RESEARCH REVIEW

Microplastics in terrestrial ecosystems: Moving beyond the state of the art to minimize the risk of ecological surprise

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Abstract

Microplastic (plastic particles measuring <5mm) pollution is ubiquitous. Unlike in other well-studied ecosystems, for example, marine and freshwater environments, microplastics in terrestrial systems are relatively understudied. Their potential impacts on terrestrial environments, in particular the risk of causing ecological surprise, must be better understood and quantified. Ecological surprise occurs when ecosystem behavior deviates radically from expectations and generally has negative consequences for ecosystem services. The properties and behavior of microplastics within terrestrial environments may increase their likelihood of causing ecological surprises as they (a) are highly persistent global pollutants that will last for centuries, (b) can interact with the abiotic environment in a complex manner, (c) can impact terrestrial organisms directly or indirectly and (d) interact with other contaminants and can facilitate their transport. Here, we compiled findings of previous research on microplastics in terrestrial environments. We systematically focused on studies addressing different facets of microplastics related to their distribution, dispersion, impact on soil characteristics and functions, levels of biological organization of tested terrestrial biota (single species vs. assemblages), scale of experimental study and corresponding ecotoxicological effects. Our systematic assessment of previous microplastic research revealed that most studies have been conducted on single species under laboratory conditions with short-term exposures; few studies were conducted under more realistic long-term field conditions and/or with multi-species assemblages. Studies targeting multi-species assemblages primarily considered soil bacterial communities and showed that microplastics can alter essential nutrient cycling functions. More ecologically meaningful studies of terrestrial microplastics encompassing multi-species assemblages, critical ecological processes (e.g., biogeochemical cycles and pollination) and interactions with other anthropogenic stressors must be conducted. Addressing these knowledge gaps will provide a better understanding of microplastics as emerging global stressors and should lower the risk of ecological surprise in terrestrial ecosystems.

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1 | INTRODUCTION

Ecological surprise occurs when observed ecosystem response deviates from anticipated behavior (Christensen et al., 2006; Filbee-Dexter et al., 2017; Holling, 1996). The rapid and irreversible environmental changes associated with some ecological surprises can seriously disrupt ecosystem functioning and potentially threaten human well-being (Boyd, 2008; Christensen et al., 2006; Walters & Maguire, 1996). Recent examples of catastrophic ecological surprise include the collapse of North Atlantic cod stocks (Walters & Maguire, 1996) and coastal fisheries around the world (Hilborn et al., 2003), as well as forest dieback in the Amazon (Boyd, 2008) and elsewhere (Doak et al., 2008). Despite the vast literature on microplastics, it is still not clear whether they pose a significant environmental threat (see, e.g., Burton, 2017; Leslie & Depledge, 2020; Wardman et al., 2020). This lack of clarity may be a precursor to ecological surprise.
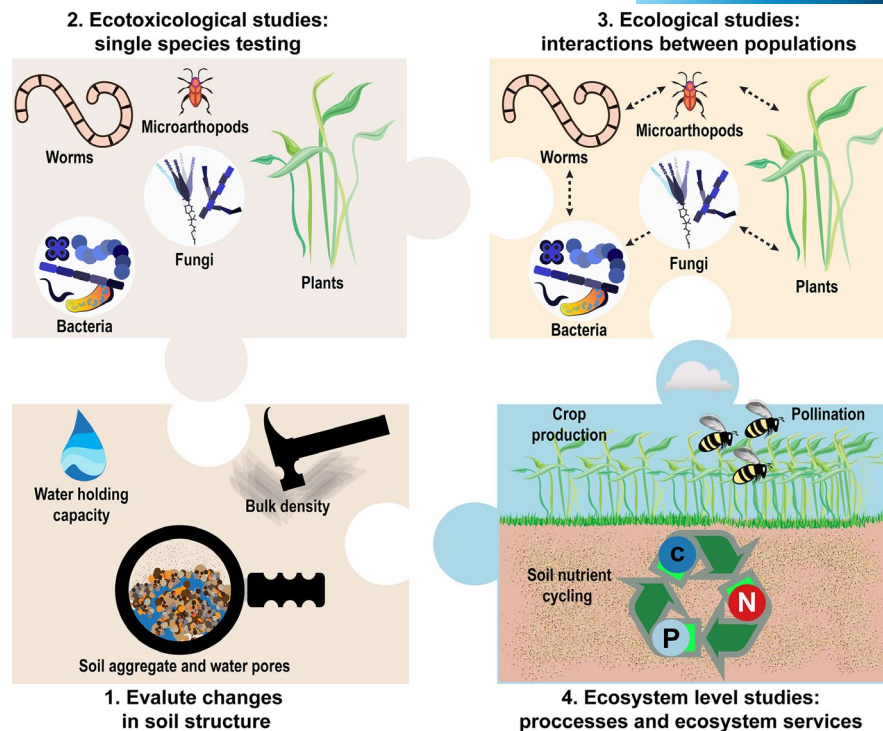
Doak et al. (2008) identified four causes of ecological surprise: (1) complex webs of interaction within real ecological communities; (2) community variability in time and space; (3) the multi-dimensional nature inherent to ecosystems that are interlinked by feedback mechanisms and (4) the cascading effects of shifting abiotic conditions in altering species distributions and their subsequent interactions. To reduce the potential for undesirable ecological surprise associated with microplastics in the terrestrial environment, we must move beyond an exclusive focus on short-term single species testing under controlled conditions to a new paradigm emphasizing the quantification of ecologically meaningful effects on multi-species assemblages in realistic natural conditions and over longer time scales (Rillig & Lehmann, 2020). This transition to more ecologically relevant studies is needed if we are to adequately account for the multiple possible interactions and interconnections occurring between the abiotic and hierarchical biotic structures that gradually increase in complexity from individual species to communities and ecosystem processes and functions (Rillig & Lehmann, 2020; Rykiel, 1985).

Microplastics, all small plastic particles measuring less than 5 mm (including nano scale plastics <0.1 μm), are ubiquitous in soils (Nizzetto et al., 2016). The potential for microplastic pollution to lead to ecological surprises is related to their widespread distribution and high abundance (Hale et al., 2020). Microplastics alter soil physical and chemical properties and have been identified as an emerging threat (Boots et al., 2019; de Souza Machado, Lau, et al., 2018; Liang et al., 2019; Liu et al., 2017; Zang et al., 2020). They can readily be ingested by organisms (Thompson et al., 2004) and can act as a novel substrate for deleterious organism groups (e.g., antibiotic-resistant bacteria; Bartkova et al., 2021; Imran et al., 2019), thus raising concerns for potential negative effects on biodiversity and ecosystem functioning (Browne et al., 2007; Reid et al., 2019; Rillig & Lehmann, 2020; Sutherland et al., 2010). To date, our understanding of possible

terrestrial ecosystem-level effects of microplastics is minimal as the vast array of possible interactions occurring between and within abiotic and biotic components of the soil environment (Figure 1) hinder meaningful predictions at the ecosystem level (Girardello et al., 2019; Tscharntke et al., 2005).

Microplastics were first documented in the Sargasso Sea by Carpenter and Smith (1972). However, the interest of the scientific community to investigate their potential effects on marine organisms came later and was inspired by the study of Thompson et al. (2004). Since then, attention has spread to freshwater (de Sá et al., 2018; Hoffman & Hittinger, 2017; Li et al., 2018) and terrestrial environments (de Souza Machado, Kloas, et al., 2018; Rillig & Lehmann, 2020; Windsor et al., 2019). Microplastic research in the terrestrial environment is still at an early stage, despite multiple lines of evidence indicating terrestrial systems as the major sinks of microplastics (Büks & Kaupenjohann, 2020; Evangeliou et al., 2020; Nizzetto et al., 2016). The development of terrestrial microplastics research has followed a similar trajectory to work on aquatic ecosystems. Researchers attempted first to quantify the scale of contamination (Corradini et al., 2019; Huerta Lwanga, Mendoza Vega, et al., 2017; Nizzetto et al., 2016; Piehl et al., 2018). Subsequently, they investigated possible deleterious effects on biota (de Souza Machado et al., 2019; de Souza Machado, Lau, et al., 2018; Huerta Lwanga et al., 2016; Rillig, Ziersch, et al., 2017; Zhou, Liu, et al., 2020) and used ecotoxicological approaches to reveal potential modes of toxic action (Rillig & Lehmann, 2020). Studies reported oxidative stress (Jiang et al., 2019; Prendergast-Miller et al., 2019; Zheng et al., 2019), reproductive impairment (Judy et al., 2019; Lahive et al., 2019; van Gestel & Selonen, 2018) and behavioral changes (Huerta Lwanga et al., 2016; Song et al., 2019) in terrestrial organisms (e.g., earthworms, snails and plants); similar responses were also observed in aquatic biota exposed to microplastics (de Sá et al., 2018; Lusher, 2015). Recent studies suggesting the effects of microplastics on soil carbon (Rillig et al., 2021) and possibly nitrogen (Rong et al., 2021) cycling are especially concerning.

Microplastics can enter the terrestrial environment through different pathways, including landfills (Geyer et al., 2017), atmospheric fallout (0.343 million tons per year globally; Evangeliou et al., 2020) and from different agricultural practices, for example, the use of plastic mulching (Büks & Kaupenjohann, 2020; Crossman et al., 2020). Agroecosystems are major entry routes for microplastics to the terrestrial environment, in particular when sludge (biosolids) from wastewater treatment are applied to the soil (Corradini et al., 2019). Microplastics present in wastewater are efficiently retained (c. 99%) in the sludge produced during treatment and the use of sewage sludge to amend soils is estimated to add between 63–430 and 44–300 thousand tons of microplastics to agroecosystems in Europe and North America annually (Nizzetto et al., 2016). Sewage

2. Ecotoxicological studies: single species testing

Worms Microarthopods Fungi Plants Bacteria

Water holding capacity Bulk density Soil aggregate and water pores

1. Evalute changes in soil structure

3. Ecological studies: interactions between populations

Worms Microarthopods Fungi Plants Bacteria

Crop production Pollination Soil nutrient cycling C N P

4. Ecosystem level studies: proccesses and ecosystem services

FIGURE 1 Conceptual diagram on the current state of knowledge regarding the effects of microplastics in the terrestrial environment. The effects of microplastics on (1) the physical and chemical properties of soil (de Souza Machado et al., 2019; de Souza Machado, Lau, et al., 2018; Zhang & Liu, 2018) and (2) on single terrestrial species using short term ecotoxicological assays (Boots et al., 2019; Lahive et al., 2019; Prendergast-Miller et al., 2019; Song et al., 2019) have recently been studied. However, ecologically oriented studies targeting the potential implications of microplastics and the risk for surprises in more natural settings that consist of (3) multiple populations of species that collectively form communities and the (4) functions those multiple species performed are currently scarce

sludge applications in terrestrial agroecosystems alone exceed the estimated amount (93–236 thousand tons of microplastics) exported to oceans annually (Van Sebille et al., 2015).

Given the importance of the soil for supporting biodiversity and ecosystem services (Bardgett & van der Putten, 2014), particularly the irreplaceable role of agroecosystems in food production, it is imperative to evaluate the ecological effects of microplastics in terrestrial environments as any adverse effects could threaten sustainability and the well-being of future generations (Rillig, Ingraffia, et al., 2017). Additional studies on effects of microplastics on terrestrial food web stability, critical ecological processes (e.g., biogeochemical cycles and pollination) and interactions with other anthropogenic stressors must be conducted. Other contaminants, for example, dichlorodiphenyl-trichloroethane (DDT; Colborn et al., 1993), per- and polyfluoroalkyl substances (PFAS; Koch et al., 2020) and antibiotics (Ferri et al., 2017) that share similar characteristics as microplastics (i.e., high persistency and widespread presence in the environment) have caused unexpected ecosystem responses including the near extinction of iconic predatory bird species (ospreys and bald eagles; Wiemeyer et al., 1975, 1984) and even put human health at risk (Fry, 1995; Sunderland et al., 2019). Unlike other chemical contaminants, microplastics have unique properties: make up of diverse polymers, broad size ranges including the nano scale (<0.1 μm), vary in shape and contain diverse chemical additives (Rochman et al., 2019), which increase their likelihood to impact ecosystems and cause ecological surprises.

Here, we review findings about microplastics in the terrestrial environment that may help to minimize the likelihood of ecological surprises. Our main objectives were to (a) summarize the characteristics of microplastics (shape and size) observed in terrestrial ecosystems, (b) document species- and community-level effects on terrestrial organisms, (c) summarize changes in soil physicochemical properties associated with the presence of microplastics, (d) link changes in soil physicochemical properties to their effects on biogeochemical processes, (e) evaluate the dispersion and degradation of microplastics in the terrestrial environment, (f) highlight the role of microplastics as vectors for other contaminants and (g) identify future research needs to uncover and understand potential ecological consequences which should avoid related surprises.

2 | METHODOLOGICAL APPROACH

A survey of peer-reviewed publications was performed on March 21st, 2021, using Thompson Reuters database ISI web of Science (Versions 5.34 and 5.35, respectively). We initially tried different combined keywords, including "*microplastic, soil,*" "*microplastic, terrestrial,*" "*microplastic, colloids,*" "*microplastic, ecosystem services,*" "*nanoplastic, soil*" and "*nanoplastic, terrestrial.*" The first paired keywords "*microplastic, soil*" gave the highest number of hits and largely covered the results (identical publications) obtained from the remaining

search terms leading to a total of 921 candidate publications. The abstracts from these 921 candidate publications were carefully read to further identify relevant studies that covered at least one of the following selection criteria: (a) degradation of microplastic in terrestrial environment, (b) dispersion of microplastic in terrestrial environment, (c) effects of microplastic in terrestrial environment and (d) reviews addressing microplastic research in terrestrial ecosystems. Publications covering either aquatic environments or method development for quantifying microplastics were excluded. Among the 921 candidate publications, 264 fulfilled our selection criteria and relevant information including microplastic properties (i.e., type, shape and size range) were recorded. Information on methodological approaches to investigate microplastic effects on terrestrial biota, for example, targeted biological entities (single species testing vs. multiple-species assemblages), study scale (lab; small-scale experiments in controlled conditions, vs. field; large-scale experiments in realistic conditions), study duration and exposure concentrations were extracted. Whenever possible, microplastic concentrations were standardized as percent weight of microplastic per weight of soil (% w/w) to facilitate comparison between studies. As some studies assessed multiple response variables in combination with a range of microplastic properties, the number of data points (989) exceeded the number of publications (264).

Plastics were classified according to their polymer types: acrylic polymers, polyamide, polyester, polyethylene, polypropylene, polystyrene and polyvinylchloride. High- and low-density polyethylene, polylactic acid and polyethylene terephthalate were classified as polyethylene (de Sá et al., 2018). When specified, microplastic shapes were recorded as beads (spherical), fragments, fibers, particles (irregular ovoidal shape) and pellets (short rod shape). When reported, microplastic sizes were classified using the following ranges: <1, 1–10, 10–70, 70–250, 250–650, 650–1000 and 1000–5000 μm. Microplastics belonging to the nano range (<0.1 μm) were included in the smallest size class (<1 μm). Ecotoxicological responses included the following: biomolecular changes, biosynthesis, gene expression, genotoxicity, oxidative stress, histopathological effects, behavior, physical effects, mortality, reproductive success, transgenerational effects, community structure and community functioning. The role of microplastics as vectors for other contaminants and pathogens including bacteria carrying antimicrobial resistance (AMR) genes was also considered.

3 | OVERVIEW OF RELEVANT RESEARCH AND MICROPLASTIC PROPERTIES IN THE TERRESTRIAL ENVIRONMENT

Most publications focused on literature reviews and effect studies followed by papers addressing the distribution and degradation of microplastics in the terrestrial environment (Figure 2a). Overall, these publications reflect the high heterogeneity of microplastic polymers (Figure 2b), shapes (Figure 2c) and sizes (Figure 2d). The type of plastic polymers used, and their shape differed among study

categories (degradation, distribution and effect). The frequency of plastic polymers reported in effect studies reflects a combination of global production patterns (Geyer et al., 2017) and the type of polymer commonly found in sewage sludge (Mahon et al., 2017), where polystyrene (35%), polyethylene (22%), polyester (20%) and polyvinylchloride (6%) were more abundant. Fragments were the only shape type that were consistently present in all study categories (Figure 2c). Fragments, mixed shapes and particles were reported in a large proportion of studies documenting microplastic distribution, whereas beads and fragments were reported in a large proportion of studies that primarily focused on effects. The five smallest size classes, <1, 1–10, 10–70, 70–250 and 250–650 μm, were reported in 87% of studies (Figure 2d). Drawing general conclusions about shape and size of microplastics in the terrestrial environment is challenging as microplastics are mostly studied in relation to their different points of entry (i.e., application of sewage sludge on agricultural land, mulching, seed coating, inadequate waste management and atmospheric deposition). Therefore, any information about microplastic prevalence and fate is context-specific. For example, both fibers and fragments (Corradini et al., 2019; van den Berg et al., 2020; Zhang, Xie, et al., 2020) can predominate in soils receiving sludge amendments, whereas fragments are commonly found in sites affected by atmospheric deposition (Klein & Fischer, 2019) and poor waste management practices (Huerta Lwanga, Mendoza Vega, et al., 2017).

4 | OVERVIEW ON THE DISPERSAL OF MICROPLASTICS BY TERRESTRIAL ORGANISMS AND ECOTOXICOLOGICAL EFFECTS

Most studies considering the role of terrestrial organisms in dispersing microplastics (86%) and their biological effects (50%) have been performed using single species (Figure 3a,b). These studies document the role of, for example, plants (corn; *Zea mays*, soybean; *Glycine max* and ryegrass; *Lolium multiflorum*, Li et al., 2021), earthworms (*Lumbricus terrestris*; M. Yu et al., 2019), snails (*Helix aspersa*, *H. aperta* and *H. pomatia*; Panebianco et al., 2019) and birds (*Gallus gallus domesticus*; Huerta Lwanga, Mendoza Vega, et al., 2017; *Falco tinnunculus*, *Buteo buteo* and *Milvus migrans lineatus* Zhao et al., 2016) as vectors for dispersal of microplastics.

We found only one study that explicitly addressed the role of interactions between multiple species assemblages in promoting microplastic dispersal (Zhu, Bi, et al., 2018). The authors (Zhu, Bi, et al., 2018) showed that predator–prey interactions in microarthropod assemblages can also enhance the dispersion of microplastics.

Microplastic concentrations used in ecotoxicological studies had a median of 0.5% w/w, a mode of 1% w/w and ranged from a minimum of 0.000002% w/w to a maximum of 60% w/w. Early terrestrial studies (e.g., Huerta Lwanga et al., 2016) used high exposure concentrations (up to 60% w/w) while more recent studies have used more environmentally realistic levels mimicking the range of concentrations representative of near-pristine remote areas (0.0005% w/w,
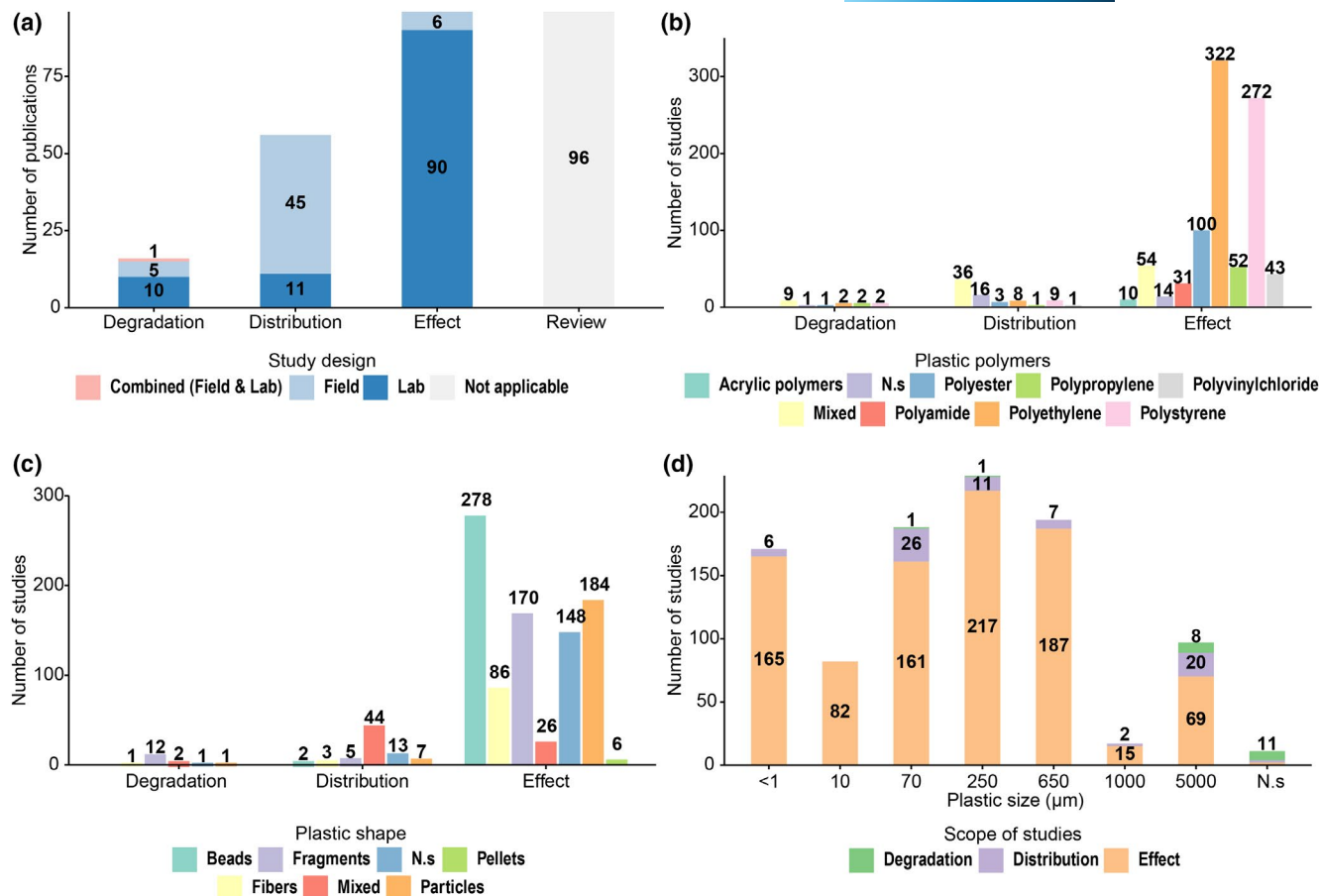
FIGURE 2 Number of publications and characteristics of studied microplastics within the terrestrial environment. The number of peer-reviewed publications covering the three main topic areas: degradation, dispersion, effects and reviews (a) and the associated properties of microplastics; type of polymer (b), shape (c) and size range (d). In some studies, the shape and the plastic polymers were not specified (denote as N.s in the legend)

average value reported from Swiss floodplain soils, see Scheurer & Bigalke, 2018) to industrial hotspots (up to 6.75% w/w; Fuller & Gautam, 2016).

Ecotoxicological effects of microplastics were more frequently studied in certain organism groups (Figure 3c,d): bacteria (27.8%; concentration: 0.001%–28% w/w), plants (23.4%; 0.0002%–20% w/w) and worms (19.6%; concentration: 0.0000015%–60% w/w). Laboratory experiments were conducted over relatively short time scales (median of 30 days), whereas field trials were conducted over longer time frames (median of 287 days, see Figure 4a,b). Slightly more than half (53%) of these studies reported significant effects of exposure to microplastics (Figure 4b).
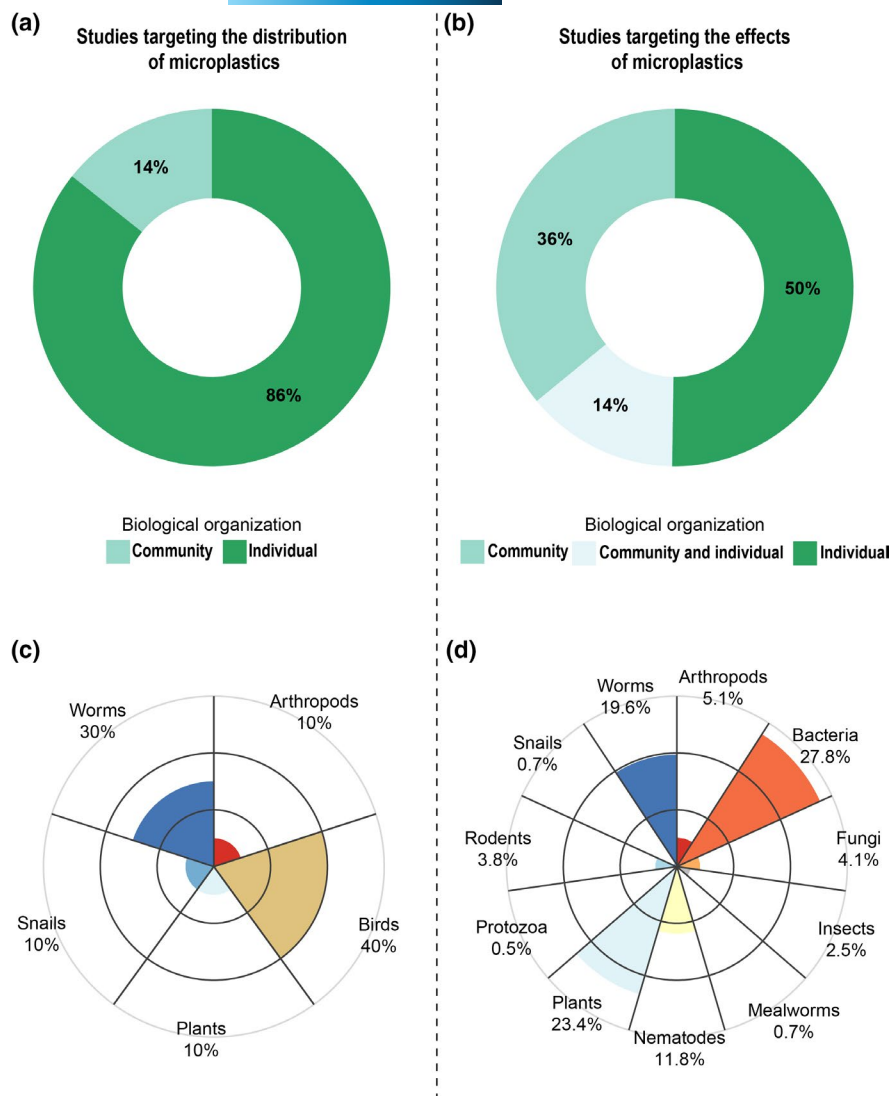
Some ecotoxicological responses were more widely reported than others, for example, physical changes (i.e., alterations in the morphological characteristics of the organisms such as growth rate, size, length and weight), mortality and behavioral effects (Figure 5). Microplastics are rarely lethal for terrestrial organisms, but rather induce general oxidative stress responses including gene expression (Yu et al., 2020) and ultimately influence the regulation and activation of antioxidant enzymes including, for example, superoxide dismutase, catalase and glutathione s-transferase (Chen et al., 2020;

Jiang et al., 2019; Le et al., 2018; Prendergast-Miller et al., 2019; Song et al., 2019; Wang et al., 2019). Those reactions can be interpreted as stress responses or defense mechanisms (Jeong & Choi, 2019).

Effect studies that evaluate community endpoints (structure and function) predominantly involve soil microbial assemblages. The soil microbial community fulfills vital ecological roles (Gattinger et al., 2008; Sofo et al., 2014) for nutrient cycling and is essential for plant survival. Any changes in soil microbial communities have the potential for ecological surprise as they can alter soil quality (Bardgett & van der Putten, 2014) and potentially affect carbon cycling (Rillig et al., 2021).

5 | MICROPLASTIC EFFECTS ON TERRESTRIAL PLANTS

There are a limited number of studies of microplastic effects on terrestrial plants and it is hard to draw any general conclusions. With the exception of Lozano and Rillig (2020), most studies have targeted single species, including, for example, perennial ryegrass

(a)

Studies targeting the distribution of microplastics

14%

86%

Biological organization

Community   Individual

(b)

Studies targeting the effects of microplastics

36%

50%

14%

Biological organization

Community   Community and individual   Individual

FIGURE 3 An overview of the studies reporting distribution of microplastics by terrestrial organisms and microplastic effects on terrestrial organisms. Comparing the level of biological organization, i.e., individual species versus community assemblages across studies focusing on the distribution (a) and the effects of microplastics (b) and the terrestrial organism groups studied (c and d) for each respective category

(c)

Worms 30%

Arthropods 10%

Snails 10%

Birds 40%

Plants 10%

(d)

Worms 19.6%

Arthropods 5.1%

Snails 0.7%

Bacteria 27.8%

Rodents 3.8%

Fungi 4.1%

Protozoa 0.5%

Insects 2.5%

Plants 23.4%

Mealworms 0.7%

Nematodes 11.8%

(Boots et al., 2019), common wheat (Judy et al., 2019), broad bean (Jiang et al., 2019), carrots (Lozano et al., 2021) and spring onion (de Souza Machado et al., 2019). The experimental conditions used in these studies vary widely with respect to soil type (loamy sandy soil and clay collected at different locations), duration of exposure (2–273 days) and microplastic concentrations (0.001%–2% w/w). This lack of standardization makes any cross-study comparisons challenging. When multiple studies can be compared because they share some common aspects of experimental setup, conflicting results are often reported. For example, Boots et al. (2019) found that germination of perennial ryegrass was impaired when exposed to acrylic polymers and polyester at concentrations of 0.1% w/w, whereas Judy et al. (2019) did not find any negative effects on germination of common wheat using identical concentrations of polyester, polyethylene and polyvinylchloride, suggesting that species respond differently. The presence of microplastics can trigger different effects on different plant organs. Boots et al. (2019) reported that perennial ryegrass exposed to polyethylene microplastics had a significant increase in root biomass but no significant change in shoot biomass. Size of microplastics also appears to matter. Jiang et al. (2019) showed that

identical concentrations (0.001%, 0.005% and 0.01% w/w) of different sized polystyrene microplastic beads caused size-dependent responses in broad bean. Exposure to larger beads (5 μm) significantly reduced root length but did not induce any genotoxicity effect, whereas smaller beads (0.1 μm) triggered significant genotoxicity effects while root length was unaffected (Jiang et al., 2019).

A better understanding of the interactions between plants and mycorrhizal fungi may help to elucidate some of the mechanisms behind the morphological changes in plants associated with exposure to microplastics (de Souza Machado et al., 2019). A study assessing effects of six microplastics polymers (polyamide: 2% w/w, polyester: 0.2% w/w, high-density polyethylene: 2% w/w, polypropylene: 2% w/w, polystyrene: 2% w/w and polyethylene terephthalate: 2% w/w) with three different shapes (fibers, beads and fragments) on plant (spring onion; *Allium fistulosum*)–mycorrhizal fungi interactions showed polymer and shape-specific effects on symbiosis (de Souza Machado et al., 2019). Compared to the other polymers, polyester fibers substantially increased mycorrhizal colonization (by 8-fold) around the roots (de Souza Machado et al., 2019). The increase in colonization was attributed to localized changes in soil structure
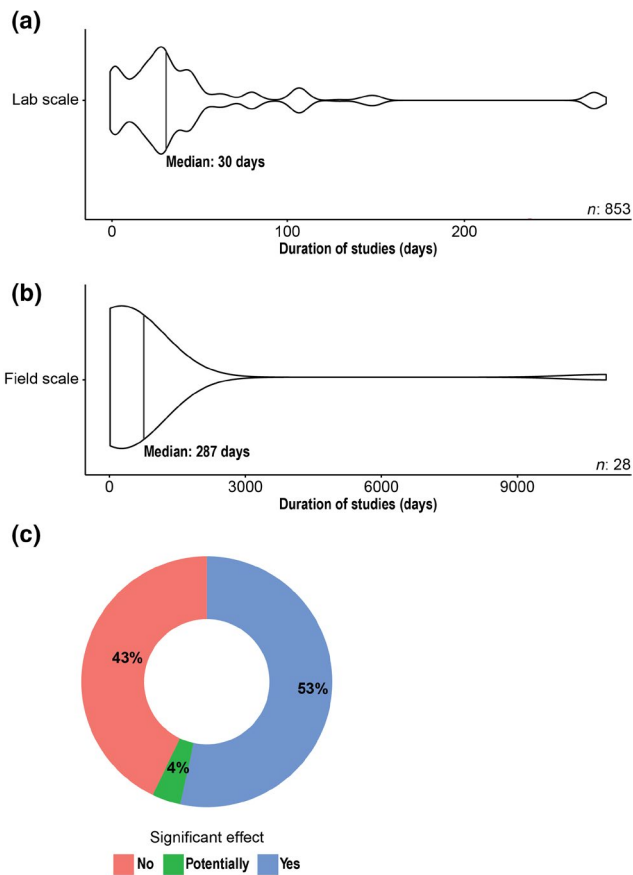
FIGURE 4 The duration of the experiments at the laboratory (a) and the field scale (b) shown as violin plots, and the percentage of ecotoxicological studies reporting significant effects (c)

(an increase soil density and water stable aggregates) surrounding the roots and subsequently improved the yield of spring onion bulbs (de Souza Machado et al., 2019). These findings clearly demonstrate that microplastics can indirectly affect the viability of agroecosystems by influencing multi-species interactions and plant growth processes.

6 | MICROPLASTIC EFFECTS ON SOIL STRUCTURE AND FUNCTION

6.1 | Changes in soil physicochemical properties

Soil is a critical environmental matrix that provides habitat and sustains ecosystem functions and services (Bardgett & van der Putten, 2014). The presence of microplastics can significantly alter soil physicochemical properties, including the distribution of water stable aggregates, bulk density, water retention capacity and pH (Table 1). These changes in physicochemical properties are dependent on microplastic shape, polymer and concentration (de Souza Machado et al., 2019; de Souza Machado, Lau, et al., 2018; Lozano et al., 2021; Zhang, Zhang, et al., 2019). Specifically, fibers have been

observed to trigger stronger effects on soil bulk density, distribution of soil aggregates and the formation of macropores (>30 µm; de Souza Machado et al., 2019; de Souza Machado, Lau, et al., 2018; Zhang, Zhang, et al., 2019; Zhang & Liu, 2018). Moreover, some of the changes in soil physicochemical properties can be modulated by interactions with soil dwelling organisms, for example, plants and fungi (de Souza Machado, Lau, et al., 2018; Liang et al., 2019; Wang et al., 2020).

6.2 | Interactions with soil biota

Interactions between microplastics and the soil biota can influence changes in soil physicochemical characteristics. Liang et al. (2019) showed that the effects of microplastics on water stable aggregates varied when the soil was inoculated with different strains of saprobic fungi. For instance, the interactions between microplastics and two individual fungal strains substantially reduced the percentage of water stable aggregates, whereas the opposite effect was observed for a third strain. In another study, Boots et al. (2019) found that the addition of microplastics (polyethylene polymer) on their own significantly decreased soil pH. However, in treatments that combined microplastic addition and a plant (perennial ryegrass), no decrease in pH was observed. Interactions between different microplastic polymers (polyamide, polyester, polyethylene, polypropylene and polystyrene) and spring onion on soil characteristics (bulk density and water-holding capacity) were investigated by de Souza Machado et al. (2019). They demonstrated differences in outcomes across treatments that included microplastic–plant interactions compared to microplastics alone (de Souza Machado et al., 2019). For instance, microplastic-induced reductions in soil bulk density (with the exception of polypropylene) and increase in water-holding capacity were less pronounced when interactions between microplastics and plants were considered (de Souza Machado et al., 2019). These findings highlight the potential for ecological surprise when microplastics are added to the soil. The complexity of interactions between microplastics, plants and the soil biota in relatively simple experimental setups underscores the need to address interactions between microplastics, plants, fungi and other soil organisms for understanding changes in soil physicochemical properties and potential effects on agroecosystem integrity.

6.3 | Linking changes in soil structure and function to presence of microplastics

Changes in soil structures (bulk density, water stable aggregates, water-holding capacity, pore volume) associated with the presence of microplastics can directly affect soil microbial communities, which are the main drivers of nitrogen cycling (Rong et al., 2021), carbon processing (Rillig et al., 2021) and other biogeochemical processes essential for human survival (Bender et al., 2016). Multiple studies have shown that microplastics can affect soil microbial
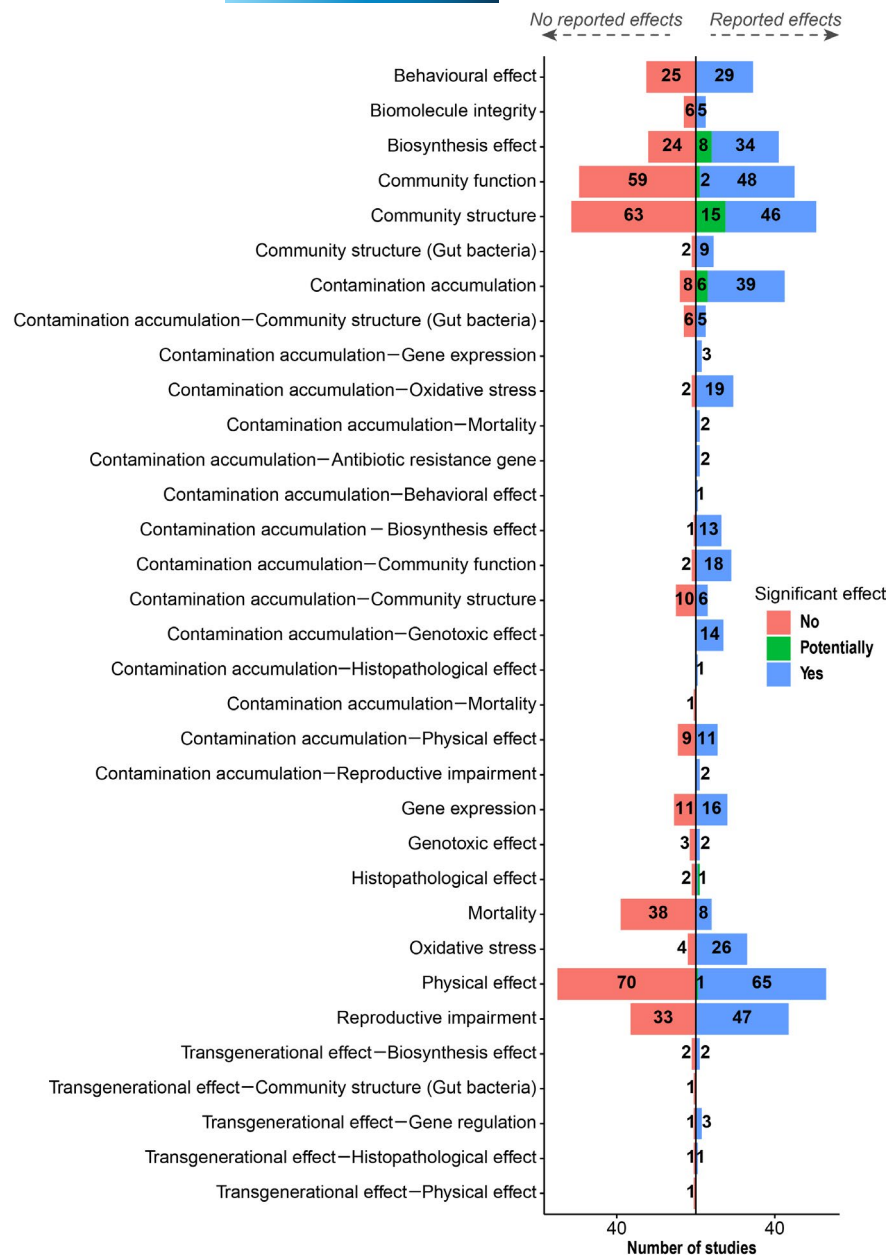
FIGURE 5 Ecotoxicological endpoints reported in studies of microplastic effects on terrestrial organisms. Each bar shows the total number of studies

activity and may have important implications for nutrient cycling (de Souza Machado et al., 2019; de Souza Machado, Lau, et al., 2018; Liu et al., 2017; Rong et al., 2021). Liu et al. (2017) demonstrated that the increase in microbial activity (based on the enzymatic activities of fluorescein diacetate hydrolase) associated with the presence of microplastics was correlated with an increase in soil-water dissolved nutrient concentrations. This suggests that microplastics can alter the performances of microbial communities and influence the accumulation of dissolved nutrients (Liu et al., 2017). Others reported mixed results on the performances of soil microbial communities, for example, respiration and nitrification were both negatively and positively influenced when exposed to microplastics (Judy et al., 2019) and Yan et al. (2020) reported changes in microbial community diversity but no effect on soil nutrients. Moreover, the response of microbial assemblages to low concentrations of microplastics (0.05%–0.25% w/w) is particularly intriguing as they seem to produce stronger effects than higher exposure levels (de Souza Machado, Lau, et al., 2018; Judy et al., 2019). The mechanism behind this response is not fully understood and might be linked to a combination of factors including the changes that microplastics triggered in the soil structure and the ability to microbes to colonize microplastic (de Souza Machado, Lau, et al., 2018).

There is conflicting evidence about the role of microplastics in the terrestrial carbon cycle (Gao et al., 2021; Ren et al., 2020; Rillig et al., 2021). Carbon is the main element in microplastics and unlike more labile carbon sources that originate from natural processes (photosynthesis or biomass production), this recalcitrant carbon pool is better able to resist degradation, thus affecting soil carbon storage (Rillig & Lehmann, 2020). The relative long persistency of microplastics in the soil gives microbial communities an opportunity to

TABLE 1 Summary of studies reporting changes in soil physical and chemical properties

Polymer	Shape	Concentration	Size (µm)	Scale of study	Induced significant changes in soil physicochemical parameters	Reference
Not specified	Fibers, fragments, particles	7100–42,960 particles/kg	50–1000	Field	72% of plastics are associated with soil aggregates and change the proportion of macro (>2 mm) and micro-aggregates (0.25–0.05 mm)	Zhang and Liu 2018
Polyester	Fibers	0.1, 0.3 (% w/w)	1170–4780	Field	Increase in volumes pores >30 µm	Zhang, Zhang, et al. (2019)
Polyester	Fibers	0.1, 0.3 (% w/w)	1170–4780	Lab	Increase in macro-aggregates (>2 mm)	Zhang, Zhang, et al. (2019); Zhang, Zhao, et al. (2019)
Acrylic polymers	Fibers	0.05, 0.10, 0.20, 0.40 (% w/w)	1260–9100	Lab	Decrease soil bulk density; Decrease in water stable aggregate	de Souza Machado, Kloas, et al. (2018); de Souza Machado, Lau, et al. (2018)
Polyamide	Beads	0.25, 0. 50, 1.00, 2.00 (% w/w)	15–20	Lab	Decrease soil bulk density; Decrease in water stable aggregate; Increase evapotranspiration	de Souza Machado et al. (2019); de Souza Machado, Kloas, et al. (2018); de Souza Machado, Lau, et al. (2018)
Polyester	Fibers	0.05, 0.10, 0.20, 0.40 (% w/w)	1540–6300	Lab	Decrease in soil bulk density; Increased water-holding capacity; Decrease in water stable aggregate; Increased evapotranspiration	de Souza Machado et al. (2019); de Souza Machado, Kloas, et al. (2018); de Souza Machado, Lau, et al. (2018)
Polyethylene	Fragments	0.25, 0. 50, 1.00, 2.00 (% w/w)	160–1200	Lab	Decrease in soil bulk density; Increase in water stable aggregates	de Souza Machado et al. (2019); de Souza Machado, Lau, et al. (2018)
Polypropylene	Fragments	2.0 (% w/w)	647–754	Lab	Decrease in soil bulk density; Increased evapotranspiration	de Souza Machado et al. (2019)
Polystyrene	Fragments	2.0 (% w/w)	547–555	Lab	Decrease in soil bulk density; Decrease in water stable aggregates; Increased evapotranspiration	de Souza Machado et al. (2019)
Polyethylene	Not specified	0.1 (% w/w)	0.48–316	Lab	Decrease in soil pH; Change in size distribution of water stale aggregates causing a decrease in macro-aggregates (<2 mm) relative to an increase in micro-aggregates (0.25–0.125 mm)	Boots et al. (2019)
Polyester	Not specified	0.1 (% w/w)	0.6–363	Lab	Alter the size distribution of water stable aggregates causing a decrease in macro-aggregates (<2 mm) relative to an increase in micro-aggregates (0.25–0.125 mm)	Boots et al. (2019)
Not specified	Fibers	0.001 (% w/w)	2000–7000	Lab	Alter the size distribution of water stable aggregates causing a decrease in macro-aggregates (<2 mm) relative to an increase in micro-aggregates (0.25–0.125 mm)	Boots et al. (2019)

evolve and exploit these additional sources of carbon (Rillig, 2018). Microbial communities are known to develop abilities to degrade other recalcitrant organic compounds, for example, polycyclic aromatic hydrocarbons (PAHs) through multiple metabolic pathways (Ghosal et al., 2016; Posada-Baquero & Ortega-Calvo, 2011), suggesting that they also have the ability to degrade microplastics. The above findings provide strong incentives to further study the impacts of microplastics on soil microbial community functions related to biogeochemical cycling. Notably, the quantitative implications of these findings for carbon cycling have not yet been explicitly accounted for (Huerta Lwanga et al., 2018; Rillig et al., 2021; Tian et al., 2017; Zhang, Zhao, et al., 2019).

7 | MICROPLASTICS DISPERSION AND FRAGMENTATION IN SOIL

Once microplastics reach the soil, multiple abiotic and biotic factors can influence their dispersion and fragmentation. Dispersion of microplastics can be enhanced by physical features of the soil, for example, the presence of macropores and fissures (appearing during dry season), which act as a conduit between the surface and deep layers (Rillig, Ingraffia, et al., 2017). Physical disturbance caused by agricultural practices (plowing, tilling and crop harvesting) can also facilitate the transfer of microplastics from the soil surface into deeper soil layers (Rillig, Ingraffia, et al., 2017). Soil biota can also transport microplastic and the outcomes vary according to the niche occupied by the organisms (Maaß et al., 2017). In laboratory conditions, larger soil-dwelling organisms such as earthworms transport microplastics over relatively larger distances (reaching 10 cm; Rillig, Ziersch, et al., 2017) while smaller organisms have more restricted ranges, which was about 3 cm for collembola (Maaß et al., 2017). Earthworms are efficient carriers of microplastics of less than 50 µm in size and facilitate the incorporation of plastics particles in the soil (Huerta Lwanga et al., 2016; Huerta Lwanga, Gertsen, et al., 2017). They contribute to the spreading of microplastics through their casts following ingestion and by their burrowing activities (Huerta Lwanga et al., 2016; Huerta Lwanga, Gertsen, et al., 2017). Predator–prey interactions can also enhance the dispersion of microplastics (Zhu, Bi, et al., 2018). Specifically, predatory mites (*Hypoaspis aculeifer*) trigger predator avoidance behavior in prey species (collembolan; *Folsomia candida* and oribatid mite; *Damaeus exspinosus*) and the increased prey activity concomitantly led to increased microplastics dispersal at a local scale.

Microplastics undergo ageing and weathering processes in the soil (i.e., fragmentation and degradation), which leads to a gradual decrease in particle size (Ng et al., 2018). Weathering can be more rapid in soil than water (Duan et al., 2021). As plastic is generally resistant to degradation (Zhang et al., 2021), microplastics in soil may be more prone to breakdown by fragmentation (Lambert et al., 2014; Zhou, Wang, et al., 2020). Microplastics in the uppermost soil layers are more susceptible to photo- and thermo-oxidation (Ng et al., 2018). Photooxidation occurs when plastic polymers absorb high-energy radiation (especially ultraviolet radiation 290–400 nm), which excites electrons to higher energy levels and eventually gives rise to the formation of free oxygen radicals (Lambert et al., 2014). Once free oxygen radicals are formed, they trigger bond (C–H and C–C) cleavages in the polymer chain resulting in decreased tensile strength and molecular weight (Singh & Sharma, 2008). Thermal oxidation causes bond scissions of the polymer chains when overheated (Lambert et al., 2014). Thermal oxidation affects the entire polymer and results in a decrease in crystallinity, loss of tensile strength and an increase in crack formation (Arkatkar et al., 2009). Other fragmentation processes that can occur in deeper soil layers include high impact mechanical disturbances associated with tillage that can shred microplastic into smaller pieces (Astner et al., 2019) and contact with water that can increase embrittlement (Julienne et al., 2019). Water-induced weathering can further contribute to fragmentation; prolonged contact with water triggers the formation of micro-cavities on the surface of polymers that eventually form larger cracks (Julienne et al., 2019).

Soil biota can also contribute to reducing the size of microplastics. Some fungi and bacteria can use the backbone (C–H) structures of microplastic as a carbon source (Huerta Lwanga et al., 2018; Tian et al., 2017). Bacterial communities isolated from the digestive tract of earthworms (*Lumbricus terrestris*), which were previously exposed to microplastics could reduce the size polyethylene microplastic (Huerta Lwanga et al., 2018). The isolated bacterial community consisted of Actinobacteria (*Microbacterium awajiense*, *Rhodococcus jostii*, *Mycobacterium vanbaalenii* and *Streptomyces fulvissimus*) and Firmicutes (*Bacillus simplex* and *Bacillus* sp.). The bacterial community was able to drastically reduce the size distribution of microplastics and even led to the formation of nanoplastics after a 4-week incubation period (Huerta Lwanga et al., 2018). Decrease in size (approximately 10%) was also observed after microplastic fibers were ingested by snails (*Achatina fulica*); however, the contributions of snail gut bacteria or other digestive processes to the observed size reduction are currently unknown (Song et al., 2019).

8 | TROJAN HORSE FOR OTHER CONTAMINANTS

Microplastics can act as a vector for various extrinsic contaminants, herbicides (Hüffer et al., 2019), hydrophobic organic pollutants (Wang et al., 2019) and heavy metals (e.g., zinc; Hodson et al., 2017), and thereby influence their distribution in soils (Ming Zhang & Xu, 2020). Microplastics also contain diverse intrinsic contaminants that are added during manufacturing (Campanale et al., 2020; Hahladakis et al., 2018). Results of studies on microplastics as a vector for extrinsic contaminants in terrestrial environments showed a more complex picture than the relatively clear role of microplastics as vectors for other contaminants observed in aquatic ecosystems (Hartmann et al., 2019). Wang et al. (2019) reported that polyethylene and polystyrene microplastics neither acted as carriers for hydrophobic organic contaminants nor enhanced their

uptake by earthworms (*Eisenia fetida*). However, Hodson et al. (2017) showed that microplastics can act as a vector for zinc in earthworms (*Lumbricus terrestris*). Hüffer et al. (2019) demonstrated that polyethylene microplastics can reduce the sorption capacity of soil organic matter for two herbicides (atrazine and 4-(2,4-dichlorophenoxy) butyric acid), thereby increasing their mobility and the likelihood that they will interact with non-target organisms (Hüffer et al., 2019).

Intrinsic contaminants include substances (up to 70% w/w) added to plastic polymers to improve their durability, flexibility and other desirable properties (Geyer et al., 2017; Hahladakis et al., 2018). Once in the environment, intrinsic contaminants may leach from the plastic matrix where they can have adverse effect on biota and ecosystem function (Stone et al., 2020). Phthalates and bisphenol A are of special concern as they are well-known endocrine disruptors (Erkekoglu & Kocer-Gumusel, 2016). The combined effects of microplastics and their additives are incompletely known (Campanale et al., 2020). However, the risk posed by endocrine disrupting additives alone on the reproductive health of terrestrial organisms is well founded (de Souza Machado, Kloas, et al., 2018; Zhang, Yang, et al., 2020). These results highlight the importance of additional studies addressing interactions between microplastics and other contaminants that depend on both their chemical properties and the prevailing environmental conditions (Hüffer et al., 2019; Wang et al., 2019).

9 | THE NEED TO MOVE TOWARD ECOLOGICAL STUDIES

9.1 | Microplastic as a stressor

To date, most studies addressing the effects of microplastics in terrestrial systems have been based on single species testing performed in controlled laboratory conditions and over relatively short exposure times (Figures 3b and 4a). Scaling the mortality, physical and reproductive impairment observed under controlled, short-term artificial conditions used in ecotoxicological assays to the ecosystem level is challenging and often falls short (Forbes et al., 2017). The scaling challenges arise from multiple sources including the community structure of real ecosystems, the complex structure of the natural environment and the time scale over which effects are manifested (Forbes et al., 2016, 2017; Kéfi et al., 2016). The complex structures of ecosystems consisting of multiple populations of species that collectively form communities where numerous interactions and self-reinforcing feedbacks that vary across time and space makes it hard to properly contextualize results from single species studies (Forbes et al., 2016, 2017; Kéfi et al., 2016; Windsor et al., 2018). Therefore, to reduce the potential for ecological surprise, it is imperative to design studies which can detect ecological effects of microplastics in realistic terrestrial (model) ecosystems. With some exceptions, for example, Yan et al. (2020), who worked with seven vascular plant species, most studies of community effects have focused on microbes. Many of these studies showed profound changes in microbial community composition when exposed to microplastics

(de Souza Machado et al., 2019; de Souza Machado, Lau, et al., 2018; Judy et al., 2019; Liu et al., 2017; Ng et al., 2021; Ren et al., 2020). Microplastics have the potential to generate effects that stretch beyond changes in species richness and community structures and impact ecological functions that species fulfill, commonly known as biodiversity–ecosystem functioning relationship (Hooper et al., 2005; Loreau et al., 2001). Multiple lines of evidence consistently show a coupling between biodiversity and ecosystem functioning (Balvanera et al., 2006; Cardinale et al., 2011; Gonzalez et al., 2020). Adopting the biodiversity–ecosystem functioning approach is necessary if we are to adequately evaluate whether or not microplastics in the terrestrial environment pose a risk of ecological surprises that can lead to failure in key ecosystem functions including nutrient recycling, pollination, seed dispersal and energy transfer trophic interactions, that are benchmarks of healthy terrestrial ecosystems (Brockerhoff et al., 2017; Sekercioglu, 2010).

9.2 | Risk of ecological surprises and implications for agroecosystems

The potential consequences of ecological surprise are especially acute in terrestrial agroecosystems, given their role in providing the vast majority of the human food supply. Current agricultural practices, plastic mulching and the application sewage sludge as fertilizers, are considered as major entry routes for microplastics into terrestrial ecosystems (Büks & Kaupenjohann, 2020; Corradini et al., 2019; Nizzetto et al., 2016). However, microplastics do not occur in isolation; other environmental pressures resulting from natural disturbances, for example, drought, flood and pest infestation (Food & Agriculture Organisation, 2017), anthropogenic activities, for example, global warming (Gornall et al., 2010), pesticide use (Aktar et al., 2009), invasive species (Paini et al., 2016) and heavy metal contamination (Li et al., 2019) also occur in agroecosystems. As microplastics weather and fragment into smaller sized particles, their surface area and surface reactivity increase, thus allowing multiple interactions with other contaminants including their co-transport (Hale et al., 2020). Microplastics alone are likely to cause sub-lethal effects (Figure 5) but the interactions with other co-occurring stressors can result in non-additive effects where their cumulative impacts become difficult to predict and can potentially lead to ecological surprises (Christensen et al., 2006). Previous studies have demonstrated that interactive effects of stressors can lead to unexpected effects in terrestrial ecosystems. For example, the interactive effects of temperature, precipitation, ozone and carbon dioxide on hard-wood forests led to an increase of 20% in the net ecosystem carbon exchange, which defied the predicted decrease (29%) when each factor was modelled separately (Hanson et al., 2005). This increase in the net ecosystem carbon exchange indicates that more carbon will be sequestered in tree biomass than being emitted by respiration in the future where temperature and winter precipitations are expected to increase by 4°C and 20%, respectively (Hanson et al., 2005). In another example, the effects of nutrient enrichment

were found to intensify drought effects on grassland communities (Tilman & Downing, 1994). Therefore, it is crucial to investigate how plausible interactions between microplastics and other stressors can impact terrestrial biodiversity and the ecological functions they fulfill. This is particularly important for agroecosystems, as crop production is dependent on multiple interacting ecological functions including nutrient recycling, symbiosis and pollination that are performed by terrestrial organisms (de Souza Machado et al., 2019; Oliveira et al., 2019). Studying the cumulative effects of microplastics and other stressors in realistic ecological contexts is needed if we are to disentangle their impacts on biodiversity–ecosystem functioning, and to subsequently evaluate the risk of ecological surprises in agroecosystems (Christensen et al., 2006).

9.3 | Trophic transfer and food web stability

While studies of the transfer of microplastics across trophic levels in terrestrial system are scarce, microplastics have been found in the digestive tracts of carnivorous terrestrial birds, which suggests that they can be transferred up the food chain (Carlin et al., 2020; D'Souza et al., 2020; Zhao et al., 2016). Bioaccumulation of microplastics in terrestrial organisms was first reported by Huerta Lwanga, Gertsen, et al. (2017) and Huerta Lwanga, Mendoza, et al. (2017). The authors documented a gradual increase in microplastic concentrations from the soil (0.87 ± 1.9 particles/g) to earthworm casts (14.8 ± 28.8 particles/g) and finally in chicken feces (129.8 ± 82.3 particles/g). However, significant knowledge gaps remain regarding the potential effects of microplastics on terrestrial food web structure (the flow of energy and materials; White et al., 2007) and food web stability (maintaining longer food chains; Sterner et al., 1997). Terrestrial and aquatic food webs are generally intertwined by organisms, which are mobile and occupy large spatial area (e.g., birds and winged insects), and can seamlessly transit ecotones (Knight et al., 2005). A recent study (D'Souza et al., 2020) showed that microplastics can be transferred from aquatic to terrestrial ecosystems, when aquatic insects (Ephemeroptera and Trichoptera; 26 ± 3 particles/g) are consumed by passerines (*Cinclus cinclus*; regurgitated pellets 16 ± 3 particles/g, feces 8 ± 2 particles/g).

10 | UNCERTAINTIES ABOUT NANOPLASTICS IN TERRESTRIAL ENVIRONMENTS

The effects of nanoplastics, a subfraction of microplastics smaller than 0.1 µm, on terrestrial ecosystems are even more uncertain (Wahl et al., 2021). Apart from their smaller size, nanoplastics have different surface properties, that is, large surface areas, and higher degrees of curvature, reactivity and charge (Wagner & Reemtsma, 2019). These properties enable them to cross biological membranes and adsorb other contaminants (de Souza Machado, Kloas, et al.,

2018; Roach et al., 2006). The combination of different physical and chemical properties of nanoplastics and the almost complete absence of studies on their behavior in the terrestrial environment increases the risk for ecological surprise through direct or indirect deleterious effects on terrestrial organisms (de Souza Machado, Kloas, et al., 2018; Oliveira et al., 2019) which may then impact food production. The available studies are reviewed by Wang et al. (2021) who note that toxic effects on organisms and reduced plant growth have been observed in the laboratory. Outcomes included a range sub-lethal effects leading to reduction in biomass, oxidative stress, behavioral changes, genotoxicity, alterations in gut microbiota and reproductive impairment (Jiang et al., 2019; Kim et al., 2020; Le et al., 2018; Zhu, Fang, et al., 2018). However, these short-term, single-species studies were performed using a single polymer (polystyrene) under laboratory conditions and can thus be subject to the same uncertainties as the majority of studies on microplastics in terrestrial environments.

We currently have limited knowledge about the modes of action and effects for many nanoplastics (Wagner & Reemtsma, 2019). However, these knowledge gaps can be overcome using existing knowledge about the ecological consequences of nanomaterials. Nanomaterials and nanoplastics share similar characteristics. They are of a similar size, vary in shape and have comparable surface properties (Bundschuh et al., 2018; Pitkethly, 2004). Owing to these similarities in properties, some of the progress achieved over the last decade of research on the fate and effects of nanomaterials in the environment (Bundschuh et al., 2018) can potentially be applicable to nanoplastics. The range of effects reported for nanomaterials on different terrestrial organisms include oxidative stress (Rico et al., 2015), shifts in bacterial community and functioning (Ge et al., 2011), impaired reproduction of springtails (McKee et al., 2017) and hampered development in plants (Colman et al., 2013; Lee et al., 2010). In addition, uptake and translocation of nanomaterials in plants (Tripathi et al., 2017; Wang et al., 2012) and the effect of environmental factors (e.g., soil texture, organic matter concentration and pH) on nanomaterial bioavailability to soil organisms (Bundschuh et al., 2018; Cornelis et al., 2014; Tourinho et al., 2012) are relatively well described. Therefore, it is likely that some of the effects of nanoplastics on the terrestrial environment can be predicted using previous research on nanomaterials (Hüffer et al., 2017).

11 | CONCLUSIONS AND OUTLOOK

Research on microplastics in the terrestrial environment is still in its infancy but has gained momentum during recent years. Here, we summarized the findings of previous research on microplastics in terrestrial environments and highlighted the need to move away from single-species testing and begin to address ecological effects. Such a transition is needed if we are to avoid ecological surprises through a better understanding of the potential implication of microplastics on terrestrial biodiversity and the terrestrial ecosystem functions

relevant for sustaining crop production in agroecosystems. Based on our discussion above, we identify some pressing research priorities:

- Reevaluation of the risk posed by sewage sludge amendment of agroecosystems. Recent evidence from aquatic ecosystems showed that biofilms, which developed on the surface of microplastics can include bacteria with AMR genes (Guo et al., 2020; Yang et al., 2020). Besides creating an entry route to the environment for microplastics, sewage sludge amendment can promote the spread of bacteria with AMR genes in agroecosystems and represent a risk for global health.
- Establishment of long-term agricultural experimental plots to provide a better understanding of microplastic effects on crop production and the possible bottom-up effects of microplastics on soil structure and microbial communities (Boots et al., 2019; de Souza Machado et al., 2019). Long-term agricultural experimental plots have been instrumental in the past to optimize the use of fertilizers and herbicides and help to understand the damage caused by pathogens (Johnston & Poulton, 2018).
- Evaluation of the role microplastics as a stressor and their potential long-term impacts on terrestrial communities (microbes, invertebrates, mammals and birds) and food web stability.
- Evaluation of the cumulative effects of microplastics and other stressors on terrestrial biodiversity and ecosystem functioning.

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DATA AVAILABILITY STATEMENT

Data sharing is not applicable to this article as no new data were created or analyzed in this study.

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