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Underestimated and ignored? The impacts of microplastic on soil invertebrates—Current scientific knowledge and research needs

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The accumulation of plastics in the environment is a major problem in the Anthropocene. As most plastic is produced, used and discarded on land, ~4–23 times more plastics are deposited in soils than in the oceans. However, there is far too little knowledge on the ecological consequences of plastic pollution, especially for soil ecosystems. Microplastics (<5 mm), whether derived from larger plastic pieces through physical, chemical and biological degradation or produced as primary particles, is of considerable interest, as they can be ingested by organisms at the basis of the trophic net and transferred to higher trophic levels. Nonetheless, although the assessment of microplastic effects on soil invertebrates is of undeniable relevance, most studies have focussed on nano- and microplastics in aquatic environments. This review examines the current state of knowledge regarding the effects of microplastics on soil invertebrates. As part of the soil biota, these organisms are of utmost importance for carbon cycling, respiration and biodiversity. Based on strict quality criteria, the data of 45 papers reporting ecotoxicological effects on soil invertebrates were analyzed, considering various test organisms and types of microplastic (in terms of polymer, shape and size). However, although different impacts were demonstrated, a deduction of general effect tendencies of microplastics in soils was difficult due to the scarcity of data and the use of diverse methodological setups. Moreover, almost all experiments were based on short-term single-species testing involving only a small number of species and single microplastic types. The review concludes with a discussion of the remaining knowledge gap and the needs for a standardized approach allowing an ecologically relevant risk assessment of the impacts of microplastic on invertebrates in terrestrial ecosystems.

KEYWORDS

invertebrates, mesofauna, microplastic, nanoplastic, soil, toxicity

1 Introduction

Environmental contamination with plastic has rapidly become a major problem of global proportion. The mass production of plastics started in the 1940s and has steadily increased (Thompson et al., 2009a; Cole et al., 2011; Hohn et al., 2020; PlasticsEurope, 2020), given the numerous desirable features of this material, including its low-cost, durability, lightweight and resistance to biodegradation. However, the recalcitrance of plastic has led to its accumulation in the environment and therefore its environmental risk (Thompson et al., 2005; Gautam et al., 2007; Shah et al., 2008; Barnes et al., 2009; Thompson et al., 2009b; Imhof et al., 2013). In 2020, ~367 million tonnes of plastics were produced globally (PlasticsEurope, 2021). About 54% of anthropogenic waste discarded into the environment consists of plastic (Hoellein et al., 2014). In the environment plastics can be found in different particle sizes: macroplastic (>25 mm), mesoplastic (5–25 mm), microplastic (<5 mm) and nanoplastics (<100 nm) (Zalasiewicz et al., 2016; Horton et al., 2017; Alimi et al., 2018). The focus of this study is microplastics, which are highly abundant in the environment, consist of a wide range of polymers (e.g. polyethylene, polystyrene, polypropylene) and forming various shapes (e.g. beads, fragments, fibers or films) (He et al., 2018). Microplastics are also defined according to their source: Primary microplastics are plastic particles produced directly for commercial or industrial use and most commonly consists of polypropylene, polyethylene and polystyrene (Horton et al., 2017; Helmberger et al., 2020). Secondary microplastics emerge through the physical, chemical and biological degradation of macroplastics (Thompson et al., 2005; Ryan et al., 2009; Gewert et al., 2015). Both plastic types can reach the soil via improper waste management, tire abrasion or the application of sewage sludge and waste water on agricultural fields, which is a common practice throughout the world (Blaesing & Amelung, 2018; Li et al., 2018; Corradini et al., 2019).

Plastic production, usage and disposal mainly happen in terrestrial ecosystems rather than in oceans. It was estimated that 80% of the litter found in oceans derives from terrestrial sources (Jambeck et al., 2015). Terrestrial ecosystems have therefore a high risk of being contaminated with plastic debris, and, consequently, have to be considered as long-term sinks or at least intermediate transport pathways (Zubris & Richards, 2005; Rillig, 2012). In fact, soils contain 4–23 times more plastic than oceans (Horton et al., 2017). It is thus surprising that most studies have focussed on marine or freshwater systems rather than soil ecosystems (Andrady, 2011; Browne et al., 2011; Wright et al., 2013; Erkes-Medrano et al., 2015; He et al., 2018; Wang et al., 2019). The first study of microplastics in soil was published in 2012 (Rillig, 2012). The scarcity of studies has continued as a Web of Science query revealed that in 2021 there were 418 publications examining the effects of microplastics in the marine environment, with far fewer studies of freshwater systems (124 publications) and even fewer studies that focussed on soil (113 publications) (Figure 1). If the several false positive hits provided by the Web of science search for microplastic research in soils are excluded, a small number of 17 publications remain for the year 2021, which states a relative count of 0–3% between 2012 and 2021.

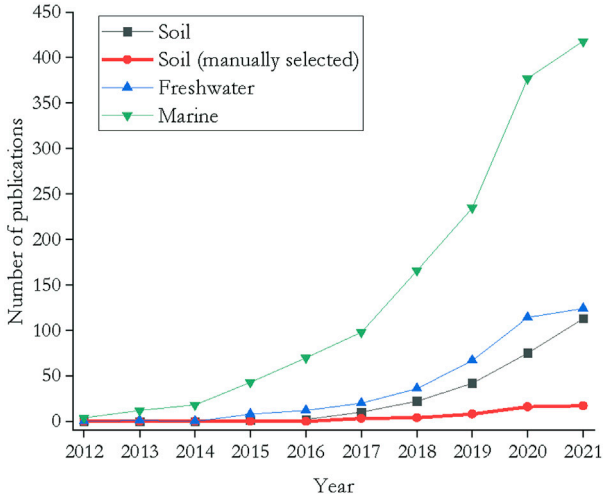

FIGURE 1
Number of studies published between 2012 and 2021 that examined the impact of micro- and nanoplastics in different environments. Search terms: for marine ecosystems (microplastic* OR nanoplastic*) AND (marine* OR sea*) AND (invertebrate* OR mesofauna* OR meiobenthos* OR organism*); for freshwater systems (microplastic* OR nanoplastic*) AND (freshwater* OR limn*) AND (invertebrate* OR mesofauna* OR meiobenthos* OR organism*); for soil (microplastic* OR nanoplastic*) AND (soil* OR terrestr*) AND (invertebrate* OR mesofauna* OR organism*). A manual selection of the hits was performed by excluding all publications that did not match the search terms (exclusion of false positives). Query performed on 12 October 2021.

Soil is proposed to provide most of the living biomass and to be one of the most biodiverse habitats, harbouring about 25% of all species on Earth. Soil organisms are key drivers for ecosystem functions and processes, such as nutrient cycling and water transfer (Bar-On et al., 2018; Bardgett & van der Putten, 2014; Bowker et al., 2010; Lavelle & Spain, 2001; Porazinska et al., 2003; Roger-Estrade et al., 2010; van den Hoogen et al., 2019). The heterotrophic soil biota are responsible for up to half of the soil respiration and soil animals including the mesofauna considerably determine decomposition rates of organic matter (Cisneros-Dozal et al., 2006; Frouz, 2018).

Arrived in the soil, plastics exert both short- and long-term impacts on soil organisms (Steinmetz et al., 2016). The ingestion of the omnipresent micro- and nanoplastic by soil invertebrates is of particular concern as these organisms form the base of the trophic net and allow the transfer of plastic particles to higher trophic levels (Duis & Coors, 2016; Carbery et al., 2018). The

physical or mechanical effects of microplastics occur outside the body of the organism or after ingestion (Anbumani & Kakkar, 2018; Padervand et al., 2020). The strength of the effect was shown to inversely correlate with the plastic size, with smaller particles more likely to be ingested (da Costa et al., 2016; Horton et al., 2017). However, independent of particle size, the ecotoxicological effects of microplastics could be accurately predicted by their total surface area (shown for PS beads; Mueller et al., 2020). Furthermore, in addition to direct toxicity, indirect effects, via the interaction of microplastics with the food of the tested organisms, have been reported (e.g. Mueller et al., 2020b; Rauchschwalbe et al., 2021; Besseling et al., 2013; Wright et al., 2013; Scherer et al., 2018; Oganowski et al., 2018). Finally, the additives used in plastic production, such as plasticizers and stabilizers, can leach into the soil and accumulate in the food chain, leading to chemical effects and biomagnification (Teuten et al., 2009; Haegerbaeumer et al., 2019a; Halle et al., 2020).

The ecotoxicological effects of microplastic on soil biota have been examined in several studies but drawing conclusions from those studies is hindered by differences in experimental methods and the concentration, type, shape and color of the tested microplastics (e.g. Boots et al., 2019). In addition, detailed quantitative analyses of the data are lacking (e.g. He et al., 2018; Wang et al., 2019; Kim Y.-N. et al., 2020). Therefore, in this review we evaluated the current state of knowledge regarding the effects of microplastics on soil invertebrates. To identify future research needs, the reviewed studies were differentiated in terms of (i) the test organism and toxicity endpoints and (ii) the type, shape, size and concentration of the microplastic. By this, we could compare effect threshold concentrations for various types of plastics and organisms and provided insights into possible toxicity mechanisms.

2 Methods of the literature research

The literature search was performed to identify all published results regarding ecotoxicological impacts of microplastics on soil invertebrates. The Web of Science and Google Scholar search was executed on the 12 November 2021 using 'Harzing's Publish or Perish' (Windows GUI Edition) 8.1.3625.7987 under usage of the following search term: TOPIC ((microplastic* OR nanoplastic*) AND (soil* OR terrestri*) AND (invertebrate* OR mesofauna* OR organism*)). As the first study of the effects of microplastics on soil organisms was published in 2012 (Rillig, 2012), the query was performed for the years 2012–2021. The publications in the resulting list were assessed based on several quality criteria. First, the publications within the raw list (I) were evaluated manually for their title and abstract to remove false positives (II). In the final step, the following quality criteria were applied: the use of soil as the testing medium in at least one experiment; a detailed description of type (shape and/or polymer), size and concentration of the microplastic and, if applied, the concentration of environmental chemicals (e.g. heavy metals); and the use of negative controls and statistical analyses (III). The steps comprising the literature search and the quality criteria are summarized in Figure 2.

Bar charts (OriginPro 2020 64-bit) were created to condense the general characteristics of the reviewed publications: experimental setup/soil, microplastic type/shape, the taxonomic group of the test organisms and the effect categories cellular response, (gut) microbiota, mortality, development, behavior, reproduction and community structure.

As not indicating an impact, any results regarding the ingestion, egestion, bioturbation of microplastic and the microplastic-related accumulation of other environmental chemicals were separately examined (Tables 1–3).

The effect-related results were condensed in lists (Tables 4–6) summarizing the tested species, associated taxa, the microplastic type (polymer, shape), size and concentration, the quantity and quality of the observed effects on the test organism, the corresponding author and the publication date. For clarity the effects were grouped within the above-mentioned parameter categories, whereby community structure was only physically affected, and can therefore be found in Table 4 only.

2.1 Chemical vs physical interactions

In this paper, we differentiate between chemical effects and the physical interactions of the studied organisms with microplastics. When considered together, they constitute an observed adverse effect (OAE) (according to ECETOC, 2018):

$$OAE = E_{iT} + E_{PInt}$$

where E_{iT} is the intrinsic toxic effect and E_{PInt} the physical interaction.

We provide following definitions for these terms:

Physical interactions are related to the physical properties of microplastics, including size, shape, rigidity, specific density, surface area and structure. In contrast to dissolved chemicals, insoluble, particulate microplastics can interact directly with the organism to cause physical injury or congestion. Indirect impacts can also occur, such as when microplastics interact with the soil to cause food dilution (Table 4).

The adverse effects of chemicals are related to the ability of the compounds to enter the cells of organisms, interact with

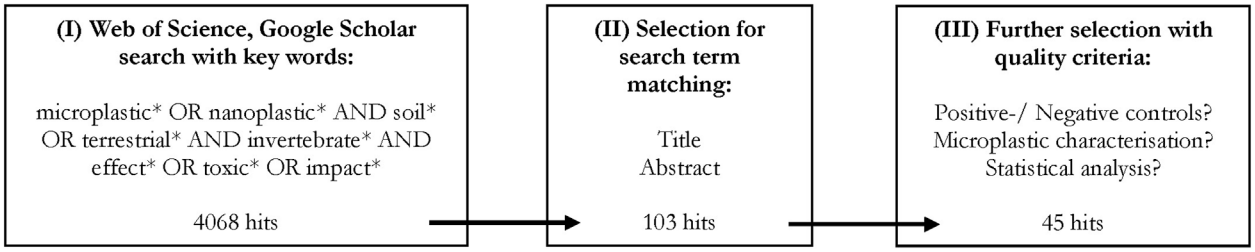

FIGURE 2
Step-by-step procedure used in the literature search. Web of Science and Google Scholar search performed on the 12.11.2021. With Harzing's Publish or Perish' (Windows GUI Edition) 8.1.3625.7987.

chemical receptors and thus cause a molecular, sub-organismic or organismic response. For microplastics, chemical effects arise from the leaching of chemicals associated with the microplastics, including additives (e.g. plasticizers or flame retardants), monomers, or compounds absorbed from the environment (e.g. polychlorinated biphenyls (PCBs), phenanthrene) (Tables 5, 6) (Teuten et al., 2009; Carbery et al., 2018).

3 Results

3.1 General characterization of the reviewed publications

From the initial raw list of 4,068 publications, 45 recently published papers (since 2017) were finally included in this literature review (Figure 2).

The publications describe experiments with exposure times between 3 min and 287 days. Physical effects of microplastics on soil invertebrates were examined in 32 of the 45 publications. Eighteen studies considered the effects of leached chemicals and environmental chemicals associated with microplastics. Overall, seven different effect endpoints were evaluated, with the most frequent being mortality, cellular response and development, analyzed in 29, 25, and 24 publications respectively, followed by reproduction and behavior (16 and 15 papers). The influence of microplastics on the invertebrate's (gut) microbiota was examined in six papers and the effects on the structure of the soil invertebrate communities in two papers (Figure 3).

Microplastics consisting of 13 different polymers were applied, namely polyethylene (PE, including high-density PE: HDPE, low-density PE: LDPE, linear low-density PE: LLDPE), polystyrene (PS), polyvinyl chloride (PVC), synthetic clothing fibers, polyethylene terephthalate (PET), polyester, polyamide (PA; nylon), polyacrylonitrile (PAN), synthetic clothing fibers (SCF), melamine-formaldehyde fragrance encapsulates (MFR), antifouling paint particles (APP) and biodegradable polylactic acid (PLA). PE was the most frequently studied polymer (22 publications) followed by PS, which was used in 11 publications. PP, tires abrasions, PVC and PET were applied in three to six publications each (Figure 4A).

Four different microplastic shapes were evaluated for their effects on soil invertebrates: spheres, fragments, fibers and films. Fragments were tested in 22 studies, spheres in 10, fibers in 9, and plastic films in one. In eight publications the microplastic shape was not specified (Figure 4B).

In the microplastic exposure experiments, the amounts of dry soil ranged from <1 g (four publications) to >1 kg (five publications). Most approaches used 100–600 g (21 publications), followed by 10–100 g (12 publications). In three papers the amount of soil was not specified. Only publications were considered in which the organisms were exposed to microplastics at least partly via the soil matrix (Figure 4C).

Most studies made use of field soils (28 publications), although the LUFA standard soil type 2.2 and OECD artificial soil were also commonly applied in nine and six publications, respectively. Additionally, a further artificial soil or other grounds were used (Figure 4D).

In general, 13 species of soil invertebrates were studied for their response to the addition of microplastics. In 34 publications annelids were tested, with the standard test organisms *Eisenia fetida* and *Folsomia candida* used in 15 and five publications respectively, collembolans in 12, *Porcellio scaber* (isopoda) in four, and the mite *Oppia nitens* in one. As representative of the nematodes and molluscs, *Caenorhabditis elegans* and *Achatina fulica* were used in three and one publication as test organism, respectively. In two studies, a natural occurring soil community was exposed to microplastics (Figure 5).

3.2 The uptake and bioturbation of microplastics through soil invertebrates

While ingestion, egestion and bioturbation are not strictly ecotoxicological effects, they contribute to the adverse effects of

TABLE 1 Ingestion, egestion and distribution of microplastics through soil invertebrates.

Taxa	*Species*	Polymer	Shape	Size	Concentration	Parameter		Comment	References
Annelida	*Lumbricus terrestris*	LDPE	Fragments	<150 μm	7–60% in dry surface litter w/w	Distribution	+	Amount of LDPE and organic matter in burrows	Huerta Lwanga et al. (2017)
							+	smaller LDPE particles in burrows than in surface litter; size-dependent transport	
		PE	Spheres	710–2800 μm	75–2625 particles (750 mg PE/2.5 kg fresh soil)	Distribution	+	Transport of PE in deeper soil layers	Rillig et al. (2017)
							+	transport of smallest PE class in deeper soil layers; transport depth- and size-dependent Further transport mechanism: adherence of PE to *L. terrestris*' body	
						Egestion	✓	Size-dependent; only 710–1400 μm PE was found in casts	
		PET	Fibers	633.7 ± 282.8 μm × 30 μm	50–5000 μg g^{-1}	Ingestion	✓		Lahive et al. (2021)
						Egestion	✓		
							-	500 μg g^{-1}	
							-	Fiber concentration in the faeces than in the soil	
		PS	Spheres	187 nm	22–2206 μg g^{-1}	Ingestion	✓		
						Egestion	✓		
							-	Sphere concentration in the faeces than in the soil	
	Eisenia andrei	LLDPE	Fragments	250–1000 μm	62.5–1000 mg kg^{-1}	Ingestion	✓		Rodriguez-Seijo et al. (2017)
		MFR	Spheres	10–25 μm	50 mg kg^{-1}	Ingestion	✓		Kuehr et al. (2021)
		PP (of face masks)	Fibers, fragments	<300 μm	1000 mg kg^{-1}	Egestion	✓		Kwak & An (2021a)
	Eisenia fetida	HDPE	Fragments	28–1464 μm	0.25% w/w	Ingestion	✓		Zhou et al. (2020)
		PE	Spheres	180–300 μm	1000 mg kg^{-1}	Ingestion	✓		Kwak and An (2021b)
		PE	Not sp	≤300 μm	0.1–10% w/w	Ingestion	✓	≥1% PE	Wang et al. (2019)
		PP	Fragments	8–1660 μm	0.25% w/w	Ingestion	✓		Li et al. (2021b)
		PP	Spheres	<150 μm	0.03–0.9% w/w	Ingestion	✓	Concentration-dependent Accumulation of PP in tissues; concentration-dependent	Zhou et al. (2020)
						Egestion	✓	Concentration-dependent	
		PS	Spheres	100 and 1300 nm	100 and 1000 μg kg^{-1}	Ingestion	✓	Concentration in intestines higher for 1300 nm than for 100 nm PS	Jiang et al. (2020)
		PS	Not sp	≤250 μm	0.1–10% w/w	Ingestion	✓	≥1% PS	Wang et al. (2019)

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TABLE 1 (*Continued*) Ingestion, egestion and distribution of microplastics through soil invertebrates.

Taxa	*Species*	Polymer	Shape	Size	Concentration	Parameter		Comment	References
		PS	Not sp	100 nm–100 μm	10 mg kg^{-1}	Ingestion	✓	Size-dependent: 10 μm > 100 μm >1 μm > 100 nm	Xu et al. (2021a)
		Tire	Fragments	<25 μm–2 mm	1–20% w/w	Egestion	✓	Concentration-dependent	Sheng et al. (2021)
	Enchytraeus crypticus	Nylon	Fragments	13–150 μm	2–12% w/w	Ingestion	✓	Size-dependent	Lahive et al. (2019)
		PE	Fibers	12 μm–24 mm	0.02–1.5% w/w; in soil/food	Ingestion	✓	Concentration-dependent	Selonen et al. (2020)
						Egestion	✓	Concentration-dependent	
	Metaphire guillelmi	HDPE	Fragments	25 μm	0.25% w/w	Ingestion	✓		Cheng et al. (2021)
		PP	Fragments	13 μm		Ingestion	✓		
Collembola	*Cryptopygus antarcticus*	PS	Foam	NA	NA	Ingestion	✓		Bergami et al. (2020)
	Folsomia candida	PP (of face masks)	Fibers, fragments	<300 μm	3 mg yeast per 1 mg PP	Ingestion	✓		Kwak and An (2021a)
						Egestion	✓		
		PVC	Not sp	80–250 μm	1 g kg^{-1}	Ingestion	✓		Zhu et al. (2018)
Isopoda	*Porcellio scaber*	PE	Fibers	12 μm–24 mm	0.02–1.3% w/w	Ingestion	✓	Concentration-dependent	Selonen et al. (2020)
Mollusca	*Achatina fulica*	PET	Fibers	1257.8 × 76.3 μm	0.01–0.17 g kg^{-1}	Ingestion	✓	PET observed in several digestive organs	Song et al. (2019)
						Egestion	✓		
							-	Excretion rate; concentration-dependent	
								Deterioration of PET after digestion	

not sp. = not specified. See section 3.1 for polymer abbreviations. Significant results represent comparison with the control treatment unless stated otherwise. The concentrations of the applied microplastic are those in dry soil unless stated otherwise. + = significant higher/more. . ., - = significant lower/fewer. . ., ✓= ingestion/egestion of microplastic particles.

TABLE 2 Accumulation of leached chemicals.

Taxa	*Species*	Material	Shape	Size	Conc	Leaching	Effect	Chemical accumulation	References
Annelida	*Eisenia fetida*	EPS	Fragments	<830–2000 μm	0.25% w/w	14, 21 days, 28 days	+	in HBCDD-EPS than in HBCDD treatment; highest values after 28 days	Li et al. (2019)
		Tire particles	Fragments	<25–2000 μm	1, 5, 10, 20% w/w	14, 28 days	+	Zn, Cd, Pb; time-dependent The smaller the tire particles, the higher the metal concentration	Sheng et al. (2021)
	Metaphire guillelmi	EPS	Fragments	<830–2000 μm	0.25% w/w	14, 21 days, 28 days	+	in HBCDD-EPS than in HBCDD treatment; highest values after 28 days	Li et al. (2019)

Abbreviations: dw = dry weight. The concentrations (conc.) of the applied microplastic are stated for dry soil. All effects are reported in comparison with the control treatment unless stated otherwise. + = significantly higher chemical accumulation.

TABLE 3 Microplastic influence on the accumulation of environmental chemicals.

Taxa	*Species*	Material	Shape	Size	Conc	Chem	Chem. con	Leaching	Effect	Chemical accumulation	References
Annelida	*Eisenia fetida*	PE	Not sp	<300 μm	7–30% w/w	Cd	2, 10 mg kg^{-1}	28 days	+	in tissues; correlation with PE/Cd conc. In soil	Huang et al. (2021)
		PE	Not sp	30, 100 μm	0.01, 0.05, 0.1% w/w	Cu	100 mg kg^{-1}		+		Li et al. (2021a)
									+	in Cu-PE than in Cu treatment; time-, PE conc.-dependent	
						Ni	40 mg kg^{-1}		+		
									+	Ni-PE than in Ni treatment	
		PP	Spheres	<150 μm	0.03–0.9% w/w	Cd	8.4 mg kg^{-1}	42 days	+	in Cd-PP than in Cd treatment	Zhou et al. (2020)
		PS	Not sp	100 nm–100 μm	10, 100 mg kg^{-1}	As, Cd	18.8, 0.2 mg kg^{-1}	3–21 days	+		Xu et al. (2021b)
									+	in MP than in NP treatments	
		PS	Not sp		10 mg kg^{-1}	PHE	5 mg kg^{-1}	21 days	+		Xu et al. (2021a)
									+	in PHE-PS than in PHE treatment	
		PVC	Not sp	<125 μm	0.1–1000 mg kg^{-1}	PFOA	10 mg kg^{-1}	28 days, 56 days	+	≥500 mg kg^{-1} PVC (bioconcentration factor)	Sobhani et al. (2021a)
						PFOS			+	≥1 mg kg^{-1} PVC (bioconcentration factor)	
						PFOA/ PFOS	5/5 mg kg^{-1}		+	1 mg kg^{-1} PVC (bioconcentration factor)	
	Enchytraeus crypticus	PA	Not sp	30 μm	1000 mg kg^{-1} dry	TC	20 mg kg^{-1}	21 days	+		Ma et al. (2020)
										no sig. Different between TC and TC-PA treatment	
		PVC	Not sp						+		
										no sig. Different between TC and TC-PVC treatment	
	Lumbricus terrestris	HDPE	Fragments	600 μm–2.04 mm	0.35% w/w	Zn	236–4,505 mg kg^{-1}	28 days	0 0	in gut, chloragog of HDPE (gut)	Hodson et al. (2017)

Abbreviations: MP = microplastic, NP = nanoplastic, not sp. = not specified, PHE = phenanthrene, TC = tetracycline, d = days. The concentrations (conc.) of the applied microplastic are stated for dry soil. All effects are stated in comparison with the control treatment unless stated otherwise. + = significant higher chemical accumulation. . ., 0 = no chemical accumulation. . .

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TABLE 4 Effects of macro-, micro- and nanoplastics on soil invertebrates.

Taxa	*Species*	Polymer	Shape	Size	Conc	Parameter	Effect	Endpoint	References
Acari	*Oppia nitens*	PE	Fibers	4–24 mm	0.5% w/w	Mortality	0		Selonen et al. (2020)
						Reproduction	0		
					0.5% w/w; in food	Mortality	0		
						Reproduction	0		
Annelida	*Aporrectodea rosea*	Biodegradable PLA	Not sp	0.6–363 μm	0.1% w/w	Development	0		Boots et al. (2019)
						Mortality	0		
		HDPE	Not sp	0.48–316 μm	0.1% w/w	Development	–		
						Mortality	0		
		Synthetic clothing fibers	Fibers	<2, 2–7 and >7 mm	0.001% w/w	Development	0		
						Mortality	0		
	Lumbricus terrestris	LDPE	Fragments	<150 μm	7–60% surface litter$_{dw}$ w/w	Behavior	+	burrow formation (7% LDPE)	Huerta Lwanga et al. (2017)
							+	bioturbation activity (7% LDPE)	
							0	burrow length/volume	
							+	burrow weight/wall density (7%, 45% LDPE)	
						Development	–	7% LDPE	
		PE	Spheres	710–2800 μm	75–2625 particles (750 mg MP/2.5 kg fresh soil)	Development	0		Rillig et al. (2017)
						Mortality	0		
		PET	Fibers	633.7 ± 282.8 μm × 30 μm	50, 500, 5000 μg g^{-1}	Behavior	0	avoidance behavior	Lahive et al. (2021)
						Development	0		
		Polyester	Fibers	361.6 ± 387 μm, × 40.7 ± 3.8 μm	0, 0.1, 1% w/w (0, 0.3 and 3 g)	Behavior	0	avoidance behavior	Prendergast-Miller et al. (2019)
						Cellular response	+	*hsp70* expression (1% polyester)	
							–	*mt-2* expression; dose-dependent increase	
							0	*sod-1* expression	
						Development	0		
						Mortality	0		
	Eisenia andrei	LLDPE	Fragments	250–1000 μm	62.5–1000 mg kg^{-1}	Cellular response	+	inflammatory reactions; LOEC 125 mg kg^{-1} dry soil	Rodriguez-Seijo et al. (2017)
							0	FTIR-ATR spectra	
						Development	0		

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TABLE 4 (*Continued*) Effects of macro-, micro- and nanoplastics on soil invertebrates.

Taxa	*Species*	Polymer	Shape	Size	Conc	Parameter	Effect	Endpoint	References
						Mortality	0		
						Reproduction	0		
		PP (of face masks)	Fibers, fragments	<300 μm	1000 mg kg^{-1}	Cellular response	–	intracellular esterase activity	Kwak & An, (2021a)
							0	oxidative stress (coelomocytes)	
							0	lysosomal stability	
							–	mature sperms, spermids (seminal vesicle score reduced to 0.8)	
							0	number of mature oocytes	
							0	tissue damage	
						Mortality	0		
	Eisenia fetida	LDPE	Beads	250 μm–1000 mm	180–200 beads in 500 mg kg^{-1}	Cellular response	0	TBARS content	Rodriguez-Seijo et al., (2018b)
						Development	0		
						Mortality	0		
				5 mm	8 beads in 500 mg kg^{-1}	Cellular response	0	TBARS content	
						Development	0		
						Mortality	0		
		LDPE (unaged)	Film	550–1000 μm	0.25% w/w	Cellular response	+	Hsp70, CRT, TCTP gene expression levels	Cheng et al., (2020)
							0	ANN gene expression level	
							0	8-OHdG level	
							+	ROS level	
							–	CAT, GST activity	
							+	MDA content	
							0	SOD activity	
						Mortality	0		
		LDPE (aged)	Film			Cellular response	+	ANN, CRT, Hsp70, TCTP gene expression levels	
							+	8-OHdG level	
							+	ROS level	
							–	CAT, GST, SOD activity	
							0	MDA content	
						Mortality	0		
		PE	Spheres	180–212, 250–300 μm	1000 mg kg^{-1}	Cellular response	0	ROS level	Kwak & An, (2021b)
							–	esterase activity (coelomocytes)	
							0	oogenesis	
							–	sperm density, mature sperm bundles	

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TABLE 4 (*Continued*) Effects of macro-, micro- and nanoplastics on soil invertebrates.

Taxa	*Species*	Polymer	Shape	Size	Conc	Parameter	Effect	Endpoint	References
							+	damage to sperm plasma membranes	
							+	disarranged male germ cells	
							+	disorder of tissue structure/intestinal tissue	
						Mortality	0		
		PE	Not sp	100 μm	0.05% w/w	Cellular response	+	CAT, POD, SOD activity, MDA content	Li et al., (2021a)
		PE	Not sp	≤300 μm	1, 5, 10, 20% w/w	Cellular response	+	CAT (20% PE), POD activity	Wang et al. (2019)
							–	GST, SOD activity (both 20% PE)	
							0	MDA conc	
						Development	0	growth	
		PP	Spheres	<150 μm	0.03, 0.3, 0.6, 0.9% w/w	Cellular response	+	LPO level	Zhou et al. (2020)
							+	GSH content; dose-dependent increase	
						Development	–	growth rate after 14 days (≥0.6% PP)	
						Mortality	+	after 42 days (≥0.3% PP)	
		PS	Spheres	100 and 1300 nm	100 and 1000 μg kg^{-1}	Cellular response	+	DNA damage (1000 μg kg^{-1} PS)	Jiang et al. (2020)
							+	DNA damage (1300 nm-sized PS) enlarged cells with irregular shapes/altered size of cell nuclei, increased intestinal cell lysis; more pronounced with 1300 nm PS (1000 μg kg^{-1})	
							–	SOD activity	
							+	GSH level (100 nm-, 1300 nm-sized (100 μg kg^{-1}) PS)	
						Development	+	growth rate	
						Mortality	–	100 μg kg^{-1}	
							+	1000 μg kg^{-1} (100 nm-sized PS)	
		PS (com.)	Fragments	65–125 μm	0.01–0.5% w/w	Cellular response	+	DNA damage in F0 (0.1, 0.5% PS), F1	Sobhani et al., (2021b)
						Development	0	growth	
						Mortality	0		
						Reproduction	–	F0, F1; dose-dependent decrease	
		PS (pure)	Fragments	65–125 μm		Cellular response	+	DNA damage in F0 (0.1, 0.5% PS), F1	
						Development	0	growth	
						Mortality	0		
						Reproduction	–	F0, F1; dose-dependent decrease	
		PS	Not sp	≤250 μm	1–20% w/w	Cellular response	+	CAT activity (≥5% PS)	Wang et al. (2019)
							0	GST activity	

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TABLE 4 (*Continued*) Effects of macro-, micro- and nanoplastics on soil invertebrates.

Taxa	*Species*	Polymer	Shape	Size	Conc	Parameter	Effect	Endpoint	References
							+	MDA conc. (20% PS); POD activity, dose-dependent increase	
							–	SOD activity (20% PS)	
						Development	0	growth	
		PS	Not sp	100 nm–100 μm	10 mg kg^{-1}	Cellular response	+	DNA damage	Xu et al., (2021a)
							–	CAT, GST gene expression level	
							+	Hsp70, MT, SOD, TCTP gene expressidsupon level	
						Development	–		
						Mortality	0		
		HDPE	Fragments	28–145, 133–415, 400–1464 μm	0.25% w/w	Cellular response	+	8-OHdG content	Li et al., (2021b)
							+	disturbance in metabolic pathways related to oxidative stress, inflammation, neurotoxicity	
							–	CAT activity (28–415 μm HDPE), GST, SOD activity	
								MDA content	
						Mortality	0		
		PP	Fragments	8–125, 71–383, 761–1660 μm	0.25% w/w	Cellular response	+	8-OHdG content	
							+	disturbances in metabolic pathways related to oxidative stress, inflammation, neurotoxicity	
							–	CAT, GST, SOD activity	
							–	MDA content	
						Mortality	0		
		PVC	Not sp	<125 μm	0.1–1000 mg kg^{-1}	Development	0		Sobhani et al., (2021a)
						Mortality	0		
						Reproduction	–	number of juveniles (1000 mg kg^{-1} PVC)	
		LDPE	Fragments	250–1000 μm	62.5–1000 mg kg^{-1}	Cellular response	0	FTIR-ATR spectra	Rodriguez-Seijo et al., (2018a)
								sig. Differences in NMR biochemical profile (62.5 mg kg^{-1} LDPE)	
							–	CAT activity (125 mg kg^{-1} LDPE); dose-dependent decrease	
							+	GST activity (1000 mg kg^{-1} LDPE); TBARS conc. (≥250 mg kg^{-1} LDPE); LDH activity (1000 mg kg^{-1} LDPE), correlation with GST activity; always dose-dependent increase	
						Development	0		
						Mortality	0		

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TABLE 4 (*Continued*) Effects of macro-, micro- and nanoplastics on soil invertebrates.

Taxa	*Species*	Polymer	Shape	Size	Conc	Parameter	Effect	Endpoint	References
	Enchytraeus crypticus	HDPE	Fragments	4 mm	0, 2, 4, 8% w/w	Cellular response	+	CAT, GST activity (4, 8% HDPE); always dose-dependent increase	Pflugmacher et al. (2020)
					0/2, 0/4, 0/8% w/w	Behavior	+	avoidance of the polluted site	
						Cellular response	+	CAT (8% HDPE), GST activity (4, 8% HDPE)	
					2/4, 2/8, 4/8% w/w	Behavior	+	avoidance of polluted site	
						Cellular response	+	CAT activity (8% HDPE) (2/8% pairing)	
							+	GST activity	
		PA	Not sp	30 μm	1000 mg kg^{-1}	Cellular response	+	abundance of ARGs	Ma et al., (2020)
							0	abundance of MGEs	
						Microbiota		sig. change of microbial composition	
							–	alpha diversity	
						Mortality	0		
						Reproduction	+		
		PE	Fibers	12 μm–2.87 mm	0.02–1.3% w/w	Mortality	0		Selonen et al., (2020)
						Reproduction	0		
					0.02–1.5% w/w; in food	Mortality	0		
						Reproduction	0		
				4–24 mm	0.02–1.3% w/w	Mortality	0		
						Reproduction	–	NOEC 0.06%	
					0.5% w/w; in food	Mortality	0		
						Reproduction	0		
		PVC	Not sp	30 μm	1000 mg kg^{-1}	Cellular response	+	abundance of ARGs	Ma et al. (2020)
							0	abundance of MGEs	
						Development	+		
						Microbiota		sig. change of microbial composition	
							–	alpha diversity	
						Mortality	0		
						Reproduction	–		
		PVC	Fragments	106–150 μm	9% w/w	Mortality	0		Lahive et al. (2019)
						Reproduction	0		
		Nylon	Fragments	13–150 μm	2–12% w/w	Mortality	0		

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TABLE 4 (*Continued*) Effects of macro-, micro- and nanoplastics on soil invertebrates.

Taxa	*Species*	Polymer	Shape	Size	Conc	Parameter	Effect	Endpoint	References
						Reproduction	–	13–18 µm (EC_{50} 108 ± 8.5 g kg^{-1}) and 90–150 µm; dose-dependent decrease	
	Metaphire guillelmi	HDPE	Fragments	25 µm	0.25% w/w	Gut microbiota	0	alpha diversity (richness, diversity)	Cheng et al. (2021)
						Mortality	0		
		PP	Fragments	13 µm		Gut microbiota	0	alpha diversity (richness, diversity)	
						Mortality	0		
Arthropoda	*Arthropod community*	LDPE	Fragments	0.3–400 µm	5, 10, 15 g m^{-2}	Community structure		sig. change in microarthropod community structure	Lin et al. (2020)
							–	abundance of dipteran larvae (15 g m^{-2} LDPE: −30.5 ± 9.3%), lepidopteran larvae (15 g m^{-2} LDPE: −41.5 ± 12.2%), ants (Hymenoptera) (15 g m^{-2} LDPE: −62.5 ± 7.5%), oribatid mites (15 g m^{-2} LDPE: −15.3 ± 5.7%)	
							0	non-oribatid mites	
		PE	Fibers	2–3 mm	0.4% w/w	Behavior	0	feeding activity of microarthropod community	Barreto et al. (2020)
							+	decomposition rate i.c.w. C, PP treatments	
						Community structure	0	oribatid mite species richness	
							0	MP type/length on abundance of oribatid mites (adults, immatures, adults + immatures) and microarthropod community composition	
							0	abundance of Mesostigmata, Prostigamata, Astigmata, Acari, other invertebrates, total microarthropodes	
		PP	Fibers	5–6 mm		Behavior	0	feeding activity of microarthropod community	
							0	decomposition rate	
						Community structure	0	oribatid mite species richness	
							0	MP type/length on abundance of oribatid mites (adults, immatures, adults + immatures) and microarthropod community composition	
							0	abundance of Mesostigmata, Prostigamata, Astigmata, Acari, other invertebrates, total microarthropodes	
Collembola	*Collembolan community*	LDPE	Fragments	0.3–400 µm	5, 10, 15 g m^{-2}	Community structure	0	abundance	Lin et al. (2020)
		PE	Fibers	2–3 mm ×22.92 ± 0.17 µm	0.4% w/w	Community structure	0	abundance	Barreto et al. (2020)
							0	MP type/length on abundance	
		PP	Fibers				0	abundance	

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TABLE 4 (*Continued*) Effects of macro-, micro- and nanoplastics on soil invertebrates.

Taxa	*Species*	Polymer	Shape	Size	Conc	Parameter	Effect	Endpoint	References
				5–6 mm× 33.33 ± 0.07 µm		Community structure	0	MP type/length on abundance	
	Folsomia candida	PE	Fibers	12 µm–24 mm	0.02–1.3% w/w	Mortality	0		Selonen et al. (2020)
						Reproduction	0		
					0.02–1.5% w/w; in food	Mortality	0		
						Reproduction	0		
				4–24 mm	0.02–1.3% w/w	Mortality	0		
						Reproduction	0		
					0.5% w/w; in food	Mortality	0		
						Reproduction	0		
			Spheres	<50, 50–500 µm	0.5, 1% w/w	Behavior	+	avoidance behavior; dose-dependent increase	Ju et al. (2019)
						Mortality	0		
					0.5% w/w	Gut microbiota		sig. change of microbial composition in guts	
							–	bacterial alpha diversity	
					0.005–1% w/w	Reproduction	–	reproduction rate (0.1% PE); dose-dependent decrease; calculated EC_{50} = 0.29%	
						Mortality	–	number of adults (1% PE)	
		PP (of face masks)	Fibers, fragments	<300 µm	1000 mg kg^{-1}	Cellular response	0	esterase activity	Kwak & An, (2021a)
							0	oxidative stress	
						Development	–	growth (decreased to 92.9%)	
						Mortality	0		
						Reproduction	–	decrease to 48.2%	
		NA	NA	NA	NA	Behavior	0	light avoidance	
		PVC	Not sp	80–250 µm	1000 mg kg^{-1}	Cellular response	+	level of $\delta^{15}N$ and $\delta^{13}C$ in tissues	Zhu et al. (2018)
							0	C:N ratio in tissue	
						Development	–		
						Gut microbiota		sig. change of microbial composition in guts	
							+	bacterial alpha diversity	
						Mortality	0		
						Reproduction	–		
	Lobella sokamensis	PE	Spheres	25–262 µm	1000 mg kg^{-1}	Behavior	–	movement activity	Kim & An, (2019)
		PS	Fragments	5–1155 µm		Behavior	–	movement activity	
			Spheres	0.5 ± 0.01 µm	4, 8 mg kg^{-1}	Behavior	–	movement activity	
Isopoda	*Porcellio scaber*	PE	Fibers	12–2870 × 6 µm	0.05–1.5% w/w		0	total/differential haemocyte count	Dolar et al. (2021)

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TABLE 4 (*Continued*) Effects of macro-, micro- and nanoplastics on soil invertebrates.

Taxa	*Species*	Polymer	Shape	Size	Conc	Parameter	Effect	Endpoint	References
						Cellular response	0	viability of haemocytes	
					0.5% w/w	Cellular response	+	PO-like activity	
		PE	Fibers	12 μm–24 mm	0.02–1.3% w/w	Behavior	0	feeding activity; dose-dependent decrease	Selonen et al. (2020)
						Cellular response	0	carbohydrate, lipid, protein content	
						Development	0		
						Mortality	0		
				4–24 mm		Behavior	0	feeding activity	
						Cellular response	0	carbohydrate, lipid, protein content	
						Development	0		
						Mortality	0		
		Tire particles	Fragments	<180 μm	0.05, 0.5, 1.5% w/w	Cellular response	+	total haemocyte count (0.05% TPs)	Dolar et al. (2021)
							0	differential haemocyte count	
							0	viability of haemocytes	
					0.05, 1.5% w/w	Cellular response	0	PO-like activity	
Mollusca	*Achatina fulica*	PET	Fibers		0.01–0.71 g kg^{-1}	Behavior	–	food intake (0.14, 0.71 g kg^{-1} PET)	Song et al. (2019)
						Cellular response		tissue damages in stomach and intestines (0.14, 0.71 g kg^{-1} PET)	
							0	liver, kidney histology	
							–	GPx content, T-AOC (0.71 g kg^{-1} PET)	
							+	MDA content (0.71 g kg^{-1} PET)	
						Egestion	–		
						Development	0	shell diameter or length	
						Mortality	0		
Nematoda	*Caenorhabditis elegans*	HDPE	Fragments	<250–1000 μm	0.01–1%	Reproduction	–	number of juveniles (1% HDPE)	Kim et al., (2020b)
		LDPE	Films	<630 μm		Reproduction	0	number of juveniles	
		PAN	Fibers		0.001–0.1%	Reproduction	–	number of juveniles (0.1% PAN)	
		PET	Fragments	<250–630 μm	0.01, 0.1, 1%	Reproduction	–	number of juveniles (0.1, 1% PET)	
		PP	Fragments	<250–1000 μm		Reproduction	–	number of juveniles (1% PP, <250 μm)	
		PS	Fragments			Reproduction	–	number of juveniles (1% PS)	
		PS	Spheres	60–76 nm	0.01–100 mg L^{-1}	Reproduction	–	100 mg l^{-1} PS	Kim et al., (2020a)

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TABLE 4 (*Continued*) Effects of macro-, micro- and nanoplastics on soil invertebrates.

Taxa	*Species*	Polymer	Shape	Size	Conc	Parameter	Effect	Endpoint	References
					0.01–100 mg kg^{-1}	Reproduction	–	100 mg kg^{-1} PS (EC_{50} > 100 mg kg^{-1}); toxic effect dependent on soil physicochemical properties	
				482–510 nm	0.01–100 mg L^{-1}	Reproduction	–	100 mg l^{-1} PS	
					0.01–100 mg kg^{-1}	Reproduction	–	10, 100 mg kg^{-1} PS (EC_{50} 14.23 mg kg^{-1}); toxic effect dependent on soil physicochemical properties	
		TP	Fragments	34–265 µm	1–10000 mg kg^{-1}	Development		short-term exposure (48 h)	Kim et al. (2021)
							–	growth (≥100 mg kg^{-1} TPs)	
						Mortality		short-term exposure (48 h)	
							0	survival rate	
								long-term exposure (10 days)	
							–	survival rate after 8 days (≥10 mg kg^{-1} TPs).; reduced to 27–45% after 10 d	
						Reproduction		Short-term exposure (48 h)	
							–	brood size (≥1000 mg kg^{-1} TPs)	
	Nematode community	LDPE	Fragments	0.3–400 µm	5, 10, 15 g m^{-2}	Community structure		sig. change in nematode trophic structure	Lin et al. (2020)
							–	nematodes abundance (5 g m^{-2} LDPE: −15.4 ± 5.9%, 10 g m^{-2} LDPE: −18.2 ± 4.3%, 15 g m^{-2} LDPE: −19.7 ± 3.4%)	
							–	abundance of omnivorous/predatory (15 g m^{-2} LDPE)/plant-feeding (10, 15 g m^{-2} LDPE) nematodes	
							0	abundance of fungal-/bacterial feeding nematodes	

The parameters are listed in alphabetical order. Abbreviations: conc. = concentration, dw = dry weight, fw = fresh weight, i. c.w. = in comparison with, MP = microplastics, not sp. = not specified, 8-OHdG (indicator for DNA damage) = 8-hydroxy-2′-deoxyguanosine, sig. = significant. The concentrations of the applied microplastic are stated for dry soil unless stated otherwise. 0 = no effect on. . ., + = significant higher/more, - = significant lower/fewer. The parameter development applies to the biomass (fw) unless stated otherwise. The abbreviations of evaluated genes and enzymes are spelled in full in the text above describing the physical effects of microplastics on soil invertebrates. All effects are stated in comparison with the control treatment unless stated otherwise.

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TABLE 5 Effects of plastic leachates on soil invertebrates.

Taxa	*Species*	Material	Shape	Size	Conc	Leaching	Parameter		Effects	References
Annelida	*Eisenia andrei*	APP	Fragments	<63 μm	0.01, 0.14%	3–56 d	Development	–	biomass (fw, 0.14% APPs after 56 days)	Soroldoni et al. (2021)
					0.01, 0.14, 1.5%		Mortality	+	with 1.5% APPs after 3 d	
							Reproduction	–	number of juveniles (0.01% APPs after 56 days and overall)	
								–	number of cocoons (0.01% APPs after 42 days)	
									no production of juveniles or cocoons with 0.014% or 1.5% APPs	
	Eisenia fetida	TP	Fragments	13–1400 μm	0.0048–3.0%	21 d	Gut microbiota	0	alpha diversity of bacteria	Ding et al. (2020)
									community of bacteria, fungi significantly changed	
								+	more sensitive towards TPs than soil microbiota	
							Mortality	–	survival rate (≥0.024% TPs)	
							Reproduction	–	with ≥0.12% TPs; dose-dependent decrease; reproduction logarithmic correlated with TP conc.	
		TP	Fragments	<25 μm	1–20%	14, 28 d	Cellular response	+	CAT activity, MDA conc.; GST activity after 28days; POD activity (≥10% TPs)	Sheng et al. (2021)
								–	SOD activity (≥10% TPs)	
				25–50 μm			Cellular response	+	CAT, POD activity (≥10% TPs); GST activity (≥5% TPs after 28 days); MDA conc	
								0	SOD activity	
				50–350 μm			Cellular response	+	CAT activity (≥10% TPs after 14 days); GST activity (≥5% TPs after 28 days); MDA conc. (≥10% TPs); POD activity	
								0	SOD activity	
				350 μm–2 mm			Cellular response	0	CAT, GST, SOD activity	
								+	MDA conc. (20% TPs after 28 days); POD activity (≥10% TPs after 28 days)	
	Enchytraeus crypticus	TP	Fragments	<180 μm	0.02–1.5%	21 d	Reproduction	–	number of juveniles (0.02, 1.5% TPs)	Selonen et al. (2021)
							Mortality	0		
		TP (in food)	Fragments				Reproduction	0	number of juveniles	
							Mortality	0		
Collembola	*Folsomia candida*	TP	Fragments			21 d	Reproduction	0	number of juveniles	Selonen et al. (2021)
							Mortality	0		
		TP (in food)	Fragments				Reproduction	0	number of juveniles	
							Mortality	0		
Isopoda	*Porcellio scaber*	LDPE (virgin)	Fragments	39.8 ± 8.8 μm <125 μm	0.02–1.5%	21 d	Cellular response	+	granulocytes (0.05, 1.5% LDPE)	Kokalj et al. (2021)
								0	haemocyte viability, total haemocyte count	
								–	semigranulocytes (0.05, 1.5% LDPE)	

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TABLE 5 (*Continued*) Effects of plastic leachates on soil invertebrates.

Taxa	*Species*	Material	Shape	Size	Conc	Leaching	Parameter		Effects	References
							Behavior	–	feeding rate (0.05, 1.5% LDPE)	
								0	feeding rate i.c.w. recycled LDPE	
							Mortality	0	i.c.w. C, recycled LDPE	
		LDPE (recycled)	Fragments	205 ± 144 µm <418 µm			Cellular response	0	granulocytes, haemocyte viability, total haemocyte count	
								–	semigranulocytes (0.05, 1.5% LDPE)	
							Behavior	0	feeding rate i.c.w. C, virgin LDPE	
							Mortality	0	i.c.w. C, virgin LDPE	
		TP	Fragments	<180 µm	0.02–1.5%	21 d	Behavior	0	feeding activity	Selonen et al. (2021)
							Cellular response	–	AChE activity (EC_{50} = 1.2%, NOEC = 0.5%)	
								0	electron transfer system	
Nematoda	*Caenorhabditis elegans*	HDPE	Fragments	<250–1000 µm	0.01, 0.1, 1%	24 h	Reproduction	–	number of juveniles (1% HDPE)	Kim et al., (2020b)
								0	number of juveniles (HDPE extract)	
								–	number of juveniles (HDPE extract bound to glass beads; corresponds to 1% PET)	
		HDPE				24 h (additive extracted HDPE)	Reproduction	–	number of juveniles after one extraction cycle (0.01, 0.1% HDPE)	
								0	number of juveniles after two extraction cycles	
		LDPE	Films	<630 µm		24 h	Reproduction	0	number of juveniles	
								0	number of juveniles (LDPE extract)	
								0	number of juveniles (LDPE extract bound to glass beads; corresponds to 1% LDPE)	
		LDPE			1%	24 h (additive extracted PP)	Reproduction	0	number of juveniles after one extraction cycle	
		LDPE			0.01, 0.1, 1%	6, 12, 18 days (one wet-dry cycle every 6 days)	Reproduction	–	number of juveniles after 6, 12, 18 days	
									higher toxicity with repeating wet-dry cycles	
		PAN	Fibers	<630 µm	0.001, 0.01, 0.1%	24 h	Reproduction	–	number of juveniles (0.1% PAN)	
								0	number of juveniles (PAN extract)	
								0	number of juveniles (PAN extract bound to glass beads; corresponds to 0.1% PAN)	
		PAN			0.1%	24 h (additive extracted PAN)	Reproduction	0	number of juveniles after one extraction cycle	
		PAN			0.001, 0.01, 0.1%	6, 12, 18 days (one wet-dry cycle every 6 days)	Reproduction	–	number of juveniles after 6 days (0.001, 0.01% PAN), 12days, 18 days	
									higher toxicity with repeating wet-dry cycles	

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TABLE 5 (*Continued*) Effects of plastic leachates on soil invertebrates.

Taxa	*Species*	Material	Shape	Size	Conc	Leaching	Parameter		Effects	References
		PET	Fragments	<250–630 μm	0.01, 0.1, 1%	24 h	Reproduction	–	number of juveniles (0.1, 1% PET)	
								0	number of juveniles (PET extract)	
								–	number of juveniles (PET extract bound to glass beads; corresponds to 1% PET)	
		PET			1%	24 h (additive extracted PET)	Reproduction	–	number of juveniles after one extraction cycle	
								0	number of juveniles after two extraction cycles	
		PP	Fragments	<250–1000 μm	0.01, 0.1, 1%	24 h	Reproduction	–	number of juveniles (1% PP, <250 μm); size-dependent effect	
								0	number of juveniles (PP extract)	
								–	number of juveniles (PP extract bound to glass beads; corresponds to 1% PP, <250 μm)	
		PP			1%	24 h (additive extracted PP)	Reproduction	0	number of juveniles after extraction cycles	
		PS	Fragments	<250–1000 μm	0.01, 0.1, 1%	24 h	Reproduction	–	number of juveniles (1% PS)	
								0	number of juveniles (PS extract)	
								–	number of juveniles (PS extract bound to glass beads; corresponds to 1% PS)	
		PS			1%	24 h (additive extracted PS)	Reproduction	0	number of juveniles after extraction cycles	
		TP	Fragments	34–265 μm	1–10000 mg kg^{-1}	0, 30, 75 days preinc. of MP with soil	Development		Short-term exposure (48 h)	Kim et al. (2021)
								–	growth (≥100 mg kg^{-1} TP)	
								–	growth (≥1 mg kg^{-1} TP); with preinc. of TP in soil for 30 and 75 d	
							Mortality		Short-term exposure (48 h)	
								0	survival rate	
							Reproduction		Short-term exposure (48 h)	
								–	brood size (reduced by 3–33%)	
								–	brood size (≥1000 mg kg^{-1} TP)	
								–	brood size (≥1 mg kg^{-1} TP); with preinc. of TP in soil for 30 and 75 days	
								–	pregnant individuals; preinc.-time-dependent decrease	
						0, 75 days preinc. of MP with soil	Mortality		long-term exposure (10 days)	
								–	survival rate (≥10 mg kg^{-1} TP after 8 days); survival rate reduced to 27–45% after 10 days	
								–	survival rate (10000 mg kg^{-1} TP after 6 days); with preinc. Of TP in soil for 75days; survival rate reduced to 17–50% after 10 d	

The parameters are listed in alphabetical order. Abbreviations: APP = antifouling paint particle, conc. = concentration, d = days, fw = fresh weight, i. c.w. = in comparison with, preinc. = preincubation, TP = tire particle. The concentrations of the applied microplastic are stated as % dry soil w/w or mg kg^{-1} dry soil. All effects are based on comparisons with the control treatment unless stated otherwise.

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TABLE 6 Influence of microplastics on the bioavailability of environmental chemicals in soil invertebrates.

Taxa	*Species*	Material	Shape	Size	Conc	Chem	Chem. con	Leaching	Parameter		Effects	References
Annelida	*Eisenia fetida*	LDPE (unaged)	Film	550–1000 μm	0.25% w/w	ATZ	0.02, 2.0 mg kg^{-1}	28 d	Cellular response	+	8-OHdG level	Cheng et al. (2020)
										+	ANN, CRT, Hsp70 gene expression levels	
										0	TCTP gene expression level	
										+	ROS level i.c.w. C, LDPE (aged/unaged) or ATZ treatment	
										–	CAT, GST, SOD activity	
										–	MDA content (ATZ 0.02 mg kg^{-1}-LDPE treatment)	
											IBR values higher	
									Mortality	0		
		LDPE (aged)	Film						Cellular response		sig. change of gene expression levels (ANN, CRT, TCTP, Hsp70)	
											IBR value slightly higher in ATZ 0.02 mg kg^{-1}-LDPE (aged) treatment	
										+	8-OHdG level	
										+	ROS level; i.c.w. C, LDPE (aged/unaged) or ATZ treatment	
										–	CAT, GST, SOD activity	
										–	MDA content (ATZ 2.0 mg kg^{-1}-LDPE (aged) treatment)	
											IBR values higher	
									Mortality	0		
		LDPE	Pellets	<1 mm	180–200 beads/500 mg kg^{-1}	CPF	4 L ha^{-1}	14 d	Behavior	+	earthworms at the bottom of the containers	Rodriguez-Seijo et al. (2018b)
									Cellular response	0	AChE activity	
										+	TBARS content	
									Development	–		
									Mortality	0		
				5 mm	8 beads/500 mg kg^{-1}				Behavior	+	earthworms at the bottom of the containers	
									Cellular response	–	AChE activity	
										–	TBARS content	
									Development	0		
									Mortality	0		

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TABLE 6 (*Continued*) Influence of microplastics on the bioavailability of environmental chemicals in soil invertebrates.

Taxa	*Species*	Material	Shape	Size	Conc	Chem	Chem. con	Leaching	Parameter		Effects	References
		PE	Not sp	<300 μm	7–30% w/w	Cd	2, 10 mg kg^{-1}	48 h	Behavior	+	avoidance rate	Huang et al. (2021)
								28 d	Cellular response	+	GSH, MDA content; always dose (PE)-dependent increase	
										–	POD, SOD activity	
									Development	–		
					7, 15% w/w		2, 10 mg kg^{-1}	28 d	Reproduction	–	cocoon reproduction	
					7–30% w/w		10 mg kg^{-1}		Cellular response	+	sperm damage; dose (PE)-dependent increase	
										+	epidermal necrosis, muscle fibroses; dose (PE)-dependent increase	
										+	abnormities of intestinal epithelial/chlorogenic tissue (vesicle enlargement, layer fibrosis, tissue necrosis/disintegration)	
		PE	Not sp	30 μm	0.05% w/w	Cu	100 mg kg^{-1}	7, 14, 21 d	Cellular response	+	CAT, POD, SOD activity; i.c.w. C, PE treatments	Li et al. (2021a)
										+	MDA content i.c.w. C, PE treatment; time-dependent increase	
				100 μm	0.01, 0.05, 0.1% w/w				Cellular response	+	CAT, POD, SOD activity; i.c.w. C, PE treatments	
										+	MDA content (i.c.w. C, PE treatments); time-dependent increase	
				30 μm	0.05% w/w	Ni	40 mg kg^{-1}		Cellular response	+	CAT, POD, SOD activity; i.c.w. C, PE treatments	
										+	MDA content (i.c.w. C, PE (30 μm; 100 μm, 0.05%) treatment); time-dependent increase	
				100 μm	0.01, 0.05, 0.1% w/w				Cellular response	+	CAT, POD, SOD activity; i.c.w. C, PP treatment	
										+	MDA content (i.c.w. C, PE treatment); time-dependent increase	
										–	MDA content i.c.w. 30 μm PE (0.05%)	
		PP	Spheres	<150 μm	0.03–0.9% w/w	Cd	8.4 mg kg^{-1}	42 d	Cellular response	+	GSH content, LPO level i.c.w. C, PP treatments	Zhou et al. (2020)
									Development	–	growth rate i.c.w. C, PP treatment	
									Mortality	+	after 42 d	
		PS	Not sp	10 μm	10, 100 mg kg^{-1}	As, Cd	0.2, 18.8 mg kg^{-1}	3–21 d	Cellular response	–	CAT, MT mRNA level	Xu et al. (2021b)
										+	GST, SOD mRNA level	

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TABLE 6 (*Continued*) Influence of microplastics on the bioavailability of environmental chemicals in soil invertebrates.

Taxa	*Species*	Material	Shape	Size	Conc	Chem	Chem. con	Leaching	Parameter		Effects	References
											abnormal epithelial cells, incomplete coelom tissue, exfoliation in intestinal epithelium	
										+	CAT activity after 3 days	
										–	CAT activity (10 mg kg^{-1} after 14days, 100 mg kg^{-1} after 21 days)	
										+	MDA content, SOD activity after ≥3 days	
										–	MT activity	
											sig. change in protein expression and metabolic profiles	
									Development	–	100 mg kg^{-1}	
									Mortality	0		
				100 μm					Cellular response	–	CAT, MT mRNA level	
										+	GST, SOD mRNA level	
										+	CAT activity after 3 days	
										–	CAT activity after ≥14 days	
										+	MDA content, SOD activity after ≥3 days	
										+	MT activity	
											sig. change in protein expression and metabolic profiles	
											abnormal epithelial cells, incomplete coelom tissue, exfoliation in intestinal epithelium	
									Development	0		
									Mortality	0		
				100 nm					Cellular response	–	CAT, MT mRNA level	
										+	GST, SOD mRNA level	
										+	CAT activity after 3, 7ays, 14 days (10 mg kg^{-1})	
										–	CAT activity after 21 dayays (10 mg kg^{-1})	
										+	MDA content after ≥3 days	
										+	MT activity	
										+	SOD activity after ≥14 days	
											sig. change in protein expression and metabolic profiles	

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TABLE 6 (*Continued*) Influence of microplastics on the bioavailability of environmental chemicals in soil invertebrates.

Taxa	*Species*	Material	Shape	Size	Conc	Chem	Chem. con	Leaching	Parameter		Effects	References
											abnormal epithelial cells, incomplete coelom tissue, exfoliation in intestinal epithelium	
									Development	0		
									Mortality	0		
		PS	Not sp	100 μm	10 mg kg^{-1}	PHE	5 mg kg^{-1}	21 d	Cellular response	+	DNA damage i.c.w. C, PS, PHE treatment	Xu et al. (2021a)
										–	CAT, GST mRNA level	
										–	CAT mRNA level i.c.w. PS treatment	
										+	Hsp70, MT, SOD, TSTP mRNA level i.c.w. C, PS treatments	
										–	CAT, POD, SOD activity	
										+	GSH content after 21 days	
										+	MDA content, TPC	
									Development	–		
									Gut microbiota		sig. change of composition	
									Mortality	0		
				10 μm					Cellular response	+	DNA damage i.c.w. C, PS, PHE and all other PHE-PS treatments	
										–	CAT, GST mRNA level i.c.w. C, PS treatments	
										+	Hsp70, MT, SOD, TSTP mRNA level i.c.w. C, PS treatments	
										–	CAT, POD, SOD activity	
										–	GSH content	
										+	MDA content, TPC	
									Development	–		
									Gut microbiota		sig. change of composition	
									Mortality	0		
				1 μm					Cellular response	+	DNA damage i.c.w. PS, PHE treatments	
										–	CAT, GST mRNA level i.c.w. C, PS treatments	
										+	Hsp70, MT, SOD, TSTP mRNA level i.c.w. C, PS treatments	

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TABLE 6 (*Continued*) Influence of microplastics on the bioavailability of environmental chemicals in soil invertebrates.

Taxa	*Species*	Material	Shape	Size	Conc	Chem	Chem. con	Leaching	Parameter		Effects	References
										–	CAT activity; GSH content after 21 days	
										+	MDA content	
										–	POD, SOD activity after 21 days	
										+	TPC after 21 d	
									Development	–		
									Gut microbiota		sig. change of composition	
									Mortality	0		
				100 nm					Cellular response	+	DNA damage in i.c.w. PS, PHE treatments	
										–	CAT, GST mRNA level i.c.w. C, PS treatments	
										+	Hsp70, MT, SOD, TCTP mRNA level i.c.w. C, PS treatments	
										–	CAT activity	
										–	POD/SOD activity after 21 days	
										–	GSH content after 14 days	
										+	MDA content	
										+	TP content after 14 d	
									Development	–		
									Gut microbiota		sig. change of composition	
									Mortality	0		
		PVC	Not sp	<125 μm	0.1–1000 mg kg^{-1}	PFOA	10 mg kg^{-1}	28days, 56 d	Development	0		Sobhani et al. (2021a)
									Mortality	0		
									Reproduction	–	number of juveniles (≥500 mg kg^{-1} PVC)	
						PFOS			Development	0		
									Mortality	0		
									Reproduction	–	number of juveniles (≥500 mg kg^{-1} PVC)	
						PFOA/ PFOS	5/5 mg kg^{-1}		Development	0		
									Mortality	0		

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TABLE 6 (*Continued*) Influence of microplastics on the bioavailability of environmental chemicals in soil invertebrates.

Taxa	*Species*	Material	Shape	Size	Conc	Chem	Chem. con	Leaching	Parameter		Effects	References
									Reproduction	–	number of juveniles (≥500 mg kg^{-1} PVC)	
	Enchytraeus crypticus	PA	Not sp	30 µm	1000 mg kg^{-1}	TC	20 mg kg^{-1}	21 d	Cellular response	+	abundance of ARGs and MGEs	Ma et al. (2020)
									Microbiota		sig. change of microbial composition	
										–	alpha diversity	
									Mortality	0		
									Reproduction	–		
		PVC							Cellular response	+	abundance of ARGs and MGEs	
										+		
									Development	+		
									Microbiota		sig. change of microbial composition	
										–	alpha diversity	
									Mortality	0		
									Reproduction	–		
	Lumbricus terrestris	HDPE	Fragments	600 µm–2.04 mm	0.35% w/w	Zn	236–4,505 mg kg^{-1}	28 d	Behavior	0		Hodson et al. (2017)
									Development	0		
									Egestion	0	of Zn	
									Mortality	0		
Arthropoda	*Porcellio scaber*	PE	Fibers	12–2870 µm	0.5% w/w	CPF	0.2–2.0 mg kg^{-1}	21 d	Cellular response	–	AChE activity (≥0.8 mg kg^{-1})	Dolar et al. (2021)
										+	THC (2.0 mg kg^{-1})	
										+	proportion of hyalinocytes (0.8 mg kg^{-1} CP-PE i.c.w. CP treatment)	
										0	proportion of granulocytes, semigranulocytes, haemocyte viability	
		Tire particles	Fragments	<180 µm					Cellular response	0	THC	
										+	proportion of granulocytes (0.4 mg kg–1)	
										–	proportion of semigranulocytes (0.2, 0.6 mg kg^{-1})	
										0	proportion of hyalinocytes, haemocyte viability	

The parameters are listed in alphabetical order. Abbreviations: ATZ = atrazine, C = control, CPF = chlorpyrifos, d = days, i. c.w. = in comparison with, not sp. = not specified PHE = phenanthrene. The concentrations of the applied microplastic are stated for dry soil unless stated otherwise. The parameter development applies to biomass (fw) unless stated otherwise. The chemical abbreviations are defined in the text. All effects are based on comparison with the control treatment unless stated otherwise.

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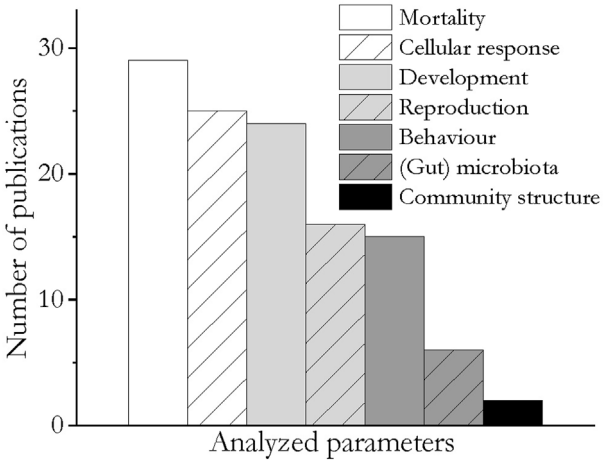

FIGURE 3
Bar chart of the parameters analyzed in the reviewed publications.

microplastics in soil (Table 1) via their incorporation and transport by soil invertebrates (Duis & Coors, 2016; Rillig et al., 2017). These processes were considered in 16 of 45 papers, 14 of which focussed on ingestion and/or egestion and the remaining two on bioturbation (Table 1). Microplastics of several different types, shapes and concentrations and ranging in size from 100 nm to 2800 µm were shown to be ingested and egested by annelids, arthropods and molluscs. Among the Annelida, *L. terrestris*, *E. andrei*, *E. fetida*, *E. crypticus* and *M. guillelmi* ingested and/or egested PE, PS, PP and MFR spheres with sizes between 100 nm and 2800 µm (Rillig et al., 2017; Jiang et al., 2020; Zhou et al., 2020; Kuehr et al., 2021; Lahive et al., 2021). Other studies demonstrated that annelids ingested and/or egested fragments (8–2000 µm) of LLDPE, PP, tires, nylon, HDPE (Rodriguez-Seijo et al., 2017; Lahive et al., 2019; Zhou et al., 2020; Kwak & An, 2021a; Li M. et al., 2021; Cheng et al., 2021; Sheng et al., 2021), fibers (12–2400 µm) of PET, PP, PE (Selonen et al., 2020; Kwak and An, 2021a; Lahive et al., 2021), PE beads (180–300 µm; Kwak & An, 2021b) and particles (100 nm–300 µm) of PE and PS of unspecific shape (Wang et al., 2019; Xu et al., 2021a). The arthropods *F. candida* (Zhu et al., 2018; Kwak and An, 2021a) and *P. scaber* (Selonen et al., 2020) ingested and/or egested PP fiber-fragment mixtures (<300 µm), PVC particles (80–250 µm) and PE fibers (12–2400 µm). The study of Bergami et al. was the only one that analyzed the uptake of PS foam (size not specified) in a natural occurring population of the collembolan *C. antarcticus* (Bergami et al., 2020). The study of Song et al. (2019) was the only one that focussed on a molluscan species *A. fulica*, which was shown to ingest and egest PET fibers (12578.8 µm in length) (Table 1).

In most studies, the ingestion/egestion of microplastics was dose- or size-dependent (e.g. Selonen et al., 2020). Evidence of the retention of micro- and nanoplastics in the tissues and organs of soil invertebrates was provided in several cases (Jiang et al., 2020; Zhou et al., 2020; Lahive et al., 2021) (Table1).

Bioturbation by microplastics was analyzed in *L. terrestris*. Exposure to LDPE fragments increased the concentration of microplastic found in the earthworm's burrows. The transport of LDPE and PE particles was size-dependent, as the fragments found in the burrows were smaller than those in the surface litter and in deeper soil layers (Huerta Lwanga et al., 2017; Rillig et al., 2017) (Table 1).

3.3 Accumulation of microplastic-associated chemicals

The accumulation of chemicals in tissues of soil invertebrates through microplastic exposure is also not a strictly ecotoxicological effect. Nevertheless, in contrast to microplastic particles, the body burdens of dissolved chemicals in organisms are clearly related to their adverse effects (McCarty et al., 2013) and therefore relevant in risk assessments (Schaefer et al., 2015). Microplastics can act as vectors for environmental chemicals in two ways. (1) by the leaching of their stabilizers, plasticizers and other associated chemical compounds into the environmental medium, allowing their subsequent ingestion by organisms and thus their bioaccumulation (Teuten et al., 2009; Halle et al., 2020) and (2) by absorbing environmental chemicals (Table 3), such as heavy metals, pesticides, or persistent organic pollutants (POPs), which are then ingested with the microplastics by soil organisms (Akdogan and Guven, 2019; Torres et al., 2021). The accumulation of microplastic-associated chemicals has been evaluated only in annelid species. Those studies generally demonstrated that microplastics are a potential source of hazardous leachates (Table 2) and facilitate the accumulation of toxic chemicals (Table 3).

3.3.1 Accumulation of leached chemicals

Two publications analyzed the accumulation of leached chemicals. A significant bioaccumulation of endogenous hexabromocyclododecanes (HBCDDs) from EPS (expanded polystyrene) was found in the digestive fluid of *E. fetida*, and *M. guillelmi* after 28 days. Accumulation increased with the decreasing size of EPS particles and prolonged exposure time (Li et al., 2019). Tire fragments induced an accumulation of metals (Zn, Cd, Pb) in *E. fetida* after 14 and 28 days. In that study, the metal concentrations in the earthworms' tissues were related to microplastic size and exposure time (Sheng et al., 2021) (Table 2).

3.3.2 Accumulation of environmental chemicals

The influence of microplastics on the bioavailability of environmental chemicals was examined in eight studies, using *E. fetida*, *L. terrestris* and *E. crypticus* as the test species (Table 3). After the co-exposure of *E. fetida* to PE particles and Cd and Cu

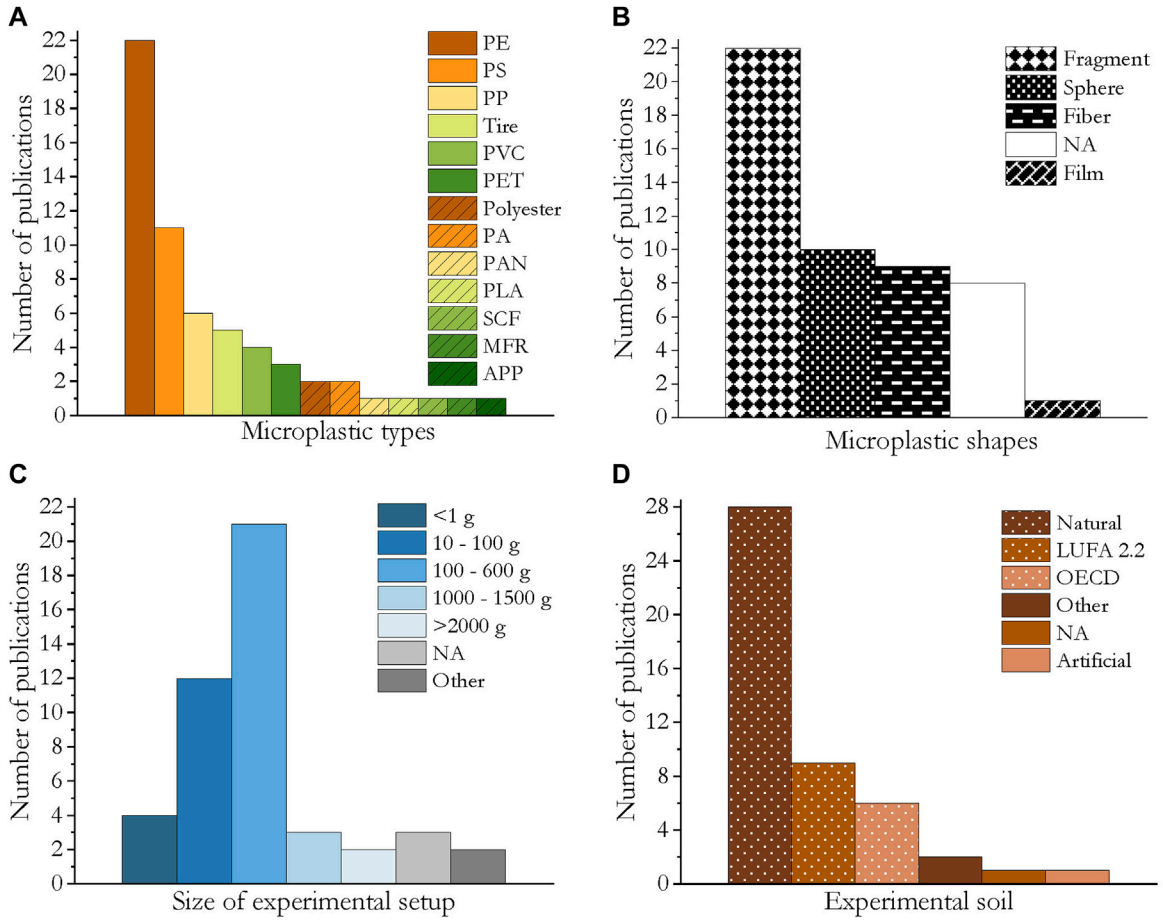

FIGURE 4
Bar charts of the experimental design and the characteristics of the applied microplastics in the reviewed publications. **(A)** Applied microplastic types. **(B)** Microplastic shapes (NA = microplastic shape not stated); abbreviations: SCF = synthetic clothing fiber, MFR = melamine-formaldehyde resin. **(C)** Size of experimental setup (NA = not stated.) **(D)** Experimental soil (NA = soil type not stated).

for 28 days, the levels of the metals in the worm's tissues were significantly increased (Li B. et al., 2021; Huang et al., 2021). Exposure to PS particles in Cd- and As-contaminated soil resulted in significantly elevated metal levels in *E. fetida's* tissues after 21 days. Microplastics induced a higher accumulation than did nanosized plastics (Xu et al., 2021b). Zhou and co-authors reported a significantly higher accumulation of Cd in the tissues of *E. fetida* when PE beads were added (Zhou et al., 2020). The bioaccumulation of phenanthrene in the tissues of *E. fetida* was higher in the presence than the absence of PS particles (Xu et al., 2021a). Perfluorooctanoic acid (PFOA), perfluorooctane sulfonate (PFOS) (both 10 mg kg^{-1}) and a mixture of PFOA/PFOS (5/5 mg kg^{-1}) accumulated in the tissues of *E. fetida*. Bioaccumulation was dependent on the concentration of PVC: 1000 mg PVC kg^{-1} significantly increased the incorporation of PFOA/PFOS, at least 500 mg PVC kg^{-1} was needed for a significant accumulation of PFOA, but only ≥1 mg PVC kg^{-1} facilitated the significant accumulation of PFOS (Sobhani et al., 2021a) (Table 3).

However, in other studies there was no increase in the accumulation of chemicals in the presence of microplastics. The addition of PA or PVC did not enhance the accumulation of tetracycline in *E. crypticus* after 21 days (Ma et al., 2020). In a co-contamination study with HDPE fragments, Zn levels in the gut or chloragog cells of *L. terrestris* were not elevated after 28 days (Hodson et al., 2017) (Table 3).

3.4 Physical effects of microplastics on soil invertebrates

The morphological features of microplastics, including their size, shape or surface structure, might pose mechanical hazards

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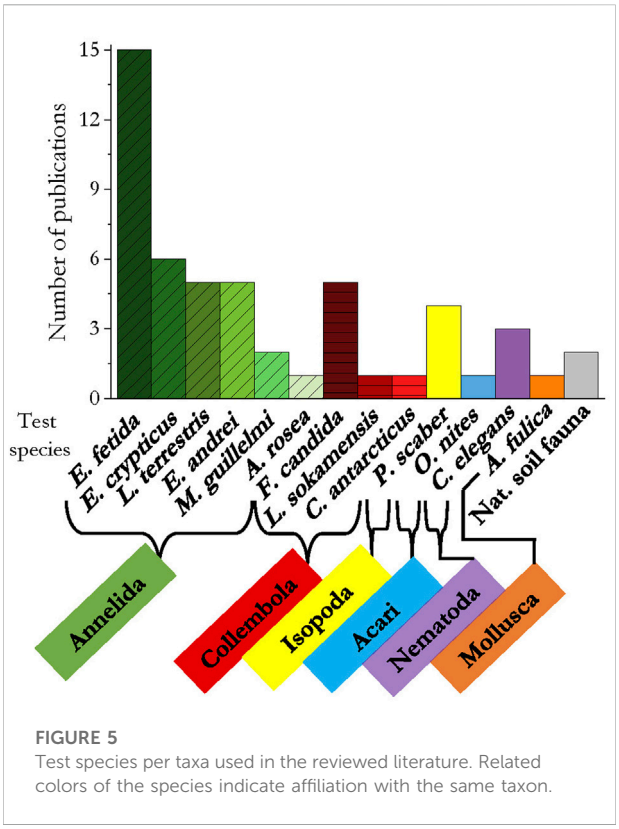

FIGURE 5
Test species per taxa used in the reviewed literature. Related colors of the species indicate affiliation with the same taxon.

to organisms (Barnes et al., 2009), in turn affecting their cellular responses, microbiota, mortality, development, behavior and reproduction. Of the 45 reviewed papers, 34 dealt with physical effects (Figure 4). It should be noted that while the respective authors assumed a physical impact, additional effects of possibly leached endogenous chemicals from the experimentally applied microplastics could not be excluded.

In the following, the different effect endpoints considered as indicators of physical impacts of microplastics are considered from a sub-organismic to organismic to community level.

3.4.1 Cellular response

Microplastics can induce changes in cellular responses via genetic damages/alterations, gene expression (levels), metabolic pathways, enzyme activity/levels, the production of reactive oxygen species (ROS), miscellaneous inflammatory reactions, cell structure/lysis, or the lipid, carbohydrate and protein content of tissues (Table 4). In most experiments, a negative influence of microplastics on the cellular responses, including oxidative stress reactions and histopathological changes, of several organisms was determined (Table 4). The expression levels of several genes and oxidative indicators were used as indicators, including annetocin (ANN), calreticulin (CRT), catalase (CAT), glutathione (GSH), glutathione S-transferase (GST), glutathione peroxidase (GPx), heat shock protein 70 (Hsp70), metallothionein (MT), translationally controlled tumor protein (TCTP), thiobarbituric acid reactive substances (TBARS), lactate dehydrogenase (LDH), lipid hydroperoxide (LPO), malondialdehyde (MDA), phenoloxidase (PO) and superoxide dismutase (SOD). Changes in the levels of those biochemical markers indicated an oxidative stress response. For example, the radical scavengers CAT, SOD and the detoxifying enzyme GST were shown to be important for inactivating ROS and obviating cell damages (Liu et al., 2012; Zhang et al., 2014; Hu et al., 2016). High levels of ROS induce detectable changes in the synthesis or modification of enzymes responsible for inactivation (Liu et al., 2011; Wang et al., 2016; Cheng et al., 2020).

The effects of microplastics were analyzed in the annelids *E. crypticus*, *E. andrei* and, most often, *E. fetida*. Exposure to HDPE, LDPE, PP and PS induced DNA damages in the F0 and the F1 generation of *E. fetida* (Cheng et al., 2020; Jiang et al., 2020; Xu et al., 2021a; Li M. et al., 2021; Sobhani et al., 2021b). The extent of the damage was larger at higher particle concentrations and larger bead sizes (Jiang et al., 2020).

The levels of GSH, ROS, LDH, LPO, MDA, and TBARS, the activity of CAT, GST, POS, SOD and the transcript levels of ANN, CRT, Hsp70, MT, SOD and TCTP increased significantly in annelids exposed to several shapes of LDPE, PE, PP and PS (Rodriguez-Seijo et al., 2018a; Wang et al., 2019; Cheng et al., 2020; Jiang et al., 2020; Zhou et al., 2020; Li B. et al., 2021; Xu et al., 2021a). Conversely, exposure to HDPE, LDPE, PE, PP, PS significantly decreased CAT, SOD, and GST activity, MDA levels and CAT and GST transcript levels (Rodriguez-Seijo et al., 2018a; Wang et al., 2019; Cheng et al., 2020; Jiang et al., 2020; Xu et al., 2021a; Li M. et al., 2021). Finally, LDPE and PE had no impact on SOD and GST activity or on TBARS, MDA or ROS levels in *E. fetida* (Rodriguez-Seijo et al., 2018b; Wang et al., 2019; Cheng et al., 2020; Kwak and An, 2021b).

On the tissue level, PS spheres induced the enlargement and deformation of *E. fetida* cells, altered the size of cell nuclei and caused cell lysis after 14 days (Jiang et al., 2020). The cell viability of *E. fetida* was affected by PE beads, as the esterase activity of the coelomocytes was significantly lowered. While female reproductive organs were not affected, male reproductive organs were significantly damaged (Kwak & An, 2021b).

In *E. andrei* a LOEC of 62.5 mg LLDPE fragments kg^{-1} dry soil caused histopathological damage, by significantly inducing fibrosis, inflammatory infiltrates and congestion (Rodriguez-Seijo et al., 2017). In *E. andrei* and *E. fetida*, these effects were dose-dependent (Rodriguez-Seijo et al., 2017; Rodriguez-Seijo et al., 2018a). After exposure to a facemask-derived PP fiber-fragment mixture, spermatogenesis in *E. andrei* was significantly damaged. The cell viability of coelomocytes was affected, but there were no signs of oxidative stress nor was tissue damage, including bleeding, swelling and thinning, observed. Neither lysosomal stability nor the number of mature oocytes was impaired (Kwak & An, 2021a). Oxidative reactions were observed in *E. crypticus* as well, as HDPE fragments (4 mm)

provoked a dose-dependent increase in GST and CAT activity (Pflugmacher et al., 2020). In the same organism, PA and PVC triggered an increase in the abundance of ARGs (antibiotic resistance genes), whereas MGEs (mobile genetic elements) were not affected (Ma et al., 2020). In *L. terrestris,* exposure to polyester microfibers caused a dose-dependent increase in the transcript level of *mt-2*. The expression of the *hsp70* was significantly reduced but there was no effect on *sod-1* expression (Prendergast-Miller et al., 2019).

The results for microplastic effects in springtail species are ambiguous. There was no evidence of oxidative stress, as esterase activity in *F. candida* was not affected by exposure to a facemask-derived PP fiber and fragment mixture (Kwak & An, 2021a). In *F. candida's* tissues, PVC treatment for 28 and 56 days resulted in a significant enrichment of $\delta^{15}N$ and $\delta^{13}C$ while the C:N ratio itself was not altered. The nitrogen and carbon isotopes were used as indicator of the springtail's feeding habits and metabolic turnover rates (Zhu et al., 2018). In the springtail *P. scaber*, neither the viability of haemocytes nor the lipid, carbohydrate and protein content was affected by PE fibers, whereas PO-like activity was increased in the presence of 0.5% PE. Tire fragments had no effect on the differential haemocyte count, haemocyte viability or PO-like activity but the total haemocyte count (THC) significantly increased (Selonen et al., 2020; Dolar et al., 2021).

Oxidative stress was also demonstrated in *A. fulica*, as GPx levels and total antioxidant capacity (T-AOC) significantly decreased and the MDA content was significantly increased after exposure of the mollusc to PET fibers for 28 days. Exposure also resulted in mechanical tissue damage in the stomach, intestines and the villi of the gastric wall of the snail, whereas kidney and liver were unaffected (Song et al., 2019).

3.4.2 Microbiota

Only four publications focussed on (gut) microbiota (Table 4). Different types of microplastics (PA, PVC, PE) were reported to significantly change the microbial composition in soil invertebrates (*F. candida*, *E. crypticus*). Bacterial alpha diversity was significantly decreased in microplastic treatments (Zhu et al., 2018; Ju et al., 2019; Ma et al., 2020). In comparison, there was no effect on the bacterial alpha diversity and density of *M. guillemi* by either HDPE or PP fragments (Cheng et al., 2021).

3.4.3 Mortality

Compared to all other analyzed parameters, the mortality of soil invertebrates, examined in 22 publications (65%), was the least sensitive towards microplastics exposure. In four studies, microplastics decreased survival at higher particle contents or a prolonged exposure time. The majority of studies revealed no effect of several microplastic types on the mortality of annelids (Boots et al., 2019; Cheng et al., 2020, 2021; Hodson et al., 2017; Kwak & An, 2021a, b; Lahive et al., 2019; Li M. et al., 2021; Ma et al., 2020; Prendergast-Miller et al., 2019; Rillig et al., 2017; Rodriguez-Seijo et al., 2017, ; Selonen et al., 2020; Sobhani et al., 2021a; Xu et al., 2021a) or *P. scaber*, *O, nitens*, *F. candida* or *A. fulica* (Kwak and An, 2021a; Selonen et al., 2020; Song et al., 2019; Zhu et al., 2018; Table 4). Jiang et al., 2020 found a significantly higher mortality of *E. fetida* exposed to 1000 µg PS kg^{-1}). In another study, exposure to PP spheres (<150 µm) at concentrations >0.3% soil dry weight increased the mortality of *E. fetida* after 42 days (Zhou et al., 2020). A long-term experiment with natural soil demonstrated significantly higher dose- and time dependent mortality in the annelid species *L. terrestris* when exposed to LDPE for 60 days (Huerta Lwanga et al., 2016). In a 1-week experiment, Ju et al., 2019 showed no lethal effects of PE spheres on *F. candida*, whereas in an experiment lasting 28 days mortality was significantly higher. The mortality of *C. elegans* was not affected by exposure to tire microplastics for 2 days whereas survival was significantly reduced after a 10-day exposure (Kim et al., 2021) (Table 4).

3.4.4 Development

Developmental alterations caused by microplastics include effects on growth (rate), biomass and body length and were examined in 19 publications. No general impact tendencies of microplastic on soil invertebrates could be derived, as contradictory results were found. In *A. rosea* and *L. terrestris* both negative and no effects on development were reported. Similarly in *E. fetida* and *F. candida* both no impact or a striking elevation/reduction of the growth rate was determined. Tire fragments decreased the growth in *C. elegans* (Kim et al., 2021). LLDPE, PE and PET had no influence on the biomass of *E. andrei*, *P. scaber* and *A. fulica* (Rodriguez-Seijo et al., 2017; Selonen et al., 2020; Song et al., 2019; Table 4).

In *A. rosea* exposed for 30 days to HDPE, biomass was significantly reduced. The growth and weight of *L. terrestris* exposed to LDPE and PS were significantly reduced after 7, 14 and 60 days; the effect on the growth rate was dose-dependent (Huerta Lwanga et al., 2016). By contrast, HDPE, PE, PET, PLA, polyester fibers and synthetic clothing fibers had no effect on the weight of either *L. terrestris* (Hodson et al., 2017; Rillig et al., 2017; Prendergast-Miller et al., 2019; Lahive et al., 2021) or *A. rosea* (Boots et al., 2019).

LDPE pellets, PVC particles, PS fragments (pure and commercial)/particles and PE particles had no significant effects on the weight of *E. fetida* (Rodriguez-Seijo et al., 2018a; Sobhani et al., 2021a, Wang et al., 2019), whereas Jiang et al., 2020 reported a significantly increased growth rate after exposure of the worm to PS spheres for 14 days. Another study showed that exposure to PP spheres (<150 µm) at a concentration of ≥0.6% soil dry weight and to PS particles (100 nm–100 µm) reduced the growth and weight of *E. fetida* after 14 and 21 days, respectively (Zhou et al., 2020; Xu et al., 2021a). However, in *E. crypticus* exposed to PVC particles for

21 days, body weight was significantly increased (Ma et al., 2020). The collembolan *F. candida* was not affected by exposure to PVC (Zhu et al., 2018) whereas face-mask-derived PP microplastics significantly reduced its growth, by 92.9% after 28 days (Kwak and An, 2021a).

3.4.5 Behavior

Alterations in behavior by microplastics comprise changes in movement, foraging activity and the stimulation of avoidance behavior. Microplastics were shown to have significant impacts on the movement activities of soil invertebrates. Ambiguous results were found focussing on the feeding activity and avoidance behavior (Table 4).

The addition of PE/PS spheres and PE fragments significantly reduced the movement of *L. sokamensis* (Kim & An, 2019). Significantly more burrow formation and bioturbation activity were provoked in *L. terrestris* exposed to 7% LDPE over 2 weeks (Huerta Lwanga et al., 2017).

The food intake of the mollusc *A. fulica* was reduced by the ingestion of PET fibers (Song et al., 2019).

Barreto and co-authors analyzed the impacts of PE and PP fibers on a natural soil community. The feeding activity of microarthropods was not affected after 4 weeks, whereas exposure to PE but not PP increased the decomposition rate (Barreto et al., 2020). In a study of avoidance behavior, *F. candida* significantly evaded PE spheres in a dose-dependent manner (Ju et al., 2019). In a two-sided test arena, *E. crypticus* clearly avoided the side containing a higher HDPE concentration (0/2, 0/4, 0/8, 2/4, 2/8, 4/8% dw) (Pflugmacher et al., 2020). In comparison, PE/PET/polyester fibers, PP fragments and fibers triggered neither avoidance behavior in *L. terrestris* (Prendergast-Miller et al., 2019; Lahive et al., 2021) or *F. candida* (Kwak and An, 2021a) nor a change in the feeding activity of *P. scaber* (Selonen et al., 2020).

3.4.6 Reproduction

Reproduction was examined in 12 publications. There was no significant reduction in *L. terrestris*, *P. scaber*, *O. nitens*, *E. andrei*, *E. crypticus, F. candida* and *C. elegans* (Huerta Lwanga et al., 2016, Kim et al., 2020b, Rodriguez-Seijo et al., 2017 and Selonen et al., 2020; Table 4). Lahive et al. (2019) observed a polymer-dependent impact on *E. crypticus* reproduction with PVC particles having no effects, while after 21 days of exposure polyamide particles at very high concentrations (>90 g/kg soil) decreased reproduction significantly (Table 4). In *E. crypticus* exposed to PA particles, the reproduction rate was elevated but it decreased significantly in response to PVC (Ma et al., 2020). After 4 weeks the number of *E. fetida* juveniles was significantly reduced by 1000 mg PVC particles kg^{-1} (Sobhani et al., 2021a). In a long-term experiment, a significant a dose-dependent decrease in the reproduction of F0 and F1 generations of *E. fetida* was determined when the worm was exposed to pure or commercial PS fragments, thus revealing transgenerational effects of microplastic exposure (Sobhani et al., 2021b). A significant inhibition by PE and PP on the reproduction rate of *F. candida* after 28 days was reported as well (Ju et al., 2019; Kwak and An, 2021a). PS beads, PET, HDPE, PP, PS, tire fragments and PAN fibers significantly reduced the number of offspring in *C. elegans* (Kim et al., 2020a; Kim et al., 2021).

3.4.7 Community structure

The effects of microplastic on the community structure of soil invertebrates was analyzed in two studies. The composition of the trophic structure, species richness, and the abundance of taxa served as indicators.

In a natural soil community exposed to LDPE fragments for 287 days in a field experiment, significant impacts on nematode trophic structure and a reduction in the overall abundance of nematodes were determined (Table 4). The abundances of omnivorous, predatory and plant-feeding nematodes declined, whereas fungal- and bacterial-feeding members were unaffected. In general, microarthropod structure was significantly altered by the addition of LDPE. The abundance of dipteran/lepidopteran larvae, ants and oribatid mites significantly decreased. Only the abundance of non-oribatid mites and collembolans remained unaffected (Lin et al., 2020). However, Barreto and co-authors found no effects of microplastics on either the species richness or the abundance of several taxa in a natural soil community, tested in a climate chamber (Barreto et al., 2020).

3.5 Chemical effects of microplastic-associated compounds on soil invertebrates

As discussed above, the leaching of chemical compounds associated with microplastic (e.g. plasticizers or flame retardants) or absorbed from the environment (e.g. polychlorinated biphenyls (PCBs), phenanthrene) can result in harmful impacts on organisms (Teuten et al., 2009; Carbery et al., 2018). While most of the following publications assumed a chemical impact, additional effects of the physical properties of the applied microplastic could not be excluded.

3.5.1 Effects of leaching compounds

Seven publications examined the effects of chemical leachates from microplastics (Table 5). Three different annelid species and *P. scaber*, *F. candida* and *C. elegans* were exposed for 3–56 days.

3.5.1.1 Cellular response

The cellular responses of soil organisms towards microplastic leachates were examined in *P. scaber* and *E. fetida* (Table 5). Oxidative stress was induced in *E. fetida* exposed to tire particles. POD, CAT, GST activity and MDA levels were elevated, whereas SOD activity decreased (Sheng et al., 2021).

Tire particles did not affect electron transfer in *P. scaber*, whereas acetylcholinesterase (AChE) activity was significantly decreased, with a NOEC of 0.5% and an EC_{50} of 1.2% (Selonen et al., 2021). When *P. scaber* was exposed to virgin LDPE, neither the total haemocyte count nor haemocyte viability were affected. However, the proportion of granulocytes increased and that of semigranulocytes decreased. Recycled LDPE did not influence the total haemocyte count, haemocyte viability or the proportion of granulocytes, but the proportion of semigranulocytes decreased in response to virgin LDPE (Kokalj et al., 2021).

3.5.1.2 Gut microbiota

One publication analyzed the effect of leachates on one annelid species (Table 5). Applied tire fragments did not affect the alpha diversity of the gut microbiota in *E. fetida*, whereas both the bacterial and the fungal community were significantly changed after exposure for 21 days. The study showed that the gut microbiota was more sensitive than soil bacteria to tire particles (Ding et al., 2020).

3.5.1.3 Mortality

Ambiguous results on the mortality of soil invertebrates were reported in five publications. Neither tire particles, nor virgin/recycled LDPE fragments and 'their leachates affected the survival of *P. scaber*, *F. candida* or *E. crypticus* (Kokalj et al., 2021; Selonen et al., 2021), whereas the survival rate of *E. fetida* and *E. andrei* exposed to tire fragments and APPs decreased significantly (Ding et al., 2020; Soroldoni et al., 2021). The mortality of *C. elegans* was not affected by exposure of the nematode to tire fragments for 2 days, independent of the preincubation time of the microplastics with the testing soil. However, in a longer experiment survival was significantly reduced after 8 days. Preincubation of the microplastic with the soil for 75 days resulted in a significant inhibition already after 6 days (Kim et al., 2021) (Table 5).

3.5.1.4 Development

Two publications found impacts of leachates on the development of invertebrates (Table 5). Antifouling paint particles (0.14%, dry soil w/w) scraped off from a boat's hull reduced the biomass of *E. andrei* after 56 days (Soroldini et al., 2021). In *C. elegans* exposed to ≥100 mg tire fragments kg^{-1} for 2 days, growth was significantly reduced. A significant inhibition of growth at lower concentrations (1 mg kg^{-1} and higher) was obtained when the tire particles were preincubated with the testing soil for 30 or 75 days, indicating an effect of leaching compounds (Kim et al., 2021).

3.5.1.5 Behavior

The behavior of soil invertebrates exposed to leachates has been analyzed only in *P. scaber* (Table 5). Exposure of the woodlouse to 0.5 and 1.5% virgin LDPE fragments for 21 days significantly reduced its feeding rate whereas recycled LDPE or tire fragments had no effect on the feeding rate or activity (Kokalj et al., 2021; Selonen et al., 2021).

3.5.1.6 Reproduction

Five studies examined the impact of microplastic leachates on three annelid species, the nematode *C. elegans* and the collembolan *F. candida* (Table 5). Annelida were clearly impacted, as the exposure of *E. fetida* to tire fragments (≥0.12% microplastic content) for 21 days caused a dose-dependent decrease in reproduction. A logarithmic correlation between reproduction and the concentration of tire particles was determined (Ding et al., 2020). The number of juveniles and cocoons produced by *E. andrei* was reduced after 56 and 42 days, respectively, when the worm was exposed to 0.01% APPs (Soroldoni et al., 2021). Reproduction in *E. crypticus* was not affected by microplastic mixed with food, whereas 0.02 and 1.5% tire particles in soil significantly decreased the number of offspring (Selonen et al., 2021.)

The exposure of *C. elegans* to extractable additive solutions of PE, PAN, PP for 24 h did not affect reproduction, whereas extracts derived from PET significantly reduced the number of offspring. After the removal of those additives by extraction, the toxic effect was lost. The adverse impact of LDPE films and PAN fibers on *C. elegans* reproduction increased with prolonged exposure and frequent wet-dry cycles of the testing soil (Kim et al., 2020b). When the nematode was exposed to tire fragments for 2 days its brood size was significantly reduced. This decreasing effect was aggravated when the microplastic was preincubated with the testing soil (Kim et al., 2021). Finally, the number of juveniles in *F. candida* was not affected by tire particles added to soil or food (Selonen et al., 2021).

3.5.2 Influence of microplastic on the toxicity of environmental chemicals

Eleven of the reviewed papers examined the vector effects of microplastics on soil invertebrates, in three annelid species in ten publications and the arthropod *P. scaber* in one publication (Table 6). The impacts of the microplastics in co-exposure experiments with atrazine, chlorpyrifos, phenanthrene, PFOA, PFOS, tetracycline and the metals As, Cd, Cu, Ni and Zn were analyzed. The exposure time ranged between 7 and 56 days.

3.5.2.1 Cellular response

Microplastic-chemical-co-exposures triggered several oxidative stress responses in *E. fetida, E. crypticus* and *P. scaber*. *E. fetida* was the most well-studied (Table 6). Its co-exposures to PE-Cd, PS- phenanthrene and LDPE-atrazine induced DNA damage (Cheng et al., 2020; Xu et al., 2021a; Huang et al., 2021). LDPE-atrazine, LDPE-chlorpyrifos, PP-Cd, PE-Cd, -Cu or -Ni, and PS-phenanthrene, -Cd or -As triggered increases in LPO and ROS levels, GSH, MDA and TBARS content, total protein, IBR, the activity of CAT, MT, POD, SOD and GST as well as the mRNA levels of HSP70, MT,

SOD and TCTP (Cheng et al., 2020; Huang et al., 2021; Li B. et al., 2021; Rodriguez-Seijo et al., 2018b; Xu et al., 2021a; Zhou et al., 2020). In general, cellular responses were aggravated by co-exposure compared to the chemical or microplastic treatment alone (Ma et al., 2020; Zhou et al., 2020; Li B. et al., 2021; Xu et al., 2021a).

By contrast, the levels/activities of several indicators of oxidative stress were decreased in *E. fetida* after its co-exposure for 28 days. Thus LDPE-atrazine, LDPE-chlorpyrifos, PE-Cd and PS-phenanthrene, -Cd or -As resulted in significantly lower AChE, CAT, GST, POD, SOD activities, GSH, MDA, TBARS contents and CAT, GST MT mRNA levels (Cheng et al., 2020; Huang et al., 2021; Rodriguez-Seijo et al., 2018b; Xu et al., 2021a,). No effect on the AChE activity were observed when *E. fetida* was exposed to 0.25–1 mm LDPE pellets in combination with chlorpyrifos (Rodriguez-Seijo et al., 2018b).

On tissue level, in *E. fetida* exposed for 21 days to PS particles (100 µm, 10 µm, 100 nm) in Cd- and As-contaminated soil cellular damage such as abnormal epithelial cells, incomplete coelom tissues and exfoliated intestinal epithelium were observed (Xu et al., 2021b; Huang et al., 2021). The exposure of *E. crypticus* to PA or PVC particles in combination with tetracycline for 21 days resulted in a significantly higher amount of ARGs and MGEs (Ma et al., 2020).

AChE activity was significantly lower in *P. scaber* exposed to PE fibers (12–2870 µm) and 0.2–2 mg kg^{-1} CP for 3 weeks. Both the total haemocyte count (THC) and the proportion of hyalinocytes were elevated whereas the proportion of granulocytes, semigranulocytes and the viability of haemocytes were not affected. The co-exposure of *P. scaber* to tire fragments (<180 µm) and CP for 3 weeks increased the proportion of granulocytes and decreased that of semigranulocytes. Haemocyte viability and the proportion of hyalinocytes were not affected (Dolar et al., 2021).

3.5.2.2 Gut microbiota

Only two publications focussed on the impacts of co-exposure on the gut microbiota. Thus, the exposure of *E. fetida* to PS of several sizes and to phenanthrene significantly changed the composition of the gut microbial community (Xu et al., 2021a). Similar results were found in *E. crypticus*, in which PA/PVC-tetracycline treatment induced a significant change in microbial composition and a reduction in microbial alpha diversity (Ma et al., 2020) (Table 6).

3.5.2.3 Mortality

Almost all publications with experiments lasting 3–56 days reported no effects on the mortality of the test species (Table 6). Most data are available for *E. fetida*: The combinations LDPE-chlorpyrifos (Rodriguez-Seijo et al., 2018b), aged/unaged LDPE-atrazine (Cheng et al., 2020), PA-Cd or -As (Xu et al., 2021b), PS-phenanthrene (Xu et al., 2021a) and PVC-PFOA and/or PFOS (Sobhani et al., 2021a) did not influence the annelid's mortality. However, co-exposure to Cd-PS spheres for >42 days significantly reduced survival. In another study the effect of PS-phenanthrene was mostly mediated by the PS component, as phenanthrene addition did not aggravate the effect (Zhou et al., 2020).

3.5.2.4 Development

Eight publications examined the effects of environmental chemicals on developmental traits, with *E. fetida* as the test organism in most of those studies. After a maximum of 28 days of co-exposure to PE-Cd or PS-phenanthrene, -As or -Cd, the weight of the worm was significantly reduced (Xu et al., 2021a Huang et al., 2021) (Table 3). Additionally, in *E. fetida* exposed to PP-Cd for 42 days the growth rate was significantly decreased. The decline was greater in the co-exposure treatment than in the Cd alone treatment (Zhou et al., 2020). However, the exposure of *E. fetida* to LDPE pellets contaminated with chlorpyrifos caused significant weight loss after 28 days (Rodriguez-Seijo et al., 2018b). Similar results were found for *L. terrestris* exposed to PVC-tetracycline for 21 days (Ma et al., 2020) whereas the co-exposure of *E. fetida* or *L. terrestris* to LDPE beads and chlorpyrifos, PS particles and Cd or As, PVC-PFOA and/or PFOS and HDPE-Zn had no impact on the weight of either organism (Hodson et al., 2017; Rodriguez-Seijo et al., 2018b; Sobhani et al., 2021a; Xu et al., 2021b) (Table 6).

3.5.2.5 Behavior

The effects of environmental chemicals and microplastic on the behavior of soil invertebrates were investigated in three studies. Significant avoidance behavior was reported for *E. fetida* exposed to PE-Cd or to LDPE pellets contaminated with chlorpyrifos (Rodriguez-Seijo et al., 2018b; Huang et al., 2021) whereas *L. terrestris* did not avoid contaminated HDPE fragments (Hodson et al., 2017) (Table 6).

3.5.2.6 Reproduction

Co-exposure treatments significantly reduced the reproduction of soil invertebrates (Table 6). Cocoon production by *E. fetida* exposed to Cd and PE particles decreased (Huang et al., 2021), and PVC combined with PFOA, PFOS or a mixture of both significantly reduced the number of juveniles (Sobhani et al., 2021a). In *E. crypticus* exposed for 21 days to PVC or PA combined with tetracycline, reproduction was significantly reduced (Ma et al., 2020).

4 Discussion

Soils can be hotspots of microplastic pollution, as here dumping, decay, fragmentation and accumulation of plastic takes place. Moreover, soils play a key role in the transition to aquatic environments. Consequently, soil invertebrates are likely to be exposed to elevated concentrations of microplastics, potentially leading to adverse effects on soil ecosystems and

thus higher trophic levels. Due to patchy data availability and the heterogenous, non-standardized methodologies used in the different studies, general microplastic-specific effect mechanism in soils could not be deduced. However, our literature review suggests trends that should be the basis of future research.

4.1 Variability in species sensitivity

In general, there was little diversity in the test organisms, as 64% of all studies analyzed only annelid species. As comparisons of species sensitivity with environmental thresholds are the basis of risk assessments, including the risk posed by the specific or unspecific impacts of microplastics (Koelmans et al., 2017), toxicity data must be required for a larger number of soil invertebrate species, to cover at least the most dominant ones.

Nonetheless, the published data revealed differences in the sensitivity of different soil invertebrates to microplastics. Annelida tended to be more sensitive than Collembola. The levels of stress biomarkers were increased in the annelid *E. fetida* exposed to 0.05% PE particles (Li B. et al., 2021) (Table 4). Cellular responses, development, mortality, and reproduction were negatively impacted in *E. andrei*, *E. fetida* and *E. crypticus* exposed to APP and tire fragments and/or its possible leachates (Ding et al., 2020; Selonen et al., 2021; Sheng et al., 2021; Soroldoni et al., 2021) (Table 5). In comparison, a concentration of at least 0.1% was needed to reduce the reproduction rate in the springtail *F. candida* (Ju et al., 2019) (Table 4). In *F. candida*, neither mortality nor reproduction was affected by the leachates of tire fragments (Selonen et al., 2021) (Table 5). However, studies with other chemicals, such as pesticides, showed that the toxicity ranking among soil invertebrates is highly substance specific (Frampton et al., 2006). For example, reproduction in *F. candida* was more sensitive than *E. fetida* to the insecticide toxaphene (EC_{50} = 3.6 mg kg^{-1} soil dry weight vs EC_{50} = 54.5 mg kg^{-1} soil dry weight) (Bezchlebova et al., 2007) and to Ag nanoparticles (Heckmann et al., 2011; Zhang and Filser, 2020) whereas the effect of nickel was comparably (*E. fetida*: EC_{50} = 362 mg Ni kg^{-1} soil dry weight; *F. candida*: EC_{50} = 391–461 mg Ni kg^{-1} soil dry weight) (Lock and Janssen, 2002).

Considering the large heterogeneity of microplastics as a contaminant group, also here a substantial variability in the sensitivity of soil organisms can be expected. Furthermore, the lack of standardized test protocols (including with respect to concentration: mass, and particle numbers; dry or wet weight) hampers reliable comparisons between studies (Cunningham and Sigwart, 2019; De Ruijter et al., 2020). The development of standardized experimental and analytical methods will enable the comparability and reproducibility of results and in turn allow the identification of valid trends in the sensitivities within soil communities.

4.2 Role of the properties of microplastics (polymer/shape/size)

The toxicity of microplastics differing in their polymer composition, shape and type and of mixtures thereof remains to be reliably determined. None of the reviewed papers except one simultaneously tested the effect of more than one microplastic type or shape (Kim and An, 2021a). While one-third of all publications evaluated the interference of PE, PS and PP with soil invertebrates (Figure 4A; Tables 4–6), there are roughly 30,000 EU-approved types of plastic and several plastic types have already been detected in a diversity of soils around the world, including their spatial temporal dynamics (Geyer et al., 2017; Horton et al., 2017; Huerta Lwanga et al., 2018; Büks and Kaupenjohann, 2020; Wong et al., 2020).

Given the limited data on the impacts of microplastics on soil invertebrates, systematic investigations of the influence of particle properties on the ecotoxicological behavior of microplastic in soil are rare and the determination of toxicity mechanisms related to microplastic properties accordingly difficult. Studies using bacteria (*Aliivibrio fischeri*), plants (*Nelumbo nucifera*) or fish (*Danio rerio*) suggest that PVC is more toxic than PS (Lei et al., 2018; Zimmermann et al., 2019; Esterhuizen and Kim, 2022), but this could not be shown for soil invertebrates. Several studies demonstrated that PS affects the cellular responses of soil organisms (Sobhani et al., 2021b; Jiang et al., 2020; Wang et al., 2019; Xu et al., 2021a; Table 4), but a direct comparison of the effects of PS vs those of other polymers is difficult, as in the respective studies the PS particles were smaller (<250 µm) than particles of other microplastic types (e.g. up to 2.4 mm for PE). Particle size is known to play a decisive role in the toxic effects of microplastics on soil invertebrates (e.g. Lei et al., 2018; Mueller et al., 2020), as ingestion is dependent on the size of the organism's mouth opening (Fueser et al., 2019; Koelmans et al., 2020).

Fragments and fibers are the most common shape found in soil environments (Büks and Kaupenjohann, 2020). In several studies, fragments were shown to negatively impact cellular responses (Rodriguez-Seijo et al., 2017, Sobhani et al., 2021b; Li M. et al., 2021; Table 4) which suggests shape-specific effects. Unfortunately, none of the reviewed publications directly compared the effects of the microplastic shapes with a standardized polymer type, size and concentration. Sublethal effects in *Daphnia magna* were found to be related to microplastic size, polymer type and shape, with beads and fragments being more toxic than fibers (Schwarzer et al., 2022). Further studies are needed to evaluate the role of microplastic shape in the toxicity of microplastics to soil invertebrates (Büks and Kaupenjohann, 2020).

4.3 Distinguishing physical from chemical effects

To distinguish between the major effect pathways of microplastics, studies were categorized into those examining

physical and chemical effects of microplastics. The former accounted for 71% of the reviewed publications. In fact, this ascription neglects that in most studies, due to their simultaneous occurrence, the two toxicity pathways are not readily distinguished. For example, some publications reported DNA damages after microplastic exposure, but this could not be unambiguously attributed to strictly chemical vs strictly physical effects (e.g. Cheng et al., 2020; Sobhani et al., 2021b; Jiang et al., 2020; Table 4). Thus, test setups are required that the experimental separation of these two effect pathways, such as by testing extracted leachates separately from the particles to test for strictly chemical effects (Kim et al., 2020b) and by testing (non-plastic) reference particles with characteristics similar to those of microplastics to account for the physical effects of particles (Koelmans et al., 2022).

4.4 Experimental approaches

Most of the studies evaluating the potential adverse effects of microplastics on soil invertebrates consisted of single-species toxicity tests. Only two studies assessed the impacts of microplastics on natural soil communities (Barreto et al., 2020; Lin et al., 2020; Table 4). Although lower-tier testing is a valuable approach for obtaining reliable toxicity thresholds, assuming standardized methods are used, realistic exposure scenarios are best achieved in community-based or long-term population-based approaches, such as in micro- or mesocosm studies. These systems have several advantages over acute, single-species tests:

1) Whole-community tests allow the toxicity of microplastics to be synchronously assessed in several species, thus significantly expanding the species database on microplastic toxicity.
2) The impacts of microplastics on whole food webs and the transfer of microplastics from one trophic level to another are unclear but can only be elucidated in multispecies systems (Haegerbaeumer et al., 2019b). Several studies have shown that microplastic particles can exert their toxic effects on organisms not only directly, but also indirectly, such as via the dilution of food through or the formation of food-microplastic agglomerates (Hanna et al., 2018; Mueller et al., 2020; Rauchschwalbe et al., 2021). Higher-tier studies, especially terrestrial model ecosystems (TMEs), can reveal both the direct and indirect effects of microplastics on several taxa, trophic, population levels or developmental stages of organisms (Gestel et al., 2020; Mueller et al., 2020).
3) In publications dealing with chemical effects, it could be reported that the exposure time influenced the magnitude of leaching or vector impact (Xu et al., 2021b; Soroldoni et al., 2021). Moreover, as microplastics are highly persistent in the environment (Thompson et al., 2005; Thompson et al., 2009a; Barnes et al., 2009; Horton et al., 2017; Selonen et al., 2020) chronic effects on the soil fauna can be expected, such that risk assessments should be conducted in long-term studies, as has been done in freshwater and marine ecosystems (e.g. Santana et al., 2018; Chisada et al., 2019; Redendo-Hasselerharm et al., 2020; Rauchschwalbe et al., 2022). As transgenerational effects of microplastics on soil invertebrates have been reported (Sobhani et al., 2021b), these should be further analyzed in model ecosystems to evaluate long-term effects of microplastic in soil.

The experimental approaches in the reviewed studies were highly diverse, ranging from single to multiple species set ups, and therefore so were their findings. Terrestrial and aquatic studies with environmental chemicals often yielded more sensitive responses of soil and sediment communities when in microcosms than as single-species tests (Clements et al., 2013; Haegerbaeumer et al., 2016; Yin et al., 2018; Boots et al., 2019). However, a microcosm study using freshwater nematodes showed that the sensitivity of a native nematode community to microplastics in the sediment of the microcosms (Rauchschwalbe et al., 2022) was similar to that determined in a single-species sediment toxicity test (Höss et al., 2022). Analogous to the risk assessment of other environmental chemicals, lower-tier testing with representative species is necessary to determine reliable toxicity thresholds, with higher-tier testing using microcosms or semi-field approaches allowing the validation of these thresholds under more realistic exposure conditions.

4.5 Differences between different soils

Since microplastics can affect the physical properties of soil, including soil structure, pH, size distribution of water-stable soil aggregates and the availability of nutrients, the impact of microplastics on soil invertebrates may depend on the nature of the tested soil (Liu et al., 2017; Machado et al., 2018; Boots et al., 2019; Machado et al., 2019). For example, the toxicity of the fungicide mancozeb on the reproduction of the two soil invertebrates was shown to be highly dependent on the soil type (Carniel et al., 2019). Among the reviewed studies, only one investigated the toxic effect of PS spheres on *C. elegans* in different soil types. The results showed that the physicochemical properties of the soil influenced the response of *C. elegans* to soil microplastics (Kim et al., 2020a). However, a comparable study in freshwater sediments found no influence of physico-chemical properties on the response of this nematode (Höss et al., 2022). Microplastic exposure in OECD soil, but not natural field soil (LUFA) significantly increased the mortality of *E. fetida* (Zhou et al., 2020; Xu et al., 2021a). Nonetheless, although these two studies used microplastics of similar concentration and size, the different experimental set ups hamper a direct comparison of the tested soils.

4.6 Risk assessment: Comparing effects with exposure

The use of effect assessment data, as presented in this review, in a microplastic risk assessment requires realistic exposure estimates, i.e. predicted environmental concentrations. However, for the smaller size fractions of microplastics (<20 μm), reliable data on environmental concentrations are still missing, due to the obvious analytical challenges posed by the presence of these particles in complex matrices, such as soils. In agricultural or horticultural sites, microplastic concentrations of up to 4.5 mg kg^{-1} dry soil have been determined, whereby and concentrations at industrial used sites exceeded such values up to four times (Büks & Kaupenjohann, 2020). Based on the lowest effect concentrations resulting in toxicity, 100 μg kg^{-1} of dry soil, as demonstrated in *E. fetida* (Jiang et al., 2020; Table 4), an environmental risk for soil invertebrates at microplastic hotspots cannot be ruled out. Moreover, including the small fraction, still not reliably covered by analytical studies, an even higher risk might be realistic.

A second important component in a risk assessment is the identification of reliable toxicity thresholds, such as those derived from dose-response curves (effect concentrations: ECx; LOEC, NOEC). However, for microplastics, toxicity thresholds comparable to those available for pesticides and other chemicals are scarce (Table 4; Rodriguez-Seijo et al., 2017; Selonen et al., 2021). A study consisting of an alignment of available microplastic effect thresholds for a variety of organisms (Koehlmans et al., 2020) was unable to include findings for soil invertebrates, due to the lack of threshold data. Therefore, studies producing effect thresholds on the basis of dose-response curves are needed for soil ecosystems.

5 Recommendations

- A larger number of soil invertebrate species should be included in the routine testing of microplastics in soil.
- Robust toxicity thresholds (i.e. effect concentrations) determined on the basis of dose-response testing remain to be established for microplastics. Chronic toxicity endpoints should be included to account for subtle effects and the time-dependent leaching of chemicals from plastic particles.
- Toxicity thresholds for microplastics, based on single-species testing, should be verified by testing under realistic exposure scenarios, including multiple species (whole communities), trophic interactions (and thus also indirect effects) and long-term exposure.
- Test systems are required that distinguish between chemical and physical effects. This includes the testing of extracted leachates and the use of non-plastic reference particles as a negative control for physical particle effects.
- As also reported for microplastics research in aquatic environments, harmonized, standardized methods are urgently needed in tests of the effects of microplastics on soil organisms, as these will allow comparisons of reported effects.

Author contributions

ACFM created the first draft of the manuscript, prepared data organization/collection and contributed to manuscript revision. AH guided manuscript revision and contributed to the organization of the database. SH was responsible for study conceptualization, manuscript writing and editing, and WT for study conceptualization and manuscript revision.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Publisher's note

References

Akdogan, Z., and Guven, B. (2019). Microplastics in the environment: A critical review of current understanding and identification of future research needs. *Environ. Pollut.* 254, 113011. UNSP 113011. doi:10.1016/j.envpol.2019.113011

Alimi, O. S., Budarz, J. F., Hernandez, L. M., and Tufenkji, N. (2018). Microplastics and nanoplastics in aquatic environments: Aggregation, deposition, and enhanced contaminant transport. *Environ. Sci. Technol.* 52, 1704–1724. doi:10.1021/acs.est.7b05559

Anbumani, S., and Kakkar, P. (2018). Ecotoxicological effects of microplastics on biota: A review. *Environ. Sci. Pollut. Res.* 25, 14373–14396. doi:10.1007/s11356-018-1999-x

Andrady, A. L. (2011). Microplastics in the marine environment. *Mar. Pollut. Bull.* 62, 1596–1605. doi:10.1016/j.marpolbul.2011.05.030

Bar-On, Y. M., Phillips, R., and Milo, R. (2018). The biomass distribution on Earth. *Proc. Natl. Acad. Sci. U. S. A.* 115, 6506–6511. doi:10.1073/pnas.1711842115

Bardgett, R. D., and van der Putten, W. H. (2014). Belowground biodiversity and ecosystem functioning. *Nature* 515, 505–511. doi:10.1038/nature13855

Barnes, D. K. A., Galgani, F., Thompson, R. C., and Barlaz, M. (2009). Accumulation and fragmentation of plastic debris in global environments. *Phil. Trans. R. Soc. B* 364, 1985. doi:10.1098/rstb.2008.0205

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

Bergami, E., Caruso, T., Birarda, G., Vaccari, L., and Corsi, I. (2020). Plastics everywhere: First evidence of polystyrene fragments inside the common antarctic collembolan *Cryptopygus antarcticus*. *Biol. Lett.* 16, 20200093. doi:10.1098/rsbl.2020.0093

Besseling, E., Wegner, A., Foekema, E. M., van den Heuvel-Greve, M. J., and Koelmans, A. A. (2013). Effects of microplastic on fitness and PCB bioaccumulation by the lugworm *Arenicola marina* (L.). *Environ. Sci. Technol.* 47, 593–600. doi:10.1021/es302763x

Bezchlebová, J., Černohlávková, J., Lána, J., Sochová, I., Kobetičová, K., and Hofman, J. (2007). Effects of toxaphene on soil organisms. *Ecotoxicol. Environ. Saf.* 68, 326–334. doi:10.1016/j.ecoenv.2007.05.009

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

Boots, B., Russell, C. W., and Green, D. S. (2019). Effects of microplastics in soil ecosystems: Above and below ground. *Environ. Sci. Technol.* 53, 11496–11506. doi:10.1021/acs.est.9b03304

Bowker, M. A., Maestre, F. T., and Escolar, C. (2010). Biological crusts as a model system for examining the biodiversity-ecosystem function relationship in soils. *Soil Biol. Biochem.* 42, 405–417. doi:10.1016/j.soilbio.2009.10.025

Browne, M. A., Crump, P., Niven, S. J., Teuten, E., Tonkin, A., Galloway, T., et al. (2011). Accumulation of microplastic on shorelines woldwide: Sources and sinks. *Environ. Sci. Technol.* 45, 9175–9179. doi:10.1021/es201811s

Büks, F., and Kaupenjohann, M. (2020). Global concentrations of microplastics in soils – A review. *SOIL* 6, 649–662. doi:10.5194/soil-6-649-2020

Carbery, M., O'Connor, W., and Thavamani, P. (2018). Trophic transfer of microplastics and mixed contaminants in the marine food web and implications for human health. *Environ. Int.* 115, 400–409. doi:10.1016/j.envint.2018.03.007

Carniel, C. L. S., Niemeyer, J. C., Iuñes de Oliveira Filho, L. C., Alexandre, D., Gebler, L., and Klauberg-Filho, O. (2019). The fungicide mancozeb affects soil invertebrates in two subtropical Brazilian soils. *Chemosphere* 232, 180–185. doi:10.1016/j.chemosphere.2019.05.179

Cheng, Y., Song, W., Tian, H., Zhang, K., Li, B., Du, Z., et al. (2021). The effects of high-density polyethylene and polypropylene microplastics on the soil and earthworm *Metaphire guillelmi* gut microbiota. *Chemosphere* 267, 129219. doi:10.1016/j.chemosphere.2020.129219

Cheng, Y., Zhu, L., Song, W., Jiang, C., Li, B., Du, Z., et al. (2020). Combined effects of mulch film-derived microplastics and atrazine on oxidative stress and gene expression in earthworm (*Eisenia fetida*). *Sci. Total Environ.* 746, 141280. doi:10.1016/j.scitotenv.2020.141280

Chisada, S., Yoshida, M., and Karita, K. (2019). Ingestion of polyethylene microbeads affects the growth and reproduction of medaka, *Oryzias latipes*. *Environ. Pollut.* 254, 113094. doi:10.1016/j.envpol.2019.113094

Cisneros-Dozal, L. M., Trumbore, S., and Hanson, P. J. (2006). Partitioning sources of soil-respired CO_2 and their seasonal variation using a unique radiocarbon tracer. *Glob. Chang. Biol.* 12, 194–204. doi:10.1111/j.1365-2486.2005.001061.x

Clements, W. H., Cadmus, P., and Brinkman, S. F. (2013). Responses of aquatic insects to Cu and Zn in stream microcosms: Understanding differences between single species tests and field responses. *Environ. Sci. Technol.* 47, 7506–7513. doi:10.1021/es401255h

Cole, M., Lindeque, P., Halsband, C., and Galloway, T. S. (2011). Microplastics as contaminants in the marine environment: A review. *Mar. Pollut. Bull.* 62, 2588–2597. doi:10.1016/j.marpolbul.2011.09.025

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

Cunningham, E. M., and Sigwart, J. D. (2019). Environmentally accurate microplastic levels and their absence from exposure studies. *Integr. Comp. Biol.* 59, 1485–1496. doi:10.1093/icb/icz068

da Costa, J. P., Santos, P. S. M., Duarte, A. C., and Rocha-Santos, T. (2016). Nano) plastics in the environment - sources, fates and effects. *Sci. Total Environ.* 566, 15–26. doi:10.1016/j.scitotenv.2016.05.041

de Ruijter, V. N., Redondo-Hasselerharm, P. E., Gouin, T., and Koelmans, A. A. (2020). Quality criteria for microplastic effect studies in the context of risk assessment: A critical review. *Environ. Sci. Technol.* 54, 11692–11705. doi:10.1021/acs.est.0c03057

Ding, J., Dong, Z., Hongtao, W., Lassen, S., Chen, Q., Li, G., et al. (2020). Dysbiosis in the gut microbiota of soil fauna explains the toxicity of tire tread particles. *Environ. Sci. Technol.* 54, 12.doi:10.1021/acs.est.0c00917

Dolar, A., Selonen, S., Gestel, C., Perc, V., Drobne, D., and Kokalj, A. (2021). Microplastics, chlorpyrifos and their mixtures modulate immune processes in the terrestrial crustacean *Porcellio scaber*. *Sci. Total Environ.* 772, 144900. doi:10.1016/j.scitotenv.2020.144900

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

ECETOC (2018). *An evaluation of the challenges and limitations associated with aquatic toxicity and bioaccumulation studies for sparingly soluble and manufactured particulate substances - technical Report No. 132*. Brussels, Belgium: European Centre for Ecotoxicology and Toxicology of Chemicals.

Eerkes-Medrano, D., Thompson, R. C., and Aldridge, D. C. (2015). Microplastics in freshwater systems: A review of the emerging threats, identification of knowledge gaps and prioritisation of research needs. *Water Res.* 75, 63–82. doi:10.1016/j.watres.2015.02.012

Esterhuizen, M., and Kim, Y. J. (2022). Effects of polypropylene, polyvinyl chloride, polyethylene terephthalate, polyurethane, high-density polyethylene, and polystyrene microplastic on *Nelumbo nucifera* (Lotus) in water and sediment. *Environ. Sci. Pollut. Res.* 29, 17580–17590. doi:10.1007/s11356-021-17033-0

Frampton, G. K., Jänsch, S., Scott-Fordsmand, J. J., Römbke, J., and van den Brink, P. J. (2006). Effects of pesticides on soil invertebrates in laboratory studies: A review and analysis using species sensitivity distributions. *Environ. Toxicol. Chem.* 25, 2480–2489. doi:10.1897/05-438R.1

Frouz, J. (2018). Effects of soil macro- and mesofauna on litter decomposition and soil organic matter stabilization. *Geoderma* 332, 161–172. doi:10.1016/j.geoderma.2017.08.039

Fueser, H., Mueller, M.-T., Weiss, L., Hoess, S., and Traunspurger, W. (2019). Ingestion of microplastics by nematodes depends on feeding strategy and buccal cavity size. *Environ. Pollut.* 255, 113227. doi:10.1016/j.envpol.2019.113227

Gautam, R., Bassi, A. S., and Yanful, E. K. (2007). A review of biodegradation of synthetic plastic and foams. *Appl. Biochem. Biotechnol.* 141, 85–108. doi:10.1007/s12010-007-9212-6

Gestel, C., Mommer, L., Montanarella, L., Pieper, S., Coulson, M., Toschki, A., et al. (2020). Soil biodiversity: State-of-the-Art and possible implementation in chemical risk assessment. *Integr. Environ. Assess. Manag.* 17, 541–551. doi:10.1002/ieam.4371

Gewert, B., Plassmann, M. M., and MacLeod, M. (2015). Pathways for degradation of plastic polymers floating in the marine environment. *Environ. Sci. Process. Impacts* 17, 1513–1521. doi:10.1039/c5em00207a

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

Haegerbaeumer, A., Höss, S., Ristau, K., Claus, E., Möhlenkamp, C., Heininger, P., et al. (2016). A comparative approach using ecotoxicological methods from single-species bioassays to model ecosystems. *Environ. Toxicol. Chem.* 35, 2987–2997. doi:10.1002/etc.3482

Haegerbaeumer, A., Mueller, M.-T., Fueser, H., and Traunspurger, W. (2019a). Impacts of micro- and nano-sized plastic particles on benthic invertebrates: A literature review and gap analysis. *Front. Environ. Sci.* 7, 17. doi:10.3389/fenvs.2019.00017

Haegerbaeumer, A., Raschke, R., Reiff, N., Traunspurger, W., and Höss, S. (2019b). Comparing the effects of fludioxonil on non-target soil invertebrates using ecotoxicological methods from single-species bioassays to model ecosystems. *Ecotoxicol. Environ. Saf.* 183, 109596. doi:10.1016/j.ecoenv.2019.109596

Halle, L. L., Palmqvist, A., Kampmann, K., and Khan, F. R. (2020). Ecotoxicology of micronized tire rubber: Past, present and future considerations. *Sci. Total Environ.* 706, 135694. doi:10.1016/j.scitotenv.2019.135694

Hanna, S. K., Bustos, A. R. M., Peterson, A. W., Reipa, V., Scanlan, L. D., Coskun, S. H., et al. (2018). Agglomeration of *Escherichia coli* with positively charged nanoparticles can lead to artifacts in a standard *Caenorhabditis elegans* toxicity assay. *Environ. Sci. Technol.* 52, 5968–5978. doi:10.1021/acs.est.7b06099

He, D., Luo, Y., Lu, S., Liu, M., Song, Y., and Lei, L. (2018). Microplastics in soils: Analytical methods, pollution characteristics and ecological risks. *TrAC Trends Anal. Chem.* 109, 163–172. doi:10.1016/j.trac.2018.10.006

Heckmann, L.-H., Hovgaard, M. B., Sutherland, D. S., Autrup, H., Besenbacher, F., and Scott-Fordsmand, J. J. (2011). Limit-test toxicity screening of selected inorganic nanoparticles to the earthworm *Eisenia fetida*. *Ecotoxicology* 20, 226–233. doi:10.1007/s10646-010-0574-0

Helmberger, M. S., Tiemann, L. K., and Grieshop, M. J. (2020). Towards an ecology of soil microplastics. *Funct. Ecol.* 34, 550–560. doi:10.1111/1365-2435.13495

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

Hoellein, T., Rojas, M., Pink, A., Gasior, J., and Kelly, J. (2014). Anthropogenic litter in urban freshwater ecosystems: Distribution and microbial interactions. *PLoS ONE* 9, e98485. doi:10.1371/journal.pone.0098485

Hohn, S., Acevedo-Trejos, E., Abrams, J. F., Fulgencio de Moura, J., Spranz, R., and Merico, A. (2020). The long-term legacy of plastic mass production. *Sci. Total Environ.* 746, 141115. doi:10.1016/j.scitotenv.2020.141115

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

Höss, S., Rauchschwalbe, M.-T., Füser, H., and Traunspurger, W. (2022). Food availability is crucial for effects of 1-µm polystyrene beads on the nematode *Caenorhabditis elegans* in freshwater sediments. *Chemosphere* 298, 134101. doi:10.1016/j.chemosphere.2022.134101

Hu, S., Zhang, W., Li, J., Lin, K., and Ji, R. (2016). Antioxidant and gene expression responses of *Eisenia fetida* following repeated exposure to BDE209 and Pb in a soil-earthworm system. *Sci. Total Environ.* 556, 163–168. doi:10.1016/j.scitotenv.2016.02.194

Huang, C., Ge, Y., Yue, S., Zhao, L., and Qiao, Y. (2021). Microplastics aggravate the joint toxicity to earthworm *Eisenia fetida* with cadmium by altering its availability. *Sci. Total Environ.* 753, 142042. doi:10.1016/j.scitotenv.2020.142042

Huerta Lwanga, E., Gertsen, H., Gooren, H., Peters, P., Salanki, T., van der Ploeg, M., et al. (2017). Incorporation of microplastics from litter into burrows of *Lumbricus terrestris*. *Environ. Pollut.* 220, 523–531. doi:10.1016/j.envpol.2016.09.096

Huerta Lwanga, E., Gertsen, H., Gooren, H., Peters, P., Salanki, T., van der Ploeg, M., et al. (2016). Microplastics in the terrestrial ecosystem: Implications for *Lumbricus terrestris* (Oligochaeta, lumbricidae). *Environ. Sci. Technol.* 50, 2685–2691. doi:10.1021/acs.est.5b05478

Huerta Lwanga, E., Thapa, B., Yang, X., Gertsen, H., Salanki, T., Geissen, V., et al. (2018). Decay of low-density polyethylene by bacteria extracted from earthworm's guts: A potential for soil restoration. *Sci. Total Environ.* 624, 753–757. doi:10.1016/j.scitotenv.2017.12.144

Imhof, H. K., Ivleva, N. P., Schmid, J., Niessner, R., and Laforsch, C. (2013). Contamination of beach sediments of a subalpine lake with microplastic particles. *Curr. Biol.* 23, R867–R868. doi:10.1016/j.cub.2013.09.001

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

Jiang, X., Chang, Y., Zhang, T., Qiao, Y., Klobucar, G., and Li, M. (2020). Toxicological effects of polystyrene microplastics on earthworm *(Eisenia fetida)*. *Environ. Pollut.* 259, 113896. doi:10.1016/j.envpol.2019.113896

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

Kim, S. W., and An, Y.-J. (2019). Soil microplastics inhibit the movement of springtail species. *Environ. Int.* 126, 699–706. doi:10.1016/j.envint.2019.02.067

Kim, S. W., Kim, D., Jeong, S.-W., and An, Y.-J. (2020a). Size-dependent effects of polystyrene plastic particles on the nematode *Caenorhabditis elegans* as related to soil physicochemical properties. *Environ. Pollut.* 258, 113740. doi:10.1016/j.envpol.2019.113740

Kim, S. W., Leifheit, E., Maaß, S., and Rillig, M. (2021). Time-dependent toxicity of tire particles on soil nematodes. *Front. Environ. Sci.* 9, 681939. doi:10.3389/fenvs.2021.744668

Kim, S. W., Waldman, W. R., Kim, T.-Y., and Rillig, M. C. (2020b). Effects of different microplastics on nematodes in the soil environment: Tracking the extractable Additives using an ecotoxicological approach. *Environ. Sci. Technol.* 54, 13868–13878. doi:10.1021/acs.est.0c04641

Kim, Y.-N., Yoon, J.-H., and Kim, K.-H. (2020c). Microplastic contamination in soil environment -a review. *Soil Sci. Ann.* 71, 300–308. doi:10.37501/soilsa/131646

Koelmans, A. A., Besseling, E., Foekema, E., Kooi, M., Mintenig, S., Ossendorp, B. C., et al. (2017). Risks of plastic debris: Unravelling fact, opinion, perception, and belief. *Environ. Sci. Technol.* 51, 11513–11519. doi:10.1021/acs.est.7b02219

Koelmans, A. A., Redondo-Hasselerharm, P. E., Mohamed Nor, N. H., and Kooi, M. (2020). Solving the nonalignment of methods and approaches used in microplastic research to consistently characterize risk. *Environ. Sci. Technol.* 54, 12307–12315. doi:10.1021/acs.est.0c02982

Koelmans, A. A., Redondo-Hasselerharm, P. E., Nor, N. H. M., de Ruijter, V. N., Mintenig, S. M., and Kooi, M. (2022). Risk assessment of microplastic particles. *Nat. Rev. Mat.* 7, 138–152. doi:10.1038/s41578-021-00411-y

Kokalj, A. J., Dolar, A., Titova, J., Visnapuu, M., Skrlep, L., Drobne, D., et al. (2021). Long term exposure to virgin and recycled LDPE microplastics induced minor effects in the freshwater and terrestrial Crustaceans *Daphnia magna* and *Porcellio scaber*. *Polymers* 13, 771. doi:10.3390/polym13050771

Kuehr, S., Windisch, H., Schlechtriem, C., Leon, G., Gasparini, G., and Gimeno, S. (2021). Are fragrance encapsulates taken up by aquatic and terrestrial invertebrate species? *Environ. Toxicol. Chem.* 41, 931–943. doi:10.1002/etc.5202

Kwak, J. I., and An, Y.-J. (2021a). Microplastic digestion generates fragmented nanoplastics in soils and damages earthworm spermatogenesis and coelomocyte viability. *J. Hazard. Mater.* 402, 124034. doi:10.1016/j.jhazmat.2020.124034

Kwak, J. I., and An, Y.-J. (2021b). Post COVID-19 pandemic: Biofragmentation and soil ecotoxicological effects of microplastics derived from face masks. *J. Hazard. Mat.* 416, 126169. doi:10.1016/j.jhazmat.2021.126169

Lahive, E., Cross, R., Saarloos, A., Horton, A., Svendsen, C., Hufenus, R., et al. (2021). Earthworms ingest microplastic fibres and nanoplastics with effects on egestion rate and long-term retention. *Sci. Total Environ.* 807, 151022. doi:10.1016/j.scitotenv.2021.151022

Lahive, E., Walton, A., Horton, A. A., Spurgeon, D. J., and Svendsen, C. (2019). Microplastic particles reduce reproduction in the terrestrial worm *Enchytraeus crypticus* in a soil exposure. *Environ. Pollut.* 255, 113174. doi:10.1016/j.envpol.2019.113174

Lavelle, P., and Spain, A. (2001). *Soil ecology*. Springer Science & Business Media.

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

Li, B., Lan, Z., Wang, L., Sun, H., Yao, Y., Zhang, K., et al. (2019). The release and earthworm bioaccumulation of endogenous hexabromocyclododecanes (HBCDDs) from expanded polystyrene foam microparticles. *Environ. Pollut.* 255, 113163. doi:10.1016/j.envpol.2019.113163

Li, B., Song, W., Cheng, Y., Zhang, K., Tian, H., Du, Z., et al. (2021a). Ecotoxicological effects of different size ranges of industrial-grade polyethylene and polypropylene microplastics on earthworms *Eisenia fetida*. *Sci. Total Environ.* 783, 147007. doi:10.1016/j.scitotenv.2021.147007

Li, M., Liu, Y., Xu, G., Wang, Y., and Yu, Y. (2021b). Impacts of polyethylene microplastics on bioavailability and toxicity of metals in soil. *Sci. Total Environ.* 760, 144037. doi:10.1016/j.scitotenv.2020.144037

Li, X., Chen, L., Mei, Q., Dong, B., Dai, X., Ding, G., et al. (2018). Microplastics in sewage sludge from the wastewater treatment plants in China. *Water Res.* 142, 75–85. doi:10.1016/j.watres.2018.05.034

Lin, D., Yang, G., Dou, P., Qian, S., Zhao, L., Yang, Y., et al. (2020). Microplastics negatively affect soil fauna but stimulate microbial activity: Insights from a field-based microplastic addition experiment. *Proc. R. Soc. B* 287, 20201268. doi:10.1098/rspb.2020.1268

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

Liu, S., Zhou, Q., and Chen, C. (2012). Antioxidant enzyme activities and lipid peroxidation in earthworm *Eisenia fetida* exposed to 1, 3, 4, 6, 7, 8-hexahydro-4, 6, 6, 7, 8, 8-hexamethyl-cyclopenta-gamma-2-benzopyr an. *Environ. Toxicol.* 27, 472–479. doi:10.1002/tox.20661

Liu, S., Zhou, Q., and Wang, Y. (2011). Ecotoxicological responses of the earthworm *Eisenia fetida* exposed to soil contaminated with HHCB. *Chemosphere* 83, 1080–1086. doi:10.1016/j.chemosphere.2011.01.046

Lock, K., and Janssen, C. R. (2002). Ecotoxicity of nickel to *Eisenia fetida, Enchytraeus albidus* and *Folsomia candida*. *Chemosphere* 46, 197–200. doi:10.1016/S0045-6535(01)00112-6

Ma, J., Sheng, G. D., and O'Connor, P. (2020). Microplastics combined with tetracycline in soils facilitate the formation of antibiotic resistance in the *Enchytraeus crypticus* microbiome. *Environ. Pollut.* 264, 114689. doi:10.1016/j.envpol.2020.114689

Machado, A. A. de S., Lau, C. W., Kloas, W., Bergmann, J., Bacheher, J. B., Faltin, E., et al. (2019). Microplastics can change soil properties and affect plant performance. *Environ. Sci. Technol.* 53, 6044–6052. doi:10.1021/acs.est.9b01339

Machado, A. A. de S., Lau, C. W., Till, J., Kloas, W., Lehmann, A., Becker, R., et al. (2018). Impacts of microplastics on the soil biophysical environment. *Environ. Sci. Technol.* 52, 9656–9665. doi:10.1021/acs.est.8b02212

McCarty, L. S., Arnot, J. A., and Mackay, D. (2013). Evaluation of critical body residue data for acute narcosis in aquatic organisms. *Environ. Toxicol. Chem.* 32, 2301–2314. doi:10.1002/etc.2289

Mueller, M.-T., Fueser, H., Trac, L. N., Mayer, P., Traunspurger, W., and Hoess, S. (2020). Surface-related toxicity of polystyrene beads to nematodes and the role of food availability. *Environ. Sci. Technol.* 54, 1790–1798. doi:10.1021/acs.est.9b06583

Ogonowski, M., Gerdes, Z., and Gorokhova, E. (2018). What we know and what we think we know about microplastic effects – a critical perspective. *Curr. Opin. Environ. Sci. Health* 1, 41–46. doi:10.1016/j.coesh.2017.09.001

Padervand, M., Lichtfouse, E., Robert, D., and Wang, C. (2020). Removal of microplastics from the environment. A review. *Environ. Chem. Lett.* 18, 807–828. doi:10.1007/s10311-020-00983-1

Pflugmacher, S., Huttunen, J., Von Wolff, M. A., Penttinen, O.-P., Kim, Y., Kim, S., et al. (2020). *Enchytraeus crypticus* avoid soil spiked with microplastic. *Toxics* 8, 10. doi:10.3390/toxics8010010

PlasticsEurope (2021). *Plastics - the facts 2021. An analysis of European plastics production*. Brussels, Belgium: Demand and Waste Data. Available at: https://plasticseurope.org/de/knowledge-hub/plastics-the-facts-2020-2/.

Porazinska, D. L., Bardgett, R. D., Blaauw, M. B., Hunt, H. W., Parsons, A. N., Seastedt, T. R., et al. (2003). Relationships at the aboveground–belowground interface: Plants, soil biota, and soil processes. *Ecol. Monogr.* 73, 377–395. doi:10.1890/0012-9615(2003)073[0377:rataip]2.0.co;2

Prendergast-Miller, M. T., Katsiamides, A., Abbass, M., Sturzenbaum, S. R., Thorpe, K. L., and Hodson, M. E. (2019). Polyester-derived microfibre impacts on the soil-dwelling earthworm *Lumbricus terrestris*. *Environ. Pollut.* 251, 453–459. doi:10.1016/j.envpol.2019.05.037

Rauchschwalbe, M.-T., Fueser, H., Traunspurger, W., and Höss, S. (2021). Bacterial consumption by nematodes is disturbed by the presence of polystyrene beads: The roles of food dilution and pharyngeal pumping. *Environ. Pollut.* 273, 116471. doi:10.1016/j.envpol.2021.116471

Rauchschwalbe, M.-T., Höss, S., Hägerbäumer, A., and Traunspurger, W. (2022). Long-term exposure of a free-living freshwater micro- and meiobenthos community to microplastic mixtures in microcosms. *Sci. Total Environ.* 827, 154207. doi:10.1016/j.scitotenv.2022.154207

Redondo-Hasselerharm, P. E., Gort, G., Peeters, E. T. H. M., and Koelmans, A. A. (2020). Nano- and microplastics affect the composition of freshwater benthic communities in the long term. *Sci. Adv.* 6, eaay4054. doi:10.1126/sciadv.aay4054

Rillig, M. C. (2012). Microplastic in terrestrial ecosystems and the soil? *Environ. Sci. Technol.* 46, 6453–6454. doi:10.1021/es302011r

Rillig, M. C., Ziersch, L., and Hempel, S. (2017). Microplastic transport in soil by earthworms. *Sci. Rep.* 7, 1362. doi:10.1038/s41598-017-01594-7

Rodríguez-Seijo, A., da Costa, J. P., Rocha-Santos, T., Duarte, A. C., and Pereira, R. (2018a). Oxidative stress, energy metabolism and molecular responses of earthworms (*Eisenia fetida*) exposed to low-density polyethylene microplastics. *Environ. Sci. Pollut. Res.* 25, 33599–33610. doi:10.1007/s11356-018-3317-z

Rodriguez-Seijo, A., Lourenco, J., Rocha-Santos, T. a. P., da Costa, J., Duarte, A. C., Vala, H., et al. (2017). Histopathological and molecular effects of microplastics in *Eisenia andrei Bouche*. *Environ. Pollut.* 220, 495–503. doi:10.1016/j.envpol.2016.09.092

Rodríguez-Seijo, A., Santos, B., Silva, E. F. d., Cachada, A., and Pereira, R. (2018b). Low-density polyethylene microplastics as a source and carriers of agrochemicals to soil and earthworms. *Environ. Chem.* 16, 8–17. doi:10.1071/en18162

Roger-Estrade, J., Anger, C., Bertrand, M., and Richard, G. (2010). Tillage and soil ecology: Partners for sustainable agriculture. *Soil Tillage Res.* 111, 33–40. doi:10.1016/j.still.2010.08.010

Ryan, P. G., Moore, C. J., van Franeker, J. A., and Moloney, C. L. (2009). Monitoring the abundance of plastic debris in the marine environment. *Phil. Trans. R. Soc. B* 364, 1999. doi:10.1098/rstb.2008.0207

Santana, M. F. M., Moreira, F. T., Pereira, C. D. S., Abessa, D. M. S., and Turra, A. (2018). Continuous exposure to microplastics does not cause physiological effects in the cultivated mussel *Perna perna*. *Arch. Environ. Contam. Toxicol.* 74, 594–604. doi:10.1007/s00244-018-0504-3

Schaefer, S., Buchmeier, G., Claus, E., Duester, L., Heininger, P., Körner, A., et al. (2015). Bioaccumulation in aquatic systems: Methodological approaches, monitoring and assessment. *Environ. Sci. Eur.* 27, 5. doi:10.1186/s12302-014-0036-z

Scherer, C., Weber, A., Lambert, S., and Wagner, M. (2018). "Interactions of microplastics with freshwater biota," in *Handbook of environmental chemistry*, 153–180.

Schwarzer, M., Brehm, J., Vollmer, M., Jasinski, J., Xu, C., Zainuddin, S., et al. (2022). Shape, size, and polymer dependent effects of microplastics on Daphnia magna. *J. Hazard. Mater.* 426, 128136. doi:10.1016/j.jhazmat.2021.128136

Selonen, S., Dolar, A., Jemec Kokalj, A., Sackey, L. N. A., Skalar, T., Cruz Fernandes, V., et al. (2021). Exploring the impacts of microplastics and associated chemicals in the terrestrial environment – exposure of soil invertebrates to tire particles. *Environ. Res.* 201, 111495. doi:10.1016/j.envres.2021.111495

Selonen, S., Dolar, A., Kokalj, A. J., Skalar, T., Dolcet, L. P., Hurley, R., et al. (2020). Exploring the impacts of plastics in soil - the effects of polyester textile fibers on soil invertebrates. *Sci. Total Environ.* 700, 134451. UNSP 134451. doi:10.1016/j.scitotenv.2019.134451

Shah, A. A., Hasan, F., Hameed, A., and Ahmed, S. (2008). Biological degradation of plastics: A comprehensive review. *Biotechnol. Adv.* 26, 246–265. doi:10.1016/j.biotechadv.2007.12.005

Sheng, Y., Liu, Y., Wang, K., Cizdziel, J. V., Wu, Y., and Zhou, Y. (2021). Ecotoxicological effects of micronized car tire wear particles and their heavy metals on the earthworm (*Eisenia fetida*) in soil. *Sci. Total Environ.* 793, 148613. doi:10.1016/j.scitotenv.2021.148613

Sobhani, Z., Fang, C., Naidu, R., and Megharaj, M. (2021a). Microplastics as a vector of toxic chemicals in soil: Enhanced uptake of perfluorooctane sulfonate and perfluorooctanoic acid by earthworms through sorption and reproductive toxicity. *Environ. Technol. Innovation* 22, 101476. doi:10.1016/j.eti.2021.101476

Sobhani, Z., Panneerselvan, L., Fang, C., Naidu, R., and Mallavarapu, M. (2021b). Chronic and transgenerational effects of polystyrene microplastics at environmentally relevant concentrations in earthworms (*Eisenia fetida*). *Environ. Toxicol. Chem.* 40, 2240–2246. doi:10.1002/etc.5072

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

Soroldoni, S., Honscha, L., Reis, F., Duarte, F. A., Da Silva Júnior, F., and Pinho, G. (2021). Antifouling paint particles in soils: Toxic impact that goes beyond the aquatic environment. *Ecotoxicology* 30, 1161–1169. doi:10.1007/s10646-021-02418-1

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

Teuten, E. L., Saquing, J. M., Knappe, D. R. U., Barlaz, M. A., Jonsson, S., Bjorn, A., et al. (2009). Transport and release of chemicals from plastics to the environment and to wildlife. *Phil. Trans. R. Soc. B* 364, 2027–2045. doi:10.1098/rstb.2008.0284

Thompson, R. C., Moore, C. J., vom Saal, F. S., and Swan, S. H. (2009a). Plastics, the environment and human health: Current consensus and future trends. *Phil. Trans. R. Soc. B* 364, 2153–2166. doi:10.1098/rstb.2009.0053

Thompson, R. C., Swan, S. H., Moore, C. J., and vom Saal, F. S. (2009b1973–1976). Our plastic age. *Phil. Trans. R. Soc. B* 364, 1973–1976. doi:10.1098/rstb.2009.0054

Thompson, R., Moore, C., Andrady, A., Gregory, M., Takada, H., and Weisberg, S. (2005). New directions in plastic debris. *Science* 310, 1117. doi:10.1126/science.310.5751.1117b

Torres, F. G., Carolina Dioses-Salinas, D., Ivan Pizarro-Ortega, C., and De-la-Torre, G. E. (2021). Sorption of chemical contaminants on degradable and non-degradable microplastics: Recent progress and research trends. *Sci. Total Environ.* 757, 143875. doi:10.1016/j.scitotenv.2020.143875

van den Hoogen, J., Geisen, S., Routh, D., Ferris, H., Traunspurger, W., Wardle, D. A., et al. (2019). Soil nematode abundance and functional group composition at a global scale. *Nature* 572, 194–198. doi:10.1038/s41586-019-1418-6

Wang, J., Li, Y., Lu, L., Zheng, M., Zhang, X., Tian, H., et al. (2019). Polystyrene microplastics cause tissue damages, sex-specific reproductive disruption and transgenerational effects in marine medaka (*Oryzias melastigma*). *Environ. Pollut.* 254, 113024. doi:10.1016/j.envpol.2019.113024

Wang, J., Wang, J., Wang, G., Zhu, L., and Wang, J. (2016). DNA damage and oxidative stress induced by imidacloprid exposure in the earthworm *Eisenia fetida*. *Chemosphere* 144, 510–517. doi:10.1016/j.chemosphere.2015.09.004

Wong, J. K. H., Lee, K. K., Tang, K. H. D., and Yap, P.-S. (2020). Microplastics in the freshwater and terrestrial environments: Prevalence, fates, impacts and sustainable solutions. *Sci. Total Environ.* 719, 137512. doi:10.1016/j.scitotenv.2020.137512

Wright, S. L., Thompson, R. C., and Galloway, T. S. (2013). The physical impacts of microplastics on marine organisms: A review. *Environ. Pollut.* 178, 483–492. doi:10.1016/j.envpol.2013.02.031

Xu, G., Liu, Y., Song, X., Li, M., and Yu, Y. (2021a). Size effects of microplastics on accumulation and elimination of phenanthrene in earthworms. *J. Hazard. Mater.* 403, 123966. doi:10.1016/j.jhazmat.2020.123966

Xu, G., Yang, Y., and Yu, Y. (2021b). Size effects of polystyrene microplastics on the accumulation and toxicity of (semi-)metals in earthworms. *Environ. Pollut.* 291, 118194. doi:10.1016/j.envpol.2021.118194

Yin, X. H., Brock, T. C. M., Barone, L. E., Belgers, J. D. M., Boerwinkel, M.-C., Buijse, L., et al. (2018). Exposure and effects of sediment-spiked fludioxonil on macroinvertebrates and zooplankton in outdoor aquatic microcosms. *Sci. Total Environ.* 610, 1222–1238. doi:10.1016/j.scitotenv.2017.08.155

Zalasiewicz, J., Waters, C. N., Ivar do Sul, J. A., Corcoran, P. L., Barnosky, A. D., Cearreta, A., et al. (2016). The geological cycle of plastics and their use as a stratigraphic indicator of the Anthropocene. *Anthropocene* 13, 4–17. doi:10.1016/j.ancene.2016.01.002

Zhang, L., Ji, F., Li, M., Cui, Y., and Wu, B. (2014). Short-term effects of Dechlorane Plus on the earthworm *Eisenia fetida* determined by a systems biology approach. *J. Hazard. Mat.* 273, 239–246. doi:10.1016/j.jhazmat.2014.03.018

Zhang, X., and Filser, J. (2020). Low concentration effects and different outcome in repeated reproduction tests with silver nanoparticles, silver nitrate and *Folsomia candida* (Collembola). *Environ. Sci. Eur.* 32, 136–138. doi:10.1186/s12302-020-00413-7

Zhou, Y., Liu, X., and Wang, J. (2020). Ecotoxicological effects of microplastics and cadmium on the earthworm Eisenia foetida. *J. Hazard. Mater.* 392, 122273. doi:10.1016/j.jhazmat.2020.122273

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

Zimmermann, L., Dierkes, G., Ternes, T. A., Völker, C., and Wagner, M. (2019). Benchmarking the *in vitro* toxicity and chemical composition of plastic consumer products. *Environ. Sci. Technol.* 53, 11467–11477. doi:10.1021/acs.est.9b02293

Zubris, K. A. V., and Richards, B. K. (2005). Synthetic fibers as an indicator of land application of sludge. *Environ. Pollut.* 138, 201–211. doi:10.1016/j.envpol.2005.04.013