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REVIEW

Comprehensive investigation on microplastics from source to sink

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Abstract

This paper provides a comprehensive review on microplastic from source to sink and reviews the current state of knowledge of the topic by focusing on the articles published within the last five years on identification, quantification, analyses, and effects of microplastics on soil and aqueous environments. Microplastics are materials formed either by the degradation of the plastic into smaller micro sized particles or obtained directly in daily products such as cosmetics, toothpastes, domestic cleaning products, etc. Hence, the origin of microplastics is either a primary or secondary microplastic source. The lack of information and research conducted on microplastics in soil compared to water influenced many disparities. These include variations in defining microplastics to lack of conclusive methodologies in analysis of microplastics in soil which therefore lead to gaps in identification of plastic source and comprehension of plastic pollution in soil. The effect of microplastics on different aquatic vertebrates, mammals, and humans is studied and, in most cases, various negative effects were observed in the organism's physiology. In addition to innovative control methods, there is a growing focus on exploring bioplastics as a potential substitute for traditional plastics. Numerous studies suggest that the environmental impact is more manageable with the production and use of bioplastics. Nonetheless, additional research is needed to confirm the viability of bioplastics as a potential solution.

Graphical abstract

Keywords Microplastics · Soil and marine · Analytical methodology · Bioplastics · Degradation

Abbreviations

ATR-FTIR	Attenuated total reference FTIP
BP	Bioplastic
BPA	Bisphenol A
CAT	Catalase
cTC	Conventional thermophilic composting
DEHA	bis(2-ethylhexyl) adipate
DFM	Disposable facemasks
DNA	Deoxyribose nucleic acid
DPHP	bis(2-propylheptyl) phthalate
FCM	Food contact material
FE-SEM	Field emission scanning electron microscope

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FPA-FTIR	Focal plane array FTIR
FTIR	Fourier transform infrared spectroscopy
GHG	Greenhouse gas
H_2O_2	Hydrogen peroxide
H_2SO_4	Sulfuric acid
HA	Humic acid
HCl	Hydrochloric acid
HGMS	High gradient magnetic separation
HNO_3	Nitric acid
hTC	Hyperthermophilic composting
IR	Infrared
KOH	Potassium hydroxide
LCA	Life cycle assessment
magPOM-SILP	Magnetic polyoxometalate-supported ionic liquid phase (magPOM-SILP)
MP	Microplastics
MPSS	Munich plastic sediment separator
MT	Metric tons
NaCl	Sodium chloride
NaOH	Sodium hydroxide
NIR	Near infrared
NIVA	Norwegian institute for water research
NO_2	Nitrogen dioxide
O_2	Oxygen
O_3	Ozone
OCP	Organochlorine pesticides
PA	Polyamide
PAH	Polycyclic aromatic hydrocarbons
PAM	Polyacrylamide
PB	Polybutylene
PBDE	Polybrominated diphenyls ethers
PBMC	Preripheral blood mononuclear cells
PBS	Polybutylene succinate
PC	Polycarbonate
PCB	Polychlorinated biphenyls
PE	Polyethylene
PET	Polyethylene terepthalate
PHA	Polyhydroxyalkanoates
PLA	Polylactic acid
PMP	Primary microplastics
PP	Polypropylene
PS	Polystyrene
PTT	Polytrimethylene terephthalate
PVC	Polyvinylchloride
Pyr–GC–MS	Pyrolysis–gas chromatography–mass spectrometry
ROS	Reactive oxygen species
SMP	Secondary microplastics
SO_2	Sulfur dioxide
SOD	Superoxide dismutase
SOM	Soil organic matter
TBBPA	Tetrabromobisphenol A
TED-GC–MS	Thermal extraction desorption-gas chromatography–mass spectrometry
TGA–MS	Thermogravimetric analysis–mass spectrometry
UV	Ultraviolet
VOC	Volatile organic compounds
WWTP	Wastewater treatment plant

Introduction

There has been an exponential growth on the production of plastics since 1950 (Geneva Environment Network 2023) and the world is currently producing more than 380 million tonnes of plastic waste annually out of which majority is single-use plastic waste (242 million tonnes) which infiltrates the environment (Abdulraheem 2018). The high production rate, chemically inert nature, and inefficient modes of disposal are some of the factors that make plastics difficult to be eliminated from the ecological system (Espinosa et al. 2016).

Microplastics (MPs), defined as insoluble plastic fragments measuring less than 5 mm in length, have emerged as a topic of great interest since their introduction by Thompson et al. in 2004. These tiny particles are pervasive in aquatic systems, soil, and the air, with water being the predominant environmental medium studied (79%), followed by soil (39%) and air/dust (7%), which have gained attention more recently according to a scoping review by Casillas et al. (2023).

The detrimental impact of MPs can be attributed to two main factors. First, they have the capacity to adsorb environmental pollutants such as polychlorinated biphenyls (PCBs) and organochlorine pesticides (OCP) (Luo et al. 2021a, b). Second, plastics themselves often contain additives such as plasticizers (phthalates), flame retardants (polybrominated diphenyl ethers (PBDE)), coloring agents, and more. These additives are ingested by organisms, allowing the pollutants to enter the food chain and cause reproductive and physiological damage (Kilponen 2016). While some studies suggest that MP bioaccumulation occurs at every trophic level, current field data do not support MP biomagnification in marine food webs. Confounding factors identified in certain laboratory studies indicate that trophic transfer may be confused by the use of exaggerated exposure circumstances (Miller et al. 2020). To address the pressing environmental issues associated with MPs and plastic pollution, exploring bio-based polymer alternatives has emerged as a promising solution since they would provide similar functionality and applications as conventional plastics while being derived from renewable resources and potentially exhibiting better biodegradability or compostability under certain conditions (Ostle et al. 2019).

Biobased polymers are derived from renewable resources, such as corn starch, sugarcane, or cellulose, and undergo processing to form small plasticchen particles (Ostle et al. 2019). These biomaterials provide several advantages in the pursuit of sustainable production and consumption. By utilizing renewable resources, they optimize resource efficiency reducing reliance on fossil fuels. The manufacturing process follows the principle of the cascade, where biomass is first used to manufacture products, and subsequently, its waste can be utilized for energy generation. This approach not only decreases carbon footprint and greenhouse gas (GHG) emissions but also promotes a circular economy by minimizing waste. On breaking down, like their conventional counterparts, these bio-based polymers give rise to bio-based MPs that share similar characteristics and potential environmental impacts as conventional microplastics but differ in their origin and composition (Ostle et al. 2019). However, it is important to note that the environmental fate, behavior, and potential risks associated with bio-based microplastics are still subjects of ongoing research and debate (Koelmans 2016). The actual biodegradability and environmental impact of bio-based microplastics can vary depending on factors such as polymer composition, size, exposure conditions, and the availability of suitable disposal infrastructure (Koelmans 2016).

Research is being conducted to better understand the behavior, distribution, and potential ecological effects of bio-based microplastics, as well as their role in the overall plastic pollution issue. It is essential to ensure that the production and use of bio-based microplastics are guided by sustainable practices, including proper waste management and disposal, to minimize any potential negative environmental impacts (Koelmans 2016).

Recognizing the urgency of the plastic pollution problem, governments and organizations have invested in research and innovation related to circular economy principles and bioplastics, including bio-based microplastics. The European Policy, for instance, has taken significant steps towards reducing single-use petrochemical plastics and funding initiatives focused on sorting bioplastics from plastic waste (Matthews et al. 2021). Furthermore, the expansion and development of bio-based products, including bio-based microplastics, offer a potential solution to the concerns surrounding MPs, provided there is persistent widespread public awareness and demand for sustainable development.

The present work offers a comprehensive perspective on the multifaced issue of MP pollution, expertly addressing its effects on the environment, innovative quantification methods, and promising control measures (Su 2023). While previous review articles have indeed explored these subjects, this work stands out for its in-depth analysis of the latest research and its pragmatic approach to tackling this pressing problem. In sum, this comprehensive review distinguishes itself through its critical evaluation of cutting-edge research, ultimately contributing to a more profound understanding of MP pollution and advancing sustainable solutions.

Literature retrieval

This literature review was performed in 2022 using Google Scholar, PubMed, and Scopus as the academic databases. The literature on MPs and bioplastics was retrieved from articles published mainly in the last five years. Specific keywords were used in reputed journals to filter out the required data for this review. Most commonly keywords that were identified are “microplastics”, “soil”, “aqueous environment”, and “bioplastics”. Any duplicates were removed from the results after which the initial screening was conducted by going through the abstracts and removing any other irrelevant articles. The full texts were then assessed based on this study’s objectives.

Figure 1 shows the number of peer reviewed articles that have been published from 2004 to 2023 on MP. The data has been extracted from a Web of Science database by using ‘microplastics’ as the keyword. Before 2011, the number of published papers was less than 40 articles and are hence not clearly visible on the graph. However, there has been a rapid growth in publication rate on MP after 2018. In 2018 there were 1052 papers that have been published on MP and in 2022 the number has reached 7430 papers.

Sources of MPs

Primary microplastics (PMPs) are manufactured in microscopic size for household or industrial uses and enter the environment in the form of small particles (Boucher and Friot 2017). In contrast, anthropogenic activities result in secondary microplastics (SMPs) originating from homes as domestic waste or industrial waste and undergoing fragmentation due to mechanical or photooxidative pathways (Jiang 2018).

Water system

Examples of primary household sources are Polyethylene (PE), Polypropylene (PP), and Polystyrene (PS) based particles present in cleaning and cosmetic products such as moisturizers, shampoos, cosmetics, shaving products, etc. Despite numerous sites with wastewater treatment plants (WWTPs) to filter out MPs, in many places without applying such amenities the MPs pass through the drainage and enter aquatic systems via sewage discharge (Jiang 2018). Another source of domestic PMPs is washing clothes made of PE, acrylics, polyamide (PA), and other synthetic materials at

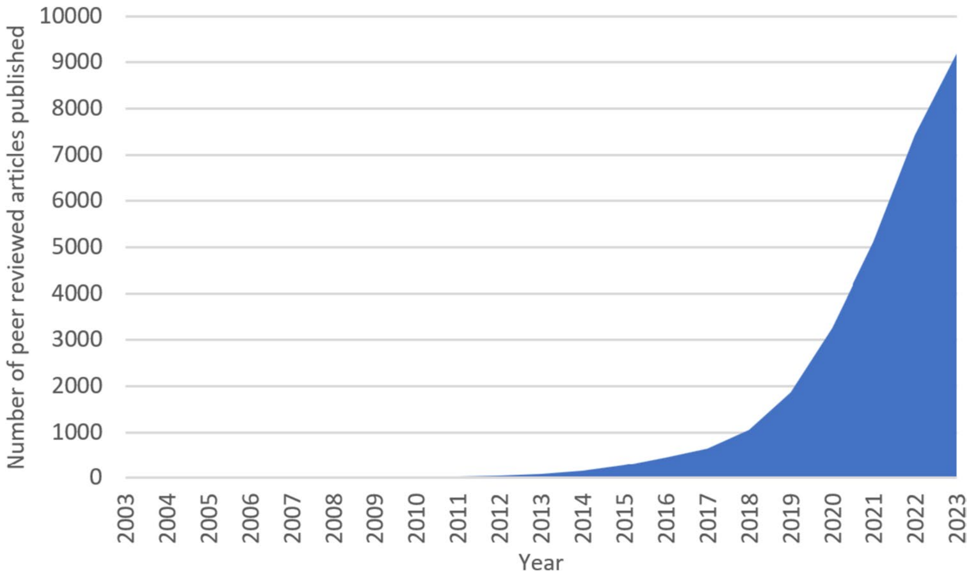

Fig. 1 Number of peer reviewed papers that have been published from 2003 to 2023

home that results in approximately 18,000,000 microfibers for an average of 6 kg load (Galvao et al. 2020). Despite the removal of chemicals in WWTPs, MPs are released into water sources or recycled back to households through tap water as the source of tap water in most countries is through either surface or ground water (Isaac and Kandasubramanian 2021). Industrial primary sources such as feedstock used in the preparation of plastics, plastic resin powders/pellets, and scrubbers that are used to blast clean surfaces, could accidentally being discharged due to spills or disposed into aquatic systems (Jiang 2018). The most important method of land based SMPs source in aquatic bodies is the loss from inappropriate managing of landfill sites and during waste collection. Other SMPs sources entering the aquatic systems are through agricultural activities, natural phenomena (hurricanes, tsunamis, strong sea waves, etc.), inappropriate management of landfill sites, etc. (Westphalen and Abdelrasoul 2017). The origin of MPs and how they enter the food chain via aquatic bodies has been shown in Fig. 2.

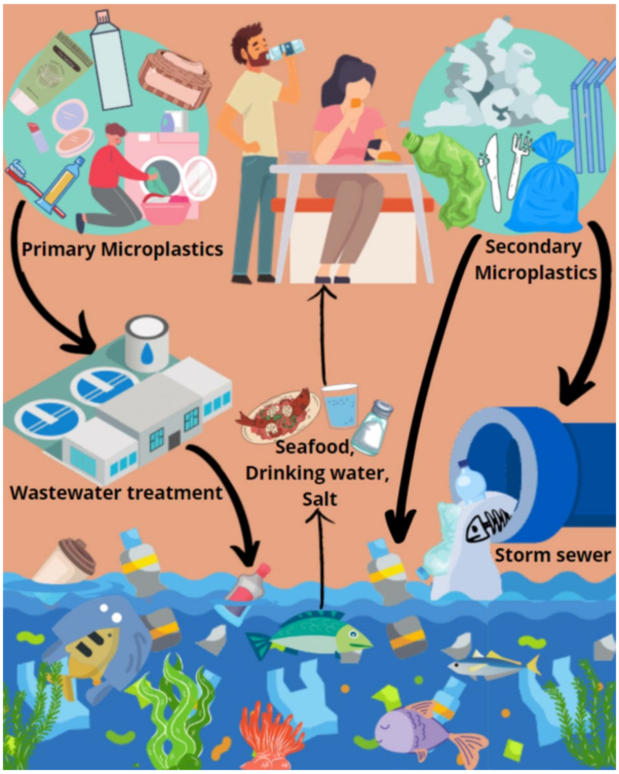

Fig. 2 Microplastic origin in the aquatic environment and how it enters the food chain

Land-based activities, such as littering, landfills, waste dumping, WWTPs, urban runoffs, etc., account for 80% of the source of MPs, whereas water-based activities such as fishing, aquaculture, and water transportation methods account for the remaining portion of MPs in aquatic systems (Kilponen 2016). Hence, a major source of MPs in the aquatic environment is through land-based activities and it is essential to understand the relation between the MPs in soil and water systems.

The year 2019 has been noted as the start of the worldwide COVID-19 pandemic which had a major impact on the release of MP in aquatic systems (Chen et al. 2020a, b, c, d). It was found that the pandemic has caused an increase in the use of plastic products due to mandatory regulations on the use of disposable face masks (DFM) which are made of polymeric materials such as PP, PU, polyacrylonitrile, PS, polycarbonate, PE, or polyester (Aragaw 2020). A study conducted by Peng et al. 2021 has found that the pandemic has resulted in 8 million tonnes of pandemic associated waste, out of which 25,000 tonnes have entered the aquatic systems.

Jiang et al., (2022) have studied the release behavior of microfibers from various kinds of DFM. It was found that each type of DFM has released more than 47 fibers per day under the experimental conditions. According to the study conducted by Zambrano-Monserrate et al. in 2020, the COVID-19 pandemic has resulted in indirect consequences for MP entry into aquatic systems. This includes the temporary halt of recycling programs in certain U.S. cities due to concerns about the virus spreading among personnel at waste recycling facilities. In a few European countries, sustainable waste management strategies have been completely restricted because of the same fear which has only resulted in a larger amount of plastic entering aquatic systems due to insufficient management. Jiang et al. 2022 also suggested the use of reusable face masks, biodegradable face masks, reusing DFM, and adopting methods of pyrolysis to reduce the impact of DFM on the environment.

Soil system

The PMPs could also be considered as by-products of larger plastic products such as tire dust from vehicles, discharges from paints, artificial turfs, etc. due to abrasions (Boucher and Friot 2017). Rise in urbanization and economic development, along with improper waste management procedures and lack of awareness of disposal by individuals, leads to littering and therefore MPs accumulation in soil. Most of the generated wastes are left out in open environments or in unassigned dumpsites (Blasing and Amelung 2018). It was noted that 4.97 billion tonnes of plastic garbage have been disposed in open environments and landfills between 1950 and 2015 (Geyer et al. 2017). Yang et al., (2021) noted that obtaining the exact quantity of microplastic debris reaching the soil via littering is inconclusive and hard to quantify since MPs could be introduced to the soil from other locations via natural pathways such as wind, runoff, and flooding.

Another SMP contributor is Plastic film mulch which mainly comprises of PE and other polymers such as Polybutylene (PB), Polyvinylchloride (PVC), PS and PP that often are used to improve crop productivity, Film mulch can conserve soil moisture, modify soil temperature, and suppress weed growth (Huang et al. 2020). As food demand increases globally, the need for mulching would also rise and cause a greater threat to soil ecosystems, and subsequently higher trophic levels in the food chain (Huang et al. 2020). They might be carriers of other contaminants such as heavy metals, pathogens and organic pollutants. Furthermore, it is hard to eliminate mulch from agricultural sites due to possessing highly labour intensive and time-consuming during removal (Yang et al. 2021). The issue arises when mulch is exposed to the UV radiation where the large pieces of mulch become fragile and photodegrade (Yang et al. 2021). Mechanical abrasion due to cultivation and exposure to weather conditions such as freezing or thawing, results in further degradation and consequently leading to preservation in soil. (Piehl et al. 2018).

Sewage sludge is another major source of MPs in soil. It mostly consists of microbeads from cosmetic and cleaning products, fibers from textile fabrics, and plastic waste released from plastic processing plants. This occurs due to improper disposal of MPs in the drainage system. The MPs eventually end up in WWTPs where some would enter the surrounding environment as part of the discharged effluent water while others are collected in the sewage sludge (solid waste) (Franco et al. 2023). It must be noted that the WWTPs are unable to specifically remove MPs from wastewater, but they are mainly captured once the wastewater passes through various treatment stages (Franco et al. 2023). It was determined from the Norwegian Institute for Water Research (NIVA) that roughly 181,679,012 MPs were transferred to sludge from wastewater each day (Nizzetto et al. 2016). According to a study conducted in 2023 by the Statistical Office of the European Communities (ESTAT) in Spain, from the cumulative sludge generated in 2018 (1210.4 thousand tonnes), about 87% (1052.7 thousand tonnes) were utilized as biosolids used in agriculture whereas the remaining was disposed to the landfills. Approximately 40% of sewage sludge is utilized in agriculture alone in the European Union, as Spain acknowledged for 65% recycling rate of its sewage sludge as fertilizer (Berg et al. 2020). The study conducted by Berg et al. (2020) showed that among 16 investigated agricultural fields in Spain, an average of 5190 microplastic kg^{-1} were obtained in fields with the application of sludge and nearly 2030 microplastics kg^{-1} in fields without the application of sludge. It is essential to note that the occurrence of MPs is specific for each region as well as for polymer type. These factors may indicate the source, sink and pathways of a particular MP which could render more specialized measures to reduce MPs generation.

Variations between soil and aquatic microplastics

By comparing the different characteristics of MPs in aquatic environment and soil, it would help indicate certain features of the sources and distribution of MPs (Yang et al. 2021). However, similarities are evident as both MPs sinks (marine and soil) contain the same chemical and physical morphologies (Eerkes-Medano et al. 2015). Furthermore, studies have revealed that polyethylene (PE) and polypropylene (PP) are the predominant polymers in both matrices, with fiber and fragment shapes being the most commonly observed in both environments. (Akdogan and Guven, 2019).

Similarly, the colours dominating the aquatic and soil biota were observed to be transparent and white microplastics. Feng et al., (2020) reported that the transparent MPs were the predominant type (46.3%) found in cultivated soil in the entire Yunnan province in China. One possibility for why such consistencies might exist, apart from the bleaching of coloured plastics, is that the MPs would enter both sinks through similar sources and pathways (Li et al. 2020). However, Yet, Yang et al., (2021) contradicted this observation and highlighted the lack of sufficient studies which could show the interconnections of the pathways. Following these claims, Du et al., (2020) have corroborated that apart from the discrepancies of analytical methodologies applied in soil and marine environments, there are different methods used in sampling and filtering of both matrices. Water samples are collected in column either by using a plankton net or filtering large volume of sample using a filter mesh size of 100 or 330 µm. This is significantly different from the soil MPs sampling methods where filtering is not even employed (Akdogan and Guven 2019). Additionally, the equipment used for detection have limited efficiency of operation as some experiments have reported high amounts of detected MPs with less than 1 mm in both matrices (Yang et al. 2021). Furthermore, it was found that due to these variations and detection limits, most of the errors due to the collected data and evaluation process have been exacerbated (Dris et al. 2017).

Analytical methodology

To effectively investigate MPs in soil, a thorough methodology must be employed. This ranges from soil sample collection to particles isolation in the soil matrix and other adhering substances whilst preventing any damage or artificial fragmentation to the MP sample. However, contrary to the aquatic systems, only a few studies have been conducted on this field for soil and thus research is still at an early stage than it ought to be, given the importance and complexity of MPs in soil. Most of the methodologies for detection of MPs in soil are derived from procedures on MPs in aquatic environments. This causes lack of standardization and harmonization which could lead to discrepancies and create a bottleneck for future studies. Therefore, development of an accurate methodology with identification of the type of plastic and pollution source is vital for the management and environmental protection frameworks.

Sample collection

Soil is easily influenced by constant human intervention, history and usage of the land and accumulation zones where MPs could be deposited (from surface runoff or windborne particulate matter). Therefore, it is important to identify the accurate sampling area and depth of soil that is to be collected. (Rillig et al. 2017). There are two primary methods of soil sample collection, composite and single site sampling depending on the degree of scattered debris of MPs in soil (Moller et al. 2020). Composite sampling consists of collecting similar size samples from several discrete locations within an area which is then combined and homogenized to a single sample (Scheurer and Bigalke 2018). This method is recommended since the concentration of MPs in soil is never uniform due to various interferences (Moller et al. 2020). Single site sampling is generally used in areas where limited human activities are present (Yang et al. 2021). To determine the number of sampling points, Moller et al. (2020) have suggested to employ statistical power analysis. However, the exact ideal replicate and volume of sample are not yet conclusive as different area of sampling units were used in studies. This has significantly restricted the sample processing and created a bottleneck for subsequent analytical methods of MPs in soil.

It is imperative to define the sampling depth since the deposition of MPs is highly dependent on activities performed on the soil. Yang et al., (2021) indicated that the surface of soil should be selected if the study is conducted on undisturbed soil (0–30 cm soil sample). It is also suggested that if concentration of MPs at varying depths of the soil is to be found, then stratified sampling should be utilized. However, the downward transportation of MPs in undisturbed soil is yet to be investigated and thus some of the collected samples might be unreliable (Rillig et al. 2017). Moller et al. (2020) also recommended collecting a large volume of samples than the required amount for quantification, as it may be needed for subsequent purposes such as sample backup, determining moisture content and sample recovery analysis.

As a measure of quality control, guidelines introduced by the US Environmental Protection Agency have suggested taking control samples that are of a similar soil type as the main MP sample. These should be collected from a nearby vicinity unaffected by contaminants of concern (EPA 2020). This could assist in monitoring potential contamination that originated during sampling, quantifying MP's background levels and even provide a more comprehensive understanding on soil matrices which, otherwise, could have been unclear (Thomas et al. 2020).

Drying, sieving and purification

It has been observed that the collected samples have undergone natural air drying to reduce soil humidity for better analysis in the subsequent stages although few reviewed publications have oven dried the samples to accelerate the drying process (Yang et al. 2021). This is usually conducted at a temperature range of 40–70 °C, well below the thermal deformation temperature of most plastics. Yet, the prevailing drying methods and conditions analysis are contrasting. For instance, Berg et al. (2020) have dried the soil at 40 °C for 72 h whereas Liu et al., (2020) have opted at 70 °C for 24 h. However, the drying temperature of MPs samples comprising of PE and PA is kept below 50 °C as high temperatures would cause impairment (Hurley et al. 2018). It must be noted that temperatures above 40 °C may affect the polymers' physical and structural properties by glass transition, melting, or degradation (Hurley et al. 2018). Therefore, freeze drying was developed as another alternative drying option (Thomas et al. 2020). Freeze drying could effectively break soil aggregates and cell walls thereby aiding further sample preparation. Nevertheless, polymer brittleness would rise if the operating temperatures were below its glass transition temperature. Also, frost wedging may fragment the sample and cellular organic matter may be released. Furthermore, freeze drying is a slower process than air or oven drying process and often limited by the size of the freeze dryer (Thomas et al. 2020).

After drying stage, the soil sample is passed through a stainless-steel sieve (1–2 mm, 5 mm sieve size) to separate microplastics using various screening methods based on whether the soil is agglomerated or contains grass and other residues (Wang et al. 2019). A sieving cascade may alleviate the amount of required work, but the technique is yet to indicate how excessive sieving increases particle fragmentation, especially when the sample is an aged or freeze-dried fractions (Thomas et al. 2020).

The soil sample may consist of a large quantity of soil organic matter (SOM) which is a complex matrix at different levels of decomposition. In several conducted studies, organic matter was reported to hinder microplastic analysis. Therefore, the removal or reduction of such components remains a necessary issue during sample preparation (Thomas et al. 2020). Thus, the MP sample would need to undergo the purification process which is a vital stage for the subsequent analysis. The most common reagents used for this stage are HNO_3, NaOH, H_2SO_4 and H_2O_2. Nuelle et al., (2014) have claimed that although H_2O_2 is widely used, the efficiency remains inconclusive as it is a time-consuming process and H_2O_2 would bleach the organic matter rather than remove it. Hurley et al. (2018) suggested that an alternate solution would be using Fenton's reagent which, in the past, was used to extract microplastics from wastewater samples. This method is more effective than the latter one as it would remove any organic material from complex substrates at a shorter duration. Other methods including acidic reagents such as HNO_3 and HCL have not been deemed suitable as they may degrade and melt the microplastic (Scheurer and Bigalke 2018). Alkaline reagents such as KOH and NaOH have been considered effective, but they displayed relatively low removal efficiency as humus and alkali insoluble compounds in the soil sample (or other complex samples like sewage sludge) were still present after the reagent application (Blasing and Amelung 2018). Thus, a proper testing on such complex matrices is required to establish the type of removal for organic matter.

Extraction

The final stage is extracting the MPs from sediments or inorganic compounds (supernatant) that were not broken down throughout the previous stage. Density separation is the most widely used technique using extraction media such as sodium chloride, calcium chloride and zinc chloride, sodium iodide, distilled water and sodium heteropolytungstate widely used (Nakajima et al. 2019). The principle underlying this method is the difference of specific gravity of soil and plastics. Most MPs are less dense than the salt solutions and hence they float to the top, leaving the denser inorganic sediment to settle at the bottom. Therefore, the soil-to-solution ratios implemented during experiments depends on the sample size and technical setup and is decisive for MP recovery (Thomas et al. 2020). However, the ratio of soil-to-density solution varied immensely from 1:2 to 1:25 (Chen et al. 2020a, b, c, d). Hence, a more harmonized method needs to be addressed.

Amongst the extraction media, sodium chloride (NaCl, 1.2 g cm^{-3}) and distilled water (1.0 g cm^{-3}), albeit its ease of access and cost, are only able to separate low density polymers such as PE, PS and PP (Li et al. 2020). For NaCl, the Na^+ may further facilitate in the dispersion of soil aggregates resulting in higher extraction efficiency (Scheurer and Bigalke 2018). Alternate extracting media is used for synthetic polymers, such as PVC (1.1–1.6 g cm^{-3}) and PET (polyethylene terephthalate) (1.3–1.6 g cm^{-3}) which have higher densities than NaCl (Moller et al. 2020). However, the type of extraction media is solely dependent on the local demand of plastics as this will determine the composition of the soil sample. Scheurer and Bigalke (2018) contended that since PVC and PET make a relatively small contribution to the larger portion of other microplastics in the sample, a sodium chloride (NaCl) solution could be utilized. Nevertheless, this argument is solely based on the local demand of plastics which determines the composition of the soil sample and thereby the extracting media. Calcium chloride has been proposed as an optimum choice for the separation of denser particles owing to its environmental friendliness and low cost (Scheurer and Bigalke 2018). However, the presence of organic floccules was observed after the separation process as Ca^+ ion can bridge the organic molecules' negative charge (Scheurer and Bigalke 2018). These floccules get attached to the filter membrane which would then impede the MP's identification process (Yang et al. 2021).

Highly dense solutions including sodium polytungstate, zinc chloride, zinc bromide, and sodium iodide are effective in separating small MP fibres despite the high cost and hazardous impacts (Nakajima et al. 2019). This situation was observed in the Munich Plastic Sediment Separator (MPSS) designed by Imhof et al. (2012) with a high recovery rate of 95–100% using zinc chloride solution without further extraction or incurring contamination or losses (Imhof et al. 2012). However, this recovery rate does not apply to the old MPs and given the properties of zinc chloride, it may corrode surfaces and result in a low separation of the less dense substances. Moreover, zinc chloride may react with sediments on the MP surface to create a foam that may disrupt the extraction process (Moller et al. 2020). Recent studies have indicated that sodium bromide is very effective in the extraction process as it is economically viable as a reagent and non-corrosive and non- hazardous, although it may have its own limitations due to the local plastic demand (Yang et al. 2021).

Multi-stage separation was an alternative strategy developed to encounter the limitations posed by separation solutions and to increase separation efficiency. In some studies, the same solution has been reused many times whilst others have opted applying lower and high-density solutions consecutively (Huang et al. 2020). Frere et al. (2017) used sodium tungstate solution (1.56 g cm^{-3}) to separate dense MP sample (0.8–1.4 g cm^{-3}) from denser particles like sand and coarse sediment grains (2.65 g cm^{-3}). A spiking experiment was utilised to verify the full recovery of the dense MPs. The study obtained a complete recovery of each polymer types (PET, PA, PVC) without any hindrance during the identification of the polymers by Raman Spectroscopy (Frere et al. 2017). Han et al. (2019) mixed saturated NaCl and NaI solution with a flotation density of 1.50 g cm^{-3} that obtained a 90% or more recovery of most of the tested MP particles wherein the authors believed that the solution can be reused 5 times after filtration.

Novel procedures of electrostatic separation also exists which permits a significant recovery rate of 90–100% for MPs ranging from 63 µm to 5 mm (Felsing et al. 2018). But this result was obtained by conducting an experiment for 3–4 h on a soil sample lacking moisture which could raise further doubts on the aggregate formation (Felsing et al. 2018). To prevent the soil aggregation, some techniques including centrifugation, aeration, ultrasonic treatment, and continuous flow can be implemented. Grbic et al. (2019) have developed magnetic extraction to recover MPs by creating surface modified hydrophobic iron nanoparticles that would bind to MPs samples, thereby allowing magnetic recovery. However, this experiment was conducted using freshwater MPs samples and has not yet been tested on soil matrices. The observed MPs have been fragmented and damaged caused by the magnet's removal (Grbic et al. 2019). However, a more promising technique to extract MP from soil was built incorporating this concept by Ramage et al. (2022) known as High Gradient Magnetic Separation (HGMS) which was able to circumvent issues associated with the former method. The HGMS proved to be more cost effective, faster and high recoveries for both high- and low-density MPs and fibres from various soil matrices. Yet one prevailing drawback of HGMS is that prior knowledge of the soil composition needs to be understood (Ramage et al. 2022). Scheurer and Bigalke, (2018) have applied the pressurized fluid extraction (PFE) method for its efficiency wherein the analysis is performed automatically without any human influences. However, there would be some limitations due to its small sample capacity and a high level of sensitivity that leads to an inaccurate quantification. It also further failed to capture information regarding particle size required for the toxicity and mobility studies of MPs (Scheurer and Bigalke 2018).

Techniques for microplastic quantification

Identification and quantification are performed after microplastics have undergone the extraction stage. The identification of MPs is determined initially through visual inspection either by naked eye or using a microscope (Wang and Wang 2018). The morphological characteristics such as colour, shape and surface texture are the primary factors to distinguish if the item is indeed an MPs. More fluorescent staining incorporating dyes such as Evans blue, Calcofluor white and Nile red can be utilised to distinguish MPs from the surrounding matrix (Helmberger et al. 2020). However, identification via visual inspection will be subjective as it depends on various factors and could impair accuracy because of misidentification due to degradation and false positives (Wang and Wang 2018). Blasing and Amelung (2018) noted a rate of 20–70% of error in heterogenous soil samples whilst conducting visual identification.

These discrepancies can be rectified and verified with spectroscopic and thermoanalytical techniques such as Raman spectroscopies, Fourier Transform infrared spectroscopy (FTIR) and Pyrolysis gas chromatography–mass spectrometry particularly for smaller sample sizes. The FTIR spectroscopy measures the amount of IR radiation absorbed by MPs sample thereby providing an analysis of its molecular composition (Chen et al. 2020a, b, c, d). Absorption peaks produced from the IR spectrum would correspond to the vibration frequencies of the samples' atomic bonds and consequently present a rapid and reliable representation of the plastic structure (Chen et al. 2020a, b, c, d). Amongst the research studies, Chen et al. (2020b) demonstrated that smaller MPs were identified by the μ-FTIR with minimum particle size of 10 μm and larger ones by ATR-FTIR (Attenuated total reflectance FTIR). The ATR-FTIR technique, however, is known to cause damage to certain MPs due to the large pressure applied by the equipment's probe. To counteract this problem, FPA-FTIR (focal plane array FTIR) can be implemented, enabling scanning of more tentative MPs types (Loder et al. 2015). However, Loder et al. (2015) noted that μ-FTIR requires a longer duration for sample measurement, (over 20 h) causing a risk of loss and contamination of coloured or small sample plastics. Raman spectroscopy is an alternate yet promising technology identifying the MPs sample and is considered more advantageous than the FTIR. It encompasses a better spatial resolution (1 μm) and requires low amounts of the MPs sample (Araujo et al. 2018). Raman spectroscopy is also known to simultaneously perform analysis on wet samples while distinguishing pigments or fillers (Zhao et al. 2018). However, studies have shown that this method is more susceptible to fluorescence interference and sample heating due to applying laser as the light source which may lead to the degradation of MP fragments (Zhao et al. 2018). Also, even though Raman spectroscopy can provide reliable information on MP identification, the process is highly time consuming (He et al. 2018). Furthermore, Paul et al. (2019) claimed that this technique tends to perform less efficient when non-coastal soils were considered and thus restricts its potential usage in the context of this research. A potential obstacle further lies with the sensitivity of FTIR and Raman spectroscopy to water, atmospheric carbon dioxide, SOM, and traces of clay present in the sample which would then require more extensive removal during the sample preparation stage (Xu et al. 2019a, b). Thomas et al. (2020) pointed out that the time needed to obtain measurements via these applications may impede screening and monitoring studies that are to be conducted.

According to a study conducted by Wang et al. (2019), mass spectrometry techniques consisting of pyrolysis–gas chromatography–mass spectrometry (Pyr–GC–MS), thermogravimetric analysis-mass spectrometry (TGA–MS), and thermal extraction desorption-gas chromatography–mass spectrometry (TED-GC–MS) can also be utilized for MP identification. Mass Spectrometry works by separating the constituents of the ionized MP sample depending on their m/z value (Wu et al. 2023). These techniques encompass several advantages, from high sensitivity and easy operation to reliable and quick analysis that can be utilised for detection of molecules along with the identification of a material's composition (Wu et al. 2023). These technologies were also shown to analyse samples with no further pre-treatment. However, Wang et al. (2019) argued that the morphology of the plastic fraction and its colour could be damaged or hampered during the evaluation of the sample. Thomas et al. (2020) indicated that there will be the issue of preparing homogenous aliquots (less than 100 mg) for these techniques if the MP sample was not extracted through using organic solvents which will then require cryomilling. Paul et al. (2019) have tested another conventional process analytic—Near infrared (NIR) spectroscopy which was successful in achieving a high throughput analysis. The electromagnetic spectrum part of NIR ranges from 667–2500 nm (in the middle of IR and UV) and has a greater capacity to penetrate deeper, and making it very viable in larger samples.

Depolymerization method with the aid of alkali-assisted heating is a recent identification technique suggested by Wang et al. (2017) where types of MPs such as polycarbonate (PC) and PET are identified and quantified by establishing their constituent compounds. However, samples consisting of a wide range of key components could cause measurement issues and thus the method would need further investigation.

Complications with sample analysis

Insufficient research conducted in the case of analytical methods will cause limited development of such technologies in future studies. The situation is even further exacerbated when observing that most of the available methodologies derived from MP analysis are in the aquatic environment. This is a critical factor as very limited quality control and external proficiency tests have been conducted in this area (Moller et al. 2020). For small sample size, the results may not be reliable and if larger samples are selected the duration of the analytical process will be too long, and expensive. Therefore, the size and quantity of sample is significantly essential for the accurate results. As strategies are varied in sample measurements, some studies indicated the volume as a basis for sample measurement and other used mass (Yang et al. 2021). The geographical location is also another essential factor since collecting and regulating the quantity of samples is related to the local plastic demand and has a direct proportional relationship with the type of MPs found in soil. Moreover, the definition of MP concentration needs to be illustrated as some studies have applied the weight of MPs per kg soil instead of the number of MPs per kg soil (Zhou et al. 2020). Zhou et al. (2020) also noted the necessity of obtaining the analytical procedure of collected samples when comparisons of MPs concentration from different regions need to be reported.

There are possibilities of losses of MPs from the soil sample especially during the sieving process. Some operations have a sieving size of 2 mm (taken as standard) but others would apply a sieving size of 1 mm which can cause that MPs greater than 1 mm to be discarded (Blasing and Amelung 2018). This would then require more consideration during identification and quantification stage as MP abundance in the sample is based on the sieve size. Furthermore, Yang et al. (2021) have pointed out that the MP size required for the FTIR spectroscopy depends on how easily it can be handled by tweezers for transferal compared to the Raman spectroscopy technique which even identifies smaller MPs sizes. Hence, it highlights the importance of all stages in the analytical methodology for sample identification. Additionally assessing pristine and aged MPs is an important factor as it plays a vital role in the MPs identification process. This is due to the fact that weathering of such aged particles may cause great losses if an improper identification method is selected (Chen et al. 2020a, b, c, d).

Key differences in analytical methods for aquatic systems

While the analytical methods for microplastics in various environmental media share common procedures, there is notable variability across different matrices. The main difference lies mostly in sampling. As discussed previously, in aquatic systems, sampling techniques involve considering a water column with depths that vary based on the research objectives. For instance, surface water sampling, a prevalent approach, aims to unravel the occurrence of microplastics (MPs) at the surface (Yu et al. 2016). Neuston nets are commonly employed for surface collection at depths ranging from 0 to 0.5 m (Anderson et al. 2017), while bongo nets are utilized for water columns of medium depth, and benthic nets are deployed for seabed sampling. Research in this field has reported a significant depth range, extending from 50 to 60 µm (surface microlayer) to 212 m (Stock et al. 2019).

Similar to soil sampling, the presence of MPs in aquatic environments depends on factors such as the collected water column, net opening area, and mesh size. Studies indicate a wide range of mesh sizes from 0.053 to 3 mm, with the net aperture playing a crucial role in determining the efficiency of microplastic capture. A flow meter is mounted at the entrance of the net to determine the filtered water volume which can thereby be used to find the MP concentration (Wagner and Lambert 2017). Alternatives such as 3D hydrodynamic—numerical modelling and acoustic doppler current profilers are other means to find the water volumes, the latter of which can be implemented for sampling different depths at one location (Stock et al. 2019).

As methodologies for assessing MPs in aquatic systems continue to evolve, the diverse array of sampling techniques reflects the complexity of studying MPs across different environmental compartments. In marine research samples obtained from below the water surface have no mention of specific sampling depths and vary according to the goal of the research conducted. For sediments near water bodies, which is regarded as a long-term sink for MPs, sampling will differ with location which can be categorized as the tideline, intertidal and supralittoral environments (Mai et al. 2018). In beaches, most studies have conducted sampling within a depth of 5 cm with equipment such as trowels, shovels, spoons used for sample extraction. Wang et al. (2018) notes that although some studies have recommended using tweezers for extracting, it may overlook smaller MPs thus underestimating the abundance of MPs in that location (Hidalgo-Ruz et al. 2012). For sublittoral zones Wang et al. (2018) suggests the usage of grabbers or box corers for superficial sediments. However, in such cases, it is important to first identify and comprehend the various sediment layers since the usage of these tools causes disturbances.

Methods of degradation

Degradation of MPs occurs continuously in the environment either by biotic or abiotic methods, resulting in their fragmentation and eventually nanoparticles formation. Degradation of these particles constantly continues to form smaller particles until the polymer is mineralized to carbon dioxide, water, and biomass by biotic means (Dawson et al. 2018). The time required to completely degrade MPs depends on the type of polymer, thickness of material, external factors, etc. A study conducted by Du et al. (2021) found that a thin MP film made of PP would degrade approximately within 144–500 h if directly exposed to the UV light in the presence of a photocatalyst. Hence, it is important to understand different mechanisms which could aid in the degradation of MP as this may further help explore remediation methods.

Abiotic degradation methods

This type of degradation deals with the break-down of plastics due to abiotic factors resulting in chemical and physical changes. Based on the type of the abiotic factor, photodegradation, thermal degradation, chemical degradation, or mechanical degradation may take place (Zhang et al. 2021). Photodegradation is the most important and common method of ageing of MPs which takes place in 3 steps; (1) initiation step, which deals with the scission of the polymer chain due to the absorption of UV rays by the carbonyl groups present on the polymer backbone and releases hydroxyl and alkoxy free radicals in the process; (2) oxidation, which occurs at the surface of the plastic; (3) termination, which results in the formation of inert products by either a recombination or disproportionation reaction (Luo et al. 2021b). This process takes place on the surface of the MP and small amounts of gas and liquids are also released (Zhang et al. 2021).

Thermal degradation of MP is similar to the photodegradation process, but the initiation of the process takes place due to the absorption of heat from the surroundings (Luo et al. 2021b). Once an adequate amount of heat is absorbed by the polymer molecules, the provided energy would break the carbon bonds in the long polymer chain and form free radicals. These radicals then react with oxygen through the process of diffusion producing hydroperoxides. The reaction is self-propagating and terminates only on the collision of two free radicals or when input of the thermal energy is cut off. The amount of heat required for the initiation of thermal degeneration is called the activation energy and varies for different polymer chains (Zhang et al. 2021). Photodegradation and thermal degradation results in changes to the size, shape, color, and properties. This type of degradation not only breaks the main polymer backbone but also breaks the cross linking and causes depolymerization which results in chemical and physical changes of the MPs (Zhang et al. 2021).

Mechanical degradation of MPs is the fragmentation of the particles due to external force such as collision applied by waves, rocks, sand, wind, etc. in both aquatic and soil environments. One type of mechanical force which is usually overlooked is the freezing and thawing of the MPs (Zhang et al. 2021). The extent of degradation due to mechanical force depends on the mechanical properties of the MPs where the breaking of polymer chains in the MPs occurs with the continuous applied mechanical forces. Elongation at break is a property about the ability of the polymer to resist change of shape with respect to the type of polymer and varies from 1 to 900% (Zhang et al. 2021). Plastics with a lower elongation at break are likely to degrade faster through the mechanical degradation methods. An example of mechanical degradation is the degradation of synthetic fibers when washing clothes (Isaac and Kandasubramanian 2021).

Chemical degradation of plastics refers to the fragmentation of the MPs through the photochemical due to the presence of compounds such as ozone (O_3), sulfur dioxide (SO_2), nitrogen dioxide (NO_2), volatile organic compounds (VOCs) in the air. These compounds either act as a reactant or a catalyst to form free radicals with the polymer chain present in MPs. The low concentrations of O_3 in the atmosphere react with the double carbon bonds in unsaturated polymer chains and with saturated carbon bonds but at a much slower rate, causing the polymer chain to become scission. Sulfur dioxide is highly reactive in the presence of UV radiation and results in unpaired electrons by the formation of singlet or triplet states whereas NO_2 contains an odd number of electrons resulting in high reactivity. Due to the reactive nature of both components, they attack carbon double bonds present on the polymer chain (Zhang et al. 2021).

The pH, salinity, and concentration of humic acid (HA) present in the soil or water also influence the degradation of MPs (Luo et al. 2021a, b). Extreme pH values or salinity is caused due to the presence of hydrogen ion (H^+) or hydroxide ion (OH^-) which can catalyze the degradation process in polymer chains that are capable of undergoing hydrolysis (e.g.: polyamides) (Zhang et al. 2021).

Biotic degradation methods

Biotic degradation of MPs refers to the type of deterioration that takes place due to various activities such as biting, chewing or digestive fragmentation by the organisms on the polymer chain. The organisms that mainly take part in this process in aquatic and soil systems are bacteria, fungi, and insects (Zhang et al. 2021). There are two main methods of biotic degradation: biophysical degradation and biochemical degradation (Luo et al. 2021a, b). The degradation through this method depends on the MPs characteristics such as chemical structure, functional groups, molar mass, crystallinity, additives, external factors, and the type of microorganisms involved in the degradation (Yuan et al. 2020). In biophysical degradation, microorganisms adhere to the MP surface and break it down into oligomeric fragments by hydrolysis, ionization, or protonation through the growth of the organism on its surface. Whereas in biochemical degradation microorganisms secrete an enzyme which is responsible for breaking the polymer chain into compounds containing a smaller molecular weight such as oligomers, dimers, or monomers. These small compounds can pass through the cell membrane of other microorganisms and act as a carbon source, resulting in its mineralization into carbon dioxide, methane, and water (Luo et al. 2021a, b). A summary of the different types of degradation of plastics is shown in Fig. 3.

Property changes of MPs after degradation

Crystallinity changes

The crystallinity changes in MPs depends on whether it is made up of a homopolymer or copolymer chain. In the case of most homopolymers, the crystallinity increases due to two reasons: (1) amorphous portion of the polymer is more likely to degrade as compared to the crystalline portions of the polymer, (2) Chemi-crystallization is a process that

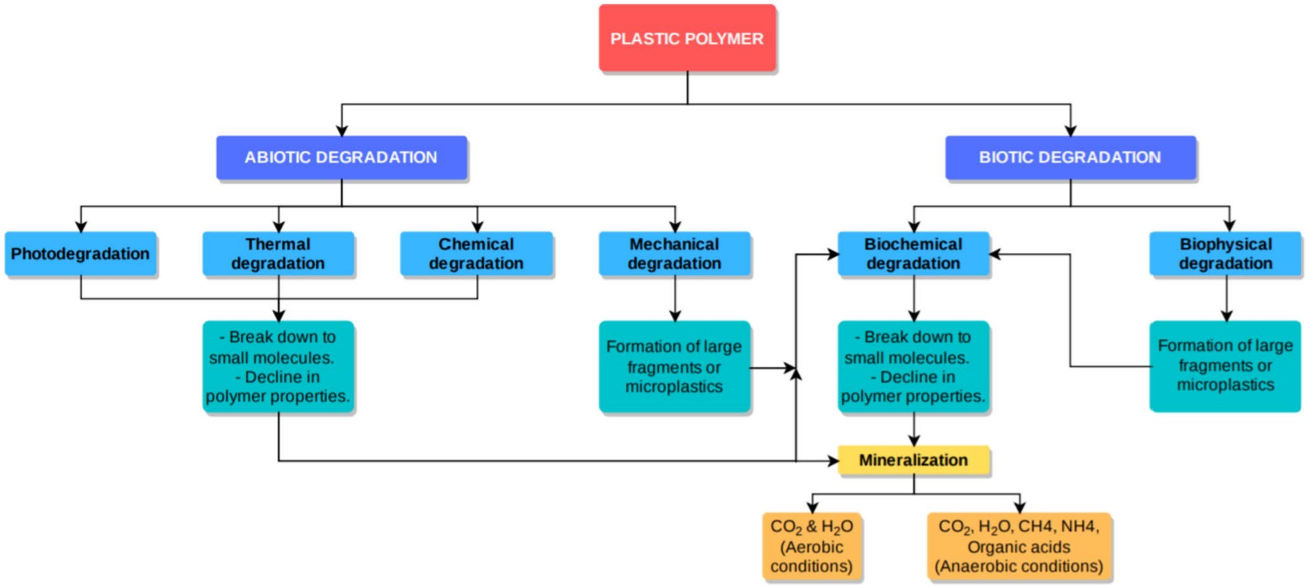

Fig. 3 Summary of different types of plastics degradation

causes the polymer chain to undergo chain scissions and cross linking allowing entangled chains (amorphous regions) to crystallize and attach to the preexisting crystalline chains (Craig et al. 2005). However, in copolymers crystallinity decreases with an increase in degradation of MPs. The comonomers present in the polymer allow the easy entry of oxygen which promotes photooxidation and thermo-oxidation since O_2 molecules are required for such degradation processes. Hence, in copolymers the rupture of the regularity domains occurs and reduces crystallinity of the MPs (Guo and Wang 2019).

Density changes

The colonization of different types of microorganisms due to degradation causes the formation of a biofilm over the MPs surface. As time passes the thickness of the biofilm increases, as does the mass of the MP, and hence an increase in the density of the MP. This results in sinking MP further down in the water body. Therefore, density is an important property of determining the settling rate of MP found in sediment samples (Guo and Wang 2019).

Surface morphological changes

These changes are seen on the surface of MPs as it happens mainly due to abiotic degradation methods. Any morphological changes first take place at the surface, as oxygen cannot penetrate through the MPs for photo and thermo-oxidation. This results in the formation of cracks at the surface. The detection, comparison and characterization of such changes can be done using Field Emission Scanning Electron Microscope (FE-SEM). Therefore, with an increase in cracks and deformations on the surface, there is a higher chance for the MPs to undergo further degradation (Guo and Wang 2019).

Color changes

The results of a study conducted by Kowalski et al. (2016) showed that Acrylonitrile Butadiene Styrene (ABS) and PS undergo discoloration from white to yellow, one month after the application of UV treatment. The yellow color is formed due to the formation of yellow colored quinone compounds. Therefore, using color to identify the compound should only be an initial visual assessment of the MPs type, prior to the spectral or chemical identification methods.

Table 1 Summary of various parameters of aquatic organisms and MPs effects on their biological health

Organism	Type of polymer	Size (µm) and concent-ration (mg/L)	Number of test species and exposure duration	Tissue	Response	References
Nematode (*Caenorhabditis elegans*)	PS	1 µm 0.01–0.1 mg/L	40 72 h	-Intestinal cells and body cavity -Affected lipids, proteins, DNA produced and caused tissue damage	•Intestinal injury •Oxidative stress[a]	Yu et al. 2020
Fresh water coral polyp (*Hydra attenuata*)	PE	< 400 µm 10,000, 20,000, 40,000, 80,000 mg/L	NA 96 h	Gastrointestinal tract	•Reduction in food uptake •Increase in time and energy for egestion	Murphy and Quinn 2018
Sea snail/mollusc (*Crepidula onyx*)	PS	2 µm Appx. 0.01 mg/L	1000 larvae 65 days	NA (requires further investigation)	•Ingestion and egestion of MP comes with an energy cost •Delays growth in younger organisms (due to requirement of more energy uptake) •Reduction in male population size (egg limitation)	Lo and Chan 2018
Transparent sea squirt (*Ciona intestinalis*)	PS	0.75–50 µm 0.125–25 mg/L	30 8 days	MP found in cytoplasm of blood cells	•Reduction in food uptake •Delays growth in younger organisms (at higher concentration due to less energy uptake) •Indigestion of MP •Translocation of MP from gut to internal extracellular compartment •Fertility reduction	Messinetti et al. 2019
Zebra mussels (*Dreissena polymorpha*)	PS	1 and 10 µm 50 mg/L	Appx 100 6 days	Gill	•Oxidative stress (affects the ribosomal function, energy metabolism, cellular trafficking, RNA binding, cytoskeleton)	Magnia et al. 2019
Mollusc (*Mytilus galloprovincialis*)	PE, PS	< 100 µm 0.5–5 mg/L	150 7 days	•Gills •Digestive glands •Hemolymph	•Immunotoxicity •Neurotoxicity •Genotoxicity •Changes in gene expression	Avio et al. 2015
Zebra fish (*Danio rerio*)	PA, PE, PP, PVC	Appx. 70 µm 0.001–10 mg/L	48 12 h	Intestinal damage and broken intestinal tissues	Intestinal injury	Lei et al. 2018
Fresh water crustacean (*Daphnia magna*)	PET	62–1400 µm 12.5, 25, 50, 100 mg/L	20 48 h	•Gastrointestinal tract	•Increase in mortality rate •Accumulation of MP in the gut	Jemec et al. 2016

Table 1 (continued)

Organism	Type of polymer	Size (µm) and concent-ration (mg/L)	Number of test species and exposure duration	Tissue	Response	References
Crab (*Eriocheir sinensis*)	PS	0.5 µm 40 mg/L	500 10 days	•Gastrointestinal tract •Liver	•Reduces growth •Damage and oxidative stress in hepatopancreas	Yu et al. 2018a, b
Fish (*Sardinella gibbosa*)	PET, PA	500–1000 µm NA	25 24 h	Gastrointestinal tract	•Reduction in food uptake •Indigestion of MP •Organism with larger body weight ingest higher amounts of MP	Hossain et al. 2019
Pond turtle (*Emys orbicularis*)	PE	4500–5000 µm 500–1000 mg/L	36 1 month	•Liver •Kidney	•Reduces level of protein, albumin, and globulin •Liver dysfunction •Kidney dysfunction	Banaee et al. 2020

[a]Oxidative stress is the disturbance in the balance between the production of oxygen species and antioxidant defenses within the organism

Effect of microplastics on organisms

Effect on aquatic vertebrates

MPs have various negative effects on different organisms which disturb their growth, reproduction abilities, feeding habits, and other physiological functions such as immune functionality and hemocyte count reduction, distorting oxidative system, respiration, etc. (Mkuye et al. 2022). Previous studies on MPs and their effects were mainly focused on the mortality rates of organisms due to entanglement and ingestion of MPs. However, recently the effects of MPs on organisms due to absorption of pollutants, pathogens, metals, and other contaminants have been examined and investigated. Unfortunately, limited information is available about the effects of MPs on complex species due to ethical considerations (Granek et al. 2020).

Table 1 shows a summary of the different experiments performed to understand the effects of MPs on aquatic organisms. The response of most organisms is noted as growth retarded effect in the juvenile cells because of the high energy uptake for the egestion of MPs, which leads to inadequate energy availability for development and growth. Oxidative stress is also a common symptom noted along with inflammation, accumulation, mortality rate, and toxicity, in some other cases. In larger complex organisms the functions of gills, liver, kidney, and gastrointestinal tract tissues are most disturbed.

It is important to investigate the effect of MPs on fish as these animals are consumed by humans. Numerous experiments have been conducted on zebrafish as their genetic makeup is similar to humans (Bhagat et al. 2020). Handy et al. (2008) reported that MPs can accumulate in the brain, liver, gut, and gills of the fish in most common cases where side effects may include vascular damage, oxidative stress, tumor formation and ion-regulatory disorders. In an experiment conducted by Lei et al. (2018) on zebrafish, the PA, PP, PE, PVC, and PS with a size of 70 µm have been used for 10 days at altering concentrations between 0.001 and 10 mg/L. It was noted to cause intestinal damage such as the breaking of the villi and electrolytes present in the intestinal walls. In a study conducted by Jin et al. (2018), the amounts of mucus secreted, protein and m-RNA formation were notably increased which further caused inflammation inside the gut due to the introduction of MPs in the experimental aquatic environment. Lu et al. (2016) demonstrated that accumulation and inflammation of fat in the liver of the organism was also detected due to the increase in superoxide dismutase (SOD) and catalase (CAT) in the blood. This resulted in an adverse change in the lipid and energy metabolism and metabolomic profiles of the organism. According to the experiment conducted by Mak et al. (2019), disturbances in the oogenesis process in females, neurotoxicity (indicated by seizures), abnormal behavior (tail bent downwards) was also observed when zebrafish were exposed to MPs for a short period of time. The transcriptomic study done by Limonta et al. (2019) on adult zebrafish has proved the presence of tissue alterations and neutrophils (immune cells that help fight infection) in the gills and intestines. This alteration is observed because of the mucosal epithelium degrading due to the ingestion of MPs and in turn deteriorates the immune response of the body. This causes the organism to utilize more energy which affects its daily activity and reduces the ability of the immune system to fight pathogens.

Effect on mammals (mice and rats)

Numerous studies are available on the direct effects of MPs on mammals such as mice and rats from either ingestion of MPs through the soil or water bodies (Kannan and Vimalkumar 2021). However, limited research has been conducted on the bioaccumulation and transfer of MPs from aquatic systems to higher trophic levels (Miller et al. 2020). Araujo and Malafaia (2021) conducted an experiment using PE to observe the effects on three trophic levels: tadpoles (*Physalaemus cuvieri*) collected in a temporary pond, juvenile fish (*tambatinga*) and male Swiss mice whose age ranges from 45 to 60 days. The result from the experiment confirmed the negative impacts of MPs on mammals. It is confirmed that MPs induce neurotoxic effects, manifesting as anxiety in organisms and a delayed anti-predatory response, including slowed locomotion, in instances of both direct and indirect exposure to MPs. This study determined that the transfer of MPs and the noxious effects on different trophic levels are most probable and may possibly cause harm to humans as well. Hence, it is essential to investigate and understand additional harmful effects on other mammals. Another study conducted by Li et al. (2020), has used 6 weeks old healthy male Wistar rats and applied PS as the MP. The rats were divided into four groups and were exposed to the MP by drinking water containing PS. Results showed that the rats underwent cardiovascular toxicity through oxidative stress, leading to heart fibrosis which finally resulted in cardiac dysfunction.

Effect on humans

According to the statistics from the Seafood source in 2018, the amount of global seafood production was estimated to be around 179 million metric tonnes (MT), out of which 23 million MT contributed to human consumption. Hence, seafood is a major pathway of MPs into human body through the process of bioaccumulation. However, inhalation and dermal contact are also possible routes of human exposure through MPs present in the air and soil (Prata et al. 2020). Transferring of PMPs from air into water bodies by wind accounts for 7% of the ocean's contamination whereas road runoff (through land) contributes to 66% of PMPs pollution (Boucher and Friot 2017), quite obviously, all originating from human activities.

In a study conducted by Cox et al. (2019), it was found that on average, a person is estimated to consume 39,000–52,000 MP particles annually from caloric intake depending on age and sex where this value increases between 74,000 and 121,000 when inhalation is considered. It is essential to understand the effects and retention capacity of MPs on humans. The MPs are toxic to humans due to carrying harmful chemicals and metals and containing noxious organic compounds such as additives and plasticizers (Campanale et al. 2019).

Physical effects

According to a study conducted by Weis (2020), most spherical MPs can pass through the gut of animals and humans without causing much damage. The most common site of entry being M-cell rich Peyer's patches in the intestine. The MPs with sharp edges may bruise the gut wall or fibrous MPs may clog up the gut. An experiment performed by Volkheimer in 1974 (and is accepted in recent studies) tested the oral ingestion of inert starch granules as large as 150 µm, to see if it would lead to the MPs persorption in the intestinal villi where there is a single layer of epithelium cells and was detected in the lumen of blood and lymph vessel of humans and animals within minutes. From the lymphatic system, the MP is sent to the liver and gall bladder which will then be released back into the small intestine along with the secreted bile (Galloway 2015). Recent studies used human cells in culture to understand the translocation of MPs with the size less than 10 µm from the gut cavity. It was observed an accumulation of MPs in liver, kidney and brain tissues. However, particles smaller than 0.1 µm can cross any cell membrane and access all organs (Yong et al. 2020). Hence, size is an important factor in determining the amount and location of the MPs absorbed by the body as well (Bouwmeester et al. 2015). Recent study has revealed that only a small fraction of the administered MPs can pass the epithelial barrier of the lungs and intestine, with the particle uptake increasing with the decrease in MP size (Vethaak and Leglar 2021). Despite the low uptake, long term exposure and potential accumulation in tissues and organs makes them a significant concern (Wright and Kelly 2017). Galloway (2015) reported that small size of MPs between the range of 0.05–0.1 µm were comparatively easier to be absorbed by the gut than MPs in the range of 0.3–3 µm. It has been found that MPs can even pass through

Table 2 Summary of negative effects microplastics on different parts of human body

Organ	Cell	Type of polymer and duration of exposure (hours)	Size (µm) and concentration of polymer (mg/L)	Response	References
Brain	- Human glioblastoma multiforme, T98G cells - Human cervical carcinoma, HeLa cells	-PE -PS 24–48 h	– 3 to 16 µm (PE) – 10 µm (PS) 0.01–10 mg/L	• Excessive generation of Reactive Oxygen Species (ROS) will result in oxidative stress -•Cytotoxicity was observed because of the oxidative stress	Schirinzi et al. (2017)
Intestine	Human colon adenocarcinoma, Caco-2 cells	PS 12 h	5 µm, 20–50 mg/L	• Mitochondrial membrane potential was disrupted by both size of MPs • Increased arsenic toxicity in the cell membrane by both size of MPs • Both size of MPs caused low toxicity on cell viability, membrane integrity, fluidity, and oxidative stress but the larger particles caused a higher impact	Wu et al. (2019)
Immune system Blood cells	- Human Mast Cell line -1, HMC-1 - Human basophilic leukemia cell line, RBL-2H3 - Human dermal fibroblast peripheral blood mononuclear cells, PBMC	PP 96 h	20, 25, and 200 µm 100, 300, 1500, 2000, 3000, 4500 mg/L	• An increase in histamine release from HMC-1 and RBL -2H3 cells • Reduction in cytokine released by PBMC • Increase in ROS production and cytotoxicity in all types of cells for smaller sized particles at higher concentration of MP	Hwang et al. 2019)
Lungs	Human lung epithelial, BEAS-2B cells	PS 48 h	3.51–4.38 µm NA	• Low levels of PS can disrupt epithelial cells, increasing risk to lung diseases • Caused cytotoxicity and inflammatory effects on the cells • Increase in ROS production causing oxidative stress	Dong et al. (2020)
Alveoli of lungs	Human lung epithelial, A549 cells	PS 0.167, 0.33, 0.5, 1, and 2 h	0.025 and 0.07 µm 1.14 mg/L (only 0.025 size) and 25 mg/L (only 0.07 size)	• Decreased cell viability • Activated inflammatory response • Changed expression of proteins associated with the cell cycle	Xu et al. (2019a, b)
Skin	Human foreskin fibroblast cells, Hs27 cells	PS 4, 24, and 48 h	µm 5, 25, and 75 mg/L	• Increase in ROS production • Genotoxicity resulting in DNA damage	Poma et al. (2019)

the placenta and the blood–brain barrier and finally be taken up by the gastrointestinal tract and lungs of the offspring (Seltenrich 2015). Most common biological effects noted in humans include oxidative stress, cytokine secretion, cellular damage, inflammation, immune reactions, DNA damage, and neurotoxic and metabolic effects (Vethaak and Leglar 2021). Table 2 shows a summary of the physical effects of MPs on different organs in the human body due to varying size and composition.

Effect of chemical additives

There are several purposes of having additives in plastics including functional additives (e.g.: stabilizers, antistatic agents, flame retardants, plasticizers, lubricants, curing agents, foaming agents, etc.), colorants, fillers (e.g.: mica, talc, clay, calcium carbonate, barium sulfate, kaolin), and reinforcements (e.g.: glass fibers, carbon fibers). The additive in rare cases, such as reactive organic additive, is chemically bound to the plastic polymer chain (COWI and DTI 2013). As MPs are consumed by aquatic life, which is later consumed by humans, it is crucial to investigate the effects of such MPs on human physiology.

Bisphenol A (BPA) is commonly used in food cans, feeding bottles for infants, PVC pipes, dental products, flame retardants in plastic products, etc. (Lamprea et al. 2018). Phthalates are the most used plasticizer found in food packaging, personal care and household products and can leach out from these products as they are not covalently bonded to the polymer chain like BPA's (Segovia-Mendoza et al. 2020). Rachon (2016) found that even low exposure of phthalates or BPA can lead to neurological, metabolic, reproductive abnormalities and carcinogenic effects in the offspring. An experiment was performed by Segovia-Mendoza et al. (2020) on mice to understand the effects of BPA and phthalates on humans. They found that it causes a higher risk of breast, colon, prostate, endometrial, cervical, and lung cancer, larger mammary tumors, promotes sexual dysfunction in both males and females, and subjects the offspring to cancer carcinogenesis as well. It showed that long-term exposure to these additives may also cause other diseases and endocrine disorders (Lazúrová and Lazúrová 2013). Nusair et al. (2019) observed that brominated flame retardants such as Tetrabromobisphenol A (TBBPA) pose a risk as they cause genotoxicity in humans. TBBPA is used in electronics, furniture, plastics, and textiles and is taken up by humans through food, respiration, and skin contact. Apart from TBBPA, Polycyclic aromatic hydrocarbons (PAH) are another organic pollutant present as a flame retardant in most plastics. Sun et al. (2021) demonstrated that low doses may trigger eye and skin irritation, nausea, vomiting, and DNA damage. On chronic exposure, it may reduce immunity and cause cataract, kidney, liver, pulmonary, and lung function impairment, lung, skin, and digestive tract cancer and jaundice. PBDE is another flame retardant which has been used in plastic products but was prohibited globally according to the Directive EEC (2003). Despite the global ban in 2003, plastics break down into MPs over a prolonged period and hence humans may still be exposed to them mainly through seafood and dust particles. The study conducted by McDonald (2002) found that it has a high bio accumulative potential which could cause PBDE induced thyroid hormone disruption, neurobehavioral deficits, and increases the risk of cancer.

Control measures

An effective approach to reduce the release of MPs is simply to decrease the use of products that are made of MPs and the products that use plastic packaging. This will result in reducing the production of PMPs and SMPs. Consumer awareness is key as they can choose to boycott products containing MPs and plastics. The best way to do is by replacing the harmful MPs with natural alternatives, which is a common method that is being implemented by companies lately. Microfibers contribute to a large part of MPs pollution in aquatic systems. In an average, for every time an individual does the laundry, about 9 million microfibers are sent to the WWTPs that are unable to filter. Therefore, it is important to manage the consumption of synthetic fibers by opting for non-synthetic, eco-friendly clothes (Rebelein et al. 2021). Removal methods from different media could be physical, chemical, or biological (Ahmed et al. 2022) which will be discussed in the next sections.

Reducing and replacing of additives

An experiment conducted by Selke (2016) tested the authenticity of so-called biodegradable additives. Interestingly, normal petroleum-based plastics and plastics containing biodegradable additives degraded in the same amount of time (both plastics contained the same polymer). Hence, the addition of such additives does not speed up the degradation process. Andrady (2017) found that alternative options for phthalates could be bis(2-propylheptyl) phthalate (DPHP) or bis(2-ethylhexyl) adipate (DEHA), which can be used as green plasticizers. An alternative to PBDE was found to be alkali metal oxides, aluminum hydroxide and phosphorus-based flame retardants. Hence, there is high scope for developing alternate eco-friendly additives which requires further research.

Wastewater treatment

A common physical method that is used for untreated wastewater is sedimentation, the efficiency of which ranges from 40.7 to 91.7% based on the type and size of MP present (Ahmed et al. 2022). Liu et al. (2019) in an experiment have reported a maximum removal efficiency of 64.4% from a wastewater stream containing MP ranging from a size of 20–4200 µm. However, a similar experiment that applied the sedimentation process was performed by Yang et al. (2019) with MP sizes ranging from 529 to 1111 µm and an overall removal efficiency of 95.16% was observed. An experiment conducted by Talvitie et al. (2017) investigated four different advanced physical methods such as a membrane bioreactor (MBR) for the primary effluent, rapid sand filter, dissolved air floatation, and the disc filter for treating the secondary effluent. The removal efficiency of each method was 99.9%, 97%, 95% and between 40 and 98.5%, respectively. Hence, applying advanced technologies would significantly reduce the amount of MP that is released into the aquatic bodies.

The two most common chemical methods that are used to treat wastewater are coagulation/flocculation and chlorination (Ahmed et al. 2022). An experiment conducted by Rajala et al. (2022) used secondary wastewater effluent to test the effects of different organic and inorganic coagulants. Ferric chloride was found to have a removal efficiency of 99.4%, polyaluminum chloride had 98.2% and polyamine had the lowest efficiency of 65%. It is worth indicating that for a larger MP size, more coagulant is required to achieve a high removal rate. Experiments conducted by Hassinen et al. (2004) and Castagnetti et al. (2011) revealed that the continuous exposure of chlorinated water to the PE pipes lead to its deterioration due to oxidative reactions. In a recent study, Liu et al. (2019) applied this chemistry to MP present in wastewater and found that after 2 h of contact with sodium hypochlorite, the MP content was reduced by 7.1%. Hence, the effect of chlorination would depend on the duration of exposure, surface area of MP, concentration of chlorine, and temperature.

Biological methods include the use of various microorganisms in activated sludge or biofilters (Ahmed et al. 2022). Facultative and anaerobic microbes that are present in the activated sludge break down the MP in wastewater into bioenergy in the absence of oxygen (Xu et al. 2020). Liu et al. (2019) observed a removal rate of 16.6% in the anaerobic tank (placed after a primary sedimentation tank). A study conducted by Liu et al. (2020) tested the efficiency of a four zone, pilot-scale biofilter using secondary effluent. The biofilm was allowed to mature prior to the beginning of the experiment to maximize the retention of MP. This allowed the removal efficiency of the biofilter to reach 79%, making the biofiltration method more effective compared to the activated sludge.

Sludge treatment

Although sewage sludge undergoes treatment in the WWTPs, the removal efficiency of MPs remains low. For instance, processes such as thermal drying and stabilization with lime induces alkaline and heat hydrolysis resulting in MPs disintegrating into tinier particles (Mahon et al. 2017). However, recent research emphasises the importance of thermophilic bacteria in the degradation of MPs during sludge treatments wherein, under a composting environment and hyperthermal conditions, the cleavage of –C–C– bonds are found to be accelerated. This concept has given rise to hyperthermophilic composting (*h*TC) which was found to have a more efficient bioconversion at a lower composting period compared to the conventional thermophilic composting (*c*TC) (Yu et al. 2018a, b). In the study conducted by Chen et al. (2020a, b, c, d), the effects of *h*TC on the in-situ biodegradation of a 200-tonne full-scale sludge (consisting of MPs) at a composting plant was investigated. The results reported the highest removal rate from conducted studies in MP biodegradation during full- scale treatment, with a MP removal rate of 43.7% observed after 45 days of treatment. This rate was noted to be 9 times greater than the results obtained for *c*TC during the experiment (Chen et al. 2020a, b, c, d).

An in-situ MPs biodegradation study has shed a new light into the possibility of implementing *h*TC in removing MPs from the wider environment (Chen et al. 2020a, b, c, d). The efficiency of treating organic wastes would facilitate the removal of MPs from various organic matter such as sewage sludge, sediments, and animal manure where MPs coexisted. Another advantage is that *h*TC can prevent the spread of contaminants through MPs as it can eliminate organic pollutants and pathogens adsorbed on the MP surface, and thus making *hTC* an economically feasible and low investment technology (Chen et al. 2020a, b, c, d).

Photocatalysis

Photocatalysis has gained attention as a promising method for MP removal and degradation. In aquatic settings, photocatalysts which are semiconductors like titanium dioxide or zinc oxide are used. When exposed to light, the semiconductors generate reactive oxygen species (ROS) in the presence of UV or visible light and effectively degrade the MP particles. The process of photocatalysis works through a series of oxidation–reduction reactions which is initiated by the absorption of photons through the catalyst's surface and generates the ROS such as hydroxyl radicals (Xie et al. 2023). This method offers advantages such as MP removal, minimized by-products, and broad applicability. However, it faces challenges in terms of light availability, catalyst immobilization, and potential ecological impacts

(Xu et al. 2023). Implementing photocatalysis in soil environments is a more recent development, marked by the complex and heterogenous nature of soil. Optimizing the distribution of photocatalysts and considering their interaction with soil environments are important. Researchers must grapple with issues related to the effective distribution of photocatalysts throughout the soil matrix and consider their interactions with diverse soil microenvironments (Ding et al. 2022). Additional attention is needed in areas such as evaluating the environmental impacts of ROS generated through photocatalysis, exploring the impact of diverse soil properties on the effectiveness of photocatalytic processes, and developing novel photocatalytic materials and methods to improve performance across various environmental conditions.

Different aqueous media

Adsorption is a physical removal method beneficial for removing MPs by incorporating novel adsorbents. A study conducted by Misra et al. (2020) achieved 100% removal of commercial PS beads (diameter = 1 and 10 µm, and concentration = 1 g/L) through synthesizing a magnetic polyoxometalate-supported ionic liquid phase (magPOM-SILP) composite. By using chemical methods for MP removal such as coagulation, in the case of drinking water, Ma et al. (2019) reported < 15% of removal efficiency for PE particles using Fe-based salts. However, the addition of polyacrylamide (PAM, 3–15 mg/L) significantly enhanced the removal efficiency of PE, particularly for anionic PAM combined with high dosage of $FeCl_3 \cdot 6H_2O$ (2 mmol/L), increasing the efficiency up to 90.9%. The study also observed that removal efficiency greatly improves for smaller MPs and is therefore size-dependent.

A study conducted by Strum et al. (2020, 2021) analyzed the removal of MPs in distilled and demineralized water by agglomeration-fixation using different organosilanes. An experiment was used to determine the removal efficiency gravimetrically. The organosilane was added to the suspension (MP and water) and stirred for 20 min for the agglomeration process to occur. The contents were then filtered to sieve out agglomerates larger than 1 mm. The filtrate was then filtered repeatedly through a filter crucible and rinsed with isopropanol to remove any organosilane residues. To accurately weigh the MP, the sample was dried, and it was found that removal efficiency varies for each MP or polymer type. The maximum removal efficiencies of PE/PP, PVC, PA and polyester at optimum conditions were found to be 98.3%,80.7%, 93.5%,76.7%, respectively.

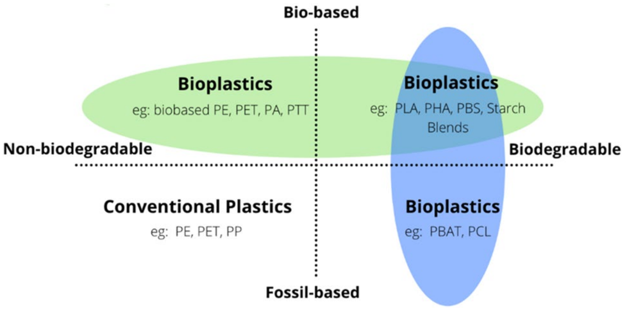

Fig. 4 Summary of different types of bioplastics

Bioplastics as an alternative

The term "bioplastics" encompasses resilient polymers that share visual and tactile similarities with traditional plastics but are sourced either from biomass or petrochemical pathways. Bioplastics manufactured from bio-based or partly bio-based routes, shown in Fig. 4, can be non-biodegradable, such as bio-based PE, PP, or PET and bio-based technical performance plastics such as polytrimethylene terephthalate (PTT). Examples of bioplastics that are both bio-based and biodegradable include polybutylene succinate (PBS), Polylactic Acid (PLA) and polyhydroxyalkanoates (PHA). Additionally, petro-chemical routes that produce biodegradable plastics such as polybutylene adipate terephthalate (PBAT) and PBS are also considered bioplastics (Spierling et al. 2018).

An example of an emerging alternative to common commodity plastics such as PE and PP, is the bio-based polymer PLA. PLA is produced from the fermentation of sugars or starch which can be manufactured into transparent and rigid polymers that closely mimic petroleum-based plastics. These are widely used in short-lived packaging such as films and in food packaging due to its permeability. A strong argument in favor of popularizing the usage of bioplastics is given by the study performed by Leblanc (2021). It was found that the benefit of using PLA bioplastics is that during composting of materials under maintained conditions, the polymers will depolymerize within about 10 days and return carbon dioxide due to decomposition after 30–40 days to the carbon cycle. Hence, not only it prevents the production and dispersal of MPs but also significantly reduces the GHGs emissions.

However, there is still a lack of in-depth toxicity studies related to BP. It is unknown how effective these materials are in terms of chemical characteristics and reduction of carcinogenic or exposure of other hazardous chemical to humans (Lambert and Wagner 2017). According to recent toxicity studies of BPs conducted by Zimmermann et al. (2019), the proportion of trials that induced toxicity were the same as that for the bio-based/biodegradable materials as is for the petrochemical-based plastics. In fact, a slightly higher percentage of BPs were found to induce a baseline toxicity when compared with the conventional plastics. However, the study also found that in terms of endocrine activity, a slightly higher percentage of conventional plastics were toxic. This study also investigated the chemical toxicity of BP products and that of its raw material. Zimmermann et al. (2019) collected data from 33 samples of end-use products and 10 samples of various types of pre-production pellets. They found that the final consumer products across all endpoints exhibit double the level of toxicity compared to that of its raw materials. Around 78% of final products versus 40% of the samples of raw materials induced a baseline toxicity, and 48% of consumer products versus 20% of the raw material samples were found inducing an oxidative stress response. Furthermore, none of the raw materials contained estrogen-like or antiandrogenic chemicals, whereas 30% of the final products were found antiandrogenic. While comparing traditional plastics with their bio-based counterparts such as PE and bio-PE, a higher number of bio-based samples were shown to induce oxidative stress. Bio-based PE was also found to inhibit the androgen receptor up to 97.4%. "Interestingly, none of the five conventional PET extracts induced relevant toxicity but one out of the two Bio-PET samples did." (Zimmermann et al. 2019). This implies that several bio-based and biodegradable materials previously thought to be much safer may not truly possess such qualities as of now. In addition, biomaterials especially used for food packaging may inherently contain toxic chemicals which can prove to be as harmful to humans as conventional plastics. However, it is possible that the presence of trace amounts of nano and microplastics existing in the samples may have influenced the authors' results.

Although a systematic approach to the assessment of toxicity remains to be developed, previous studies also reported similar findings of in-vitro toxicity for BPs. An experimental trial set up by Dang et al. (1996) similarly suggested that cellulose-based materials induce cytotoxicity in mouse fibroblasts.

Notably, various investigations into the toxicity of polylactic acid (PLA) have identified chemical leaching from diverse PLA materials employed in medical implants, causing inhibition of bacterial bioluminescence (Ramot et al. 2016). Conversely, PLA-clay nanocomposites utilized in food packaging showed no cytotoxic effects in humans (Maisanaba et al. 2014). The observed divergence in toxicity, depending on the specific product despite being composed of the same material, aligns with the outcomes of this study. For example, a coffee capsule made of PLA 5 exhibits an in vitro toxicity, whereas a single-use plastic bottle made of PLA 7 does not. Hence, it can be generalized that studies comparing different products would indicate the material-dependent toxicity since different types of chemical compositions of additives, plasticizers and other raw materials have been used (Zimmermann et al. 2019). This study revealed that the nature of BPs and other similar alternatives contain an unexpectedly high composition and variety of chemicals. They also stated that bio-based products currently available in the market do not differ in terms of chemical composition or toxicity when compared to petroleum-based plastics. This confirms that the positive connotation attached to bio-products may not necessarily convey the chemical hazards found in nature of such materials. Furthermore, when evaluating the environmental performance of BP alternatives, the central focus is only limited to either the production (e.g., reduction of carbon footprint, using renewable feedstocks) or the end-of-life stage (e.g., recycling, degradability). However, on a more positive note, six out of ten Bio-PE products did not contain toxic chemicals implying that bio-based PE formulations are available in the current market that do not contain substances that induce in vitro toxicity (Zimmermann et al. 2019). Currently, the technical performance during the usage of BPs, such as human exposure to chemicals and its toxicity, are often disregarded when evaluating the sustainability of the biomaterial (Muncke et al. 2020). It is indicated that while not all of these have impacts on human health or the environment, this highlights the challenges currently faced by researchers when aiming to assess the chemical composition and safety of plastics and other synthetic materials, especially when dealing with Food contact material. Zimmermann et al. (2019) also suggested an optimization of the chemical safety of materials using green chemistry to design out toxic characteristics during the development of new biobased or biodegradable materials. Besides human health, important aspects such as carbon, energy, water and land footprints need to be minimized to truly innovate better plastics or plastic alternatives and avoid regrettable substitutions.

The need for life cycle assessment of bioplastics

While BPs pose a glimmer of hope in the battle against plastic pollution, it is essential to acknowledge that their utilization has raised concerns regarding the potential release of MP. Several studies have indicated that BP may, in certain scenarios, pose a risk of contributing to MP pollution. Specifically, some research suggests that BPs, when exposed to environmental conditions, might not fully biodegrade as expected and instead could fragment into smaller particles, effectively becoming the source of MP. These factors are the type of BP, the environment it is exposed to, and the presence of specific microbial communities (Rosenboom et al. 2022). These findings highlight the necessity of comprehensive research into the behavior and fate of BPs, particularly in real-world conditions. It also highlights the importance of identifying specific scenarios where BPs might inadvertently exacerbate MP pollution. As such, a balanced approach to BPs is needed-one that leverages their potential benefits while remaining vigilant about the unintended consequences that could undermine our efforts to combat MP contamination. This complicated dynamic between BPs and MPs requires further investigation.

To design and develop sustainable bio-based alternatives, it is important to analyze various contributing factors such as toxicity and the direct and indirect impacts on the environment along with the lifecycle of the products. To benchmark BPs against conventional plastics, comprehensive life cycle assessments (LCA) were performed to quantify the environmental impacts that arose over the entire value chain of a product (Bishop et al. 2021). Consequently, a range of key recommendations are provided to enhance the evaluation of BP sustainability and to ensure that genuine environmental savings are achieved. (Bishop et al. 2021). While developing a model for environmental impacts, LCA studies may consider factors such as size and shape of the plastics, their degradability with respect to the environment they were disposed in, the toxic effect of the chemicals released into the environment, risks posed to wildlife by injection or entanglement and finally the polymer type and the persistence of the plastic debris (Bishop et al. 2021). Another consideration is that additives must be included in plastic LCA studies unless it is clearly noted that they contribute to under < 1% of all impact categories. There is a vital need for more studies evaluating the environmental and end-of-life impacts of additives and plasticizers. As indicated in Zimmermann et al. (2019), despite the same amount of additives and plasticizers being added to BPs and plastics, the amount of toxicity noted in the BPs is higher. Furthermore, another key recommendation suggests that since land-use is a critical aspect of BP life cycles, impacts of land-use change must be explored and accounted in the future LCA studies (Bishop et al. 2021). However, as several LCA studies expressed, the ecological damage is more controlled with the production and usage of bioplastics than of the conventional plastics (Bishop et al. 2021).

Waste management of bioplastics

According to a study conducted in 2015, only 20% of plastic waste was either reused or recycled, 20% of plastic waste was incinerated, 55% was dumped into landfills and the remaining was left unattended adding to the pollution created by plastics (Jambeck et al. 2015). However, BPs pose better waste management systems as compared to the conventional plastics and may reduce negative influences on the environment. Common end-of-life approaches for BPs are landfills, incineration, anaerobic digestion, composting, mechanical and chemical recycling. To choose

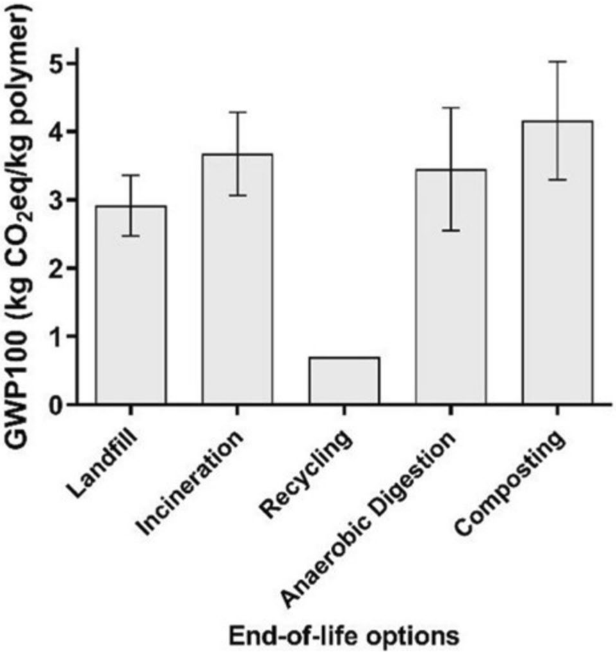

Fig. 5 Sensitivity analysis for global warming potential based on end-of-life scenarios for PLA packaging (Kakadellis and Harris 2020)

Table 3 Recyclability of plastic waste at different stages (Alhazmi et al. 2021)

Type	Description	Disadvantages
Primary	Re-extruding discarded plastic wastes from industries involves a high degree of homogeneity, hence is recycled easily	Not suitable for post-consumer plastic wastes due to non-homogeneity
Secondary	Mechanical recycling of recovered plastics from consumers, sorted and reprocessed to produce single polymer pellets or granules, to replace the virgin plastic. No alteration of chemical composition takes place	Plastic wastes must be dry and clean, ideally segregated consisting of a single polymer type
Tertiary	Complex plastic wastes undergo chemical recycling, using processes such as pyrolysis and/or hydrolysis. The polymer depolymerizes to break down into monomers and other basic chemical elements. This may be used as raw material for primary plastic production	High energy consumption. Uses chemical reagents resulting in negative environmental impacts

the best approach, it is important to evaluate the type of bio-based plastic and its characteristics. Waste disposal technologies such as landfills and composting of BPs are common end-of-life approaches due to their ease of access. However, landfilling does not add to the value chain and in fact does not facilitate PLA to break down faster than the conventional plastics. Although various recent studies clarified that PLA decomposes within 90 days, PLA requires to be properly segregated and composted in facilities with controlled temperatures up to 60 °C. The BP then degrades into carbon dioxide and water, leaving no further residue in the composter (Spierling et al. 2018). While the reuse of bio-based products is often beneficial, it is not a feasible option for short-lived packaging products. However, in some applications such as textiles, this method is prevalent for conventional and bio-based plastics on a large scale. Mechanical recycling involves the regranulation of plastics whereas chemical recycling includes treatments such as hydrolysis. Both methods are widely conducted in industry for pre-consumer recycling and BPs such as PLA. Although incineration with energy recovery is currently used for conventional plastics and some BPs, releasing of toxic gases such as furans, mercury and dioxins, the large carbon footprint, and impact on global warming caused by this process is discouraging. Hence, incineration should only be conducted if other options such as mechanical or chemical recycling are not technically or environmentally feasible (Spierling et al. 2018).

Table 3 shows the recyclability of plastic waste at different stages. Figure 5 is a sensitivity analysis for global warming potential based on end-of-life scenarios for the PLA packaging. These end-of-life options consist of landfill, incineration, anaerobic digestion, composting and the umbrella term, recycling for mechanical and chemical recycling. As shown in Fig. 5, the process of recycling is least damaging, environmentally sustainable, and hence is most preferred. However, it could be an expensive process to ensure thorough segregation and reprocessing of waste since the quality and economic value of recycled plastics heavily depend on the purity of the plastic. Chemical recycling exhibits a strong tolerance towards impurities in plastic waste such as non-plastic components or paints as the process is enabled to separate non-plastic and plastic impurities from plastic waste. However, the environmental distress caused by a large-scale pyrolysis process can be compared to that of an incineration plant. Therefore, this method should be employed only if mechanical recycling is not suitable (Alhazmi et al. 2021). Owing to the high calorific value in BP wastes obtained from various processes such as incineration, anaerobic digestion and pyrolysis, they can be used to generate energy after the complete recovery of recyclable material from the waste. The large quantities of carbon dioxide emissions of post incineration may be captured and reused to develop new bio-based products, making the process sustainable. Anaerobic digestion can also be used to manage BP waste as it can be decomposed into biogas and digestate which can be further utilized as manure. The methane released from the process may also be captured and used as fuel (Selvamurugan and Sivakumar 2019). Hence, treating bio-based plastics using appropriate end-of-life measures is vital to optimize environmental efficiency.

Conclusions

The issues presented by MPs pollution is expected to rise with current ongoing practices of waste disposal and unprecedented growth of plastic production. Resolving the significant challenge of MP pollution demands a comprehensive strategy that encompasses stages from production and consumption to the management of wastewater. Assessments of MP ingestion by aquatic organisms have revealed concerning trends. Additionally, research indicates a maximum removal efficiency of 64.4% through sedimentation, underscoring the enduring presence of MPs in untreated wastewater. These findings shed light on the substantial challenge posed by MP pollution in aquatic settings.

This issue is further exacerbated with discrepancies in sampling and analytical methods where technical inconformity further complicates any progress. The MPs that enter the aquatic and soil systems are consumed by organisms which not only damage the tissues of various wildlife, but also of humans. Due to all these reasons, MPs pose a great threat to food safety and hence there is a critical need to find sustainable alternatives to the plastics and MPs used in our daily lives. The importance of each method and technology stands out notably when considering alternatives. Notably, the use of hTC in sludge treatment showcases a significant 43.7% removal rate of MP after 45 days, surpassing cTC and demonstrating its quantitative superiority. This approach holds promise for more efficient sludge treatment, thereby aiding in reducing the environmental impact of MPs.

While biobased polymers show significant promise as a sustainable and long-term solution to the plastic problem, advanced research into the toxicity profiles and environmental impact needs to be conducted. The examination into the toxicity of both bio-based and conventional plastics emphasizes that a slightly higher percentage of bio-based plastics induce baseline toxicity, underscoring the necessity for comprehensive toxicity assessments. This insight remains crucial for evaluating potential ecological risks associated with the adoption of bioplastics. Currently, drop-in bioplastics available in the market only cater to short-lived packaging solutions as end-of-life options still require integration and optimization to truly transit from conventional plastics. Furthermore, advancements in technology and reduction of production costs on bioplastics will thereby lead to a more efficient circular economy.

Perspectives

In light of the ongoing research on microplastics and the development of sustainable alternatives, such as bioplastics, several key research gaps and future directions emerge.

(1) Understanding the Fate and Behavior of Bio-based Microplastics: As the use of bioplastics increases, it is crucial to gain a comprehensive understanding of the fate and behavior of bio-based microplastics in various environmental compartments. Further research is needed to investigate their degradation rates, potential for fragmentation, and long-term environmental impacts (Smith et al. 2018).

(2) Redefined analytical practices for MP is required. Our perspective emphasizes the urgent need to standardize analytical protocols especially for soil MP due to the additional presence of complex solid particles mixed with the MPs. The tested soil’s texture, SOM quantity and carbonate content should also be defined to allow for thorough analysis of MP recovery and interstudy comparison which would aid subsequent technologies for the analysis of MPs in soil.

(3) Assessing the Ecotoxicological Effects of Bio-based Microplastics: While we embrace BP, we advocate for rigorous assessment of their ecotoxicological impacts on aquatic and terrestrial ecosystems. Comprehensive studies should focus on determining their potential for absorption, distribution, bioaccumulation, toxicity, and sublethal effects on different trophic levels.

(4) Circular Economy Approaches for Bioplastics: As proponents of sustainability, we are resolute in our call for the exploration and deployment of circular economy models for BP. Exploring and implementing circular economy approaches for bioplastics is a key area of future work. Strategies such as closed-loop recycling systems, composting infrastructure, and the development of efficient waste management processes can contribute to achieving a more sustainable and circular plastic economy.

(5) Lifecycle Assessment and Carbon Footprint Analysis: To conduct comprehensive lifecycle assessments and carbon footprint analyses of bioplastics, we emphasize the critical need to understanding their overall environmental impact compared to conventional plastics. These assessments should consider the entire lifecycle of bioplastics, including feedstock production, manufacturing processes, use, and end-of-life scenarios.

(6) Consumer Awareness and Behavioral Change: Recognizing that the success of BP hinges on public acceptance, we endorse in-depth inquiries into consumer perceptions, preferences, and behaviours related to these innovative materials.

In conclusion, the development and utilization of bioplastics offers promising prospects for mitigating the environmental impact of conventional plastics. However, several research gaps remain in understanding fate, behavior, ecotoxicology, circular economy approaches, and consumer perspectives related to bio-based microplastics. Addressing these gaps will help informed decision-making, advance the field of bioplastics, and contribute to the development of sustainable solutions for plastic pollution.

Author contributions AS: Investigation, Methodology, Visualization, Writing—original draft. SG: Investigation, Methodology, Visualization, Writing—original draft. SS: Investigation, Methodology, Visualization, Writing—original draft. GA: Validation, Writing—review & editing. VR: Conceptualization, Supervisor, Validation, Writing—review & editing.

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