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REVIEW ARTICLE

Micro- and Nanoplastics in Environment: Degradation, Detection, and Ecological Impact

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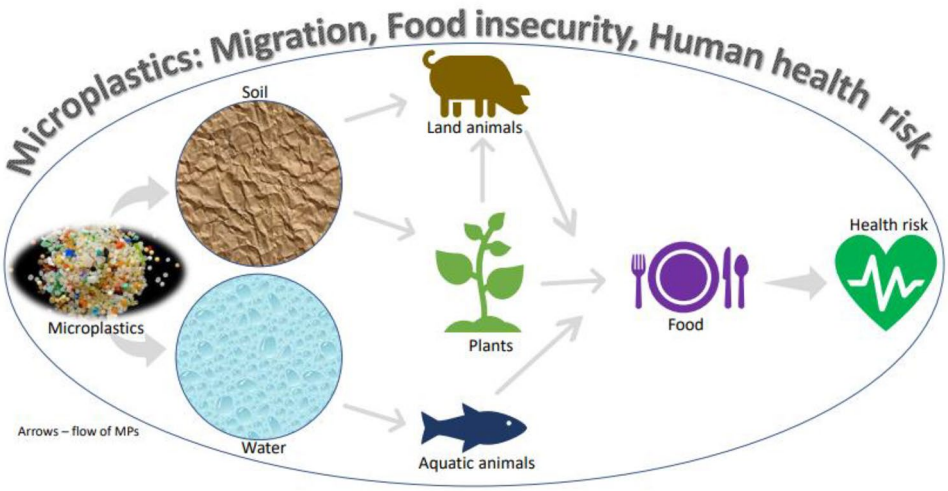
Abstract

Microplastics and also nanoplastics are tiny pieces of plastics that have been a rising source of worry due to their ubiquitous occurrence and possible environmental effect. This article dives into the presence, origins, and degrading processes that cause microscopic and nanoplastics in the natural environment, illuminating the complexities of this worldwide issue. Micro- and nanoplastics have become increasingly common in the environment during the last few decades. Microplastics have negative effects on aquatic habitats when they enter water bodies. Atmospheric deposit (microplastics are substances that have been found in the upper atmosphere, primarily originating compared to the breakdown of bigger polymers and the everyday use of car tires), splitting at sea in the marine environment, materials are confronted with constant both chemical and physical stressors, leading to dispersion into smaller pieces along with land-based runoff; storm water drainage from urban areas can transport polymer content, and particle size all impact the breakdown of micro- and nanoplastics. While plastics are known for their durability, they can be degraded through a variety of mechanisms, including mechanical weathering, photodegradation, corrosion by chemicals, biological degradation, and fragmentation. The widespread presence and persistence of micro- and nanoplastics in the surroundings has raised concerns about their potential effects on ecosystems and human health. Particles like these can be consumed by a variety of creatures, ranging from zooplankton to bigger marine animals, resulting in the spread of plastics throughout the food chain. The occurrence and degradation of micro- and nanoplastics is therefore focused in this review.

Graphical Abstract



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Highlights

- **Environmental pollution occurrence of micro- and nanoplastics is ubiquitous**
- **Some potential toxic effects of micro- and nanoplastics are on human health and the ecosystem**
- **Micro- and nanoplastics degradation depends on some influencing factors**
- **Several degrading strategies and the way forward in micro- and nanoplastics pollution were suggested.**

Keywords Occurrence · Degradation · Microplastics · Nanoplastic · Environment effects · Soil

Abbreviations

MP	Microbial plastics
NP	Nanoplastics
PS	Polystyrene
PE	Polyethylene
PLA	Polyethylene terephthalate
PET	Polyethylene terephthalate
PP	Polypropylene
UV	Ultraviolent
LDPE	Low-density polyethylene
HDPE	High-density polyethylene
GPC	Gel permeation chromatography
IR	Infrared
TGA	Thermal gravimetric analysis
AFM	Atomic force microscopy
PCL	Polycaprolactone

Introduction

Plastic is now found in nearly every aspect of our lives, from medical devices to construction materials to personal care products to everyday household items and children's toys. While plastic objects make social life easier, the downside of the plastic era is beginning to emerge (Kumar et al. 2021; Weber et al. 2022; Singh et al. 2022; Lang et al. 2023). Weber et al. (2022) found that there has been a constant flow of waste plastics into the environment due to a low recycling rate and mishandling. Microplastics (less than 5 mm) and nanoplastics (less than 100 nm) may be formed as a result of weathering, ultraviolet radiation, and biodegradation (Vighi et al. 2021; Issac and Kandasubramanian 2021; Kiran et al. 2022). Evidence suggests that the degradation of microplastics to smaller particles for mineralization is highly difficult and slow, which was previously estimated to take hundreds to thousands of years (Bhattacharya 2016; Lei et al. 2017; Hahladakis et al. 2018; Monkul and Özhan 2021; Kumar et al. 2021; Weber et al. 2022). Microplastics and nanoplastics may remain in the environment indefinitely, making it impossible to eliminate them in the near term and creating a long-term and global problem (Pavani et al. 2022). According to the majority of authorities, governments' creation of laws and regulations can aid in the reduction or even control of pollution. Some countries, such as the United Kingdom, have taken steps to limit the manufacturing and consumption of plastics, such as prohibiting the distribution and sale of plastic and prohibiting the use of plastic microbeads in personal care and cosmetic products (Wang et al. 2019a, 2020; Guo et al. 2020; Melo-Agustín et al. 2022; Stefano et al. 2022).

Several types of micro(nano)plastics have been discovered in a variety of places, including soils (Nikiema et al. 2020), oceans, river waters, air (Pignattelli et al. 2021; Auta et al. 2022), bottled water, sediments, and municipal wastewater (O'Kelly et al. 2021; Batool et al. 2022) (Fig. 1). Micro(nano)plastics can be challenging to measure if they move between different types of media. Environmental matrixes and the small particle size of micro(nano)plastics have led to the need for efficient and standard technologies for their characterization and analysis, notably for nanoplastics. Micro(nano)plastics' release, migration, and environmental occurrence have been restricted by these restrictions (Riaz et al. 2022). Microplastics have been found in bottle water at a concentration of 2649 2857 L1 (Yu et al. 2019; Nikiema et al. 2020; Pignattelli et al. 2021), sediments at a dry weight of 123.6 item per kilogram (Zhao et al. 2021), sewage sludge at a dry weight of 22.7 12.1 103 particles per kilogram (Li et al. 2022), and farmland at a dry weight of 7100–42,900 particles per kilogram. Although the concentrations of microplastics and nanoplastics in the environment remain low, they pose a substantial hazard to the ecosystem if the plastic pollution is out of control (Jambeck et al. 2015; Wang et al. 2020; Dhaka et al. 2022; Kabir et al. 2022; Radhakrishan et al. 2023).

There is a steady rise in the volume and rate of micro(nano)plastic emissions as the production and use of these polymers expands internationally. In recent years, evidence of micro(nano)plastics' detrimental effects on global biodiversity has grown more compelling. The diversity of life on Earth, including on land and in the

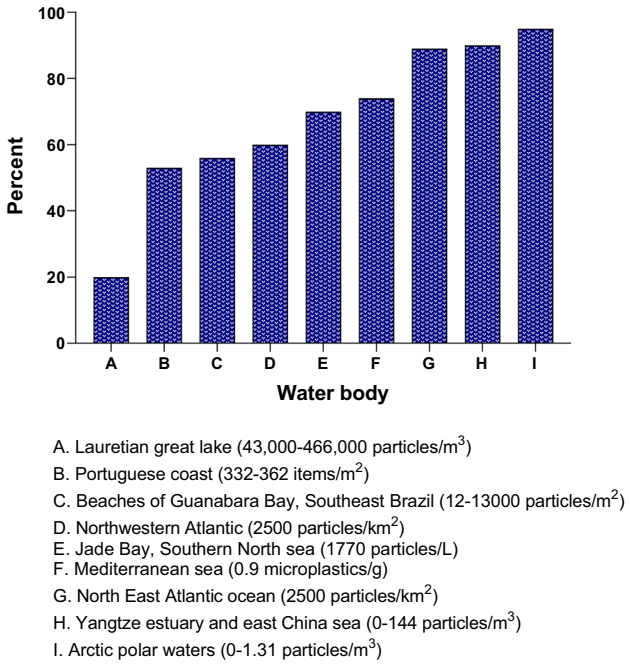

Fig. 1 Microplastic distribution in the world oceans (data from Auta et al. 2017)

oceans, is referred to as "global biodiversity." The term "biodiversity" often refers to the diversity of organisms, species, and ecosystems (Shah and Wu 2020; Ya et al. 2021; Menzel et al. 2022). Global warming, human overpopulation, hybridization, genetic pollution/erosion and food security, overexploitation, habitat degradation, and invasive species are regarded to be the most significant factors influencing biodiversity (Batool et al. 2022; Dhaka et al. 2022). Human activity has produced microplastics and nanoplastics. There are a variety of fish, birds, and marine mammals that consume micro(nano)plastics directly or indirectly because of their small particle size (Dhaka et al. 2022). This includes filter-feeding animals, such as seabirds and whales stated in a study by Wang et al. (2021, 2022). This review therefore focused on the micro- and nanoplastics pollution in the environment, the effects of this pollution on the biotic factors including humans and the ecosystem, ways of degrading these materials using current physical and chemical technologies, and the way forward in handling this pollution.

Accidental consumption of micro(nano)plastics could physically obstruct the digestive system of feeders, resulting in digestive system injury, reduced nutrition intake, or even death (Panigrahi et al. 2019; Shah and Wu 2020; Menzel et al. 2022). Polystyrene (PS) nanoplastics (39.4 nm) went past the cell membrane into the surrounding tissues, blood, and circulatory system, even into the brains of medaka, and they were difficult to get rid of (Li et al. 2020; Pizarro-Ortega et al. 2022).

Nanoplastics can also bioaccumulate and travel up the food chain (Pflugmacher et al. 2020; Pashaei et al. 2022; Fan et al. 2022). As a vector for organisms, micro(nano)plastics can also be used to move them from one place to another (de Souza Machadoet al. 2019; Mohana et al. 2021; Singh et al. 2022). Bio-invasion occurs when organisms from another environment move into a previously uninhabitable area and begin growing rapidly, destroying the ecosystem's biodiversity and ecological harmony in the process. As Giorgetti et al. (2020) found, microplastics in the water impeded treatment plants; PPCP, medicines, and personal care products.

Micro(nano)plastics have different physical and chemical properties with varying densities (Fig. 2); this property by microplastics determines the way and manner of

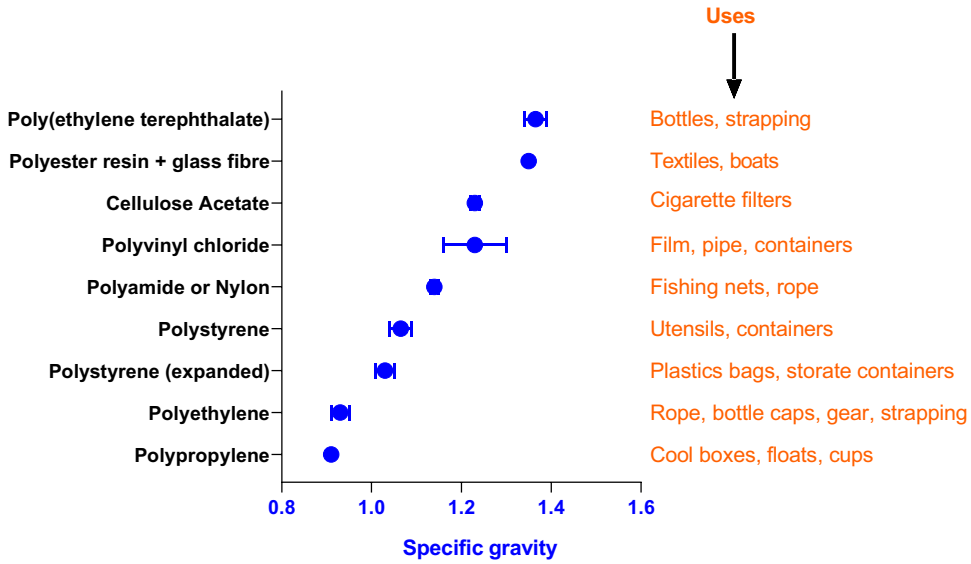

Fig. 2 Common microplastic applications and their respective densities (GESAMP 2017)

their degradation time in the environment especially in the marine.

Abiotic Degradation Pathways

Physical, referring to changes in the bulk structure, such as cracking and embrittlement; or (ii) chemical, referring to molecular changes such as bond cleavage or oxidation of long polymer chains to generate new molecules, usually with substantially shorter chains. Environmental risks from plastic degradation's soluble chemical by-products (Yu et al. 2019; Mohana et al. 2021; Xi et al. 2022) and the leaching of tiny molecules added during product formation must be taken into account. Microbial action, heat, light, or combinations of these factors can speed up chemical breakdown at near-ambient temperatures in the environment, which typically includes hydrolysis (requiring H_2O) or oxidation (requiring O_2) (Zhang et al. 2021; Guo et al. 2021; Musa et al. 2021; Li et al. 2022; Maocai et al. 2022).

Polyethylene (PE), polyethylene terephthalate (PET), and polylactic acid (PLA) are all degraded by natural abiotic processes in the following sections. It is undeniable that abiotic processes play a vital role in the breakdown of organic matter, although the majority of degradation is sparked abiotically (light, heat, acids, etc.). Smaller compounds that have been degraded by microorganisms can then be used for mineralization via the abiotic process. This section focuses on the breakdown processes for polyethylene terephthalate (PET), and polylactic acid (PLA) (Liu et al. 2022; Karkanorachaki et al. 2022; Ardestani 2022).

Polyvinyl chloride (PVC) degradation processes (O'Kelly et al., 2021; Karkanorachaki et al. 2022), polypropylene (PP) degrading mechanisms (Giorgetti et al. 2020), and polystyrene degradation mechanisms (Kumari et al. 2021) have already been examined elsewhere. In the natural environment, polyethylene (PE) degrades more slowly than other polyolefins, such as polypropylene (PP). PE's backbone chains are made entirely of C–C single bonds, which are resistant to hydrolysis and photo-oxidation, because they lack UV–visible chromophores.

Phosphorous chromophores can develop in polyethylene (PE) during its production or after weathering (Ng et al. 2018; Yu et al. 2019; Liu et al. 2022). PE's main chain or the endpoints of the chain may have a few unsaturated (CC) bonds (typically, vinyl groups in HDPE and vinylidenes in LDPE). It is easy for tropospheric radicals like O_3, NOx, and others to oxidize these sites, resulting in unstable hydroperoxides that can later be changed to UV-absorbing carbonyl groups that are more long-lasting (Hahladakis et al. 2018; Giorgetti et al. 2020; Liu et al. 2022). Due to the greater number of reactive branch points in LDPE, photo-oxidation rates were found to be higher than in HDPE. Thermal oxidative degradation of PE does not occur at temperatures below 100 °C in the absence of sunshine. There are similarities in the product distributions formed by photochemical and thermal processes due to light's role in photo-oxidative degradation simply being trigger chain reactions (Rhodes 2018; Giorgetti et al. 2020; Fan et al. 2022). (Fig. 3a). Anaerobic thermal deterioration is unlikely to occur naturally due to the high temperatures necessary (350 °C) in places lacking sunlight and oxygen (e.g., landfills).

When PET is hydrolyzed, it yields terephthalic acid and ethylene glycol, which are ultimately converted to shorter carboxylic acid- and alcohol-terminated chains. Hydrolytic cleavage of PET occurs at a fairly slow pace in the marine environment due to the near-neutral pH, but the rate is greatly accelerated under acidic environments (Conley et al. 2019; Karkanorachaki et al. 2022). During ester hydrolysis in landfills, the carboxylic acid products can cause a local pH reduction, which can lead to autocatalytic acceleration (Menéndez-Pedriza and Jaumot 2020; Ardestani 2022). This can be caused by insufficient moisture dilution when O_2 is present, heat degradation can occur via a free-radical mechanism, which is triggered when an excited carbonyl group absorbs the ester's -H. After reacting with O_2, the ensuing carbon-centered radical produces a peroxy radical, which can be used to build a new hydroperoxide and continue the chain reaction (Ardestani 2022; Liu et al. 2022). Carboxylic acids, aldehydes, and other oxygenates make up the bulk of the final product (Bergmann et al. 2019; Zhao et al. 2021).

There are carboxylic acid and vinyl-terminated chains that can be formed in the absence of O_2 by means of -H abstraction. Oxygenates with higher aldehyde concentration than carboxylic acids are formed in subsequent anaerobic processes (Bergmann et al. 2019; Hale et al. 2020; Zhao et al. 2021, Muhammed et al. 2022).

Photolytic cleavage of an ester bond in PET leads in the release of CO, CO_2, terephthalic acid, anhydrides, carboxylic acids, and esters from PET degradation (Fig. 3b). In aerobic settings, CO_2 production is higher (Bhattacharya 2016; Vighi et al. 2021; Wu et al. 2022). In both aerobic and anaerobic photodegradation processes, an excited-state carbonyl extracts an -H to give carboxylic acid and vinyl chain ends, as depicted in Fig. 3b. There are hydroxyl radicals created when the hydroperoxide O–O link is broken, and these radicals interact with the polymer backbone to form hydroxyterephthalate groups in the presence of O_2. Cross-links and new chromophores can be formed when radical intermediates and products mix, which can lead to polymer embrittlement and discoloration (although not necessarily mineralization) (Guo et al. 2020; Mohana et al. 2021; Dhaka et al. 2022).

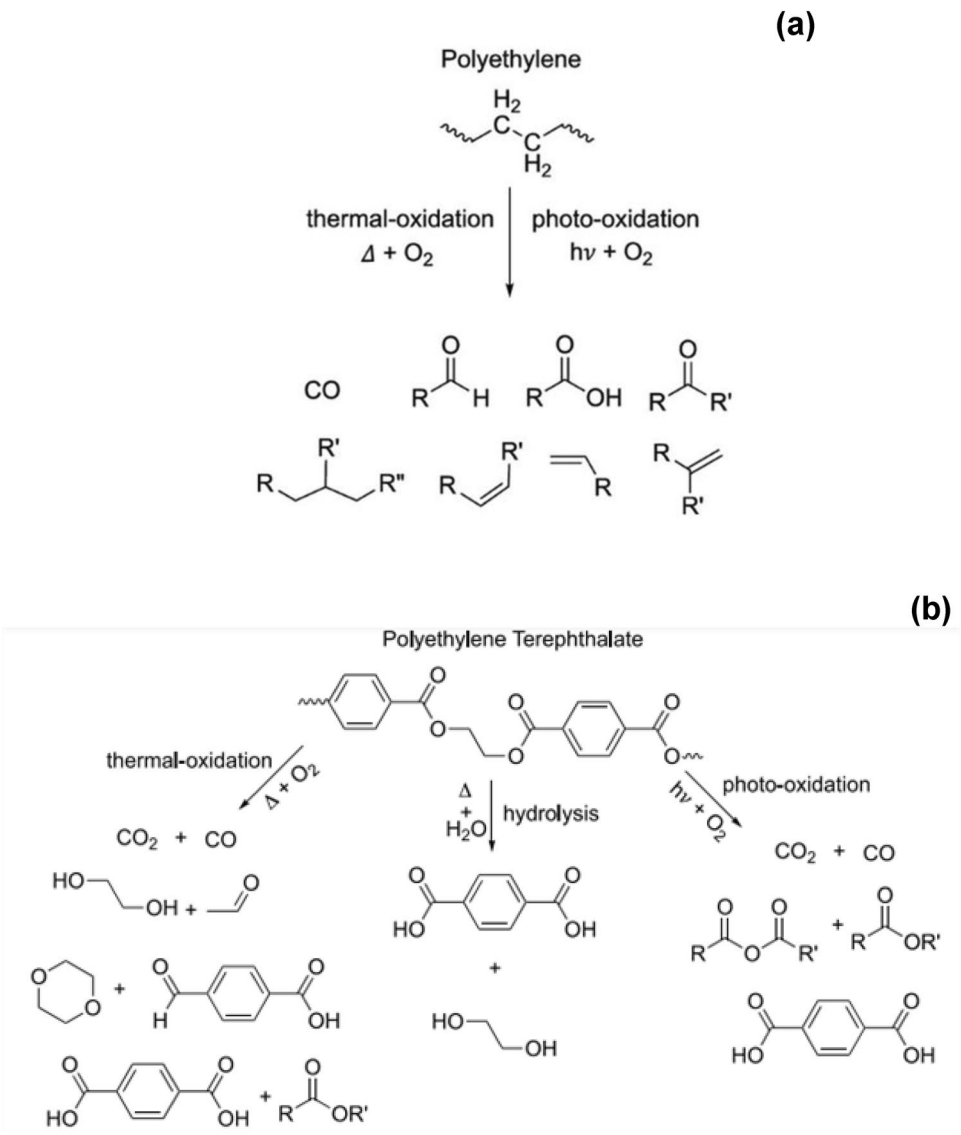

Fig. 3 Environmental impact on polyethylene plastics. **a** Polyethylene degraded by thermal or photo-oxidative methods. It produces a variety of by-products. **b** Degradation products from the three most prevalent ways polyethylene terephthalate degrades in the environment. The polymer chains R and R′ come in a variety of lengths

Methods for Assessing Plastics Degradation

Chemical changes (hydrophobicity, functional groups) in the polymer structure can be assessed using methods such as those reported for the investigation of plastics degradation, while physical changes in material qualities can be recorded using methods such as those reported (tensile strength, surface morphology, crystallinity, etc.)

Assessing Bond Cleavage

Mass Spectrometry

Polymer mass spectrometry can offer valuable information regarding polymer structure, molecular weight, polymerization degree, major functional groups, and end group structure. To examine the kinds of polymers in microplastics, mass spectrometry is frequently coupled with other approaches [Shah and Wu (2020), Wang et al. (2021, 2022), Issac and Kandasubramanian (2021)]. Mass spectrometry techniques commonly used to analyze microplastics at the moment include pyrolysis gas chromatography–mass spectrometry, thermal extraction desorption–gas chromatography–mass spectrometry, and matrix-assisted laser desorption ionization–time of fight-mass spectrometry (Ramasamy and Palanisamy 2021). Pyrolysis gas chromatography–mass spectrometry and thermal extraction desorption–gas chromatography–mass spectrometry are two important approaches for identifying microplastics by reverse analysis of microplastic thermal degradation products (Ramasamy and Palanisamy 2021). The initial step in both procedures is to put samples in an oxygen-free environment, such as inert gas. The polymers in microplastics are then thermally destroyed, resulting

in a vast number of thermal degradation products. In a chromatographic column, these products are collected and separated. Finally, mass spectrometry is used to examine the heat degradation products. However, because these two procedures may destroy samples, it is hard to get information regarding the physical properties of microplastics using these two approaches (Pashaei et al. 2022). Furthermore, various polymers may yield identical pyrolysis products, making it easy to misinterpret the kind of microplastic (Wu et al. 2021). Although the ideas, operations, benefits, and drawbacks of these two approaches are essentially the same, the two technologies have distinct features. For starters, pyrolysis gas chromatography–mass spectrometry may be used to analyze solid materials directly. Furthermore, this approach may be utilized to detect the main polymer types in microplastics as well as organic additives in microplastics at the same time. Furthermore, the amount of sample required is modest (5–200 g). Finally, there are no strict requirements for the size of microplastics when employing this approach. However, in general, pyrolysis gas chromatography–mass spectrometry can only be utilized to determine the chemical composition of single form microplastics (Liu et al. 2021; Pashaei et al. 2022). Thermal extraction desorption–gas chromatography–mass spectrometry has a shorter preparation time than pyrolysis gas chromatography–mass spectrometry. Pretreatment of samples is not always necessary. Thermal extraction desorption–gas chromatography mass spectrometry, on the other hand, has a rather limited application area. Currently, this approach is exclusively employed in the quantitative study of polyethylene microplastics. Matrix-assisted laser desorption ionization–time-of-flight-mass spectrometry is an analytical approach based on the proportional connection between ion fragment mass-to-charge ratio and time-of-flight (Pashaei et al. 2022). This approach is useful not only for identifying the major polymers in microplastics, but also for analyzing physical characterization of microplastics using imaging technology. However, because various ionization reagents are required for different types of microplastics, this approach lacks universality. As a result, whereas this approach is commonly used for determining biological macromolecules, it is rarely employed for detecting microplastics (Pashaei et al. 2022).

Mass Loss Polymer degradation can be quantified by measuring changes in mass, which is the simplest and most straightforward method. Degradation in soil (Wu et al. 2021; Melo-Agustín et al. 2022), compost (Liu et al. 2021; Pashaei et al. 2022), and labs with enriched microorganisms has all been assessed by measuring mass loss (Pignattelli et al. 2021). Degradation occurs at the plastic's surface; hence, the mass loss rate is inversely proportional to the plastic's surface area (Boots et al. 2019; Wahl et al. 2021; Xu et al. 2021; Wu et al. 2022; Batool et al. 2022). The mass of nonvolatile or insoluble polymeric material is reduced when small molecules (including but not limited to CO_2 and H_2O) are partially converted to small molecules and their subsequent volatilization or solubilization (Dong et al. 2021; Xu et al. 2021). The overall mass loss, however, confuses the liberation of tiny molecules with the flaking of bigger, insoluble fragments, including microplastics (0.55 mm) and mesoplastics (5–200 mm) (Li et al. 2021; Sharma et al. 2021). We still have a lot to learn about how plastics break down and produce microplastics. Plastic fragmentation in the water is influenced by its shape, according to new studies, and small pieces with low aspect ratios break apart more quickly, because their isotropic motion prevents biofilm development (Shah and Wu 2020; Wang et al. 2021, 2022; Issac and Kandasubramanian 2021; Ramasamy and Palanisamy 2021).

A lack of mass loss can be seen in the early stages of decomposition (Chen et al. 2020; Xu et al. 2021; Dhaka et al. 2022; Pizarro-Ortega et al. 2022). Due to oxygen incorporation and/or the attachment of microorganisms, the mass may actually rise at short exposure intervals (Samak et al. 2020). Additionally, the accumulation of decomposition debris might take place in surface cracks and pits. As a result, relevant results are rarely obtained without a lot of time in the lab. A combination of mass loss measurements and other analytical procedures mentioned below is preferable, because mass loss data alone are difficult to understand or extrapolate (Shabbir et al. 2020; Xi et al. 2022).

Changes in carbon dioxide concentrations: Despite the fact that polyesters can release some CO_2 when degraded anaerobically, CO_2 is the final destination of carbon in aerobic polymer degradation (Park and Kim 2019; Wang et al. 2020; Yuan et al. 2020; Zhao et al. 2021). Biodegradation is usually measured by the production of this compound. Methanogens and sulfate reducers degrade soluble carbon molecules in anaerobic conditions to produce CH_4 and CO_2, respectively (Paço et al. 2017; Auta et al. 2018; Park and Kim 2019; Wang et al. 2020). Analyzing the amount of CO2 released throughout the abiotic or biotic mineralization process can provide insight into the pace of polymer degradation (Kaur et al. 2021). Gas chromatography with thermal conductivity detection (GCTCD) (Yuan et al. 2020) and infrared spectroscopy (Paço et al. 2017; Auta et al. 2018; Park and Kim 2019; Faheem et al. 2020; Kumar et al. 2021; Melo-Agustín et al. 2022) and other analytical procedures such as trapping and titration can be used to measure CO_2 (Paço et al. 2017; Faheem et al. 2020).

Gel Permeation Chromatography (GPC)

Through the use of size exclusion, this technique identifies shifts in molecular weight, a key indicator of polymer degradation. This study was conducted by Sur et al. (2018)

and Băbău et al. Biotic and abiotic degradation mechanisms, which increase the concentration of chain ends and can lead to mineralization of smaller polymer chains and contribute to a decrease in molecular weight in partially degraded polymers, have both been documented (Mohanan et al. 2020; Kaur et al. 2021; Wu et al. 2022). High temperatures are required to dissolve polyolefins in a carrier solvent, which is the case with GPC. The dissolution of the polymer or the high-temperature measuring conditions must be taken care to avoid additional degradation of the product (Mohanan et al. 2020; Kaur et al. 2021; Wu et al. 2022).

Assessing Changes in Chemical Functionality

Chemical Analysis

Certain functional groups in polymers can be easily detected and quantified using spectroscopies such as nuclear magnetic resonance (NMR) and infrared (IR) (Park and Kim 2019; Singh et al. 2022; Wu et al. 2022). When HDPE is irradiated with gamma rays, 13C CP/MAS NMR shows the development of cross-linked polymer chains (through a 39.7 ppm peak) (Pignattelli et al. 2020). There are a number of compounds that can be detected by infrared spectroscopy, including polar functional groups like ketones and ester carbonyls (strong peaks at 1715 and 1735 cm^{-1}, respectively) (Jiang et al. 2019; Pignattelli et al. 2020; Campanale et al. 2020). Although a recent study suggests that the Carbonyl Index may be less accurate than other modes, such as methyl deformation, to quantify the extent of oxidation, the Carbonyl Index can be used to quantify the extent of oxidation during degradation (Campanale et al. 2020; Pignattelli et al. 2020; Li et al. 2020).

Fourier Transform Infrared (FT-IR) and Raman Spectroscopy

Although GC/MS has been reported to be a technique capable of quantitative and qualitative detection of MPs, it has certain drawbacks, including time-consuming sample preparation and a difficult operating procedure (Mohanan et al. 2020; Kaur et al. 2021). Non-destructive vibrational spectroscopy-based methods, such as FT-IR and Raman spectroscopy, have the advantage of requiring a small sample amount, enabling high-throughput screening, and, most importantly, obtaining "fingerprint" information, such as the chemical composition and crystal structure of the target, quickly and accurately (Campanale et al. 2020). In fact, the European Union expert group on marine litter and the technical subgroup on marine litter have recommended that all suspected MPs in the 1–100 mm size range have their polymer information confirmed by spectroscopic analysis as a better choice for MP identification (Jiang et al. 2019). However, as compared to FT-IR spectroscopy, Raman spectroscopy has a larger spectrum coverage and greater sensitivity to non-polar groups. Furthermore, because FT-IR is affected by water molecules, samples must be fully dried before measurement. As a result, Raman spectroscopy has increased potential for detecting MPs. SERS technique, in contrast to classical Raman spectroscopy, primarily employs the electromagnetic (EM) hot spot phenomenon in metal nanostructures to increase the Raman signal strength. Unfortunately, the hydrophobic property of MPs prevents the creation of hot spots, making the use of SERS technology to MP detection difficult. However, flexible materials, such as a filter paper-based substrate, may trap MP particles through fiber holes, solving the hydrophobic issue of MP particles properly. As a result, it is important for MP detection using the SERS approach on filter paper-based substrates (Mohanan et al. 2020). For the first time, SERS technology was used to the detection of MPs utilizing a filter paper-based flexible substrate doped with gold nanoparticles (AuNPs) in this work (Lamichhane et al. 2022; Ardestani 2022). The pores of the fiber filter paper effectively capture MPs, while the doped AuNPs form EM hot spots, amplifying the Raman "fingerprint" peaks of MPs (Ardestani 2022).

Contact Angle

When the surface density of polar functional groups, such as those generated during oxidative degradation, varies, the surface energy changes as well, and this is reflected in the contact angle with liquids (Campanale et al. 2021; Lamichhane et al. 2022; Ardestani 2022). Hydrophilic surfaces have higher surface energy and lower water contact angles because of their high wettability. Because UV radiation causes the development of polar functional groups in polymers, this leads in a decrease in contact angle. Degradation is hastened further by microbe adhesion to polymer surfaces due to increased hydrophilicity (Shabbir et al. 2020; Fan et al. 2022; Li et al. 2022).

Assessing Changes in Materials' Properties

Dynamic Mechanical Analysis (DMA)

Polymer strength is commonly measured using this method. Physical deterioration during polymer degradation can also be seen in changes in tensile strength and elongation at break (Jambeck et al. 2015; Wang et al. 2019a; Liu et al. 2021). Molecular weight loss and fracture and hole formation are also linked to changes in these mechanical characteristics (Shah et al. 2020; Colzi et al. 2022).

Thermal Analysis

A sample is heated or cooled at a predetermined rate, while its physical properties are monitored (Boots et al. 2019; Rodríguez-Seijo et al. 2019; Menzel et al. 2022). The heat capacity (Cp), melting temperature (Tm), and glass transition temperature (Tg) can all be measured using differential scanning calorimetry (DSC) (Lian et al. 2021; Batool et al. 2022). Polymer breakdown results in a drop in Tg, because shorter chains have a higher rate of movement (Lian et al. 2021; Batool et al. 2022). As a result of heating, mass changes are measured by thermal gravimetric analysis (TGA). The oxidation or loss of volatiles that occurs during heat deterioration can be determined using this method in conjunction with product analysis (Rodríguez-Seijo et al. 2019; Pflugmacher et al. 2020; Yu et al., 2021; Wahl et al. 2021).

Surface Analysis

Scanning electron microscopy and atomic force microscopy (AFM) can be used to detect changes in polymer surfaces during deterioration (Guo et al. 2021; Liu et al. 2022). These techniques can capture images of topographical changes on the polymer surface, such as holes and cracks forming, roughness increasing, or even microorganisms adhering to the surface (Isaac and Kandasubramanian 2021; Kumari et al. 2021; Zhao et al. 2021). Polymer morphological changes are observable as cracks and voids; HDPE films show surface degradation and degeneration after 6 months in the maritime environment (Yu et al. 2018; Guo et al. 2021; Wahl et al. 2021; Ardestani 2022; Fan et al. 2022).

Rates of Plastics Degradation and Extrapolated Lifetimes

Describing Degradation Rates

Depolymerization, chemical modification, alteration of physical properties, total mass loss, or complete mineralization to CO_2 and H_2O are all examples of degradation in the literature. Degradation is defined as the loss of total mass from the original polymer piece for this investigation. To be clear, this term applies well to huge plastic objects. In the marine environment, surface ablation of small plastic fragments may be crucial, according to recent studies (Lozano and Rillig 2020; Wahl et al. 2021; Vighi et al. 2021; Guo et al. 2021; Kabir et al. 2022). The starting mass is reduced by the loss of microplastic or nanoplastic particles, but the total amount of plastic remains same.

Degradation rate, or rd, is the rate at which a polymer loses mass per unit time (Qin et al. 2021; Yu et al. 2018; Guo et al. 2021; Kabir et al. 2022). We assume that the degradation rate is proportional to the surface area (SA) and that the rate constant k has dimensions kg s^{-1} m^{-2}, because degradation happens primarily on exposed surfaces. When it comes to plastics, there are several factors that influence the degradation rate, including the intrinsic properties (polymer type and molecular weight), environmental circumstances (temperature, presence of moisture and air, etc.), and the extrinsic properties (size and form) (Kabir et al. 2022).

A cylinder is a more accurate representation of the form of plastic fibers. SA 2 rh can be assumed if the aspect ratio is big (i.e., radius r/height h). Using this assumption, we can get the equations for mass loss and an estimate of the total deterioration time. To put it another way, in the absence of considerable fragmentation, crystallization, or form dependence, an HDPE film should disintegrate fully in the same amount of time as fibers of the same mass and crystallinity (Karkanorachaki et al. 2022; Ardestani 2022; Kiran et al. 2022; Wu et al. 2022). It is 390:3:1 for the film, fiber, and bead when it comes to first degradation rates (based on surface area ratios). Fiber and bead deterioration rates decline as their radii are smaller, and hence, their "average" degradation rates are even lower as well. Because of these long extrapolations, it is possible to have errors that are many times larger than those from the shorter extrapolations for film and fiber. In other words, if the relative error in kd is 20%, the film will decay in 1.8 years, the fiber in 465 years, and the bead in a thousand years if kd is at a constant 20%. Even if kd does not change significantly over the course of two millennia, we may safely predict that its errors will get even larger.

There are a few other things worth mentioning. For starters, the degree of surface roughness will probably change over time. Melt-processed polymers often start out with smooth surfaces. Surface pitting and cracking will occur as degradation continues, increasing the surface area and speeding up degradation. Surface ablation and mass loss can occur as a result of microplastic pieces being released from fissures of this type. Another element that affects degradation rates is the presence of an amorphous polymer (aSA, 0 – a – 1) on the surface of the material, which will require the addition of a scaling factor to represent the amorphous surface area fraction. Slower degradation might be expected after the amorphous polymer has been removed. It is not yet possible to quantify or simulate the effect of partial polymer degradation on cross-linking or crystallization in the amorphous zones next to crystallites (O'Kelly et al., 2021; Wu et al. 2022; Weber et al. 2022).

Polymer degradation times in the environment are clearly influenced by the material's shape and size, as well as its inherent chemical reactivity, as shown in this study (Shah and Wu 2020; Isaac and Kandasubramanian

2021; Wu et al. 2022). The SSDR can be used to compare the initial degradation rates of plastics of varying sizes and compositions, but with similar aspect ratios, so long as the surface area remains roughly constant. In the next paragraphs, we will address this assumption (Wu et al. 2022; Weber et al. 2022).

Analysis of Reported Degradation Rates

Fewer than 25 of the thousands of peer-reviewed articles that were considered for inclusion in this perspective provided all of the data necessary to calculate an SSDR (Lozano et al. 2020). Each plastic type and degrading environment (landfill/soil/compost; marine; biological; or sunlight) is grouped together in the data set. Despite the fact that these categories are not orthogonal, they can be used to represent the four main types of polymer degradation experiments that have been carried out: on land (without exposure to sunlight), in water (in freshwater or seawater with exposure to sunlight), in a lab using enzymes and microbes, or with exposure exclusively to sunlight and air. These include PLA, polyhydroxybutyrate (PHB), Poly (3-hydroxybutyrate Co. 3-Hydrochloro-Valerate) (PHBV), Mater -Bi, and polycaprolactone (PCL). Filler components (e.g., starch and PLA) and accelerating conditions (heat pretreatment, UV pretreatment, and microbial incubation) are indicated in each investigation. The presence of these components and conditions are noted. The degradation of blends of polypropylene (Patil et al. 2019; Mohana et al. 2021) and polyethylene (Nikiema et al. 2020; Vighi et al. 2021) with a biodegradable filler was examined in two articles used to build (5 or 10 wt percent starch, respectively). The scientists hypothesized that the filler enhances the deterioration rate of the plastic indirectly, potentially by increasing the surface area of the plastic when the starch component dissolves. There are several issues with this method of determining plastics breakdown rates. There is a lack of data on the degradation of PET, PP, and PS. In the end, there was just one report that satisfied our criteria for PVC degradation in any environment and reported no significant degradation after 32 years (Monkul and Özhan 2021; Kumar et al. 2021; Liu et al. 2022; Pashaei et al. 2022).

"Others" is the plastics type that includes a variety of plastics that claim to be biodegradable. On the right, you will see the range and average value for plastic types 1–6, as well as biodegradable "others." On the x-axis are data points demonstrating incredibly slow degradation rates. Columns with no data in them are shown in gray.

The SSDR values for all plastics kinds, regardless of environmental conditions, can vary by several orders of magnitude if just the nonzero values are taken into account. There are some statistically significant differences between the accelerated degradation and nonaccelerated degradation values, but they are not large enough to be statistically significant. Degradation rates, nevertheless, vary by an order of magnitude in most cases when polymer pretreatment or a filler is used to accelerate degradation filled circles). For LDPE decomposition on land, the range is a factor of 50, and this variance is most apparent. It has been stated that the fastest SSDR for LDPE can be as high as 83 µm per year for an 80-weight-per-percent PLA blend decomposing at 37 degrees Celsius (Boucher and Friot 2017). This is the lowest accelerated SSDR for LDPE that has been recorded, and it was measured after the soil was inoculated with *P. aeruginosa*. While the SSDR for pure PLA is lower than the SSDR for LDPE/PLA blend (83 m year^{-1}), both were evaluated in composting conditions at 37 °C and found to be greater than the SSDR for pure PLA. Due to this, the trial durations for the LDPE/PLA blend (28 days) and pure PLA (365 days) were different. Because blending can increase the volume percentage of amorphous regions, which have a higher degradation rate, the gap could potentially be attributable to crystallinity discrepancies (Nizzetto et al. 2016; Bergmann et al. 2019).

While it is possible to extrapolate degradation rates for plastics containing degradable additives (such as starch), this may be erroneous. Microorganisms are usually the first to destroy such additives. By combining environmental degradation (e.g., photo-oxidative, hydrolysis, etc.) with microbial action (Fig. 4), the residual polymers disintegrate much more slowly once the readily accessible filler has been consumed (Menéndez and Jaumot 2020; Weber et al. 2022). Degradation should be defined in terms of multiple stages of time-dependent dynamics (Conley et al. 2019; Pashaei et al. 2022). The length and rate of these phases can vary depending on the material's size, the type and concentration of the filler, the degradation environment and conditions, etc. Plastics' abiotic breakdown may be accelerated in comparison to the polymer without filler because of the increased surface area (Conley et al. 2019; Wu et al. 2022; Kiran et al. 2022). Blended polymer degradation kinetics should not be used to infer the mechanisms and degradation rates of pure polymers, despite the convenience of shorter time scales (Fan et al. 2022).

Breakdown in marine environments may be comparable to the degradation rates of petrochemical plastics despite the fact that biodegradable polymers such as PHB and PLA show considerable average SSDRs (59 and 21 m year^{-1}) in compost and landfill conditions (Wang et al. 2018; Bergmann et al. 2019; Nikiema et al. 2020). When comparing PLA to HDPE and LDPE, the average SSDR in the marine environment, 7.5 m year^{-1}, is comparable. However, even though PLA and other "biodegradable" plastics are supposed to decompose totally in industrial composting settings (60 °C and damp), the temperature

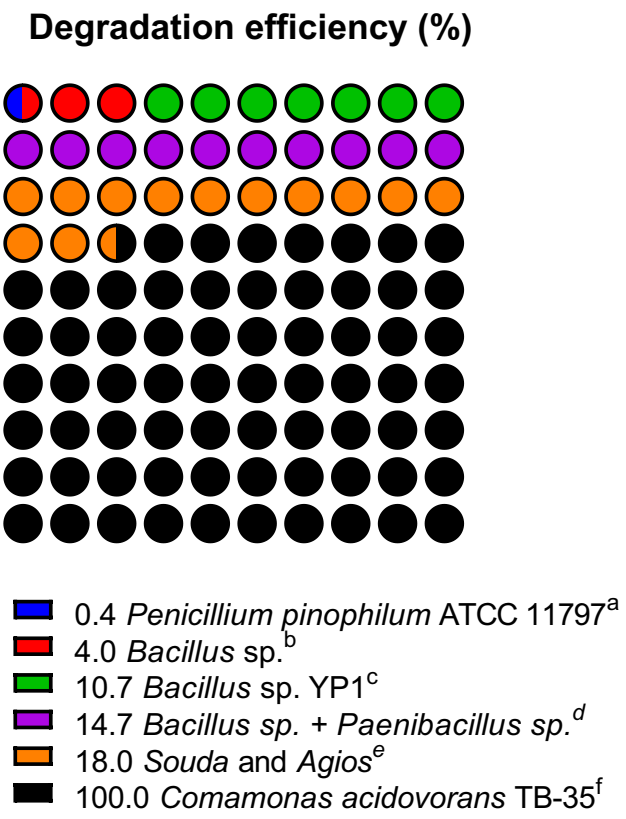

Fig. 4 Plastic degradation efficiencies of different microbial species. Details of type of plastic/environmental condition/incubation time/reference of each experiment are as follows: [a]Low-density polyethylene/in vitro/31 months/Volke-Sepúlveda et al. (2002); [b]Polypropylene/Mangrove sediment/40 days/Auta et al. 2017; [c]Polyethylene/plastic-eating waxworms/28 days/Yang et al. (2014); [d]Polypropylene/municipal landfill sediment/60 days/Park and Kim (2019); [e]High-density polyethylene/Souda bay/60 days/Tsiota et al. (2018); [f]Polyurethene/in vitro with esterases/8 days/Nakajima-Kambe et al. (1997)

in marine habitats rarely rises to the level necessary for depolymerization (20 °C being the maximum) (He et al. 2013; Rhodes 2018; Qi et al. 2018; Wang et al. 2018; Shah and Wu 2020; Qin et al. 2021). When compared to petrochemical-based plastics like HDPE, PLA's average SSDR is significantly higher (21 m year^{-1}) in soil conditions compared to petrochemical-based plastics like HDPE. There has been evidence that the temperature in landfills can get to 80–100 °C (152), which is hot enough to breakdown PLA and other plastics that contain moisture (Giorgetti et al. 2020; Shah and Wu 2020; Lamichhane et al 2022; Kasmuri et al. 2022).

Extrapolated Degradation Times

Arrhenius extrapolation of accelerated aging results and extrapolation based on initial rates measured in environmentally relevant conditions have been widely used to estimate polymer lifetimes, defined here as the time required for complete degradation (> 99% loss of the initial polymer mass). The validity of both is limited by the assumptions they make, which are often unjustified. Arrhenius-like temperature dependences are assumed in the first technique (Bergmann et al. 2019; Kasmuri et al. 2022; Kabir et al. 2022).

This extrapolation is based on the quicker deterioration rates that are more easily detected at higher temperatures, between 25 and 200 °C. For example, extrapolation of declines in tensile strength (Weber et al. 2022; Ardestani 2022; Fan et al. 2022; Liu et al. 2022) and molecular weight measurements at temperatures of 25–50 °C estimated the lifetime of poly(3-hydroxybutyrate-co-3-hydroxyvalerate) at 25 °C in distilled water to be 300 d. (Kiran et al. 2022; Singh et al. 2022). Arrhenius extrapolation, on the other hand, presupposes that the same degrading mechanism is active at all relevant temperatures, which is not the case. The Arrhenius graphs for PE and PP are nonlinear (Jambeck et al. 2015; Fan et al. 2022) suggesting a change in the mechanism and/or rate-determining step with temperature (Shah and Wu 2020; Kasmuri et al. 2022; Kabir et al. 2022).

For biodegradation rates, Arrhenius extrapolation is less useful, since enzymatic degradation occurs only at biologically relevant reaction conditions; they rarely occur at extreme temperatures. As a result, biodegradation lifetimes are often anticipated by the second method, which uses beginning degradation rates measured over a long period of time instead of a shorter period of time (Issac and Kandasubramanian 2021; Kasmuri et al. 2022; Kabir et al. 2022). If, for example, the degradation rates of LDPE–starch blends were estimated by observing the weight loss over the 125 day composting period, this rate was used to anticipate the total degradation time using a linear extrapolation, as was the case in this case (Hahladakis et al. 2018). Extrapolation based on initial data, however, could be highly misleading due to the fact that relatively rapid biodegradation of the starch component should be followed by considerably slower degradation of the remaining LDPE. The method's precision is jeopardized because of the occurrence of many deterioration stages with widely divergent rates (Guo et al. 2020; Kabir et al. 2022). As a result, accurate lifespan estimations necessitate data on degradation rates for each component of a mixture.

Estimating plastics' lives in the environment is a simple process that relies on extrapolation from starting rates. There is a need to know about the rate law, but it does not take into consideration the material's shape. When conducting TGA studies of phase changes, the shrinking core model posits that the volume (and thus the surface area) decreases over time (Wang et al. 2018; Lamichhane et al. 2022; Kiran et al. 2022). It may be more realistic, but it ignores surface

roughness variations, which can be substantial. The shrinking core model has so far only been applied to high-temperature catalytic degradation of plastics, which is not relevant to how plastics degrade naturally (Qi et al. 2018; Kabir et al. 2022).

We assume a straightforward first-order dependence on SA and further assume that SA is constant over time, since experimental examination of polymer degradation rate laws would necessitate kinetic studies lasting decades or even longer (i.e., pseudo-zeroth-order behavior). There are significant implications for these assumptions, and they can lead to extrapolation errors that add decades or centuries to one's lifespan (Qi et al. 2018; Kabir et al. 2022; Li et al. 2022). When a pseudo-zeroth-order rate law (constant surface area) is followed, a 100 m-thick film with an SSDR of 1 m $year^{-1}$ would take 100 years to degrade entirely, whereas the same film would be 99% destroyed in almost 500 years if the reaction is first-order degradation. First-order deterioration would take 68 years to reach 50% completion, whereas pseudo-zeroth-order degradation would take 50 years to accomplish the same percentage (Lamichhane et al. 2022).

Additional changes in the polymer's chemistry and morphology are expected as the process progresses toward high conversion. The rate of response can be drastically altered by the disintegration of larger components (e.g., microplastics). It is unclear exactly how mechanical forces can induce weathered or oxidized surfaces to flake, but recent studies have shown that this can lead to ablation and macrofragmentation (Boucher and Friot 2017; Isaac and Kandasubramanian, 2021 Lamichhane et al. 2022). Over the course of many weeks, the quantity of nanometer and micron-sized particles emitted by numerous polymer kinds in a weathering chamber increased exponentially (Bhattacharya 2016; Hale et al. 2020). Since synthetic fabrics and rubber tires are thought to be the primary producers of microplastics and nanoplastics (Isaac and Kandasubramanian 2021), fragmentation of rigid objects, such as bottles, bags, and pipes, may not be a significant source of these microplastics and nanoplastics.

Polymer degradation is a complex process that cannot be adequately predicted by extrapolation models. Therefore, we use the initial half-life, or the time in which the material loses half of its original mass, to compare deterioration rates (Mohana et al. 2021; Melo-Agustín et al. 2022; Mohana et al. 2021; Xu et al. 2021; Melo-Agustín et al. 2022). SSDRs based on reported minimum and maximum degradation rate values multiplied by the typical plastics thickness for the specified application give a wide range of half-lives, highlighting the substantial degree of uncertainty in the extrapolation. An initial assumption was made that decomposition proceeded in a straight line from one end to the other. The half lifetimes of LDPE plastic bags and HDPE milk bottles and laundry detergent containers in landfill/compost/soil conditions are expected to be 5 and 250 years, respectively. The projected half-lives in marine environments are 3.4 years and 58 years, respectively. Due to extremely slow decomposition rates in both landfill/compost/soil and marine environments, all figures are subject to considerable uncertainty. It may take thousands of years for heavy industrial objects like HDPE pipes to completely degrade, regardless of the environment (we note that the durability of such items is often desirable for their intended purpose) (Ramasamy and Palanisamy 2021; Mohana et al. 2021; Zhao et al. 2021; Melo-Agustín et al. 2022).

Assuming pseudo-zeroth-order kinetics, the polymer mass is converted to half of its original mass in half of the time (i.e., invariant SA). Assuming cylindrical form and decreasing radius (but not height) with time, half-lives are predicted to be twice as long as they were before. To the best of our knowledge, no published data were discovered for any of the criteria marked with (Sharma et al. 2021). The ranges in parentheses are the values that have been reported in the literature. Rounding all values to two significant figures was done. It is possible that the trial was too short to measure a substantial weight reduction if the values of 0 m $year^{-1}$ are observed in reports. When calculating average deterioration rates, these variables were taken into account (Mohana et al. 2021; Liu et al. 2022; Lamichhane et al. 2022).

Degradation accelerator (UV pretreatment, thermal pretreatment, or microbial incubation) and/or polymers with a quickly disintegrating filler are included in this category (e.g., starch, PLA, etc.). This means that the relevant investigations discovered no observable degradation, probably because the duration of the experiment was too brief, allowing only an approximated lower limit for the projected degradation period (Singh et al. 2022). $d > 2500$ Based on the sensitivity of the balance used in the investigation and the half-life of the thinnest substance (100 nm, ca. 2500 years), the value of 2500 years was calculated (Giorgetti et al. 2020; Vighi et al. 2021; Wang et al. 2022).

A constant degradation rate, constant surface area-to-volume ratio, consistent reaction order, uniform crystallinity, and a mechanism for microplastic degradation identical to the parent materials can all be assumed to get an idea of how long it will take for a material to degrade completely (Pflugmacher et al. 2020; Singh et al. 2022). Such calculations are highly speculative because of these assumptions (Singh et al. 2022). Nevertheless, it is projected that it will take 500 and 116 years, respectively, for an HDPE bottle to completely degrade in the land and marine environments.

Environmental Effects on Degradation Rates

Polymers are broken down into smaller bits (eventually to microplastics) and cleaved into smaller molecules when they are exposed to environmental elements, such as moisture,

heat, light, or microbial action (Issac and Kandasubramanian 2021). When it comes to the rate at which a plastic degrades, the type of plastic has a significant impact on the factor. Studies of petrochemical-based polymers have shown that breakdown rates in the marine environment are lower than in landfills (Shah and Wu 2020; Issac and Kandasubramanian 2021; Singh et al. 2022). Because of lower temperatures and lower dissolved oxygen concentrations in the marine environment, the variations are often attributed to (Bhattacharya 2016; Giorgetti et al. 2020; Issac and Kandasubramanian 2021; Guo et al. 2021). Even though the differences could be statistically negligible because of the considerable uncertainties in the averages, our literature study suggests that average degradation rates for HDPE and LDPE are slightly higher in marine environments than degradation on land. When compared to a landfill, ocean temperatures and oxygen concentrations may be less harmful because of the more powerful UV light.

"Heat accumulation" can occur when plastics exposed to sunlight on land are subjected to higher temperatures than the surrounding air, causing them to degrade at an accelerated rate (Lei et al. 2017; Pflugmacher et al. 2020; Kumari et al. 2021). It has been claimed that landfills and industrial composters can achieve temperatures of 80–100 °C, which can speed up the thermal–oxidative degradation and hydrolysis processes, respectively, if adequate oxygen and/or moisture are present. Under industrial composting conditions (60 °C), for example, PLA undergoes ester hydrolysis, whereas it is very slow to breakdown at lower temperatures (Hale et al. 2020; Liu et al. 2022; Lamichhane et al. 2022). As a result, PLA appears to be equally as resistant in maritime conditions, where temperatures are much below 60 °C, as their petrochemical equivalents.

Photodegradation is impeded by the lack of sun UV light in landfill/soil/compost settings. Photodegradation can be slowed by biofouling, because sunlight penetration is reduced (Boucher and Friot 2017; Pflugmacher et al. 2020; Kumari et al. 2021; Babaniyi et al. 2023b). While some plastics (such PET, PVC, and PLA) do not float in the ocean, biofouling can increase the total density of plastic pieces, resulting in them sinking (Patil et al. 2019; Nikiema et al. 2020). As plastic trash has been found to sink and float again, the process may be time-dependent. Due to the lack of sunlight required to maintain the film, defouling can occur after the fouled plastic debris has sunk, causing the density to drop and allowing the debris to resurface (Qi et al. 2018; Patil et al. 2019; Nikiema et al. 2020). Similar changes in buoyancy are likely to affect the decomposition rate of biodegradable plastics as well, according to studies on petroleum-based plastics. Biodegradable plastics appear to decompose at the same rate as petrochemical-based plastics when exposed to sunlight.

Effects of Accelerating Conditions

UV Irradiation

To begin photo-oxidation in most polymers, solar UV light is required, which leads to bond breaking and a drop in molecular weight via a radical chain process (Patil et al. 2019; Zhang et al. 2020; Nikiema et al. 2020). Due to their enhanced hydrophilicity (which improves microbial adherence), shorter chains are more susceptible to assault by microbes and mineralization (Lozano and Rillig 2020). Polyethylene's photo-oxidation and biodegradation have been shown to work together in numerous experiments (Qin et al. 2021). Carbonyl groups in polyethylene increased after exposure to UV light for 60 h, and then dropped when the photo-oxidized polyethylene was incubated with microbes, suggesting microbial breakdown of the polymer (Panigrahi et al. 2019).

Transient Thermal Treatment

Polyolefin oxidative breakdown is considerably accelerated by moderate heating in the presence of air. The inclusion of oxygen-containing functional groups in polymers enhances the ability of microbes to adhere to the surface (Boucher and Friot 2017; Patil et al. 2019; Kumari et al. 2021). LDPE, HDPE, and PP films were thermally processed for 10 days at 80 °C to increase biodegradation rates to 12, 4, 5, and 2, respectively, in a soil culture during a 12-month period, compared to 6.3, 1.8, and 0.01 m $year^{-1}$, respectively, without heat pretreatment (Ng et al. 2018). More than four-to-seven times as many marine bacteria colonized prepared LDPE and HDPE films than untreated ones after 6 months of thermal pretreatment in an oven at 80 °C. Thermally processed LDPE and HDPE had higher mass losses than untreated materials over the same time period (Menéndez-Pedriza and Jaumot 2020), with losses of 17% and 5.5%, respectively.

Humidity

Hydrolysis promotes polyester breakdown when the humidity level is high. At 60 °C and 100% relative humidity, chain scission of PET in a plastic bottle was five times greater than at 45 °C and 40% relative humidity (Menéndez-Pedriza and Jaumot 2020). Because thermal–oxidative deterioration occurs at a faster rate than hydrolysis even at temperatures over 80 °C, an increase in humidity did not significantly boost hydrolysis rates. PLA (He et al. 2013; Menéndez-Pedriza and Jaumot 2020) and polyolefins like PE (Hahladakis et al. 2018), PP (Bergmann et al. 2019), and PVC (Jambeck et al. 2015) have been demonstrated to degrade

faster in the presence of humidity (Menéndez-Pedriza and Jaumot 2020).

Effect of Micro- and Nanoplastics in the Environment and Soil

Due to their constant release and buildup in the environment, micro(nano)plastics, new developing pollutants, are found everywhere. In addition to posing a concern to human health and the environment, widespread use of micro(nano)plastics could alter the world biodiversity by increasing the amount of time organisms are exposed to them. Some studies have been conducted on the distribution of micro(nano)plastics in the environment; contamination sources; methods of testing; and environmental effects on creatures that ingest micro(nano) plastics. As a result, only a small amount of research has been done into how micro(nano)plastics affect global biodiversity, and this research is currently in its infancy as a result.

Effect of Micro(nano)plastics on Growth and Reproduction

As micro(nano)plastics have been shown to alter the growth, development, and reproductive toxicity of live creatures, they could be regarded a factor in biological growth and reproduction (Wang et al. 2019a; Ardestani 2022). Organisms can more easily absorb and collect smaller microplastic particles. There is strong evidence that PVC microplastics (Sparus aurata) have a negative impact on the growth, oxidative damage, oviposition, and biological enzyme activity of gilthead seabream (Wang et al. 2020; Babaniyi et al. 2023a). A high dose of PVC microplastics (500 mg kg^{-1} for 30 days) cannot be found in the environment. In oysters, evidence showed that PS microplastics reduced egg cell size and number, as well as sperm motility and larval number, at a concentration of 0.023 mg L-1 (2–6 m); in Gammarus pulex, evidence showed that PS microplastics reduced growth by 10% to 40% (mass concentration, microplastics/sediment, 20–500 m) (Shah and Wu 2020; Yu et al. 2019; Kabir et al. 2022); and in shrimp, evidence showed that PS microplastics reduced growth by The filter-feeding rate of the blue mussel (*Mytilus edulis* L.) might be decreased by 0.3 g L1 (30 nm) by PS nanoplastics, and this could have a higher influence on the cellular and organizational levels (Schöpfer et al. 2020). Microplastics have been shown to impede the reproduction of earthworms, as well as the vegetative and reproductive growth of wheat (Hahladakis et al. 2018). Mice exposed to microplastics (Briassoulis et al. 2015; Monkul and Özhan 2021; Yu et al. 2019; Kabir et al. 2022) showed altered energy and lipid metabolism and oxidative stress. Furthermore, micro(nano)plastics can increase sediment permeability, which alters heat transmission and so affects temperature-related biological development and growth. The sex ratio of turtle eggs during incubation could be affected by temperature fluctuations (Zhang et al. 2020; Kabir et al. 2022).

Since PS nanoplastics can easily pass through lipid membranes, they alter membrane structure and protein activity, weakening molecular diffusion rates and so modifying cellular activities. PS nanoplastics: (Lozano and Rillig 2020). It was found that the formation of coronal protein rings on the surface of nanoplastics significantly impacted the endocytosis of the nanoplastics in a study by O'Kelly et al. (2021) that used six different PS nanoparticles with three different surface chemicals (carboxyl-modified, amine-modified, and plain PS) and two sizes (50 nm and 100 nm). The blood–brain barrier, a highly selective barrier, may be breached by PS nanoplastics (39.4 nm), putting organisms' brain tissues at risk (Zhang et al. 2020; Kabir et al. 2022). As small as 70 nm and 20 g L^{-1} of PS nanoplastics have been found to significantly alter the movement of zebrafish larvae before entering the circulatory system and accumulating in the liver, where they can cause infection and abnormal lipid accumulation, leading to problems with liver function and energy metabolism (Kiran et al. 2022). Crescent algae, chlorella, and grid algae are electrostatically attracted to PS nanoparticles (20 nm) by the electrostatic effect, which reduces the absorption and utilization of carbon dioxide and photons by algae cells and limits algal growth (Yu et al. 2019; Kabir et al. 2022). Compared to negatively charged PS nanoplastics, positively charged PS nanoplastics can greatly increase the content of reactive oxygen species and free calcium ions, while simultaneously decreasing the mitochondrial membrane potential and the content of adenosine triphosphate in cells, thus affecting cell activity and proliferation (Pflugmacher et al. 2020; Kiran et al. 2022). It is possible that species and ecosystem services could be reduced and global biodiversity will shift if normal survival, growth, and reproduction of species are stopped. Toxic effects on organisms of micro(nano)plastics have been documented; however, tests were carried out using high or extremely high dosages of micro(nano)plastics in a short period of time. More research is needed to determine the long-term effects of exposure to micro(nano)plastics at low doses or in the environment.

Furthermore, the majority of studies on the toxicity and danger of micro(nano)plastics have been conducted utilizing PS. PVC, polyethylene (PE), polypropylene (PP), and polyethylene terephthalate (PET) are some of the other micro(nano)plastic compounds that may require further study (PET) (Aransiola et al. 2023b). As a result of exposure to micro(nano)plastics, microbial populations have changed microplastics have been found to increase the impact of treated wastewater on freshwater microbes,

according to studies on the microbial community (Guo et al. 2020; Kiran et al. 2022). Halobates sericeus spawning density could be increased by increasing the concentration of microplastics in the environment, and the predation of eggs and larvae could speed up the transfer of energy among different communities, influencing and changing the microbial community's structure and composition (Kumari et al. 2021; Karkanorachaki et al. 2022). Changes in the structure of the biological community caused by the accumulation of microplastics in pelagic waters have been discovered, particularly the enhancement of pathogenic bacterial reproduction (Qin et al. 2021; Kumari et al. 2021; Pashaei et al. 2022; Weber et al. 2022).

When water is exposed to microplastics (PE, PP, and PS), the bacterial community can be altered, and microplastics host a distinct bacterial community (Nizzetto et al. 2016; Giorgetti et al. 2020; Zhang et al. 2021; Giorgetti et al. 2020; Kumari et al. 2021). While the characteristics of the bacterial and fungal community were clearly linked to microplastics (Nizzetto et al. 2016). With the presence of biodegradable plastics, changes in the structure, composition, and function of the soil biological community can be clearly seen, mainly by increasing soil temperature and reducing gas exchange; at the same time, enrichment of fungal taxa and microbial activity were significantly enhanced, while the population of soil invertebrates was decreased (Wang et al. 2021, 2022; Guo et al. 2021). Even more importantly, microplastic mulches can alter soil physical structure to effect root development and plant exudation. (Giorgetti et al. 2020; Wang et al. 2021, 2022; Aransiola et al. 2023b).

Effects of MPs on the Environment

The MPs fragments are a major issue in many ecological fields, and they have been found in the environment. Other research has found that MPs have traveled everywhere from the highest peak on Earth (Mount Everest) to the depths of the ocean (Bergmann et al. 2017; Cunningham et al. 2020). Only about 20% of MPs were found to have aquatic origins, while nearly 80% came from land. Among the environmental effects of MPs is the death and injury of aquatic birds, fish, mammals, snails, worms, reptiles, and other organisms (Fig. 5) due to plastic aggregation and digestion (Guo et al. 2021). Below, we will examine the primary environmental concerns that have garnered attention in recent decades, including the effects on the land, sea, and people.

Ecological Effect of Micro(nano)plastics

Due to their high diffusion capacity, hydrophobic surface, and enormous specific surface area, micro(nano)plastics have been shown to be a novel type of biological and chemical vector. Micro(nano)plastics can enrich organic and inorganic contaminants, providing a fertile environment for some bacteria and algae. The accumulation of pollutants in organisms and the mechanisms by which they cause harm would be altered if small and medium enterprises(nano) plastics and contaminants coexisted. The application of 50 Angstroms PS nanoplastics, respectively, to flea populations increased the toxicity of phenanthrene's chemotherapeutic effects (Zhang et al. 2021). They speculate that nanoplastics. These may slow the breakdown of the compound or its by-products, potentially resulting in to higher concentrations of the pollutant in fleas. Further, the endocrine system of adult *Oryzias latipe* was found to be disrupted by PS micro(nano)plastics and contaminants (Guo et al. 2021). Polystyrene (PS) micro(nano)plastics, when broken down, can release styrene molecules and other polymer monomers, which have the potential for disrupting hormonal balance and have negative effects like decreased reproductive rate. Researchers used very high doses to simulate harmful effects in their experiments.

Extremely high concentrations of microplastics cannot occur in nature. If there are only a few micro(nano) plastics floating around, there should not be too many problems. Micro(nano)plastics have the potential to affect global biodiversity in their role as a biological vector. Due to biofilm formation via adsorption of living things, micro(nano)plastic particles are rich in organic matter and nutrients, serving as the active center for photosynthesis, break down, and regrowth of organic matter and nutrients (Conley et al. 2019). A change in the structure of the microbial community may result from the use of micro(nano) plastics to aggregate various microbial species (Bergmann et al. 2019). The ecological repercussions could include the introduction of introduced species, the increasing number of pathogenic bacteria, the overabundance of resistance genes, and the emergence of new species mediated by microbes. Even more concerning is the possibility that micro(nano) plastics facilitate the dissemination of toxic algae (Wang et al. 2020). Once these species have successfully invaded a suitable area, they will alter biodiversity by damaging marine life, reducing the biological environment's capacity to serve, and reducing the bulk water's physical and chemical quality.

Effects on the Earth's Surface Biota

MPs are a problem, because they show how plasticized our lives have become and could have negative effects on the land ecosystem (Wang et al. 2019b). While MPs have been the subject of extensive study in marine environments, researchers have only recently begun to focus on MPs in terrestrial ecosystems (He et al. 2013). Terrestrial ecosystems are more receptive to MPs' interactions with biota, which could have consequences for the environment's

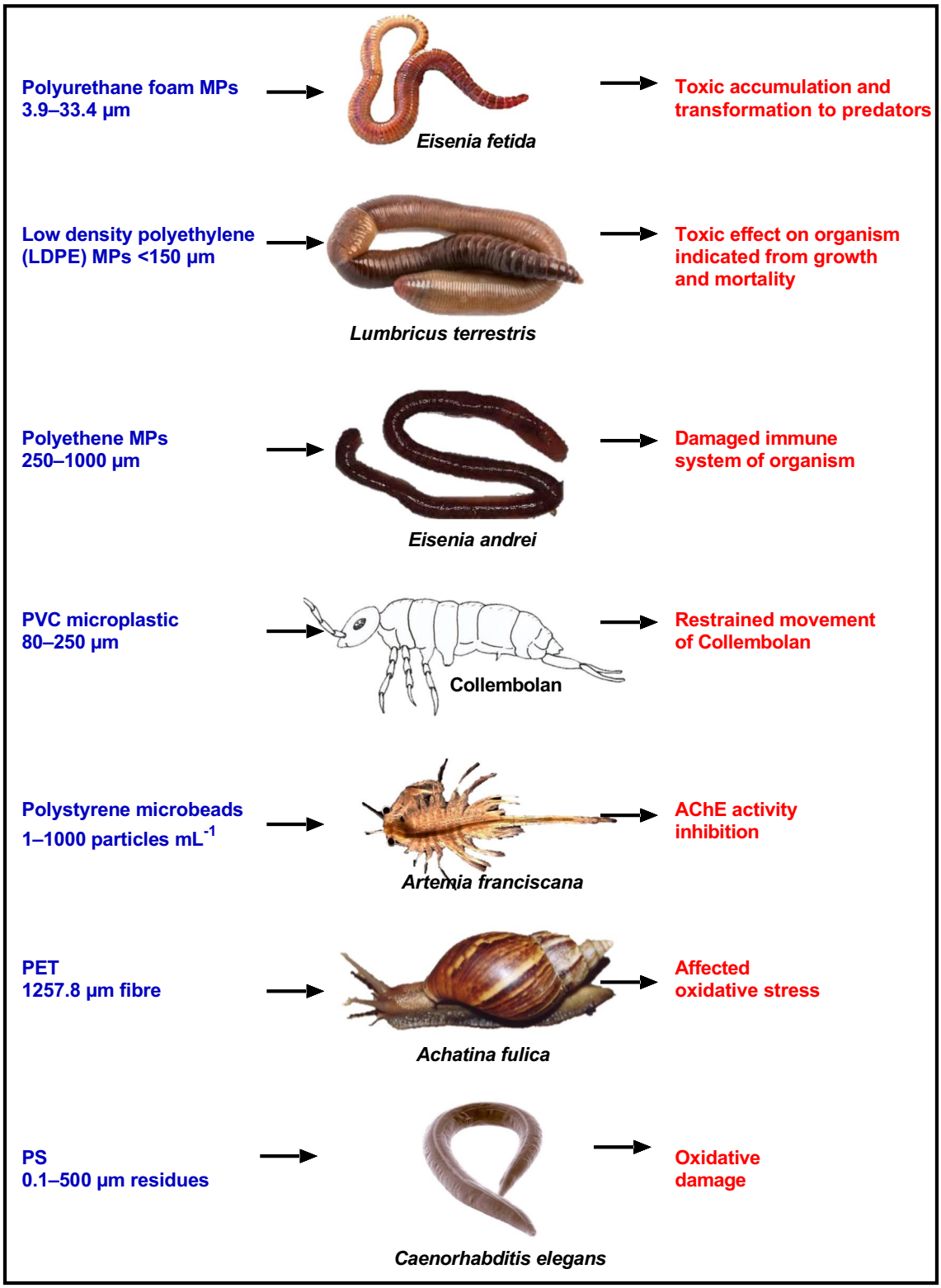

Fig. 5 Effects of MPs on different organisms in the soil (based on data from: Gaylor et al. 2013; Lwanga et al. 2017; Rodríguez-Seijo et al. 2019; Huerta et al. 2016; Song et al. 2017)

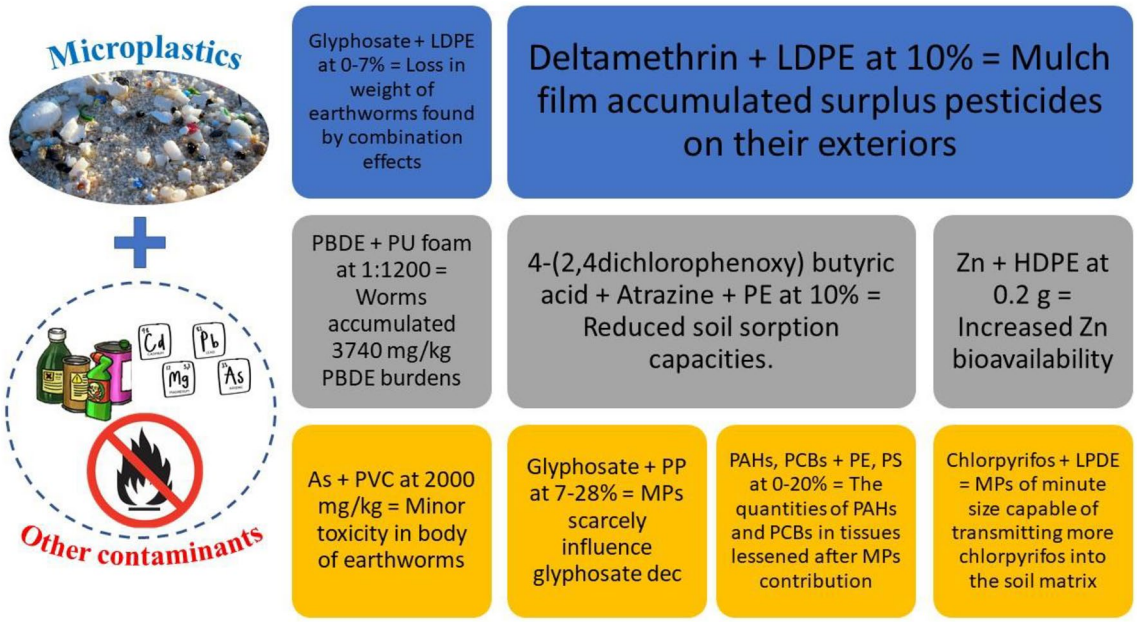

Fig. 6 The combined effects of MPs in the soil along with other contaminants (data collected from: Yang et al. 2018, 2019a, 2019b, 2019c; Ramos et al. 2015; Wang et al. 2019a,b; Huffer et al. 2019; Hodson et al. 2017)

geology and biophysics. Insights into MPs as a stressor of worldwide changes in terrestrial systems, particularly soil and air environments (Fig. 6), are presented here. MPs are present in soils everywhere, especially those used for farming (Kumar et al. 2021; Li et al. 2020; Velasco et al. 2020). They can be introduced to soil in a number of different ways (Velasco et al. 2020; Yang et al. 2019a, b), including through irrigation, sewage sludge, littering, and atmospheric deposition. Soil biota and soil characteristics, among others, play a role in controlling MP's vertical and horizontal mobility within the soil. Soil aggregates are transformed when MPs are incorporated into them (Guo et al. 2020). Due to the lack of light and oxygen in soils, MPs have the potential to live for decades. Thus, MPs may interact with soil fauna by changing the biophysical environment in which they live, thereby impacting the health and productivity of the soil. Once they build up in the soil, they can be absorbed by plants and passed on through the food web (Aransiola et al. 2023a). For instance, seeds exposed to MPs [biodegradable polylactic acid (PLA), conventional high-density polyethylene (HDPE), and MPs clothing fibers] show reduced germination and shorter shoots (Boots et al. 2019). The growth and survival of *Lumbricus terrestris* (Oligochaeta, Lumbricidae) were found to decrease when the animals were exposed to MPs (PE, 150 m) at varying concentrations (Huerta Lwanga et al. 2016). Microbial activity and water-stable aggregates were found to have a functional relationship that was altered by the presence of MPs (de Souza Machado et al. 2019). Changes in plant biomass, elemental tissue composition, root traits, and soil microbial activities were among the effects of MPs on soil health and the performance of spring onion (*Allium fistulosum*) that were identified by Machado et al. (de Souza Machado et al. 2019). Earthworms (*Lumbricus terrestris*) and other soil microorganisms can easily ingest MPs, which then accumulate in their casts through the intestine. Since burrows are the foundation of many food chains, their destruction could have far-reaching ecological consequences (Huerta Lwanga et al. 2016).

Comparable research by Liu et al. found that as MP concentrations increased, populations of soil worms and microarthropods declined (Liu et al. 2022). Microarthropod and nematode communities were profoundly altered by the introduction of polyethylene fragments in the wild.

The widespread use of face masks during the COVID-19 emergency is further evidence of environmental disorder in both the terrestrial and aquatic worlds and that the global pandemic has not reduced the threat of ecological plastic contamination (Tsiota et al. 2018; Zhuo et al. 2021).

To control COVID-19, surgical face masks have been used, and the results are clearly visible in the food chain and, eventually, in the form of chronic health problems for humans and the environment (Ardestani 2022). However, they cautioned that more research is needed to fully understand its effects on human health.

MPs have been spotted in both indoor and outdoor settings, suggesting their presence throughout the

atmosphere. Due to a lack of physical analysis and standardized sampling and identification methods, many unanswered questions remain about the occurrence, fate, transport, and effect of atmospheric MPs (Zhang et al. 2020). Dris et al. (2018) conducted the first study to look at MPs fibers in both indoor and outdoor air. Indoor concentrations were found to be anywhere from 1.0 to 60.0 fibers/m3, while outdoor concentrations were found to be a much more manageable 0.3 to 1.5 fibers/m3. Atmospheric MPs were spotted by Allen et al. in the Pristine Mountain watershed. There were 249 fragments, 73 films, and 44 fibers per square meter of catchment per day, as determined by their analysis of samples collected over the course of 5 months to represent atmospheric wet and dry deposition (Dris et al. 2018).

Also, the range of atmospheric fallout was highlighted by Dris et al. (2018) as being between 2 and 355 particles/m^2/day. Even within the same setting, the concentration of MPs in the air may vary significantly. High concentrations of MP particles were also found in rural areas of the Hamburg Metropolitan area in Germany. The movement and accumulation of MPs are influenced by a number of environmental factors. Particles containing MPs are transported to the ocean's surface air and even to distant sites as a result of these phenomena.

Impact on the Aquatic Ecosystem

Hydrodynamic mechanisms and the transmission by wind and ocean currents make MPs common in the marine environment, raising scientific concern at an exponential rate in recent decades. Seventy percent of marine plastic garbage ends up in sediments, 15% floats along shorelines, and the rest is dispersed over the ocean's surface. Fish, mussels, zooplankton, seabirds, and other marine species may unwittingly consume MPs due to their small size (Yang et al. 2019a, b).

Researchers have studied the impact of MPs after discovering them in anything from giant animals to tiny mollusks. Three species of benthic-feeding fish in Sydney Harbour, Australia, have been documented eating trash (Lei et al. 2017). They found that consumption of debris at the time of testing varied between 21 and 64% across the three species studied, and that the number of items consumed varied from 0.2 to 4.6 per fish among the species studied, with 53% of the debris being MPs. Inflammation and lipid build up in the digestive tract of Danio rerio were shown to be triggered by polystyrene (PS) MPs of both 5 m and 70 nm in size, as reported by Lie et al. (2017).

MPs' toxicity might vary with their size. *Caenorhabditis elegans* exposed to polystyrene particles of intermediate size (i.e., 1.0 m) exhibited the greatest toxicity on surviving, developing, and motor-related neurons (Lei et al. 2017). It has been suggested that MPs act as a transport mechanism for toxic chemicals, including dichlorodiphenyltrichloroethane (DDT) and hexachlorobenzene, which then accumulate in the body of the consuming organism (Kumar et al. 2017). *Calanus helgolandicus*, a crucial trophic relationship among primary producers and upper-trophic marine animals, may have its eating capability altered by this (Kumar et al. 2017).

Some aquatic animals are able to readily consume MPs and their smaller components, NPs, as shown in the research. Symptoms of poisoning include nausea, vomiting, abdominal pain, and diarrhea. The gastrointestinal toxicity of particles with a size of 70 nm polyamide, PE, PP, PVC, and PS was shown in zebrafish and nematodes, causing villi breaking as well as splitting of enterocytes (Lei et al. 2017). In addition, research has shown that ingestion of PE microplastics within the blue mussel *Mytilus edulis* L. causes noteworthy histological changes and an intense inflammatory reaction, and that 0.5 m PS microplastics induced dysbiosis is microbiota, and inflammation in the gastrointestinal tract of adult zebrafish (Pathak and Navneet 2017). The liver of the cichlid fish Eriocheir sinensis showed signs of oxidative stress after being fed MPs (Yu et al. 2018). According to research conducted by Yu et al. (2018), PS MPs induced oxidative stress in zebrafish, altering metabolic pathways and so disrupting lipid and energy metabolism.

Another major issue of worry among scientists is plastic's toxicity to aquatic habitats due to the presence of heavy metals. The many heavy metals employed in the production of plastic eventually find their way into the environment, where they wreak havoc. The plastics industry has historically relied on metals, such as cadmium, zinc, and lead as heat stabilizers and slide agents. Up to 3% of a polymer's mass may come from these metals, which have been shown to have negative consequences (Pathak and Navneet 2017). In addition, analysis of 144 samples demonstrated that PVC may be among of many primary perpetrators of heavy metal pollution from plastic debris in our seas.

Microplastics as a Public Health Concern

Massive data sets have generated significant evidence for MPs' nature, chemical structure, reactivity, and structures thanks to recent improvements in techniques that permitted the characterization of MPs in food, water, and air (Zhao et al. 2021; Auta et al. 2022). Researchers have recently been concerned about the effects of MPs on the health of humans and small vertebrates after seeing several epidemic occurrences in which MPs were identified in the organisms of those affected. A few studies have previously laid the groundwork and shown that MPs are readily excessive to many (small to big) regions of the body, but that this seldom causes serious difficulties. Numerous studies have already categorized a wide range of illnesses thought to

be caused by MPs, but there is a pressing need for more research into this area. Yan et al. found lately that persons with inflammatory bowel disease (IBD) like Crohn's and ulcerative colitis often have microplastics in their feces. Inflammation in the intestines, changes in the gut flora, and other problems have been linked to MPs in animal studies (Yang et al. 2019a, b, c). It is unclear if persons with IBD acquire more fecal microplastics as a result of their condition or whether this exposure causes or contributes to IBD, as stated by the researchers. New evidence reveals that even the smallest particles of microplastic may penetrate cell membranes and enter circulation, challenging the long-held belief that microplastics are harmless and exit the body through the digestive system. Laboratory animals and human cells exposed to microplastics show signs of cell death, inflammation, and metabolic disturbances.

Polyethylene terephthalate, polyethylene, and polymers of styrene are the most often detected plastics in human blood samples, whereas polypropylene numbers too low to be quantified (Yang et al. 2019b). In a groundbreaking human biomonitoring investigation, researchers were able to show that plastic particles may be absorbed by human bodies and pose health risks. We think these particles lead to some other, as-yet-unknown, bodily dysfunction.

In a similar vein, research by Yang et al. revealed that MPs and NPs concentrations would cause some degree of stress in cells (Yang et al. 2018). In addition, microplastics and nanoplastics may enter the human body through the digestive tract, the lungs, and the skin, all of which can have detrimental effects on human well-being (Yang et al. 2019a).

However, particle size and surface chemistry determine the exposure route of human cells (Wu et al. 2021). Researchers have been testing the mammalian model to anticipate MPs' toxicity and link it to people, since the explosive impact of MPs on human health is not established to far. After 28 days of exposure, PS MPs accumulated in the kidney, intestines, and liver of mice, causing inflammation in the liver and a lipid metabolic issue (Deng et al. 2017). According to in vitro research on PS NPs (Xia et al. 2008), cationic polystyrene nanoparticles induce apoptosis in macrophage (RAW 264.7) and epithelial (BEAS-2B) cells through reactive oxygen species (ROS) production and endoplasmic reticulum (ER) stress in mice. PS nanoplastics were shown to decrease the viability of human gastric cancer cells and to stimulate the production of inflammatory genes including IL-6 and IL-8, according to one study (Xia et al. 2008).

Furthermore, transcriptome studies showed that extended exposure to MPs affected the transcription levels of gut-related genes, as well as other critical metabolic pathways and life activities (Wu et al. 2020). Hwang and colleagues found that PS and PP particles might act as immunological stimulants, prompting immune cells to produce cytokines and so increasing the risk of disease (Wang et al. 2019a, b, 2020). Overexposure to MPs has been shown to have a variety of impacts on humans, as shown in Table 1.

The majority of studies that have shown an association between MPs and a particular illness have either a weak methodological foundation (McCormick et al. 2014) or insufficient evidence to support their findings. The negative impact of MPs' exposure for the living system has been

Table 1 Some of the potentially toxic effects of MPs and NPs on human health

Toxicity effect	Plastics	Size of plastics	Results	References
Oxidative stress	PVC PMMA (poly methyl methacrylate)	120 nm 140 nm	Increases reactive oxygen species (ROS), and reduce cell Feasibility	Mahadevan and Valiyaveettil (2021)
	Cationic PS NPs	60 nm	Increases reactive oxygen species (ROS) generation and endoplasmic reticulum (ER) stress	Chiu et al. (2015)
Gastrointestinal effect	PS NPs	50 nm and 200 nm	Alter intestinal ion transport	Campanale et al. (2020)
	PS MPs	0.5 µm and 5 µm	Increased metabolic disorder risk in the offspring	Guo et al. (2021)
	PS MPs	0.5 and 50 µm	Induce mouse hepatic lipid disorder	Yang et al. (2019a, b, c)
	PS MPs	5 µm	Reduces intestinal mucus secretion and induce gut micro- biota dysbiosis	Yang et al. (2019a, b, c)
Neurotoxicity	PS MPs	5 and 20 µm	Increase in AChE activity in the liver, and may lead to the reduction in cholinergic neurotransmission efficiency	Deng et al. (2017)
	PS NPs	38.92 nm	Decreased locomotor activity	Ya et al. (2021)

documented by a number of biological tests, at least in vitro (Mahadevan and Valiyaveettil 2021). Again, the in vitro research' lack of transferability to human complexity with multi-organelle activity is due to their reliance on a single cell type; nearly invariably with an immortal cell line. However, there is some evidence that MPs may be eliminated through the in vivo-relevant cellular excretory route after being taken up by phagocytic cells (Yang et al. 2019a, b, c). The blood vessel dilation, which is infiltration, along with congested as a result of MPs' structural construction is a different one expected consequence of MPs demonstrated in in vivo models of small animal (Guo et al. 2021), but this has not yet been reported in humans.

The degree of occupational exposure to airborne MPs is correlated with the pathology reported among MP exposed human employees (Giorgetti et al. 2020; Wang et al. 2021, 2022). This is in contrast to MPs consumed by food and drink. Prolonged exposure to MPs has been linked to an overall decline in quality of life among workers due to enhanced chronic bronchial constriction including asthma-like clinical characteristics (Fan et al. 2020). It is unclear how chronic airway irritants, such as MPs, disrupt the respiratory tract's immune tolerance; however, this process is likely dose-dependent and influenced by a wide range of human physiological factors, including genetics (Ardestani 2022).

In conclusion, well-designed clinical-epidemiological research are needed to clarify the pathogenesis, spectrum of disease, and long-term damage owing to continuous exposure to MP. Yet, we may partly presume that MPs do have the ability to worsen human physiology and homeostasis according to information relating to small mammals, invertebrates, even in vitro human cell toxicity; yet, the evidence to state this assertion is extremely poor; hence, further research is required in this issue.

Conclusion

The emergence and degradation of micro- and nanoplastics in the environment is a complicated and diverse problem with worldwide consequences. These microscopic plastic particles are ubiquitous in ecosystems, posing risks to aquatic and terrestrial species. Understanding their origins and processes of deterioration is critical for creating effective mitigation solutions. Micro- and nanoplastics are being addressed by the development of more sustainable plastic materials, improved waste management methods, and the decrease of plastic pollution at its source. Furthermore, research into the ecological and physiological implications of these particles is underway to better inform laws and regulations aimed at protecting the environment and human well-being from the ubiquitous presence of micro- and nanoplastics.

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