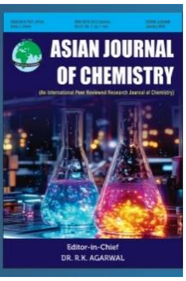
Asian Journal of Chemistry; *Vol. 37, No. 10 (2025), 2355-2365*

ASIAN JOURNAL OF CHEMISTRY

https://doi.org/10.14233/ajchem.2025.34421



REVIEW

From Pollution to Solution: Scalable Approaches to Microplastic Degradation and Sustainability

CHEVURU SAI SHREYA REDDY[1], BALAJI GANESH[1] and SIVAKAMAVALLI JEYACHANDRAN[2,*]

[1]Department of Periodontics, Saveetha Dental College and Hospitals, SIMATS, Chennai-600077, India
[2]Lab in Biotechnology and Biosignal Transduction, Department of Orthodontics, Saveetha Dental College and Hospital, SIMATS, Saveetha University, Chennai-600077, India

*Corresponding author: E-mail: sivakamavalli.sdc@saveetha.com

Received: 7 July 2025 *Accepted*: 22 August 2025 *Published online*: 30 September 2025 AJC-22118

In 21st century, it is critical for sustainable development due to natural resources depletion and the expansion of new territories, leading to increased urbanization, environmental risks and the creation of large-scale economic systems. Microplastics originating from industries, transportation and other human activities, have become a significant global aquatic pollutant, posing serious threats to marine life. Traditional methods for removing microplastics have proven ineffective due to their small size, prompting the exploration of new degradation strategies. This review highlights various approaches for microplastic degradation, including physical methods like density gradient separation, filtration systems, reverse osmosis and magnetic separation; chemical methods involving acid-alkali treatment, chemical oxidation and photocatalysis; and biological methods focusing on bioremediation through bioaugmentation and biostimulation. While these methods offer partial degradation of microplastics, often producing valuable byproducts on a small scale, they have limitations. To address these shortcomings, innovative techniques such as adsorption on algae, biochar and photocatalytic micromotors are being developed, reflecting the latest research trends in microplastic degradation.

Keywords: Sustainable development, Microplastic degradation, Marine pollution, Environmental risks.

INTRODUCTION

In current scenario, the challenges of sustainable development have become increasingly critical as the depletion of natural resources in traditional areas drives the need to explore and develop new regions. However, this expansion particularly in large-scale systems and remote territories often leads to significant environmental consequences. Among the emerging threats is microplastic pollution, one of the most pervasive and hazardous byproducts of modern development. Microplastics have been detected across a wide range of habitats globally, with aquatic ecosystems being the most affected [1,2]. Their presence highlights the unintended environmental costs associated with the rapid advancement and industrialization of new areas, underscoring the urgent need for sustainable strategies and pollution mitigation measures. The plastic particles that are < 5mm diameter are considered as microplastics. They come in a variety of materials and forms, where it is grouped into primary or secondary microplastics, depending on the source [3].

Primary microplastics are intentionally manufactured in small sizes, whereas secondary microplastics are the consequence of deterioration, weathering or the breakdown of bigger plastic into smaller particles. Secondary microplastics include textile fibres and tire wear. Industrial pellets and cosmetic microbeads are examples of primary microplastics. Their extensive industrial use has contributed to the contamination of aquatic ecosystems *via* wastewater discharge and bioaccumulation in aquatic organisms [4]. The fragmentation of macroplastic debris into microplastics increase the presence of plastic in marine environments, heightening the risk of disruption to aquatic food chains. Moreover, the accumulation of microplastic particles in the water column can reduce water transparency, limiting light penetration and affecting organisms at various levels of depths. Hence, researchers worldwide have been investing their time in formulating effective degrading methods with a strong mechanism to eliminate microplastic contamination [5]. Microplastics disintegrate through the same mechanisms that degrade macroplastics residues, due to high surface-to-volume ratio. Microplastic particles shrink much further and finally

---

become nanoplastics. The word "degradability" describes a change in the chemical structure brought on by physical, (photo-) chemical or biological interactions [5]. Degradation may take many different forms, from a single change in a molecule's structure to a series of changes that provide energy and carbon for cell development as well as the mineralization of CO_2 and H_2O. Based on a number of chemical, physical or biological conditions, the microplastics degradation may typically be differentiated into two types: abiotic and biotic [6]. Each category follows a distinct method where macroplastics degrade into smaller units of molecules (such as monomers, oligomers or other altered forms) and may even become entirely mineralized throughout the process [7]. Therefore, this review primarily focuses on the three major macroplastics degradation mechanisms in water such as physical, chemical and biological methods in detail reported so far. It also reviews the recent emerging techniques in microplastic degradation that interests and helps researchers with their innovative ideas in future for a sustainable environment.

Physical methods of removal of microplastics: Microplastics are anticipated to be more hazardous than other contaminants in aquatic settings due to their negative impacts on fish, mammals, marine birds and reptiles as a result of entrapment and bioaccumulation. Microplastics are typically thought of as persistent materials, however, depending on their nature and chemical makeup, they disintegrate more or less [8].

Removal through density separation: To isolate macroplastics from sediments, a general process is density separation. Once stirring, when compared with the medium, the less denser macroplastics might float over the suspension's top layer which causes separation from the sediments [9]. Lightweight plastic can float because the density is increased by the addition of salt. The salt most frequently suggested for density separation is NaCl. The characteristics of NaCl are its vast availability, low cost and safety for the environment. However, even though many macroplastics such as nylon, polypropylene (PP), polyethene (PE), polystyrene (PS) and polyamide, which is more commonly known as polyamide have a density of < 1.2 g/cm^3, many other macroplastics such as denser PVC and PET, which account for > 17% of the demand of world's plastic, won't float in this solution [10]. Further, the following compounds are often used for density separation: $ZnBr_2$, $CaCl_2$, NaBr, NaI and $3Na_2WO_4.9WO_3$. A standard technique was developed by Zhang *et al.* [11] using heating-assisted NaH_2PO_4 with a density that may ranging from 1.4 to 1.5 g cm^{-3} at varying temperatures of 20 to 40 ºC.

Removal through filtration: Based on the plastic size, filtration separates solid particles from a fluid and is frequently aided by a vacuum filtration system. Filtration stops macroplastics from slipping through pores in media and development of cake deposition on filters may help with the separating procedure. Based on the filters used, filtration process includes membrane and granular filtration [12]. Deposition on filters may help the separation process, during which mineralization can also occur [7].

Granular filtration: The usage of granular medium components such as glass beads, activated carbon and quartz sands, preserves solid particles during the transport and adhesion phases. Large particle transport is mostly controlled by interception and sedimentation, whereas Brownian diffusion primarily controls the movement of nanoscale macroplastics [13]. The current understanding of particle transport to collectors appears to be comprehensive, based on analyses involving various particle sizes and their deposition behavior in porous media. The flow characteristics, granular media and solution characteristics are also strongly correlated with the separation efficiency. Steric interactions, hydration forces, hydrophobic contacts, electrostatic repulsive forces and van der Waals forces are only a few of the factors that contribute to the attachment of plastic particles [14].

Membrane filtration: When a material is filtered through a membrane, the membrane acting in the form of a selective barrier, allows some molecules, ions or microscopic materials to traverse but blocking others. Membranes are classified as ultrafiltration, reverse osmosis, microfiltration and nanofiltration membranes based on pore size. Reverse osmosis mostly uses desalination, whereas microfiltration, nanofiltration and ultrafiltration are able to remove particles as big as 0.08 to 2 microns, 0.005 to 0.02 microns and 0.002 microns [15]. According to Li *et al.* [16], the dynamic membranes can effectively remove microplastics from synthetic wastewater.

Ultrafiltration: Ultrafiltration, which normally has pores that are 0.01 to 0.1 m (or 10 to 100 nm) in size, is very capable of holding particles, bacteria, viruses and proteins as well. When particle retention and organic removal are needed in water treatment, coagulation and ultrafiltration are combined [17]. When treating drinking water, uniform pore distribution membranes have been used frequently. Additional tests were carried out with ultrafiltration membranes present, where the PE completely rejects ultrafiltration membrane and forms primary fouling (cake layer) due to the particle size and bigger flocs of PE when compared to the pore size of a typical membrane [18]. Ultrafiltration capillary membranes can be used either "outside in" or "inside out". Inside-out refers to the process in which the filtrate exits the capillary from the outside while the feed stream, which contains the species to be retained, is pushed *via* the inside of capillary.

Ultrafiltration is recognized as a practical solution for water treatment due to its high separation efficiency, low energy consumption and compact plant size. It allows for the production of high-quality drinking water at an affordable cost. This low-pressure (1-10 bar) process utilizes asymmetric ultrafiltration membranes with pore sizes ranging from 1 to 100 nm, capable of removing macromolecules and particles such as fatty acids, proteins, protozoa, bacteria, viruses and other suspended solids (Figs. 1 and 2) [15]. Today, ultrafiltration combined with a coagulation stage is considered one of the primary and highly effective water treatment methods, demonstrating significant efficiency in the removal of organic matter from water. Due to the alarmingly high concentrations of microplastics in freshwater systems, it is imperative to thoroughly investigate the effectiveness of microplastic removal during coagulation and ultrafiltration processes, especially given their widespread use in producing clean drinking water.

Reverse osmosis: Reverse osmosis (RO) is utilized in industrial as well as municipal water treatment systems for purifying water through the removal of salts, heavy metals, pollutants and other toxins/impurities by utilizing nanofiltra-

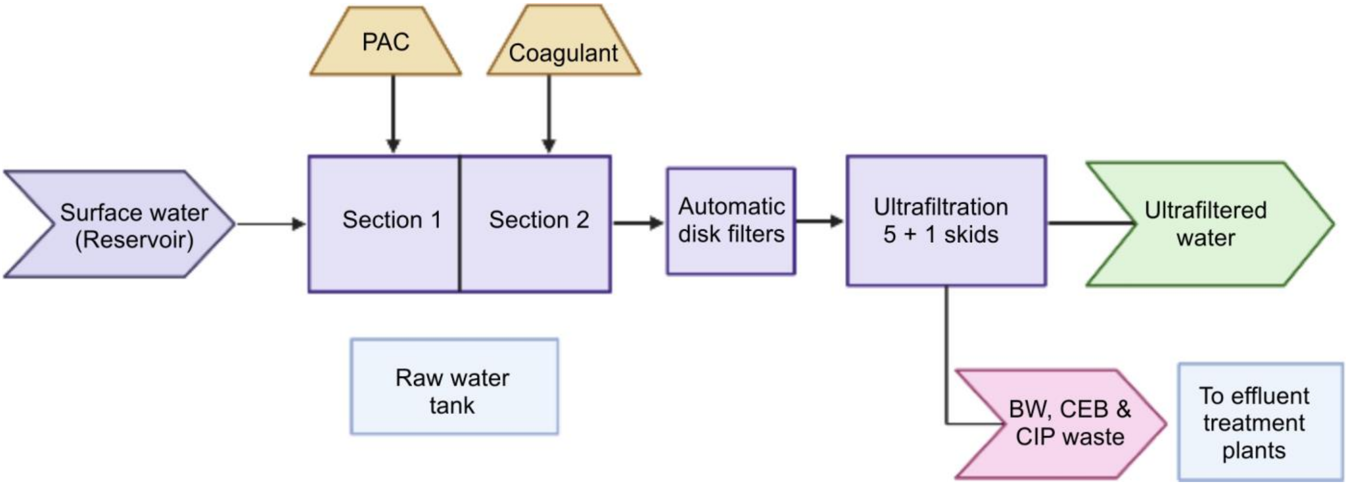


Fig. 1. Schematic process describing the ultrafiltration of microplastics

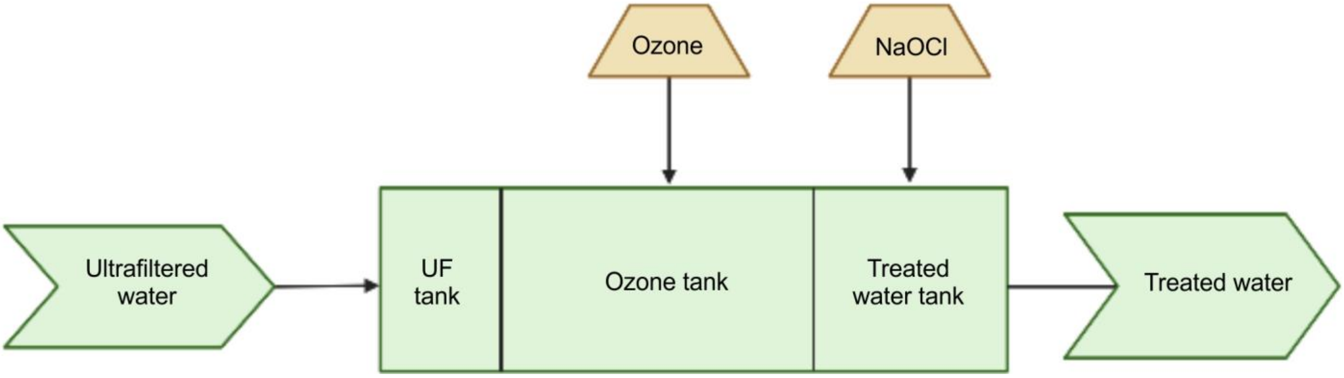


Fig. 2. Schematic process describing the ultrafiltration of microplastics

tion or non-porous membranes of pore size > 2 nm. Fouling of the RO membrane is a significant obstacle to the effective membrane performance [19]. In industrial-scale, RO desalination systems, a pre-treatment stage is necessary to regulate flux rates, manage to foul of membranes, reduce the frequency of membrane cleaning and extend the useful life of the membranes. A study assessed the performance of the RO process in removing microplastics by identifying and quantifying microplastics in samples collected from a wastewater treatment plant employing multiple advanced treatment stages including disinfection/dechlorination, flocculation, ultrafiltration, screening, sedimentation and ultimately RO. The findings revealed that despite the comprehensive treatment, microplastic fibers were still present in the samples following the reverse osmosis stage [20]. The majority of RO's more effective uses for removing microplastics are gained when it is combined with membrane bioreactor technology.

Dynamic membrane technology: Dynamic membrane technology is becoming an appealing method for municipal wastewater treatment, sludge treatment, surface water treatment, industrial wastewater treatment and oily water treatment, are all applications. In recent years, dynamic membranes have attracted significant interest due to their potential to reduce both energy consumption and operational costs in wastewater treatment processes. For removing low-density, non-biodegradable macroplastics, like plastics, that are common debris, dynamic membranes are a potential solution [16]. Since, these microparticles are difficult to remove using traditional sedimentation, downstream unit processes must spend more on operation and maintenance. According to Li *et al.* [16] dynamic membranes were produced on a 90 m long supporting mesh by filtering synthetic wastewater. The performance of dynamic membranes (DMs) has been evaluated with a focus on the effects of influent particle concentration and flow rate, both of which significantly influence membrane formation and filtration efficiency. After 20 min of filtration, the effluent's turbidity dropped to < 1 NTU, demonstrating the effectiveness of dynamic membrane in removing microparticles. The relationship between total filtration resistance and transmembrane pressure linearly increased along filtration time ($R^2 > 0.998$). TMP had a total filtration resistance ranging from 2.89×10^{-9} m^{-1} to 6.52×10^{-9} m^{-1} during the filtration of dynamic membrane and a water head range of 80 to 180 mm. This method is based on the formation of a cake layer, which acts as a secondary membrane and barrier after particles are separated by the supporting membrane. Dynamic membrane technology has gained a lot of attraction as when compared to conventional membranes, it uses cheaper materials including stainless steel mesh, woven filter cloth and non-woven fabric [21]. Since the filtering layer is made up of the pollutants of the influent, additional chemicals or other contaminants are not added.

Membrane bioreactor (MBR): In MBR systems, the separation method is related to catalysis which is fueled by biological catalysts (enzymes, bacteria) and is run through a membrane system that is majorly ultrafiltration or microfiltration [15]. Moreover, this technology's adaptability enables simple integration with other procedures (such as pervaporation and reverse osmosis) that follow the principle of green chemistry and the logic of process intensification. A study by Bayo *et al.* [22] investigated the effectiveness of MBR technology in reducing microplastic concentrations in the final effluent of an urban wastewater treatment plant over an 18-month observation period. For the influent, the average microplastic content was 4.40 ± 1.01 MP L^{-1}, whereas for the MBR, it was 0.92 ± 0.21 MP L^{-1}. Films (0.59 0.24 ± items L^{-1}), fibres (1.34 ± 0.23 items L^{-1}), beads (0.02 ± 0.01 items L^{-1}) and fragments (0.20 ± 0.09 things L^{-1}) were the predominant microplastic types identified.

Magnetic separation: To separate microparticles grafting by magnetic seed particles, magnetic fields can be used. Due to the wide-range magnetic force, this technique has the benefits of having an improved separation, reduced waste sludge and huge capacity. Magnetic extraction made up of three steps: creating magnetic seeds, boosting magnetic seeding-induced aggregation and increasing separation efficacy. As magnetic seed particles, hexadecyltrimethoxysilane-modified hydrophobic iron nanoparticles were employed. Hydrophobic (Fe) nanoparticles are attached to the surface of microparticles due to hydrophobic interaction, allowing the microparticles magnetic separation [23]. Shi *et al.* [24] used magnetic Fe_3O_4 NPs in microplastic infested water to remove microparticles. An optimal concentration of Fe_3O_4 NPs and a specific magnetization treatment time were identified. The magnetization process resulted in significant adsorption of Fe_3O_4 NPs onto the microplastics surface. This approach demonstrated the potential to effectively remove microplastics of various shapes and sizes from both seawater and pure water. The removal efficiency varied depending on the polymer type and size of the microplastics, showing a positive correlation with the density of Fe_3O_4 NPs distributed on the surfaces of microplastics following magnetization treatment. Bar magnets were employed to extract plastic debris from water samples [25]. Although the static magnetic fields are effective for separating small volumes in wastewater purification and environmental remediation, high-gradient magnetic separators (HGMS) are preferred for high-throughput systems due to the need to treat large volumes of water. In testing solution and wastewater, Tang *et al.* [26] developed magnetic carbon nanotubes that demonstrated significant attraction to all common microplastics surfaces. The removal of the microplastics/MCNT agglomerates may be completed in 300 min by using magnetic forces. Thermal processing at 600 ºC can recycle spent MCNTs and the recycled ones can still be utilized to remove around 80% of the microplastics. These characteristics of MCNTs show that they may be employed as an effective, affordable and environmentally acceptable material to remove microplastics during wastewater treatment and water environment restoration.

Oil film separation: Oil film separation of microplastics based on the hydrophobicity approach is independent of density effects. In order to remove microplastics from the aquatic sediments, Lechthaler *et al.* [27] developed an oil extraction technique and achieved a recovery of over 96% using canola oil. Castor oil was utilized to remove microplastics from the soil and aquatic samples, with an average matrix reduction of 95%, the mean spike recovery of microplastics was found to be 99%. This development could result in a breakthrough in systematic uniformity and hasten microplastic eradication tactics. In order to separate microplastics from compost samples and soils which contained PU, PC, PVC, PET, PS and PE, an olive oil-based technique has also been developed [28]. By freezing samples in polytetrafluoroethylene cylinders at a low temperature, the oil film separation was adjusted and the microplastics recovery from fabricated samples was above 90%. Solid samples might readily impede separatory funnels during separation. Before spinning the oil and environmental matrix, the oil film separation calls for further digestion for samples with high biogenic concentrations. In order to improve particle density, initially buoyant plastic particles may be bound to other denser substances such as oil/petroleum film with adherent sand grains, biofilms or faecal pellets. If plastic is denser than saltwater, once the initial buoyancy is removed, it will sink. The remaining oil traces can interfere with oil film separation and require further cleaning with hexane and ethanol.

Froth floatation: Surface wettability is a fundamental aspect of froth flotation, where hydrophobic substances tend to attach to air bubbles and rise as froth, while the hydrophilic materials are carried downwards as underflow. Post-consumer plastic typically exhibits a low-energy hydrophobic surface and is susceptible to bubble entrainment. Hydrophilic materials remain dissolved in the liquid phase, while the hydrophobic materials, aided by attached air bubbles, rise to the surface and form a froth that can be skimmed off. To separate two buoyant plastic materials in a specific liquid phase, a wetting agent can be introduced. This agent selectively adsorbs to one polymer while avoiding the other, enabling their separation [29]. As a result, the wetting chemical preferentially adheres to the superficial face of a particular plastic, which makes it hydrophilic and acts as a flotation depressant. As a result of reduced flotation, hydrophilic plastic sinks to the bottom whereas hydrophobic plastic will continue to float. The plastic which floats can be collected from the mixture's surface. The critical surface tension and the surface free energy of microplastics, indicating at which the surface tension of solid microplastics get entirely wet by the liquid, are only two of the numerous variables that must be taken into consideration throughout the froth flotation process. Bolto & Xie [30] explored the use of functionalized polymers in sewage effluent treatment *via* flotation, demonstrating its effectiveness in removing grease, oil, surfactants and algae. Moreover, proposed froth flotation as a targeted method for microplastic removal, this approach aims to reduce the burden on treatment units, improve the efficiency of wastewater treatment plants (WWTPs) and minimize the release of microplastics into effluent.

Applying the modern final-stage wastewater treatment methods, Talvitie *et al.* [31] were able to remove microplastics from WWTPs with dissolved air flotation (DAF), which reduced microplastics by 95%. Despite the fact that plastic waste disintegration is gradual, when plastic trash is exposed to heat and UV light in aquaculture settings, several functional groups of hydrophilic nature can be identified. It may render froth flotation by eliminating the initial hydrophobicity of microplastics. Microplastics can aggregate significant amounts of plasticizers, heavy metals, antibiotics and other organic pollutants due to its surface hydrophobicity and degradation. By studying the degrading process of the synthetic polymers, one can assess the floatability of microplastics. The buoyancy, wettability and transport of microplastics in coastal water are expected to be impacted by surface fouling by biofilm [32]. Bioimpact fouling on plastic's capacity to float, however, is still developing. The abundance and size of microparticles are of utmost importance in order to set an appropriate size of the bubble and air capacity that needs strong carrying capacity and adhesion efficiency for the microplastic debris.

Chemical degradation of microplastics: Microplastics generally persist in the sludge after being exposed to the degradation through different physical treatments, including multistage wastewater treatment procedures. This sludge rich in microplastics is frequently treated and either dumped or utilized as fertilizer for agricultural land. Even when toxic materials are removed through various treatment processes prior to agricultural use, the presence of microplastics is often overlooked. However, due to the absence of effective strategies to recycle or completely eliminate the collected and separated microplastic particles and debris, the physical degradation-based solutions outlined above unfortunately fail to provide a permanent solution to environmental microplastic pollution [33]. As physical degradation of microplastics often leads to their re-entry into the environment, the chemical methods have been developed to achieve permanent destruction. In these processes, catalysts play a vital role by generating reactive oxygen species (ROS), which drive the breakdown of microplastics [34]. The chemical degradation of microplastics varies depending on several factors such as additives presence, polymer type, depositional environment and the surrounding medium. For instance, microplastics exposed on beach surfaces are more likely to absorb higher levels of UV radiation compared to those buried in benthic sediments or suspended deep in the water column [35]. Studies have shown that microplastics degrade more rapidly in saltwater or simulated seawater than in freshwater due to differences in salinity, alkalinity and biological colonization between the two environments. Density separation is one of the traditional chemical methods used to extract microplastics from materials such as beach sands and sediments. This technique involves floating microplastic particles while denser materials settle in high-density solutions [36]. Various solutions including sodium polytungstate, sodium iodide and hypersaline sodium chloride, have been used to degrade microplastic polymers. However, solutions like NaCl are less effective for higher-density polymers and contamination by organic or inorganic substances can alter the density of microplastic particles, highlighting the need for more consistent degradation methods. The chemical processes involving oxidation, acidic or alkaline conditions have been developed to break down natural organic matter, leaving behind more resilient microplastic particles and minimal non-digestible residues [37].

Acid and alkali treatment: According to a chemical treatment strategy for the breakdown of the plastic polymeric chain, four exterior coffee capsule samples composed of were subjected to alkali and acid treatment. To degrade the samples, chemicals such as sulphuric acid and sodium hydroxide were added. For 86 days, two specimens underwent conditioning in 35% (v/v) conc. H_2SO_4 and two in 35% (v/v) conc. NaOH. As a result, no significant changes were observed on the surface of the PP material under NaOH treatment conditions. According to the results, H_2SO_4 exposure causes the container of coffee capsules to have sticky lamination. Instead of damaging the surface of the PP polymers, the acid solution used in this treatment has harmed the adhesive bond. Thus, polymer degradation has shown enhanced activity and is not significantly affected by the use of NaOH and H_2SO_4 as additive materials [38]. Alkaline settings shown the ability to digest organic substances when fish digestive tracts were digested using 10% KOH over a two to three-week period in an initial work, although treatment effects on plastic particles were not assessed. In another study, Cole *et al.* [39] evaluated the effectiveness of NaOH compared to acidic and enzyme-based treatments, observed that 10 M NaOH solution provided the most effective alkaline conditions for degrading PVC, nylon and polyethylene particles. Similarly, Enders *et al.* [40] tested a 4:1 mixture of HNO_3 and $HClO_4$, which resulted in the degradation of polymers such as polyamide, polyurethane, black tire rubber elastomer and to a lesser extent, PMMA, PVC and acrylonitrile butadiene styrene (ABS). Furthermore, Collard *et al.* [41] successfully digested fish stomach contents using $NaClO_4$ and HNO_3 rinses combined with ultrasonication, although this method caused a mass loss of 25% in PVC material. Although studied in digestions, HCl was determined to be the least efficient way to reduce biogenic material microplastics. These reports assess the effects of several types of microplastic particles from the marine environment being digested under acidic and alkaline conditions [42].

Chemical oxidation: The chemical oxidation process breaks down polymer microparticles by disrupting their chemical bonds, converting them into smaller molecules. These molecules can further processed into useful chemicals or completely mineralized into CO_2 and H_2O. Any monomer in the polymer can have an irregular polymer chain break, which will eventually cause microplastics to break down into commercially viable organic or inorganic compounds. Environmental conditions play a crucial role in influencing the types of polymers present, which in turn determine the degradation mechanisms and the resulting byproducts [43]. According to those variables, the oxidation-based degradation of microplastics includes electrochemical, photocatalytic and direct photodegradation. The advanced oxidation process (AOP), a highly effective chemical treatment method, has shown remarkable efficiency in degrading persistent organic pollutants in water systems. This is achieved through the generation of ROS, such as hydroxyl radicals in Fenton reactions and sulfate radicals [44]. Polymer oxidation is primarily believed to occur through photodegradation. In case of microplastics, extended exposure to sunlight, particularly UV radiation, can lead to the formation of free radicals, which pose environmental risks. This process can also cause oxygen incorporation, conformational changes or cross-linking of polymer chains and physical alterations such as flaking and cracking. The most crucial element impacting this process was thought to be UV radiation [45]. The photo-oxidation process decomposes microplastics at a slow rate, particularly in natural aquatic environments. Among these, the removal of PS microplastics is a critical research priority, as PS is one of the most commonly detected plastics in marine ecosystems. Zhu *et al.* [46] conducted a study using PS microplastics as a model to investigate the aging process in an aquatic environment under simulated sunlight (295 to 2500 nm) over a period of 150 days. Their findings revealed that PS microplastics underwent photoaging under controlled experimental conditions, shedding light on the mechanisms of ROS formation and photodegradation of microplastics under simulated sunlight.

Chemical weathering: In environments where processes like abrasion, thermal oxidation, biodegradation and photo-oxidation are limited, such as on ocean floor or in landfills, PET and other aromatic polyesters can undergo hydrolysis. This process breaks down the polymer chains into shorter segments, producing ethylene glycol and terephthalic acid as byproducts. Such process defines the term chemical weathering and is a significant process in microplastics degradation [47]. Both chemical and surface evidence prove the chemical weathering process to be significant. Upon chemical weathering, the physical characteristics of microplastics fluctuate with time, causing colour changes, mechanical property loss that causes surface fissures and microcracks and molecular weight fluctuations [48]. Cracks often referred to as crazing and discolouration can be seen under a microscope and frequently with the human eye.

FTIR spectroscopy generates spectra that serve as the most reliable indicators of compositional changes in polymers caused by chemical weathering. This technique is particularly effective in identifying functional groups such as hydroperoxides and carbonyls, which play key roles in decomposition pathways during oxidative weathering. Naik *et al.* [49] used FTIR to study the impact of accelerated photodegradation on nylon 6, HDPE, PP and high-impact PS pellets in simulated saltwater environments. The spectra of weathered and unweathered pellets were analyzed and compared following chemical weathering. Evidence of chemical changes was observed through various vibrational modes: methyl carbon stretching and C-H stretching in nylon, C=C stretching, C-H bending and stretching in HDPE, O-H and C-H stretching in high-impact PS, as well as O-H and C=O stretching in PP.

Chemical weathering of microplastics can be effectively complemented by coagulation and flocculation processes, employing inorganic and organic coagulants such as $FeCl_3$, PAC and polyamines. By enhancing the chemical and physical heterogeneity of the plastic surface, weathering microplastics enhanced the possibility of surface interactions with flocculants and coagulants. The removal of weathered microplastics can be linked to the formation of new coagulant anchoring sites on the PE material due to weathering, such as -COOH, -OH and -C=C- groups. Perren *et al.* [50] studied the degradation of microplastics through electrocoagulation and settling processes. Their findings, along with other studies, demonstrate degradation rates exceeding 90% after treatment with $FeCl_3$, $Fe_2(SO_4)_3$ or electrocoagulation using iron electrodes. Furthermore, recent research on the effectiveness of iron and aluminum coagulants in removing microplastics from an artificial matrix of drinking water revealed maximum removal rates from 17% to 36% for treatments involving $AlCl_3$ and $FeCl_3$, respectively [51].

Thus, based on these studies, the chemical treatment facilitates the efficient degradation of microplastics by employing chemical reagents as catalysts to accelerate the reaction time and initiate the initial stage of polymer breakdown. This process can precede biological treatment or serve as a means to produce secondary plasticizers. However, the chemical treatment is not suitable for large-scale applications due to its reliance on multiple chemical additives.

Biological degradation of microplastics: Biological method of degradation, often known as bioremediation for microplastic degradation is an evolving technique for large-scale administration. Bioremediation is a cost-effective, least toxic and efficient approach for removing hazardous pollutants from the environment using biological systems and their products. Microbes are the most adaptive creatures to have ever lived on earth. They are well adapted to the environment and its stresses and some of them can utilize complex polymers as a carbon source to biodegrade contaminants. However, the negative impacts of microplastic contamination can damage the quality of life of different living creatures, including humans [52]. The organisms occupying the higher trophic level in the food pyramid are affected the most. Typically carrying a negative surface charge, microplastics can attract and bind positively charged contaminants. As their size decreases, their mobility increases, enhancing their ability to interact with both organic and inorganic pollutants in the environment. This not only facilitates the transport of these pollutants but also promotes their bioaccumulation, compounding the ecological risks across the food web [53].

In environment, the microplastics presence causes the adaptation of microbes to environmental stress in many ways, including an increase or reduction in the metabolic rate and growth rate and the biosynthesis of novel chemicals to combat stress [54]. Up to 2.32×10^9 microplastic particles can be relea-sed daily by wastewater that has not undergone standard treatment. The microbial population that lives in such effluent develops an adaptive factor as a result of the microplastics presence in it [55].

Polychlorinated biphenyls (PCBs) are among the most persistent xenobiotic contaminants due to their low reactivity and high chemical stability under various environmental conditions. Their tendency to accumulate in animal fatty tissues leads to the biomagnification through the food chain [56]. Biotic and abiotic stress responses, including changes in gene expression and enzymatic activity, are key to the degradation of such contaminants. Microorganisms can enzymatically break down plastic polymers into monomers, which are then utilized as energy and carbon sources [57]. Specifically, the enzyme biphenyl dioxygenase (BphA), produced by several microbial groups, initiates PCB degradation by oxidizing them into *cis*-dihydrodiol intermediates, which are further cleaved into chlorobenzoates highlighting the crucial role of microbes in PCB biodegradation [58].

Bioremediation generally involves two main approaches (i) bioaugmentation, which is the introduction of cultured microorganisms into a contaminated environment to enhance pollutant degradation and (ii) biostimulation, which involves modifying environmental conditions to stimulate the activity of native microbial communities capable of bioremediation [59]. Despite being a promising approach for microplastics degradation, bioremediation has several limitations, including slow degradation rates, incomplete mineralization and largely unexplored mechanisms of degradation and biotransformation, which can lead to the persistence of organic pollutants. The technique remains in its early stages and the key challenges include the potential negative interactions between introduced microbial strains and native soil microbiota, as well as the inability of foreign microbes to adapt to local environmental conditions. These issues can be addressed by developing microbial consortia composed of carefully selected native strains, minimizing ecological disruption [60]. Bioaugmentation may fail as a result of cell stress, nutrient deficiency, the presence of a more preferred carbon source, competition, or insufficient biological availability.

The major steps involved in the microbial breakdown of microplastics include biodeterioration (changes in shape, size and chemical composition of polymer), biosynthesis, biofragmentation and mineralization [61]. Based on the high surface-to-volume ratio of microplastics, there is a lot of surface area available for microbial colonization. Various bacterial taxa may develop biofilm on the plastic fragments surface through extracellular polysaccharide production. Biofilm production eventually leads to a size decrease and full degradation [54]. The microbial degradation of plastic is influenced by various factors, including enzymatic processes, co-substrate and substrate concentrations, oxidative stress, temperature, bioavailability, density, functional groups, pH and other abiotic and biotic variables [57]. A major limitation of culture-dependent methods is that over 99% of microbes in natural environments are either extremely difficult to culture or uncultivable. However, advancements in technologies such as next-generation sequencing, metabolic and protein profiling and enhanced bioinformatics tool have significantly improved the understanding of microbial communities and the mechanisms involved in the bioremediation of environmental pollutants.

These omics methods include metabolomics, metagenomics, meta-transcriptomics and meta-proteomics. Furthermore, advancements in technology have significantly enhanced the evaluation and application of microbiological bioremediation methods. Omics techniques have enabled the identification of novel bacteria that would not have been detected through traditional culturing methods. The effectiveness of bioremediation is expected to improve with the proper and professional application of precise molecular approaches. Developing effective bioremediation strategies requires a comprehensive understanding of the specific microorganisms present in contaminated environments, along with their metabolic pathways and adaptive mechanisms. Even though there is still much work to be done before the bioremediation science of is completed, it currently seems to be feasible. Their application is frequently constrained by a lack of knowledge of the factors that govern and control the development, metabolism and dynamics of different microbial communities in contaminated environments [54]. Despite limited comprehensive understanding of bioremediation, a substantial number of studies have been reported. Table-1 highlights a selection of these, along with the microorganisms involved in degrading various types of microplastics.

Emerging techniques for microplastic degradation

Removal through adsorption on algae: Studies showed that the microalgae can colonize microplastic particles, altering the buoyancy of the resulting aggregates. This interaction indicates the potential of microalgae as a viable approach for microplastic removal. As a result, the sedimentation rates differ from those of unaggregated particles. The impact of microplastics on microalgae growth varies depending on microplastic size and specific characteristics of the microalgae species, such as cell wall structure and mobility [91]. The possibility exists for freshwater microalgae to colonize microplastics and create extracellular polymeric substances (EPS) that can aggregate (adhere and incorporate) microplastics. For example, *Gloeocapsa* sp. has proven to be the most effective microalga for EPS production and the subsequent aggregation of microplastics across various types, size fractions and densities. In experiments with *Scenedesmus abundans* against three types of plastics, PMMA, PS and PLA, an average removal efficiency (η) exceeding 84% was achieved for all microplastics. Sundbæk *et al.* [92] used *Fucus vesiculosus*, a marine seaweed, to explore the potential of microplastic removal through translocation into algal tissues. The study revealed that the narrow channels within the algal cells restrict the mobility of microplastics, leading to their capture. Notably, the dissected regions of the algae demonstrated an efficiency of upto 94.5% in capturing microplastics [93]. The release of anionic polysaccharide compounds from these dissected areas enhances the adhesion of plastic particles. Moreover, the presence of anionic polysaccharides in the algal cell wall promotes the sorption of positively charged microplastic particles onto the algae. The surface charge of the particles has substantial impact on the absorption of microplastics on algae [94]. Positively charged microplastics are more likely to bind to other materials.

Photocatalytic micromotors: Micromotors are tiny self-propelled machines that have caught the attention of researchers all over the world due to their use in the removal of metalloids, oil metals and different organic materials. Due to their intriguing physics and anticipated uses in various sectors, such as biomedical and environmental applications, micromotors, self-propelled nano-/microscale devices, have received a lot of interest in the past 10 years [94]. Micromotors, in particular, have enormous potential for restoring the environment, including organic deterioration. A quick on/off switch is another benefit of photocatalytic micromotors, which can also run on water as a harmless light and fuel as sustainable energy source [95]. For the removal of microplastic, Liu *et al.* [96] used a micromotor based on photocatalytic TiO_2 ($Au@mag@TiO_2$, mag = Ni, Fe). Due to the photocatalytic reaction taking place over the particles, this catalytic micromotor can move effectively in water as well as peroxide when exposed to UV light. Phoretic interactions brought on by these individual micromotors' photocatalytic activity exhibit collective features with passive particles in minimal peroxide concentrations. To address the challenge of peroxide-free

TABLE-1
DIFFERENT MICROORGANISM INVOLVED IN THE DEGRADATION OF MICROPLASTIC COMPOUNDS

Organism	Type of plastic	Ref.
Bacillus paramycoides (Ac: MT995121)	Polyethylene, polypropylene and polystyrene	[62]
Rhodococcus sp. IN306 (Ac: KX058399)	Polychlorinated biphenyls (PCBs)	[63]
Mycolicobacterium frederiksbergense IN53 (Ac: JN572675)	Total petroleum hydrocarbons (TPH)	[64]
Achromobacter xylosoxidans	Polychlorinated biphenyls (PCBs), high density polyethylene	[65]
Acinetobacter sp.	Polychlorinated biphenyls (PCBs), polyethylene	[66]
Burkholderia xenovorans LB400	Polychlorinated biphenyls (PCBs)	[67]
Janibacter sp. MS3-02	Polychlorinated biphenyls (PCBs)	[68]
Mycobacterium sp. PYR-1	Polychlorinated biphenyls (PCBs)	[69]
Ochrobactrum anthropi	Polychlorinated biphenyls (PCBs)	[70]
Pseudomonas stutzeri	Polychlorinated biphenyls (PCBs)	[71]
Rhodococcus ruber	Polychlorinated biphenyls (PCBs)	[72]
Stenotrophomonas panacihumi	Polypropylene	[73]
Ideonella sakaiensis	Polyethylene terephthalate	[73]
Paenibacillus alvei	Polypropylene (PP)	[74]
Bacillus cereus	Polyethylene (PE), polyethylene terepthalate (PET), polypropylene (PP) and polystyrene (PS)	[75]
Bacillus gottheilii	Polyethylene (PE), polyethylene terepthalate (PET), polypropylene (PP) and polystyrene (PS)	[75]
Aspergillus flavus	Polyethylene	[76]
Aspergillus tubingensis	HDPE	[77]
Zalerion maritimum	Polyethylene	[78]
Gloeophyllum trabeum	Polystyrene sulfonate	[79]
Rhizopus oryzae	Polyethylene	[80]
Curvularia lunata	LDPE	[81]
Penicillium pinophilum	LDPE	[82]
Galleria mellonella	HDPE	[83]
Yarrowia lipolytica	HDPE	[83]
Alicaligens faecalis	Polyethylene	[84]
Bacillus sp. JY14	Polyhydroxybutyrate	[85]
Mycolicibacterium phocaicum	Di-(2-ethylhexyl) phthalate (DEHP)	[86]
Alcanivorax borkumensis	LPDE, PET, PS	[87]
Brevibacillus borstelensis	Polyethylene	[88]
Pseudomonas aeruginosa	Polyethylene	[89]
Phanerochaete chrysosporium	PVC	[90]

microplastic cleanup, they also formed constructed chains by linking individual particles in the presence of an external magnetic field.

Removal by biochar: The production of biochar is an established and well-known method. According to Wang *et al.* [97], biochar has shown promise as a porous material for microbead removal. They also predicted that its possible integration into quicksand filters could offer a technologically and economically viable way to increase the capacity of WWTPs. Biochar filters have a substantially greater capacity than equivalent grain-sized sand filters investigated for the separation and immobilization of 10 m diameter microplastic spheres (over 95%). With a high economic added value, biochar, a carbonaceous substance, is frequently used in agriculture to enhance the soil's characteristics [98]. It has been demonstrated that substituting biochar for activated carbon can substantially lower process costs while retaining a comparable degree of removal efficiency in WWTPs. The pores in the biochar's microstructure range in size from 50 m to less than 1 nm. An appropriate porous structure for removing turbidity from water is provided by the complex and multiscale structure [99]. Siipola *et al.* [100] developed a cost-effective method for producing porous biochar adsorbents using bark-containing side streams from the wood refinery industry, employing steam activation in the process. Biochars activated through steam were found to be highly effective sorbents for the chemical species studied and for microplastic removal, despite having relatively modest surface areas (200-600 m^2/g). The bigger microplastic particles were successfully removed by the activated biochars.

Wang *et al.* [101] proposed effective magnetic biochar adsorbents modified with Mg and Zn for microplastic removal. The removal efficiencies for Zn-modified magnetic biochar (Zn-MBC), Mg-modified magnetic biochar (Mg-MBC) and unmodified magnetic biochar (MBC) were 99.46%, 98.75%, and 94.81%, respectively, for PS microspheres (1 µm, 100 mg/mL) in aqueous solution. However, the presence of $H_2PO_4^-$ and organic matter in real water significantly reduced the removal efficiency of Zn-MBC due to competitive adsorption. Thermal treatment enabled simultaneous regeneration of the adsorbent and degradation of microplastics. Owing to their low cost, environmental compatibility and durability, Mg/Zn-modified biochars show strong potential for practical microplastic remediation. In a related study, Ye *et al.* [102] developed porous, graphitized magnetic biochar designed to remove metals associated with microplastics. Their analysis revealed that prolonged weathering forms an organic layer on microplastic surfaces, which serves as a key site for metal sorption, with peeling and disintegration being critical steps in the process.

Conclusion

To address the growing threat of widespread microplastic pollution in the oceans largely driven by anthropogenic activities, an urgent priority is to investigate the mechanisms underlying microplastic degradation. Understanding these processes is critical for developing effective strategies and tools that support global sustainability efforts. In the environment, microplastics degrade through physical, chemical, and biological pathways, each influenced by a combination of internal properties and external conditions. Key factors affecting the rate and extent of degradation include the chemical composition, structure and additives of the plastics, as well as environmental variables such as humidity, temperature and the type of surrounding medium, be it soil, sand or water. In aquatic ecosystems, additional elements like sunlight exposure, particle depth and mechanical abrasion play significant roles in shaping degradation behaviour. Although each degradation pathway can independently reduce microplastics into nanoplastics, monomers or oligomers, synergistic interactions between multiple processes are often more effective in accelerating breakdown. Therefore, a comprehensive understanding of these degradation mechanisms, especially under simulated environmental conditions, is essential. This review aims to explore these processes in depth to inform and advance innovative solutions for mitigating microplastic pollution.

ACKNOWLEDGEMENTS

The authors are thankful to the authority of the Saveetha Dental College and Hospital, Saveetha Institute of Medical and Technical Sciences (SIMATS), Saveetha University, Chennai, India for providing necessary facilities and support. This study has been supported by Science & Engineering Research Board (SERB-SPG) research grant (File no. SPG/2021/003904).

CONFLICT OF INTEREST

The authors declare that there is no conflict of interests regarding the publication of this article.

REFERENCES

1. M. N. Issac and B. Kandasubramanian, *Environ. Sci. Pollut. Res. Int.*, **28**, 19544 (2021); https://doi.org/10.1007/s11356-021-13184-2
2. S. Khurana, S. Ali, A.K. Srivastava, A. Singh, H. Agarwal, R. Chauhan, N.C. Joshi, L. Dufossé and A. Chauhan, *J. Hazard. Mater. Adv.*, **19**, 100777 (2025); https://doi.org/10.1016/j.hazadv.2025.100777
3. A.L. Andrady, *Mar. Pollut. Bull.*, **119**, 12 (2017); https://doi.org/10.1016/j.marpolbul.2017.01.082
4. S. Sharma, S. Basu, N.P. Shetti, M.N. Nadagouda and T.M. Aminabhavi, *Chem. Eng. J.*, **408**, 127317 (2021); https://doi.org/10.1016/j.cej.2020.127317
5. R.H. Waring, R.M. Harris and S.C. Mitchell, *Maturitas*, **115**, 64 (2018); https://doi.org/10.1016/j.maturitas.2018.06.010
6. J. Yuan, J. Ma, Y. Sun, T. Zhou, Y. Zhao and F. Yu, *Sci. Total Environ.*, **715**, 136968 (2020); https://doi.org/10.1016/j.scitotenv.2020.136968
7. L. Hou, D. Kumar, C.G. Yoo, I. Gitsov and E.L.-W. Majumder, *Chem. Eng. J.*, **406**, 126715 (2021); https://doi.org/10.1016/j.cej.2020.126715
8. A. Lusher, P. Hollman and J. Mendoza-Hill, Microplastics in Fisheries and Aquaculture: Status of Knowledge pn Their Occurrence and Implications for Aquatic Organisms and Food Safety, Fisheries and Aquaculture Technical Paper No., 615 FAO, Rome (2017).
9. Y. Zhang, H. Jiang, K. Bian, H. Wang and C. Wang, *J. Environ. Chem. Eng.*, **9**, 105463 (2021); https://doi.org/10.1016/j.jece.2021.105463
10. S. Lechthaler, K. Waldschläger, G. Stauch and H. Schüttrumpf, *Environments*, **7**, 73 (2020); https://doi.org/10.3390/environments7100073ss
11. W. Zhang, K. Song, R. Ding, H. Han, L. Yao, M. Ji, Z. Chen, H. Yu, C. Wu and T. Fang, *J. Hazard. Mater.*, **419**, 126429 (2021); https://doi.org/10.1016/j.jhazmat.2021.126429
12. C.G. Bannick, R. Szewzyk, M. Ricking, S. Schniegler, N. Obermaier, A.K. Barthel, K. Altmann, P. Eisentraut and U. Braun, *Water Res.*, **149**, 650 (2019); https://doi.org/10.1016/j.watres.2018.10.045
13. Q. Abbas, B. Yousaf, Amina, M.U. Ali, M.A.M. Munir, A. El-Naggar, J. Rinklebe and M. Naushad, *Environ. Int.*, **138**, 105646 (2020); https://doi.org/10.1016/j.envint.2020.105646
14. S. Farrokhpay, *Adv. Colloid Interface Sci.*, **151**, 24 (2009); https://doi.org/10.1016/j.cis.2009.07.004
15. T. Poerio, E. Piacentini and R. Mazzei, *Molecules*, **24**, 4148 (2019); https://doi.org/10.3390/molecules24224148
16. L. Li, G. Xu, H. Yu and J. Xing, *Sci. Total Environ.*, **627**, 332 (2018); https://doi.org/10.1016/j.scitotenv.2018.01.239
17. X. Yue, Y. Yang, X. Li, J. Ren, Z. Zhou, Y. Zhang and H. Yu, *J. Environ. Chem. Eng.*, **9**, 106803 (2021); https://doi.org/10.1016/j.jece.2021.106803
18. B. Ma, W. Xue, Y. Ding, C. Hu, H. Liu and J. Qu, *J. Environ. Sci.*, **78**, 267 (2019); https://doi.org/10.1016/j.jes.2018.10.006
19. M. Asadollahi, D. Bastani and S.A. Musavi, *Desalination*, **420**, 330 (2017); https://doi.org/10.1016/j.desal.2017.05.027
20. S. Ziajahromi, P.A. Neale, L. Rintoul and F.D.L. Leusch, *Water Res.*, **112**, 93 (2017); https://doi.org/10.1016/j.watres.2017.01.042
21. Y. Zhang, Y. Zhao, H. Chu, B. Dong and X. Zhou, *Chin. Sci. Bull.*, **59**, 247 (2014); https://doi.org/10.1007/s11434-013-0048-x
22. J. Bayo, J. López-Castellanos and S. Olmos, *Mar. Pollut. Bull.*, **156**, 111211 (2020); https://doi.org/10.1016/j.marpolbul.2020.111211
23. J. Grbic, B. Nguyen, E. Guo, J.B. You, D. Sinton and C.M. Rochman, *Environ. Sci. Technol. Lett.*, **6**, 68 (2019); https://doi.org/10.1021/acs.estlett.8b00671
24. X. Shi, X. Zhang, W. Gao, Y. Zhang and D. He, *Sci. Total Environ.*, **802**, 149838 (2022); https://doi.org/10.1016/j.scitotenv.2021.149838
25. L.M.A. Martin, J. Sheng, P.V. Zimba, L. Zhu, O.O. Fadare, C. Haley, M. Wang, T.D. Phillips, J. Conkle and W. Xu, *Nanomaterials*, **12**, 2348 (2022); https://doi.org/10.3390/nano12142348
26. Y. Tang, S. Zhang, Y. Su, D. Wu, Y. Zhao and B. Xie, *Chem. Eng. J.*, **406**, 126804 (2021); https://doi.org/10.1016/j.cej.2020.126804
27. S. Lechthaler, L. Hildebrandt, G. Stauch, and H. Schüttrumpf, *Anal. Methods*, **12**, 5128 (2020); https://doi.org/10.1039/D0AY01574A
28. B. Zhou, J. Wang, H. Zhang, H. Shi, Y. Fei, S. Huang, Y. Tong, D. Wen, Y. Luo and D. Barceló, *J. Hazard. Mater.*, **388**, 121814 (2020); https://doi.org/10.1016/j.jhazmat.2019.121814
29. S. Saisinchai, *Eng. J.*, **18**, 45 (2014); https://doi.org/10.4186/ej.2014.18.1.45
30. B. Bolto and Z. Xie, *Processes*, **7**, 374 (2019); https://doi.org/10.3390/pr7060374
31. J. Talvitie, A. Mikola, A. Koistinen and O. Setälä, *Water Res.*, **123**, 401 (2017); https://doi.org/10.1016/j.watres.2017.07.005

32. A.M. Elgarahy, A. Akhdhar and K.Z. Elwakeel, *J. Environ. Chem. Eng.*, **9**, 106224 (2021); https://doi.org/10.1016/j.jece.2021.106224
33. M. Padervand, E. Lichtfouse, D. Robert and C. Wang, *Environ. Chem. Lett.*, **18**, 807 (2020); https://doi.org/10.1007/s10311-020-00983-1
34. T.A. Lastovina and A.P. Budnyk, *J. Water Process Eng.*, **43**, 102209 (2021); https://doi.org/10.1016/j.jwpe.2021.102209
35. P.L. Corcoran, Degradation of Microplastics in the Environment, In: Handbook of Microplastics in the Environment, eds.: T. Rocha-Santos, M.F. Costa and C. Mouneyrac, Springer International Publishing, Cham, pp. 531-542 (2022).
36. S. Klein, I.K. Dimzon, J. Eubeler and T.P. Knepper, eds.: M. Wagner and S. Lambert, Analysis, Occurrence and Degradation of Microplastics in the Aqueous Environment, In: Freshwater Microplastics, The Handbook of Environmental Chemistry, Springer International Publishing, Cham, pp. 51-67 (2018).
37. K. Munno, P.A. Helm, D.A. Jackson, C. Rochman and A. Sims, *Environ. Toxicol. Chem.*, **37**, 91 (2017); https://doi.org/10.1002/etc.3935
38. K.W. Meereboer, M. Misra and A.K. Mohanty, *Green Chem.*, **22**, 5519 (2020); https://doi.org/10.1039/D0GC01647K
39. M. Cole, H. Webb, P.K. Lindeque, E.S. Fileman, C. Halsband and T.S. Galloway, *Sci. Rep.*, **4**, 4528 (2014); https://doi.org/10.1038/srep04528
40. K. Enders, R. Lenz, C.A. Stedmon and T.G. Nielsen, *Mar. Pollut. Bull.*, **100**, 78 (2015); https://doi.org/10.1016/j.marpolbul.2015.09.027
41. F. Collard, B. Gilbert, G. Eppe, E. Parmentier and K. Das, *Arch. Environ. Contam. Toxicol.*, **69**, 331 (2015); https://doi.org/10.1007/s00244-015-0221-0
42. I. Gdara, J. Lawler, A. Staines and S. O'Neill, The State of the Art on the Potential Human Health Impacts of Microplastics and Nanoplastics, EPA Research Document 361, Dublin: Environmental Protection Agency (2021).
43. O.M. Rodríguez-Narvaez, A. Goonetilleke, L. Perez and E.R. Bandala, *Chem. Eng. J.*, **414**, 128692 (2021); https://doi.org/10.1016/j.cej.2021.128692
44. A.R. Lado Ribeiro, N.F.F. Moreira, G. Li Puma and A.M.T. Silva, *Chem. Eng. J.*, **363**, 155 (2019); https://doi.org/10.1016/j.cej.2019.01.080
45. H. Luo, C. Liu, D. He, J. Xu, J. Sun, J. Li and X. Pan, *J. Hazard. Mater.*, **423**, 126915 (2022); https://doi.org/10.1016/j.jhazmat.2021.126915
46. K. Zhu, H. Jia, Y. Sun, Y. Dai, C. Zhang, X. Guo, T. Wang and L. Zhu, *Water Res.*, **173**, 115564 (2020); https://doi.org/10.1016/j.watres.2020.115564
47. A.B. Silva, A.S. Bastos, C.I.L. Justino, J.P. da Costa, A.C. Duarte and T.A.P. Rocha-Santos, *Anal. Chim. Acta*, **1017**, 1 (2018); https://doi.org/10.1016/j.aca.2018.02.043
48. L.M. Matuana, S. Jin and N.M. Stark, *Polym. Degrad. Stab.*, **96**, 97 (2011); https://doi.org/10.1016/j.polymdegradstab.2010.10.003
49. R.A. Naik, L.S. Rowles III, A.I. Hossain, M. Yen, R.M. Aldossary, O.G. Apul, J. Conkle and N.B. Saleh, *Sci. Total Environ.*, **736**, 139690 (2020); https://doi.org/10.1016/j.scitotenv.2020.139690
50. W. Perren, A. Wojtasik and Q. Cai, *ACS Omega*, **3**, 3357 (2018); https://doi.org/10.1021/acsomega.7b02037
51. M. Kiendrebeogo, M.R. Karimi Estahbanati, A. Khosravanipour Mostafazadeh, P. Drogui and R.D. Tyagi, *Environ. Pollut.*, **269**, 116168 (2021); https://doi.org/10.1016/j.envpol.2020.116168
52. V.K. Gaur, S. Gupta, P. Sharma, P. Gupta, S. Varjani, J.K. Srivastava, J.-S. Chang and X.-T. Bui, *Curr. Pollut. Rep.*, **8**, 30 (2022); https://doi.org/10.1007/s40726-021-00210-7
53. I.A. Ricardo, E.A. Alberto, A.H. Silva Júnior, D.L.P. Macuvele, N. Padoin, C. Soares, H. Gracher Riella, M.C.V.M. Starling and A.G. Trovó, *Chem. Eng. J.*, **424**, 130282 (2021); https://doi.org/10.1016/j.cej.2021.130282
54. A. Debroy, N. George and G. Mukherjee, *J. Chem. Technol. Biotechnol.*, (2021); https://doi.org/10.1002/jctb.6978
55. H.L. Davies, H. Robb, K.D. Cox, G.A. Coverton, T.M. Eastham, H.J. Alexander and F. Juanes, *J. Exp. Marine Biol. Ecol.*, **542**, 151589 (2021); https://doi.org/10.1016/j.jembe.2021.151589
56. S. Elangovan, S.B.S. Pandian and S.J.G. Joshi, eds.: P.K. Arora, Polychlorinated Biphenyls (PCBs): Environmental Fate, Challenges and Bioremediation, In: Microbial Metabolism of Xenobiotic Compounds, Singapore: Springer, pp. 165-188 (2019).
57. H.K. Karapanagioti and I.K. Kalavrouziotis, Microplastics in Water and Wastewater. IWA Publishing, USA (2019).
58. C. Tu, Y. Teng, Y. Luo, X. Li, X. Sun, Z. Li, W. Liu and P. Christie, *J. Hazard. Mater.*, **186**, 1438 (2011); https://doi.org/10.1016/j.jhazmat.2010.12.008
59. G.O. Adams, P.T. Fufeyin, S.E. Okoro and I. Ehinomen, *Int. J. Environ. Bioremediat. Biodegrad.*, **3**, 28 (2015); https://doi.org/10.12691/ijebb-3-1-5
60. M. Tyagi, M.M.R. da Fonseca and C.C.C.R. de Carvalho, *Biodegradation*, **22**, 231 (2011); https://doi.org/10.1007/s10532-010-9394-4
61. S. Habib, A. Iruthayam, M.Y. Abd Shukor, S.A. Alias, J. Smykla and N.A. Yasid, *Polymers*, **12**, 2616 (2020); https://doi.org/10.3390/polym12112616
62. M.R. Pikoli, P. Astuti, F.A. Rahmah, A.F. Sari and N.A. Solihat, *Chiang Mai Univ. J. Sci.*, **49**, 1 (2022); https://doi.org/10.12982/CMUJNS.2022.005
63. K. Wojtowicz and T. Steliga, *Nafta-Gaz*, **76**, 507 (2020); https://doi.org/10.18668/NG.2020.08.03.
64. T. Steliga, K. Wojtowicz, P. Kapusta and J. Brzeszcz, *Molecules*, **25**, 709 (2020); https://doi.org/10.3390/molecules25030709
65. A.V.B. Reddy, M. Moniruzzaman and T.M. Aminabhavi, *Chem. Eng. J.*, **358**, 1186 (2019); https://doi.org/10.1016/j.cej.2018.09.205
66. S.-Y. Chung, M. Maeda, E. Song, K. Horikoshij and T. Kudo, *Biosci. Biotechnol. Biochem.*, **58**, 2111 (1994); https://doi.org/10.1271/bbb.58.2111
67. Y. Liang, R. Meggo, D. Hu, J.L. Schnoor and T.E. Mattes, *Ecol. Eng.*, **71**, 215 (2014); https://doi.org/10.1016/j.ecoleng.2014.07.046
68. I. Sierra, J.L. Valera, M.L. Marina and F. Laborda, *Chemosphere*, **53**, 609 (2003); https://doi.org/10.1016/S0045-6535(03)00418-1
69. J.D. Moody, D.R. Doerge, J.P. Freeman and C.E. Cerniglia, *Appl. Microbiol. Biotechnol.*, **58**, 364 (2002); https://doi.org/10.1007/s00253-001-0878-3
70. S. Murínová and K. Dercová, *Water Air Soil Pollut.*, **225**, 1980 (2014); https://doi.org/10.1007/s11270-014-1980-3
71. S. Zorádová, H. Dudášová, L. Lukáčová, K. Dercová and M. Čertík, *Int. Biodeterior. Biodegradation*, **65**, 1019 (2011); https://doi.org/10.1016/j.ibiod.2011.03.012
72. H. Wang, J. Hu, K. Xu, X. Tang, X. Xu and C. Shen, *Biodegradation*, **29**, 1 (2018); https://doi.org/10.1007/s10532-017-9809-6
73. H.J. Jeon and M.N. Kim, *Int. Biodeterior. Biodegradation*, **115**, 244 (2016); https://doi.org/10.1016/j.ibiod.2016.08.025
74. K. Bano, M. Kuddus, M. Rehan Zaheer and R. Rehan Zaheer, *Turk Biyokim. Derg.*, **44**, 344 (2019); https://doi.org/10.1515/tjb-2018-0207
75. H.S. Auta, C.U. Emenike and S.H. Fauziah, *Environ. Pollut.*, **231**, 1552 (2017); https://doi.org/10.1016/j.envpol.2017.09.043
76. J. Zhang, D. Gao, Q. Li, Y. Zhao, L. Li, H. Lin, Q. Bi and Y. Zhao, *Sci. Total Environ.*, **704**, 135931 (2020); https://doi.org/10.1016/j.scitotenv.2019.135931
77. R. Sangeetha Devi, V. Rajesh Kannan, D. Nivas, K. Kannan, S. Chandru and A. Robert Antony, *Mar. Pollut. Bull.*, **96**, 32 (2015); https://doi.org/10.1016/j.marpolbul.2015.05.050
78. A. Paço, K. Duarte, J.P. da Costa, P.S.M. Santos, R. Pereira, M.E. Pereira, A.C. Freitas, A.C. Duarte and T.A.P. Rocha-Santos, *Sci. Total Environ.*, **586**, 10 (2017); https://doi.org/10.1016/j.scitotenv.2017.02.017
79. M.C. Krueger, U. Hofmann, M. Moeder and D. Schlosser, *PLoS One*, **10**, e0131773 (2015); https://doi.org/10.1371/journal.pone.0131773

80. M.T. El-Sayed, G.H. Rabie and E.A. Hamed, *Environ. Dev. Sustain.*, **23**, 14556 (2021); https://doi.org/10.1007/s10668-021-01258-7
81. S. Raut, S. Raut, M. Sharma, C. Srivastav, B. Adhikari and S.K. Sen, *Indian J. Microbiol.*, **55**, 258 (2015); https://doi.org/10.1007/s12088-015-0522-z
82. T. Volke-Sepúlveda, G. Saucedo-Castaneda, M. Gutierrez-Rojas, A. Manzur and E. Favela-Torres, *J. Appl. Polym. Sci.*, **83**, 305 (2002); https://doi.org/10.1002/app.2245
83. A.R. Cowan, C.M. Costanzo, R. Benham, E.J. Loveridge and S.C. Moody, *J. Appl. Microbiol.*, **132**, 78 (2022); https://doi.org/10.1111/jam.15203
84. N. Afianti, A. Rachman, A. Hatmanti, D. Yogaswara, M. Anggiani, N. Fitriya and Y. Darmayati, *J. Ecol. Eng.*, **23**, 261 (2022); https://doi.org/10.12911/22998993/145463
85. J.Y. Cho, S. Lee Park, H.-J. Lee, S.H. Kim, M.J. Suh, S. Ham, S.K. Bhatia, R. Gurav, S.-H. Park, K. Park, D. Yoo and Y.-H. Yang, *Chemosphere*, **283**, 131172 (2021); https://doi.org/10.1016/j.chemosphere.2021.131172
86. L. Ren, G. Wang, Y. Huang, J. Guo, C. Li, Y. Jia, S. Chen, J.L. Zhou and H. Hu, *Sci. Total Environ.*, **791**, 148303 (2021); https://doi.org/10.1016/j.scitotenv.2021.148303
87. A. Delacuvellerie, V. Cyriaque, S. Gobert, S. Benali and R. Wattiez, *J. Hazard. Mater.*, **380**, 120899 (2019); https://doi.org/10.1016/j.jhazmat.2019.120899
88. D. Hadad, S. Geresh and A. Sivan, *J. Appl. Microbiol.*, **98**, 1093 (2005); https://doi.org/10.1111/j.1365-2672.2005.02553.x
89. A.O. Azghani, E.J. Miller and B.T. Peterson, *Lung*, **178**, 261 (2000); https://doi.org/10.1007/s004080000031
90. M.I. Ali, S. Ahmed, I. Javed, N. Ali, N. Atiq, A. Hameed and G. Robson, *Int. J. Environ. Sci. Technol.*, **11**, 339 (2014); https://doi.org/10.1007/s13762-013-0220-5
91. S.H. Joo, Y. Liang, M. Kim, J. Byun and H. Choi, *Environ. Challenges*, **3**, 100042 (2021); https://doi.org/10.1016/j.envc.2021.100042
92. K.B. Sundbæk, I.D.W. Koch, C.G. Villaro, N.S. Rasmussen, S.L. Holdt and N.B. Hartmann, *J. Appl. Phycol.*, **30**, 2923 (2018); https://doi.org/10.1007/s10811-018-1472-8
93. T. Atugoda, H. Piyumali, S. Liyanage, K. Mahatantila and M. Vithanage, eds.: T. Rocha-Santos, M.F. Costa and C. Mouneyrac, Fate and Behavior of Microplastics in Freshwater Systems, In: Handbook of Microplastics in the Environment, Springer Nature, pp. 1–31 (2020).
94. A. Bellingeri, E. Bergami, G. Grassi, C. Faleri, P. Redondo-Hasselerharm, A.A. Koelmans and I. Corsi, *Aquat. Toxicol.*, **210**, 179 (2019); https://doi.org/10.1016/j.aquatox.2019.02.022
95. A. Halder and Y. Sun, *Biosens. Bioelectron.*, **139**, 111334 (2019); https://doi.org/10.1016/j.bios.2019.111334
96. W. Liu, X. Chen, X. Lu, J. Wang, Y. Zhang and Z. Gu, *Adv. Funct. Mater.*, **30**, 2003195 (2020); https://doi.org/10.1002/adfm.202003195
97. Z. Wang, M. Sedighi and A. Lea-Langton, *Water Res.*, **184**, 116165 (2020); https://doi.org/10.1016/j.watres.2020.116165
98. Z.U. Zango, N.S. Sambudi, K. Jumbri, A. Ramli, N.H.H. Abu Bakar, B. Saad, M.N.H. Rozaini, H.A. Isiyaka, A.M. Osman and A. Sulieman, *Water*, **12**, 2921 (2020); https://doi.org/10.3390/w12102921
99. S. Joseph, C.I. Kammann, J.G. Shepherd, P. Conte, H.-P. Schmidt, N. Hagemann, A.M. Rich, C.E. Marjo, J. Allen, P. Munroe, D.R.G. Mitchell, S. Donne, K. Spokas and E.R. Graber, *Sci. Total Environ.*, **618**, 1210 (2018); https://doi.org/10.1016/j.scitotenv.2017.09.200
100. V. Siipola, S. Pflugmacher, H. Romar, L. Wendling and P. Koukkari, *Appl. Sci.*, **10**, 788 (2020); https://doi.org/10.3390/app10030788
101. J. Wang, C. Sun, Q.-X. Huang, Y. Chi and J.-H. Yan, *J. Hazard. Mater.*, **419**, 126486 (2021); https://doi.org/10.1016/j.jhazmat.2021.126486
102. S. Ye, M. Cheng, G. Zeng, X. Tan, H. Wu, J. Liang, M. Shen, B. Song, J. Liu, H. Yang and Y. Zhang, *Water Res.*, **179**, 115876 (2020); https://doi.org/10.1016/j.watres.2020.115876