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REFERENCES

Ballent, A., Corcoran, P. L., Madden, O., Helm, P. A., & Longstaffe, F. J. (2016). Sources and sinks of microplastics in Canadian Lake Ontario nearshore, tributary and beach sediments. *Marine Pollution Bulletin*, *110*, 383–395. https://doi.org/10.1016/j.marpolbul.2016.06.037

Barnes, D. K. A., Galgani, F., Thompson, R. C., & Barlaz, M. (2009). Accumulation and fragmentation of plastic debris in global environments. *Philosophical Transactions of the Royal Society B-Biological Sciences*, *364*, 1985–1998. https://doi.org/10.1098/rstb.2008.0205

Basel, U. O. (2016). News: Rhine one of the most microplastic polluted rivers worldwide. *Marine Pollution Bulletin*, *102*, 4–8.

Bergmann, J., Verbruggen, E., Heinze, J., Xiang, D., Chen, B., Joshi, J., & Rillig, M. C. (2016). The interplay between soil structure, roots, and microbiota as a determinant of plant-soil feedback. *Ecology and Evolution*, *6*, 7633–7644. https://doi.org/10.1002/ece3.2456

Besseling, E., Quik, J. T. K., Sun, M., & Koelmans, A. A. (2017). Fate of nano- and microplastic in freshwater systems: A modeling study. *Environmental Pollution*, *220*, 540–548. https://doi.org/10.1016/j.envpol.2016.10.001

Bouwmeester, H., Hollman, P. C. H., & Peters, R. J. B. (2015). Potential health impact of environmentally released micro- and nanoplastics in the human food production chain: Experiences from nanotoxicology. *Environmental Science & Technology*, *49*, 8932–8947. https://doi.org/10.1021/acs.est.5b01090

Clarke, B. O., & Smith, S. R. (2011). Review of 'emerging' organic contaminants in biosolids and assessment of international research priorities for the agricultural use of biosolids. *Environment International*, *37*, 226–247. https://doi.org/10.1016/j.envint.2010.06.004

Conrad, R. (1996). Soil microorganisms as controllers of atmospheric trace gases (H-2, CO, CH_4, OCS, N_2O, and NO). *Microbiological Reviews*, *60*, 609.

(Contam) EPOCITFC (2016). Presence of microplastics and nanoplastics in food, with particular focus on seafood. *EFSA Journal*, *14*, 4501. https://doi.org/10.2903/j.efsa.2016.4501

Dekiff, J. H., Remy, D., Klasmeier, J., & Fries, E. (2014). Occurrence and spatial distribution of microplastics in sediments from Norderney. *Environmental Pollution*, *186*, 248–256. https://doi.org/10.1016/j.envpol.2013.11.019

Dris, R., Gasperi, J., Saad, M., Mirande, C., & Tassin, B. (2016). Synthetic fibers in atmospheric fallout: A source of microplastics in the environment? *Marine Pollution Bulletin*, *104*, 290–293. https://doi.org/10.1016/j.marpolbul.2016.01.006

Dubovik, O., Holben, B., Eck, T. F., Smirnov, A., Kaufman, Y. J., King, M. D., . . . Slutsker, I. (2002). Variability of absorption and optical properties of key aerosol types observed in worldwide locations. *Journal of the Atmospheric Sciences*, *59*, 590–608. https://doi.org/10.1175/1520-0469(2002)059<0590:VOAAOP>2.0.CO;2

Filella, M. (2015). Questions of size and numbers in environmental research on microplastics: Methodological and conceptual aspects. *Environmental Chemistry*, *12*, 527–538. https://doi.org/10.1071/EN15012

Fischer, M., & Scholz-Bottcher, B. M. (2017). Simultaneous trace identification and quantification of common types of microplastics in environmental samples by pyrolysis-gas chromatography-mass spectrometry. *Environmental Science & Technology*, *51*, 5052–5060. https://doi.org/10.1021/acs.est.6b06362

Forte, M., Iachetta, G., Tussellino, M., Carotenuto, R., Prisco, M., De Falco, M., . . . Valiante, S. (2016). Polystyrene nanoparticles internalization in human gastric adenocarcinoma cells. *Toxicology in Vitro*, *31*, 126–136. https://doi.org/10.1016/j.tiv.2015.11.006

Fuller, S., & Gautam, A. (2016). A procedure for measuring microplastics using pressurized fluid extraction. *Environmental Science & Technology*, *50*, 5774–5780. https://doi.org/10.1021/acs.est.6b00816

Galloway, T. S., Cole, M., & Lewis, C. (2017). Interactions of microplastic debris throughout the marine ecosystem. *Nature Ecology & Evolution*, *1*, 0116. https://doi.org/10.1038/s41559-017-0116

Geyer, R., Jambeck, J. R., & Law, K. L. (2017). Production, use, and fate of all plastics ever made. *Science Advances*, *3*, e1700782. https://doi.org/10.1126/sciadv.1700782

Gil-Delgado, J. A., Guijarro, D., Gosalvez, R. U., Lopez-Iborra, G., Ponz, A., & Velasco, A. (2017). Presence of plastic particles in waterbirds faeces collected in Spanish lakes. *Environmental Pollution*, *220*, 732–736. https://doi.org/10.1016/j.envpol.2016.09.054

Hallett, P. D., Feeney, D. S., Bengough, A. G., Rillig, M. C., Scrimgeour, C. M., & Young, I. M. (2009). Disentangling the impact of AM fungi versus roots on soil structure and water transport. *Plant and Soil*, *314*, 183–196. https://doi.org/10.1007/s11104-008-9717-y

Hamoir, J., Nemmar, A., Halloy, D., Wirth, D., Vincke, G., Vanderplasschen, A., . . . Gustin, P. (2003). Effect of polystyrene particles on lung microvascular permeability in isolated perfused rabbit lungs: Role of size and surface properties. *Toxicology and Applied Pharmacology*, *190*, 278–285. https://doi.org/10.1016/S0041-008X(03)00192-3

Hidalgo-Ruz, V., Gutow, L., Thompson, R. C., & Thiel, M. (2012). Microplastics in the marine environment: A review of the methods used for identification and quantification. *Environmental Science & Technology*, *46*, 3060–3075. https://doi.org/10.1021/es2031505

Hodson, M. E., Duffus-Hodson, C. A., Clark, A., Prendergast-Miller, M. T., & Thorpe, K. L. (2017). Plastic bag derived-microplastics as a vector for metal exposure in terrestrial invertebrates. *Environmental Science & Technology*, *51*, 4714–4721. https://doi.org/10.1021/acs.est.7b00635

Holland, E. R., Mallory, M. L., & Shutler, D. (2016). Plastics and other anthropogenic debris in freshwater birds from Canada. *Science of the Total Environment*, *571*, 251–258. https://doi.org/10.1016/j.scitotenv.2016.07.158

Horton, A. A., Svendsen, C., Williams, R. J., Spurgeon, D. J., & Lahive, E. (2017). Large microplastic particles in sediments of tributaries of the River Thames, UK – Abundance, sources and methods for effective quantification. *Marine Pollution Bulletin*, *114*, 218–226. https://doi.org/10.1016/j.marpolbul.2016.09.004

Horton, A. A., Walton, A., Spurgeon, D. J., Lahive, E., & Svendsen, C. (2017). Microplastics in freshwater and terrestrial environments: Evaluating the current understanding to identify the knowledge gaps and future research priorities. *Science of the Total Environment*, *586*, 127–141. https://doi.org/10.1016/j.scitotenv.2017.01.190

Huerta Lwanga, E., Gertsen, H., Gooren, H., Peters, P., Salánki, T., van der Ploeg, M., . . . Geissen, V. (2017). Incorporation of microplastics from litter into burrows of Lumbricus terrestris. *Environmental Pollution*, *220*, 523–531. https://doi.org/10.1016/j.envpol.2016.09.096

Hussain, N., Jaitley, V., & Florence, A. T. (2001). Recent advances in the understanding of uptake of microparticulates across the gastrointestinal lymphatics. *Advanced Drug Delivery Reviews*, *50*, 107–142. https://doi.org/10.1016/S0169-409X(01)00152-1

Jambeck, J. R., Geyer, R., Wilcox, C., Siegler, T. R., Perryman, M., Andrady, A., . . . Law, K. L. (2015). Plastic waste inputs from land into the ocean. *Science*, *347*, 768–771. https://doi.org/10.1126/science.1260352

Jeong, C. B., Won, E. J., Kang, H. M., Lee, M. C., Hwang, D., Hwang, U. K., . . . Lee, J. S. (2016). Microplastic size-dependent toxicity, oxidative stress induction, and p-JNK and p-p38 activation in the monogonont rotifer (*Brachionus koreanus*). *Environmental Science & Technology*, *50*, 8849–8857. https://doi.org/10.1021/acs.est.6b01441

Kato, T., Yashiro, T., Murata, Y., Herbert, D. C., Oshikawa, K., Bando, M., . . . Sugiyama, Y. (2003). Evidence that exogenous substances can be phagocytized by alveolar epithelial cells and transported into blood capillaries. *Cell and Tissue Research*, *311*, 47–51. https://doi.org/10.1007/s00441-002-0647-3

Kirstein, I. V., Kirmizi, S., Wichels, A., Garin-Fernandez, A., Erler, R., Loder, M., & Gerdts, G. (2016). Dangerous hitchhikers? Evidence for potentially pathogenic Vibrio spp. on microplastic particles. *Marine Environmental Research*, *120*, 1–8. https://doi.org/10.1016/j.marenvres.2016.07.004

Koelmans, A. A., Bakir, A., Burton, G. A., & Janssen, C. R. (2016). Microplastic as a vector for chemicals in the aquatic environment: Critical review and model-supported reinterpretation of empirical studies. *Environmental Science & Technology*, *50*, 3315–3326. https://doi.org/10.1021/acs.est.5b06069

Lachenmeier, D. W., Kocareva, J., Noack, D., & Kuballa, T. (2015). Microplastic identification in German beer – an artefact of laboratory contamination? *Deutsche Lebensmittel-Rundschau*, *111*, 4.

Lebreton, L. C. M., Van Der Zwet, J., Damsteeg, J. W., Slat, B., Andrady, A., & Reisser, J. (2017). River plastic emissions to the world's oceans. *Nature Communications*, *8*, 10.

Lechner, A., Keckeis, H., Lumesberger-Loisl, F., Zens, B., Krusch, R., Tritthart, M., … Schludermann, E. (2014). The Danube so colourful: A potpourri of plastic litter outnumbers fish larvae in Europe's second largest river. *Environmental Pollution*, *188*, 177–181. https://doi.org/10.1016/j.envpol.2014.02.006

Lechner, A., & Ramler, D. (2015). The discharge of certain amounts of industrial microplastic from a production plant into the River Danube is permitted by the Austrian legislation. *Environmental Pollution*, *200*, 159–160. https://doi.org/10.1016/j.envpol.2015.02.019

Liebezeit, G., & Liebezeit, E. (2013). Non-pollen particulates in honey and sugar. *Food Additives and Contaminants Part a-Chemistry Analysis Control Exposure & Risk Assessment*, *30*, 2136–2140.

Liebezeit, G., & Liebezeit, E. (2014). Synthetic particles as contaminants in German beers. *Food Additives & Contaminants. Part A, Chemistry, Analysis, Control, Exposure & Risk Assessment*, *31*, 1574–1578.

Liebezeit, G., & Liebezeit, E. (2015). Origin of synthetic particles in honeys. *Polish Journal of Food and Nutrition Sciences*, *65*, 143–147.

Lusher, A. L., Welden, N. A., Sobral, P., & Cole, M. (2017). Sampling, isolating and identifying microplastics ingested by fish and invertebrates. *Analytical Methods*, *9*, 1346–1360. https://doi.org/10.1039/C6AY02415G

Lwanga, E. H., Gertsen, H., Gooren, H., Peters, P., Salánki, T., van der Ploeg, M., … Geissen, V. (2016). Microplastics in the terrestrial ecosystem: Implications for *Lumbricus terrestris* (Oligochaeta, Lumbricidae). *Environmental Science & Technology*, *50*, 2685–2691. https://doi.org/10.1021/acs.est.5b05478

Lwanga, E. H., Vega, J. M., Quej, V. K., de los Angeles Chi, J., del Cid, L. S., Chi, C., … Koelmans, A. A. (2017). Field evidence for transfer of plastic debris along a terrestrial food chain. *Scientific Reports*, *7*, 7.

Maass, S., Daphi, D., Lehmann, A., & Rillig, M. C. (2017). Transport of microplastics by two collembolan species. *Environmental Pollution*, *225*, 456–459. https://doi.org/10.1016/j.envpol.2017.03.009

Machado, AaS, Valyi, K., Rillig, M. C. (2017). Potential environmental impacts of an "underground revolution": A response to Bender et al. *Trends in Ecology & Evolution*, *32*, 8–10. https://doi.org/10.1016/j.tree.2016.10.009

Machado, AaD, Zarfl, C., Rehse, S., & Kloas, W. (2017). Low-dose effects: nonmonotonic responses for the toxicity of a *Bacillus thuringiensis* biocide to *Daphnia magna*. *Environmental Science & Technology*, *51*, 1679–1686. https://doi.org/10.1021/acs.est.6b03056

Mahler, G. J., Esch, M. B., Tako, E., Southard, T. L., Archer, S. D., Glahn, R. P., & Shuler, M. L. (2012). Oral exposure to polystyrene nanoparticles affects iron absorption. *Nature Nanotechnology*, *7*, 264-U1500.

Mahon, A. M., O'connell, B., Healy, M. G., O'connor, I., Officer, R., Nash, R., & Morrison, L. (2017). Microplastics in sewage sludge: effects of treatment. *Environmental Science & Technology*, *51*, 810–818. https://doi.org/10.1021/acs.est.6b04048

Mani, T., Hauk, A., Walter, U., & Burkhardt-Holm, P. (2015). Microplastics profile along the Rhine River. *Scientific Reports*, *5*, 7.

Manikkam, M., Tracey, R., Guerrero-Bosagna, C., & Skinner, M. K. (2013). Plastics derived endocrine disruptors (BPA, DEHP and DBP) induce epigenetic transgenerational inheritance of obesity, reproductive disease and sperm epimutations. *PLoS ONE*, *8*, 18.

Marty, M. S., Blankinship, A., Chambers, J., Constantine, L., Kloas, W., Kumar, A., … Verslycke, T. (2017). Population-relevant endpoints in the evaluation of endocrine-active substances (EAS) for ecotoxicological hazard and risk assessment. *Integrated Environmental Assessment and Management*, *13*, 317–330. https://doi.org/10.1002/ieam.1887

Mason, S. A., Garneau, D., Sutton, R., Chu, Y., Ehmann, K., Barnes, J., … Rogers, D. L. (2016). Microplastic pollution is widely detected in US municipal wastewater treatment plant effluent. *Environmental Pollution*, *218*, 1045–1054. https://doi.org/10.1016/j.envpol.2016.08.056

Mattsson, K., Johnson, E. V., Malmendal, A., Linse, S., Hansson, L. A., & Cedervall, T. (2017). Brain damage and behavioural disorders in fish induced by plastic nanoparticles delivered through the food chain. *Scientific Reports*, *7*, 7.

Mccormick, A. R., Hoellein, T. J., London, M. G., Hittie, J., Scott, J. W., & Kelly, J. J. (2016). Microplastic in surface waters of urban rivers: Concentration, sources, and associated bacterial assemblages. *Ecosphere*, *7*, 22.

Mchale, G., Newton, M. I., & Shirtcliffe, N. J. (2005). Water-repellent soil and its relationship to granularity, surface roughness and hydrophobicity: A materials science view. *European Journal of Soil Science*, *56*, 445–452. https://doi.org/10.1111/j.1365-2389.2004.00683.x

Meybeck, M. (2004). The global change of continental aquatic systems: Dominant impacts of human activities. *Water Science and Technology*, *49*, 73–83.

Mintenig, S. M., Int-Veen, I., Loder, M. G. J., Primpke, S., & Gerdts, G. (2017). Identification of microplastic in effluents of waste water treatment plants using focal plane array-based micro-Fourier-transform infrared imaging. *Water Research*, *108*, 365–372. https://doi.org/10.1016/j.watres.2016.11.015

Miyazaki, J., Kuriyama, Y., Miyamoto, A., Tokumoto, H., Konishi, Y., & Nomura, T. (2014). Adhesion and internalization of functionalized polystyrene latex nanoparticles toward the yeast *Saccharomyces cerevisiae*. *Advanced Powder Technology*, *25*, 1394–1397. https://doi.org/10.1016/j.apt.2014.06.014

Miyazaki, J., Kuriyama, Y., Tokumoto, H., Konishi, Y., & Nomura, T. (2015). Cytotoxicity and behavior of polystyrene latex nanoparticles to budding yeast. *Colloids and Surfaces A: Physicochemical and Engineering Aspects*, *469*, 287–293. https://doi.org/10.1016/j.colsurfa.2015.01.046

Mutua, P. M., Gicheru, M. M., Makanya, A. N., & Kiama, S. G. (2011). Comparative quantitative and qualitative attributes of the surface respiratory macrophages in the domestic duck and the rabbit. *International Journal of Morphology*, *29*, 353–362. https://doi.org/10.4067/S0717-95022011000200008

Nemmar, A., Hoylaerts, M. F., Hoet, P. H. M., Vermylen, J., & Nemery, B. (2003). Size effect of intratracheally instilled particles on pulmonary inflammation and vascular thrombosis. *Toxicology and Applied Pharmacology*, *186*, 38–45. https://doi.org/10.1016/S0041-008X(02)00024-8

Nizzetto, L., Bussi, G., Futter, M. N., Butterfield, D., & Whitehead, P. G. (2016). A theoretical assessment of microplastic transport in river catchments and their retention by soils and river sediments. *Environ Sci Process Impacts*, *18*, 1050–1059. https://doi.org/10.1039/C6EM00206D

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

Nomura, T., Tani, S., Yamamoto, M., Nakagawa, T., Toyoda, S., Fujisawa, E., … Konishi, Y. (2016). Cytotoxicity and colloidal behavior of polystyrene latex nanoparticles toward filamentous fungi in isotonic solutions. *Chemosphere*, *149*, 84–90. https://doi.org/10.1016/j.chemosphere.2016.01.091

Oberdorster, G. (2000). Toxicology of ultrafine particles: In vivo studies. *Philosophical Transactions of the Royal Society of London Series a-Mathematical Physical and Engineering Sciences*, *358*, 2719–2739. https://doi.org/10.1098/rsta.2000.0680

Plasticseurope (2015). Plastics — The Facts 2014/2015.

Programme UNE (2014). *Year book: Emerging issues in our global environment*. Nairobi: United Nations Environment Programme.

Rehse, S., Kloas, W., & Zarfl, C. (2016). Short-term exposure with high concentrations of pristine microplastic particles leads to immobilisation of *Daphnia magna*. *Chemosphere*, *153*, 91–99. https://doi.org/10.1016/j.chemosphere.2016.02.133

Rillig, M. C. (2005). A connection between fungal hydrophobins and soil water repellency? *Pedobiologia*, *49*, 395–399. https://doi.org/10.1016/j.pedobi.2005.04.004

Rillig, M. C. (2012). Microplastic in terrestrial ecosystems and the soil? *Environmental Science & Technology*, *46*, 6453–6454. https://doi.org/10.1021/es302011r

Rillig, M. C., Ingraffia, R., & De Souza Machado, A. A. (2017). Microplastic incorporation into soil in agroecosystems. *Frontiers in Plant Science*, *8*, 1805. https://doi.org/10.3389/fpls.2017.01805

Rillig, M. C., Ziersch, L., & Hempel, S. (2017). Microplastic transport in soil by earthworms. *Scientific Reports*, *7*, 6.

Roach, P., Farrar, D., & Perry, C. C. (2006). Surface tailoring for controlled protein adsorption: Effect of topography at the nanometer scale and chemistry. *Journal of the American Chemical Society*, *128*, 3939–3945. https://doi.org/10.1021/ja056278e

Rodriguez-Seijo, A., Lourenco, J., Rocha-Santos, TaP, Da Costa, J., Duarte, A. C., Vala, H., & Pereira, R. (2017). Histopathological and molecular effects of microplastics in *Eisenia andrei* Bouche. *Environmental Pollution*, *220*, 495–503. https://doi.org/10.1016/j.envpol.2016.09.092

Sanders, L. C., & Lord, E. M. (1989). Directed movement of latex-particles in the gynoecia of 3 species of flowering plants. *Science*, *243*, 1606–1608. https://doi.org/10.1126/science.243.4898.1606

Saxton, K. E., & Rawls, W. J. (2006). Soil water characteristic estimates by texture and organic matter for hydrologic solutions. *Soil Science Society of America Journal*, *70*, 1569–1578. https://doi.org/10.2136/sssaj2005.0117

Schmid, O., & Stoeger, T. (2016). Surface area is the biologically most effective dose metric for acute nanoparticle toxicity in the lung. *Journal of Aerosol Science*, *99*, 133–143. https://doi.org/10.1016/j.jaerosci.2015.12.006

Sohoni, P., & Sumpter, J. P. (1998). Several environmental oestrogens are also anti-androgens. *Journal of Endocrinology*, *158*, 327–339. https://doi.org/10.1677/joe.0.1580327

Steffen, W., Grinevald, J., Crutzen, P., & Mcneill, J. (2011). The Anthropocene: Conceptual and historical perspectives. *Philosophical Transactions of the Royal Society a-Mathematical Physical and Engineering Sciences*, *369*, 842–867. https://doi.org/10.1098/rsta.2010.0327

Steinmetz, Z., Wollmann, C., Schaefer, M., Buchmann, C., David, J., Tröger, J., . . . Schaumann, G. E. (2016). Plastic mulching in agriculture. Trading short-term agronomic benefits for long-term soil degradation? *Science of the Total Environment*, *550*, 690–705. https://doi.org/10.1016/j.scitotenv.2016.01.153

Sutherland, W. J., Clout, M., Cote, I. M., Daszak, P., Depledge, M. H., Depledge, M. H., . . . Watkinson, A. R. (2010). A horizon scan of global conservation issues for 2010. *Trends in Ecology & Evolution*, *25*, 1–7. https://doi.org/10.1016/j.tree.2009.10.003

Syberg, K., Khan, F. R., Selck, H., Palmqvist, A., Banta, G. T., Daley, J., . . . Duhaime, M. B. (2015). Microplastics: Addressing ecological risk through lessons learned. *Environmental Toxicology and Chemistry*, *34*, 945–953. https://doi.org/10.1002/etc.2914

Talvitie, J., Mikola, A., Setala, O., Heinonen, M., & Koistinen, A. (2017). How well is microlitter purified from wastewater? A detailed study on the stepwise removal of microlitter in a tertiary level wastewater treatment plant. *Water Research*, *109*, 164–172. https://doi.org/10.1016/j.watres.2016.11.046

Tamschick, S., Rozenblut-Koscisty, B., Ogielska, M., Kekenj, D., Gajewski, F., Krüger, A., . . . Stöck, M. (2016). The plasticizer bisphenol A affects somatic and sexual development, but differently in pipid, hylid and bufonid anurans. *Environmental Pollution*, *216*, 282–291. https://doi.org/10.1016/j.envpol.2016.05.091

Veresoglou, S. D., Halley, J. M., & Rillig, M. C. (2015). Extinction risk of soil biota. *Nature Communications*, *6*, 10.

Whitacre, D. M. (2014). *Reviews of environmental contamination and toxicology*, vol. 227. New York, NY: Springer. https://doi.org/10.1007/978-3-319-03777-6

Wick, P., Malek, A., Manser, P., Meili, D., Maeder-Althaus, X., Diener, L., . . . von Mandach, U. (2010). Barrier capacity of human placenta for nanosized materials. *Environmental Health Perspectives*, *118*, 432–436.

Yang, C. Z., Yaniger, S. I., Jordan, V. C., Klein, D. J., & Bittner, G. D. (2011). Most plastic products release estrogenic chemicals: A potential health problem that can be solved. *Environmental Health Perspectives*, *119*, 8.

Zhan, Z. W., Wang, J. D., Peng, J. P., Xie, Q. L., Huang, Y., & Gao, Y. F. (2016). Sorption of 3,3′,4,4′-tetrachlorobiphenyl by microplastics: A case study of polypropylene. *Marine Pollution Bulletin*, *110*, 559–563. https://doi.org/10.1016/j.marpolbul.2016.05.036

Zhang, K., Xiong, X., Hu, H. J., Wu, C., Bi, Y., Wu, Y., . . . Liu, J. (2017). Occurrence and characteristics of microplastic pollution in Xiangxi Bay of three gorges reservoir, China. *Environmental Science & Technology*, *51*, 3794–3801. https://doi.org/10.1021/acs.est.7b00369

Zhao, S., Zhu, L., & Li, D. (2016). Microscopic anthropogenic litter in terrestrial birds from Shanghai, China: Not only plastics but also natural fibers. *Science of the Total Environment*, *550*, 1110–1115. https://doi.org/10.1016/j.scitotenv.2016.01.112

Zheng, W., Morris, E. K., Lehmann, A., & Rillig, M. C. (2016). Interplay of soil water repellency, soil aggregation and organic carbon. A meta-analysis. *Geoderma*, *283*, 39–47. https://doi.org/10.1016/j.geoderma.2016.07.025

Zhu, D., Chen, Q.-L., An, X.-L., Yang, X. R., Christie, P., Ke, X., . . . Zhu, Y. G. (2018). Exposure of soil collembolans to microplastics perturbs their gut microbiota and alters their isotopic composition. *Soil Biology and Biochemistry*, *116*, 302–310. https://doi.org/10.1016/j.soilbio.2017.10.027

Ziková, A., Lorenz, C., Hoffmann, F., Kleiner, W., Lutz, I., Stöck, M., & Kloas, W. (2017). Endocrine disruption by environmental gestagens in amphibians – A short review supported by new in vitro data using gonads of Xenopus laevis. *Chemosphere*, *181*, 74–82. https://doi.org/10.1016/j.chemosphere.2017.04.021

Zubris, KaV, & Richards, B. K. (2005). Synthetic fibers as an indicator of land application of sludge. *Environmental Pollution*, *138*, 201–211. https://doi.org/10.1016/j.envpol.2005.04.013

SUPPORTING INFORMATION

Additional Supporting Information may be found online in the supporting information tab for this article.

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