Journal of Hazardous Materials 510 (2026) 142138

Contents lists available at ScienceDirect

Journal of Hazardous Materials

journal homepage: www.elsevier.com/locate/jhazmat

Review

When "biodegradable" is not benign: Microplastic-driven disruption of soil processes and plant-microbe interactions

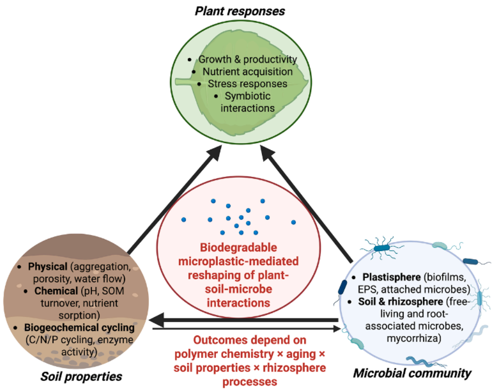
Teja Pelko [a,*], Anita Jemec Kokalj [a], Marjana Regvar [a], Marina Dermastia [b], Katarina Vogel-Mikuš [a,c]

[a] University of Ljubljana, Biotechnical Faculty, Department of Biology, Jamnikarjeva 101, Ljubljana, Slovenia
[b] National Institute of Biology, Večna pot 121, Ljubljana, Slovenia
[c] Jožef Stefan Institute, Jamova cesta 39, Ljubljana, Slovenia

HIGHLIGHTS

- Integrative framework links BMP effects across soil–microbe–plant systems.
- BMP impacts are governed by polymer traits, aging, and soil context.
- BMPs reshape soil structure, biogeochemistry, and microbial niches.
- Plant responses to BMPs are largely indirect and rhizosphere-mediated.
- Standardized, long-term studies are essential for realistic BMPs risk assessment.

GRAPHICAL ABSTRACT



ARTICLE INFO

Keywords:
soil ecosystem
biodegradable microplastics
plant microbiome
plastisphere

ABSTRACT

The increasing use of biodegradable plastics (BPs) as alternatives to conventional plastics (CPs) is leading to the accumulation of biodegradable microplastics (BMPs) in terrestrial environments. Contrary to assumptions of rapid degradation, BMPs can persist in soil long enough to interact with key biological processes. This review advances the field by proposing a mechanistic framework linking BMP aging and degradation, soil physicochemical transformations, plastisphere assembly, rhizosphere interactions, and plant responses, and by critically

---

Abbreviation list: AMF, arbuscular mycorrhizal fungi; ARG, antibiotic resistance genes; BMP, biodegradable microplastic; BP, biodegradable plastic; CAT, catalase; CMP, conventional microplastic; CP, conventional plastic; CUE, carbon use efficiency; DOC, dissolved organic carbon; DOM, dissolved organic matter; EPS, extracellular polymeric substances; GMD, geometric mean diameter; HGT, horizontal gene transfer; LDPE, low-density polyethylene; MAOM, mineral-associated organic matter; MDA, malondialdehyde; MP, microplastic; MRG, metal resistance genes; NP, nanoplastic; PBAT, polybutylene adipate-co-terephthalate; PBS, polybutylene succinate; PBSA, poly(butylene succinate-co-adipate); PCL, polycaprolactone; PE, polyethylene; PGPR, plant growth-promoting rhizobacteria; PHA, polyhydroxyalkanoates; PHB, polyhydroxybutyrates; PHBV, poly(3-hydroxybutyrate-co-3-hydroxyvalerate); PLA, polylactic acid; POD, peroxidase; POM, particulate organic matter; PS, polystyrene; PVC, polyvinyl chloride; ROS, reactive oxygen species; SOC, soil organic carbon; SOD, superoxide dismutase; SOM, soil organic matter; VF, virulence factor; WHC, water-holding capacity.

[*] Corresponding author.

E-mail address: teja.pelko@bf.uni-lj.si (T. Pelko).

https://doi.org/10.1016/j.jhazmat.2026.142138
Received 13 February 2026; Received in revised form 3 April 2026; Accepted 17 April 2026

rhizosphere
nutrient cycling

evaluating the sources of inconsistency across studies. We show that divergent effects of BMPs can be best explained by four interacting determinants: polymer chemistry and additive composition, aging-driven surface transformations, soil physicochemical properties, and rhizosphere processes including plant-mediated effects. Through these coupled pathways, BMPs can alter aggregation, pore architecture, pH, enzyme activity, and carbon and nutrient cycling, thereby reshaping the soil environment in which microorganisms and roots interact. BMP surfaces can also act as dynamic microbial niches that promote biofilm formation, shift microbial community composition and function, and under certain conditions may facilitate pollutant transport, pathogen persistence, and horizontal gene transfer. Plant responses to BMPs are predominantly indirect and emerge from rhizosphere-mediated processes, which helps explain the wide variability in reported plant responses, ranging from subtle metabolic changes to pronounced growth inhibition. However, current evidence is constrained by short-term studies and insufficient consideration of aged materials. Biodegradability should therefore not be equated with low ecological risk in soils. Progress in this field requires integrative approaches linking BMP properties, plastisphere dynamics, and plant–soil interactions over time.

1. Introduction

The growing crisis of plastic pollution has raised increasing concern about the impacts of microplastics (MPs) on terrestrial ecosystems, particularly soil processes, plant-associated microbiomes, and ecosystem functioning. MPs are commonly defined as plastic particles smaller than 5 mm [1] or 1 mm according to ISO/DIS 24187 [2] and originate primarily from the fragmentation and degradation of larger plastic materials. In terrestrial environments, MPs enter soils through multiple pathways, including agricultural practices, industrial activities, consumer products, and environmental inputs such as sewage sludge, compost, irrigation water, surface runoff, and atmospheric deposition [3]. Current reports on the levels of MPs in soils indicate that concentrations can range from trace amounts to extremely high loads, reaching up to 67,500 mg kg^{-1} (6.75% w/w) at highly contaminated industrial sites [4,5], while concentrations in agricultural soils have already been reported to exceed 1% (w/w) under certain conditions [6].

Biodegradable plastics (BPs) have been promoted as environmentally friendly alternatives to conventional plastics (CPs) and are increasingly used in agriculture and horticulture, including mulching films, drip irrigation components, seed coatings, slow-release fertilizers, biodegradable pots, and packaging materials [7]. These materials are derived from renewable (bio-based) or fossil-based feedstocks and include polymers such as polylactic acid (PLA), polyhydroxyalkanoates (PHA), polybutylene adipate-co-terephthalate (PBAT), polycaprolactone (PCL), and polybutylene succinate (PBS). Importantly, “bio-based” refers to feedstock origin, whereas “biodegradable” and “compostable” refer to end-of-life behavior under specific

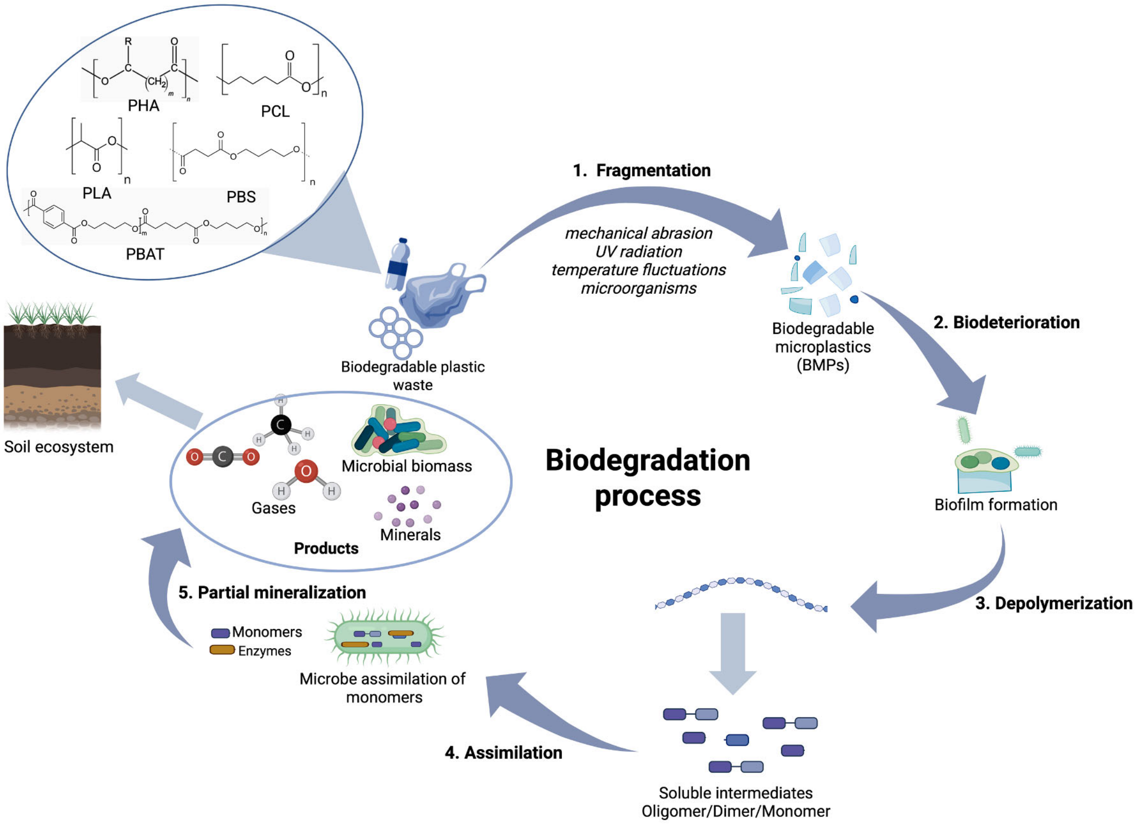


Fig. 1. Biodegradable polymers of different chemical composition (e.g., polylactic acid - PLA, polybutylene succinate PBS; polyhydroxyalkanoates – PHA; polycaprolactone – PCL; polybutylene adipate-co-terephthalate – PBAT) undergo physical fragmentation under soil conditions, leading to the formation of biodegradable microplastics (BMPs) that can persist in soil rather than undergoing immediate mineralization. BMP surfaces are subsequently colonized by microorganisms, enabling biofilm formation and enzymatic depolymerization into low-molecular-weight oligomers and monomers, which may subsequently be assimilated into microbial biomass or partially mineralized.

environmental conditions; these terms are therefore not interchangeable. Despite their "biodegradable" designation, many BPs degrade incompletely under natural soil conditions, fragmenting into biodegradable microplastics (BMPs) that can persist in soil for variable periods and may pose ecological risks comparable to, or in some cases exceeding, those associated with CP-derived MPs (CMPs) [8]. Compared with CPs, the environmental fate of BPs is more strongly governed by polymer chemistry, microbial depolymerization, and environmental conditions, which determine whether they undergo partial mineralization or persist as BMPs (Fig. 1). Polymer structure, additives, temperature, moisture, oxygen availability, and microbial activity jointly control degradation rates and the release of oligomers, monomers, and other transformation products [9–12]. As a result, materials such as PLA, PHA, PBS, PCL, and PBAT do not degrade uniformly in soil and may persist long enough to influence soil physicochemical conditions, microbial colonization, and plant-associated processes.

Once BMPs enter the soil environment, they interact directly with soil constituents, particularly microorganisms. Growing evidence indicates that BMPs can disrupt finely balanced microbial networks in soil, exerting both stimulatory and inhibitory effects on microbial activity and plant performance. While BMPs may serve as an additional carbon source and stimulate microbial metabolism, they have also been shown to reduce enzyme activities, alter microbial community composition, and impair soil nutrient cycling [13]. Moreover, BMPs can influence plant–soil feedbacks and symbiotic interactions, including those involving mycorrhizal fungi, with potential consequences for nutrient acquisition and plant resilience [14,15].

Despite increasing research on BMP–soil interactions, a comprehensive synthesis addressing BMP effects across the entire plant-associated microbiome is still lacking. The plant microbiome encompasses multiple interconnected compartments, including the rhizosphere (soil–root interface), rhizoplane and phylloplane (root and leaf surfaces), and endosphere (internal plant tissues), each hosting distinct yet interacting microbial communities that are critical for plant health and ecosystem stability [16,17]. The rhizosphere is a hotspot for nutrient exchange and microbial interactions, while endophytic communities influence plant growth, stress tolerance, and disease resistance [18]. Although less intensively studied, phylloplane microbiota also contribute to microbiome dynamics and plant functioning [19]. Collectively, these microbial communities form an integral component of the plant holobiont, enhancing nutrient acquisition, pathogen resistance, and tolerance to abiotic stresses [20].

Although BMPs and CMPs in soil, as well as plastisphere and rhizosphere processes, have been widely studied in recent years, the evidence remains fragmented across soil physics, microbial ecology, and plant physiology. Apparent inconsistencies in reported BMP effects are therefore not necessarily contradictory, but rather reflect strong modulation by polymer-specific properties, particle aging, exposure levels, soil characteristics, and soil–microbe–plant interactions. Rather than treating degradation, plastisphere formation, and plant responses as separate processes, this review integrates these dimensions to explain how BMPs influence soil functioning through interconnected physical, microbial, and plant-associated pathways. In contrast to recent reviews that address these topics independently or within broader overviews of MPs in soil, we focus specifically on BMPs and develop a unified mechanistic framework linking degradation, soil-property changes, microbial community responses, and plant performance.

Within this framework, divergent outcomes can be interpreted as emerging from the interaction of four key determinants: polymer chemistry and additive composition, aging-driven surface transformations, soil physicochemical context, and rhizosphere processes, including root exudation and plant-mediated effects. Because much of the available evidence is derived from short-term studies using pristine materials and simplified exposure scenarios, reported effects are interpreted primarily as mechanistic insights rather than direct predictions of field-scale ecological risks. By synthesizing these processes within a systems-level perspective, this review helps resolve apparent inconsistencies in the literature and identifies priorities for environmentally realistic, process-based future research.

2. Literature research strategy

The literature considered in this review was identified through searches in the Web of Science, Scopus, and ScienceDirect databases, supplemented by backward and forward citation screening of relevant review and research articles. Database searches were conducted between January 2025 and August 2025, with an additional update performed in March 2026. Search terms combined keywords related to biodegradable plastics/microplastics, soil systems, and plant–microbe interactions, including terms such as "biodegradable microplastic", "biodegradable plastic", "soil", "rhizosphere", "plastisphere", "plant", "microbial community", "nutrient cycling", "carbon", "nitrogen", "phosphorus", and "enzyme activity", together with names of major biodegradable polymers (e.g., PLA, PBAT, PBS, PHA, and PCL).

When comparisons with conventional plastics were relevant, additional searches included terms such as "conventional plastic*" and specific polymer names (e.g., PE, PP, PVC). Only peer-reviewed articles written in English were considered. Priority was given to studies directly addressing the degradation, persistence, soil effects, microbial interactions, and plant-associated responses of biodegradable plastics or biodegradable microplastics in terrestrial environments. Recent literature was emphasized, while earlier studies were included where needed to provide mechanistic or terminological context. No formal meta-analysis was conducted due to methodological heterogeneity across studies.

3. Impact of biodegradable microplastics on soil properties and processes

BMPs can alter soil functioning through a combination of direct physical and chemical interactions and indirect, microbially mediated processes. These effects manifest across multiple levels of soil organization, including soil structure, physicochemical properties, enzymatic activity, and biogeochemical cycling of carbon and nutrients. This section synthesizes current evidence on how BMPs influence soil properties and processes, providing a mechanistic basis for understanding downstream effects on soil microbial communities and plant performance.

3.1. Soil aggregate structure

Soil aggregates are essential for maintaining soil structure, regulating water and gas exchange, and supporting biological activity [21]. Increasing evidence indicates that BMPs can substantially modify aggregate size distribution and stability, but the direction and magnitude of these changes are not uniform. In some systems, BMPs (e.g., PLA) have been associated with reduced macroaggregate (>250 μm) proportions and lower stability, consistent with declines in microbial "binding agents" that cement particles into stable aggregates [22,23]. In other soils and exposure conditions, however, PLA and especially PHA have been reported to decrease microaggregate fractions while increasing macroaggregates and geometric mean diameter (GMD), often without a short-term reduction in aggregate stability [21]. Field-based evidence further suggests that BMPs may maintain aggregate stability relative to conventional polymers (e.g., polyethylene - PE), and that BMPs more strongly couple aggregate-associated carbon responses to microbial pathways [24].

These contrasting outcomes likely reflect the net balance of competing mechanisms. BMPs may destabilize aggregates through physical interference with particle cohesion and by indirectly reducing microbial-derived binding agents, thereby favoring macroaggregate breakdown and microaggregate enrichment under some conditions [23]. Conversely, partial biodegradation can stimulate microbial activity and biofilm formation, while aging-related surface transformations, such as the development of oxygen-containing functional groups and reduced hydrophobicity, may enhance interactions with soil minerals, organic matter, and microorganisms, thereby contributing to aggregate cementation [25].

Aggregate responses can also depend on environmental forcing: structural indices may decline under slow wetting yet increase under fast wetting via reduced slaking, implying that moisture dynamics can modulate or even reverse apparent stability outcomes [26]. As BMP degradation progresses, continuous generation of fragments and their variable occlusion within aggregates can further complicate outcomes; for example, small PBAT particles may become protected within aggregates while larger particles remain more exposed and may simultaneously be associated with increased macroaggregation in rhizosphere-influenced soils [27].

Overall, the effects of BMPs on soil aggregation appear to arise from a dynamic balance between destabilizing and stabilizing processes, influenced by polymer characteristics, degradation stage, and moisture dynamics. Long-term studies encompassing a wider range of exposure levels, aging states, and hydrological conditions are therefore needed to better resolve their implications for soil structural integrity.

3.2. *Soil porosity and water-holding capacity*

BMPs can significantly modify soil porosity, permeability, and water retention, with effects strongly dependent on concentration and soil texture. At low concentrations, BMPs increase soil porosity by approximately 10–15%, enhancing water infiltration and aeration [28]. In contrast, excessive BMP accumulation can block macropores, leading to reduced hydraulic conductivity and impaired soil functioning [29,30].

Effects on soil water retention follow a similar concentration-dependent pattern. Low BMP inputs tend to increase micropore volume and water-holding capacity (WHC), whereas higher concentrations disrupt water bridges between soil particles, resulting in reduced retention and enhanced desorption, particularly under semi-arid conditions [31,32]. These effects are more pronounced in loam and clay soils than in sandy soils, highlighting the central role of soil texture in mediating BMP–soil interactions [33].

By reshaping pore architecture and soil moisture dynamics, BMPs indirectly influence water balance, microbial activity, and nutrient transport, underscoring the importance of assessing their long-term impacts on soil functioning.

3.3. *Soil density*

Because of their lower density relative to mineral soil particles, MPs—including BMPs—can reduce soil bulk density when incorporated into the soil matrix, potentially resulting in a looser structure with increased aeration. Hydrophobic MPs may additionally increase soil water repellency by raising the soil–water contact angle, thereby reducing water retention [34]. Although BMPs are generally less hydrophobic than CMPs, similar effects can occur at elevated concentrations. At environmentally relevant concentrations (e.g., 4.5 mg kg^{-1} dry soil), however, changes in bulk density and water repellency are often negligible [4], indicating limited short-term impacts under typical field conditions.

Nevertheless, the long-term consequences of continuous BMP accumulation, particularly under repeated agricultural inputs, remain poorly understood and require further investigation.

3.4. *Soil pH*

The properties of MPs, including particle size and shape, can influence soil physicochemical conditions such as pH [35,36]. BMPs can actively modify soil pH through the microbial release of low-molecular-weight organic acids—such as lactic acid, succinic acid, and 3-hydroxybutyric acid—during polymer decomposition [37]. This acidification may be further amplified by BMP-induced stimulation of nitrifying bacterial communities, including *Nitrospirae*, which promote nitrification and generate H^+ ions [37,38].

Empirical studies consistently report measurable decreases in soil pH following BMP exposure, often accompanied by changes in soil chemical composition and microbial community structure [37,39]. For instance, enzymatic degradation of PLA-based BMPs leads to the accumulation of lactic acid oligomers, further intensifying soil acidification [40]. However, increases in soil pH have also been reported [36,41,42]. Increases in soil pH likely occur when alkalinizing pathways outweigh organic-acid inputs. First, many biodegradable mulch films (e.g., PBAT/PLA-based materials) contain inorganic mineral fillers (commonly carbonates such as $CaCO_3$) and other additives; as BMPs weather and fragment, leaching of these alkaline components can neutralize acidity and increase soil pH [43]. Second, BMPs can shift soil nitrogen cycling toward alkalinity-generating transformations, for example by favoring net NH_4^+ production (e.g., ammonification) or suppressing nitrification [44,45]. Compared with BMPs, CMPs generally influence soil pH less directly because they do not undergo rapid biodegradation and therefore do not typically generate the same microbially driven release of low-molecular-weight organic acids. Reported pH shifts under CMP exposure are more often attributed to indirect changes in soil physical conditions, additive release, or microbial community composition rather than decomposition of the polymer itself [35].

Overall, current evidence suggests that BMPs affect soil pH through pathways that are more directly linked to biodegradation and microbial activity than those associated with CMPs, highlighting the need to evaluate their long-term effects on soil buffering capacity under environmentally realistic conditions.

3.5. *Soil enzymatic activity*

Soil enzymes are protein molecules synthesized by soil microorganisms and plant roots and released into the soil matrix, where they catalyze a wide range of biochemical reactions [46]. They play a central role in soil organic matter (SOM) decomposition and nutrient mineralization, thereby regulating nutrient availability for plants and microorganisms and supporting ecosystem metabolism [47]. Accordingly, soil enzymatic activities are widely used as sensitive indicators of soil ecosystem functioning and biogeochemical cycling [48].

BMPs can influence soil enzymatic activity, particularly enzymes such as dehydrogenase and urease, with implications for key ecosystem processes [49]. These effects are highly context-dependent and mediated by changes in soil physical conditions and nutrient availability. While BMPs may provide an additional carbon source for microorganisms, they can also adsorb nutrients, resulting in reduced activity of enzymes involved in nitrogen cycling, such as urease [50]. Environmental conditions further modulate these responses. Under drought conditions, BMPs have been reported to increase soil respiration and enzymatic activity, likely due to improved aeration and moisture retention [14,15]. In contrast, under well-watered conditions, the same materials have been shown to suppress microbial activity and enzyme function, indicating a strong dependence on soil moisture status and aeration.

Catalase (CAT) activity in soil—produced primarily by soil microorganisms and modulated by plant root exudation—reflects the soil's capacity to decompose hydrogen peroxide (H_2O_2) and is therefore commonly used as an indicator of microbial oxidative metabolism and overall biological activity in the rhizosphere [51]. Reduced soil CAT activity following exposure to PLA-based BMPs has been reported, suggesting diminished microbial enzymatic capacity to remove H_2O_2 and a potentially more oxidative soil–plant interface [49]. Collectively, the available evidence indicates that the effects of BMPs on soil enzymatic activity are largely governed by interactions between material properties and environmental conditions, rather than reflecting uniform or unidirectional responses [14,49,50]. A plausible mechanistic interpretation is that enzyme stimulation is more likely when BMPs release readily metabolizable carbon, improve microsite aeration, or promote colonization by copiotrophic degraders, thereby increasing microbial activity and the demand for extracellular enzymes. In contrast, inhibitory effects are more likely when weathered particles adsorb nutrients or enzymes, when acidic or potentially toxic degradation products accumulate, when oxygen or moisture conditions become unfavorable, or when shifts in community composition suppress key functional taxa. Therefore, contrasting enzyme responses across studies likely reflect differences in polymer composition, additive release, aging status, soil moisture regime, and the balance between substrate-driven stimulation and physicochemical stress [14,49].

3.6. *Soil organic matter (SOM)*

Soil organic matter (SOM) is fundamental to soil fertility and contaminant dynamics and comprises particulate organic matter (POM), mineral-associated organic matter (MAOM), and dissolved organic matter (DOM) [52,53]. BMP inputs can disrupt the balance among these fractions by becoming incorporated into soil aggregates alongside organic residues, thereby contributing to the POM pool [54].

In contrast to CMPs, BMPs release monomers, oligomers, and additives during depolymerization, which can substantially increase DOM concentrations and enhance the leaching potential of these compounds [55,56]. These degradation by-products may also interact with heavy metals, altering contaminant mobility and associated ecological risks [57].

Rather than primarily changing total SOM stocks, BMPs appear to modify the distribution and composition of SOM fractions, including POM, MAOM, and DOM pools [58]. These effects are likely mediated by the incorporation of biodegradable residues into soil organic pools and by changes in DOM quantity and chemodiversity during polymer degradation [58]. The following section therefore focuses specifically on how these changes influence SOC mineralization, turnover, and broader carbon-cycling dynamics.

3.7. *Alterations in carbon cycling*

Soil organic carbon (SOC) is a central component of the global carbon cycle, regulating atmospheric CO_2 concentrations while supporting soil health and ecosystem functioning. How SOC responds to external inputs is therefore critical for understanding carbon dynamics in terrestrial ecosystems, particularly under changing climatic conditions. By associating with soil aggregates, BMPs can modify soil structure and microbial activity, thereby influencing SOC decomposition and turnover [59].

Dissolved organic carbon (DOC) plays a key role in these processes because it is readily available to microorganisms and highly mobile within the soil matrix. Several studies report higher DOC concentrations in soils containing BMPs compared to CMPs, reflecting the release of low-molecular-weight carbon compounds during depolymerization [60]. At the same time, BMPs do not always enhance SOC mineralization. In some cases, PLA and other BMPs may adsorb DOC because of their porous physical structure, thereby lowering DOC bioavailability to microorganisms and reducing SOC mineralization [61]. Thus, BMP effects on SOC mineralization are not unidirectional: depending on polymer properties and soil conditions, they may either reduce mineralization by lowering DOC bioavailability or enhance it by increasing substrate availability for microbial metabolism [56,62]. Unlike the largely inert carbon associated with petroleum-based CMPs, BMP-derived carbon can be rapidly metabolized, with part of it mineralized to CO_2 and part incorporated into microbial biomass [56,62].

Evidence from stable isotope tracing confirms microbial assimilation of BMP-derived carbon. Using ^{13}C-labeling, Zumstein et al. [56] demonstrated the production of $^{13}CO_2$ during soil incubations with PBAT monomers, while Nelson et al. [62] showed that approximately 7% of carbon from PBS was incorporated into microbial biomass. Following microbial turnover, this carbon may enter POM or MAOM pools [63]. Because MAOM is strongly stabilized by mineral interactions, it has the potential to contribute to longer-term carbon retention [64]. Despite these pathways, current evidence suggests that the contribution of BMP-derived carbon to long-term SOC storage is limited. Even under favorable conditions, only about 10% of BMP-derived carbon has been estimated to enter microbial biomass within two years [64], representing a minor addition relative to existing SOC stocks. Accordingly, sustained BMP inputs are unlikely to substantially increase SOC, particularly in carbon-saturated soils [65].

Moreover, BMP degradation can shift the quality and availability of carbon substrates, thereby stimulating microbial activity and accelerating carbon losses through enhanced mineralization [39,66]. The fate of BMP-derived carbon is further shaped by microbial carbon use efficiency (CUE), which governs the balance between biomass production and respiration [67]. Because BMPs generally represent nutrient-poor substrates, their decomposition may reduce CUE and limit carbon stabilization in soil [59]. Biodegradable polymers may also induce priming effects, including microbial nitrogen mining, in which microorganisms decompose native SOM to obtain nutrients, thereby enhancing SOC mineralization [31,68]. This mechanism may help explain why BMPs can, under some conditions, contribute to stronger SOC losses than would be expected from their biodegradable label alone.

We can therefore conclude that BMPs influence soil carbon cycling primarily by altering substrate availability and microbial processing, rather than contributing substantially to long-term carbon sequestration.

3.8. *Alterations in nitrogen cycling*

Nitrogen is a key nutrient in soil ecosystems, occurring in both organic and inorganic forms, with nitrate (NO_3^-–N) and ammonium (NH_4^+–N) representing the principal forms taken up by plants [69]. BMPs can potentially adsorb both NO_3^-–N and NH_4^+–N due to their surface chemistry [70]. Compared with CMPs, BMPs may influence soil nitrogen cycling more strongly because their oxygenated functional groups and hydrolysable ester bonds increase surface reactivity and promote the release of low-molecular-weight carbon compounds during depolymerization. These properties can enhance microbial biomass and nitrogen immobilization, alter the balance between mineralization and nitrification, and stimulate nitrogen-cycling genes in ways that are less pronounced in inert conventional polymers such as PE or polyvinyl chloride (PVC) [39,58]. For example, PLA exerted stronger effects than PVC on soil microbial assemblages and nitrogen-cycling functions, especially at higher concentrations [71]. Recent work further showed that BMPs increased total nitrogen-cycling gene abundance and stimulated gross nitrogen transformation processes relative to CMPs, likely through enhanced substrate supply and microbial activity [72].

Beyond these direct interactions, BMPs influence nitrogen cycling indirectly by altering soil physicochemical conditions and microbial community structure [73,74]. Changes in bulk density, aggregate stability, and WHC modify soil porosity and oxygen availability, thereby regulating microbial processes such as nitrification and denitrification [28,30,75]. Increased porosity generally promotes nitrification by improving oxygen diffusion to aerobic nitrifiers, whereas reduced water retention may constrain microbial activity and slow nitrogen mineralization under certain conditions [76,77]. BMP-induced changes in soil structure also affect oxygen gradients and substrate transport, influencing the spatial coupling of nitrification and denitrification processes [76,77]. In addition, BMP-associated soil acidification may favor *Acidobacteria*, including members of the *Nitrospirae*, which play important roles in nitrification [74]. At the functional level, BMP exposure has been linked to shifts in the abundance of genes involved in nitrogen cycling, with increases in genes related to nitrogen immobilization and nitrate reduction and decreases in those associated with nitrification and denitrification pathways [73]. Biofilm formation on BMP surfaces and the pronounced decomposition of these materials further shape microbial community composition and activity, thereby influencing nitrogen transformation rates and pathways [57,78].

Polymer identity is also a key determinant of these processes. PLA and related aliphatic polyesters are generally more readily hydrolyzed and metabolized than conventional polymers, which can lead to faster carbon release, stronger microbial restructuring, and clearer shifts in nitrogen-cycling communities. PBAT may behave differently from PLA because its aliphatic-aromatic structure makes it more resistant to enzymatic attack; nevertheless, PBAT degradation can still stimulate microbial activity and may promote SOM decomposition through increased microbial nitrogen demand [79]. High-dose PLA has also been shown to alter urease activity, NH_4^+–N and NO_3^-–N dynamics while reducing N_2O emissions, further indicating that biodegradable polymers can reshape nitrogen cycling through coupled carbon–enzyme–microbial pathways [80].

Taken together, BMP effects on soil nitrogen cycling are better explained by polymer-specific physicochemical traits than by biodegradability as a label alone. The magnitude and direction of these responses depend on polymer identity, degradation behavior, concentration, and soil properties, which together determine how BMPs modify substrate availability, microbial activity, and nitrogen-transformation pathways.

3.9. Alterations in phosphorus cycling

Phosphorus is an essential nutrient regulating plant growth and soil ecosystem functioning [81]. In soils, phosphorus occurs in both organic and inorganic forms, with inorganic phosphate representing the primary fraction available for plant uptake [81]. MPs can influence soil phosphorus dynamics through additive release, surface-mediated adsorption, and indirect effects on microbial activity [39,82]. However, the mechanisms are unlikely to be identical for BMPs and CMPs. CMPs are generally more inert and often affect phosphorus mainly through physical disturbance of soil structure or additive leaching, whereas BMPs can additionally affect phosphorus turnover through evolving surface chemistry, biodegradation-linked carbon release, and stronger stimulation of microbial colonization and enzyme production [39].

BMPs have been shown to stimulate acid phosphatase activity, enhancing the mineralization of organic phosphorus into inorganic forms accessible to plants [83,84]. In contrast, phosphorus-free BMPs may promote microbial growth and lead to phosphorus immobilization within microbial biomass, potentially reducing short-term phosphorus availability [39]. These contrasting responses highlight the importance of polymer composition in determining BMP effects on soil phosphorus cycling. Compared with CMPs, BMPs more readily accumulate microorganisms and support biofilm formation [82]. Interactions between negatively charged phosphate groups and positively charged functional groups of proteins—key components of biofilms—can enhance phosphorus adsorption onto BMP surfaces [82,85]. Weathering further modifies BMP surface properties by increasing surface area, reducing hydrophobicity, and increasing heterogeneity, while simultaneously releasing additional carbon substrates [86]. These changes promote microbial adhesion and biofilm development, thereby influencing phosphorus availability and turnover over time.

At the same time, recent evidence indicates that BMPs should not automatically be assumed to exert stronger effects on phosphorus availability than CMPs in all soils. In a short-term incubation, both PE and PLA affected phosphorus availability differently depending on fertilizer regime, and PE exerted a stronger influence than PLA under some conditions, with shifts in available phosphorus linked to changes in acid phosphatase activity and bacterial community composition [87]. This suggests that polymer-specific effects on phosphorus cycling arise not only from degradability itself, but also from interactions among surface chemistry, microbial enzyme responses, and background nutrient conditions.

Given the inherently low bioavailability of soil phosphorus [39,82], even modest BMP-induced shifts in microbial activity, enzyme production, and surface-mediated adsorption may have disproportionate effects on P cycling and plant P nutrition, particularly under long-term field conditions.

Alterations induced by BMPs extend across multiple dimensions of soil functioning, spanning physical structure, hydrological behavior, chemical conditions, and biogeochemical cycling. By reshaping aggregation, pore architecture, pH regulation, organic matter dynamics, and nutrient transformations, BMPs modify the environmental framework within which soil microorganisms and plant roots interact. These changes do not act in isolation but interact to redefine resource availability and habitat suitability at micro- and mesoscales. More broadly, BMP effects are best understood as the outcome of interactions between material traits and biological context. Polymer chemistry and additive composition determine degradability and the nature of released compounds; aging alters surface reactivity and microbial attachment; soil conditions regulate the accessibility of oxygen, water, and nutrients; and rhizosphere processes shape how microbial communities and plant roots respond to these altered microsites. Clarifying how such soil-level modifications propagate to microbial community assembly and plant–microbe interactions is therefore a critical next step and provides the basis for the microbial-focused discussion in the following section.

4. Biodegradable microplastics as ecological niches for microorganisms

Soil microorganisms occupy highly dynamic niches and rapidly respond to changes in habitat structure, resource accessibility, and plant-derived inputs. Within this context, BMPs represent a novel and heterogeneous component of the soil environment, providing new surfaces for attachment as well as localized sources of organic carbon. These features enable direct microbial colonization and biofilm development, giving rise to distinct microbial assemblages that differ from those in the surrounding soil matrix. This section explores how BMPs function as microbial niches, shaping plastisphere communities, influencing rhizosphere-associated microorganisms, and selecting for taxa capable of degrading biodegradable polymers.

4.1. Biofilm formation and microbial activity on biodegradable microplastic surfaces

The degradation pathways of BPs share similarities with those of CPs but differ substantially due to polymer chemistry and interactions with soil ecosystems [10]. While CPs primarily undergo abiotic degradation processes—such as photodegradation, thermal and mechanical fragmentation—accompanied by additive leaching and surface colonization by microorganisms, they generally persist in soils over long time scales [10,88]. In contrast, BPs are subject to both abiotic and biotic (microbial) degradation processes, leading to fragmentation, biofilm formation, enzymatic depolymerization, and, under favorable conditions, partial or complete mineralization to CO_2, water, and microbial biomass (Fig. 1) [10,11].

Degradation rates depend strongly on polymer composition, environmental conditions such as temperature, moisture, oxygen availability, and microbial activity, as well as the presence of additives [9]. Under industrial composting conditions (approx. 60 °C), some BPs can achieve substantial degradation within months [88]. For instance, PLA may reach approx. 60% degradation within 30 days under anaerobic conditions at 58 °C and approx. 84% within 58 days under aerobic composting [9]. In contrast, PBS requires substantially longer periods, reaching approx. 90% degradation only after 160 days under comparable conditions [9]. PBAT, a fossil-based aliphatic–aromatic polyester frequently blended with starch to improve flexibility and compostability [89], typically degrades more slowly owing to its aromatic ester bonds and linear polymer structure [90].

The biodegradability of plastics is largely governed by their molecular structure. Polymers containing oxygenated functional groups generally degrade more readily than CPs [90], while aliphatic polyesters such as PHA, PLA, PBS, and PCL are more susceptible to microbial depolymerization than aliphatic–aromatic polyesters such as PBAT, which are comparatively resistant to enzymatic attack [9,90]. Additives, including pro-oxidants, plasticizers, and photostabilizers, further modulate degradation behavior [12]. Pro-oxidants can promote oxidative chain scission under UV exposure, whereas plasticizers enhance flexibility but may reduce microbial accessibility to polymer chains [91]. The chemical nature of additives also influences microbial colonization, enzyme activity, and overall biodegradation efficiency [43]. Importantly, additives and degradation by-products can leach from plastics under natural soil conditions, with release rates enhanced by fluctuations in moisture and temperature, raising concerns about toxicity in soil ecosystems [92].

BMPs, owing to their relatively hydrophilic surfaces and higher carbon availability, readily support microbial colonization and biofilm formation (Fig. 2) [93]. Biofilm development alters particle properties by increasing surface hydrophilicity and effective density [94], which can enhance particle mobility and contaminant adsorption in soil environments [93,95]. As a result, BMP-associated biofilms may facilitate the transport of co-occurring pollutants through soil matrices [60,94,95].

Microbial enzymes embedded within these biofilms contribute to polymer degradation, although degradation efficiency varies strongly with polymer chemistry and environmental conditions [94,96]. At the same time, BMP-associated biofilms can act as reservoirs for pathogenic microorganisms and antibiotic resistance genes (ARGs), raising concerns about their potential role in contaminant and gene dissemination in soil ecosystems [97]. These findings indicate that biofilm formation on BMP surfaces creates functionally distinct microhabitats with implications for pollutant dynamics, microbial ecology, and soil health.

4.2. *Effects of biodegradable microplastics on rhizosphere microbial communities*

The rhizosphere—the soil zone directly influenced by plant roots—hosts highly diverse microbial communities, including bacteria, fungi, archaea, and protists engaged in symbiotic, commensal, or pathogenic interactions with plants [98]. Key functional groups include plant growth-promoting rhizobacteria (PGPR), such as *Pseudomonas fluorescens* and *Bacillus subtilis*, which enhance nutrient acquisition and pathogen resistance, as well as nitrogen-fixing bacteria including *Rhizobium* and *Frankia* that contribute to soil fertility [99–101]. Arbuscular mycorrhizal fungi (AMF) play a central role in phosphorus uptake and plant stress tolerance, while taxa such as *Burkholderia cepacia* and *Achromobacter xylosoxidans* contribute to nutrient cycling and disease suppression [101–103]. Because these communities are strongly shaped by root exudation and microscale soil conditions, they are particularly sensitive to environmental perturbations such as BMP exposure.

BMPs influence rhizosphere microbial communities both indirectly, by modifying soil physicochemical properties, and directly, by providing additional surfaces and labile carbon sources that support microbial colonization [104]. Root exudates further stimulate microbial activity at the soil–root interface, promoting the formation of distinct microbial assemblages associated with BMP surfaces, often referred to as the plastisphere [105]. Compared with CMPs, BMPs may therefore favor microbial taxa adapted to exploit polymer-derived substrates.

Empirical studies support this interpretation. Zhang et al. [63] showed that microbial community structure differed between soils amended with biodegradable polymers (PLA, PBS, PBAT) and those containing low-density polyethylene (LDPE), with BMPs supporting communities better adapted to utilize available carbon sources. In the same study, manure application increased bacterial alpha diversity on BMPs but reduced diversity on LDPE, highlighting strong interactions between organic amendments and plastic type. Likewise, soils amended

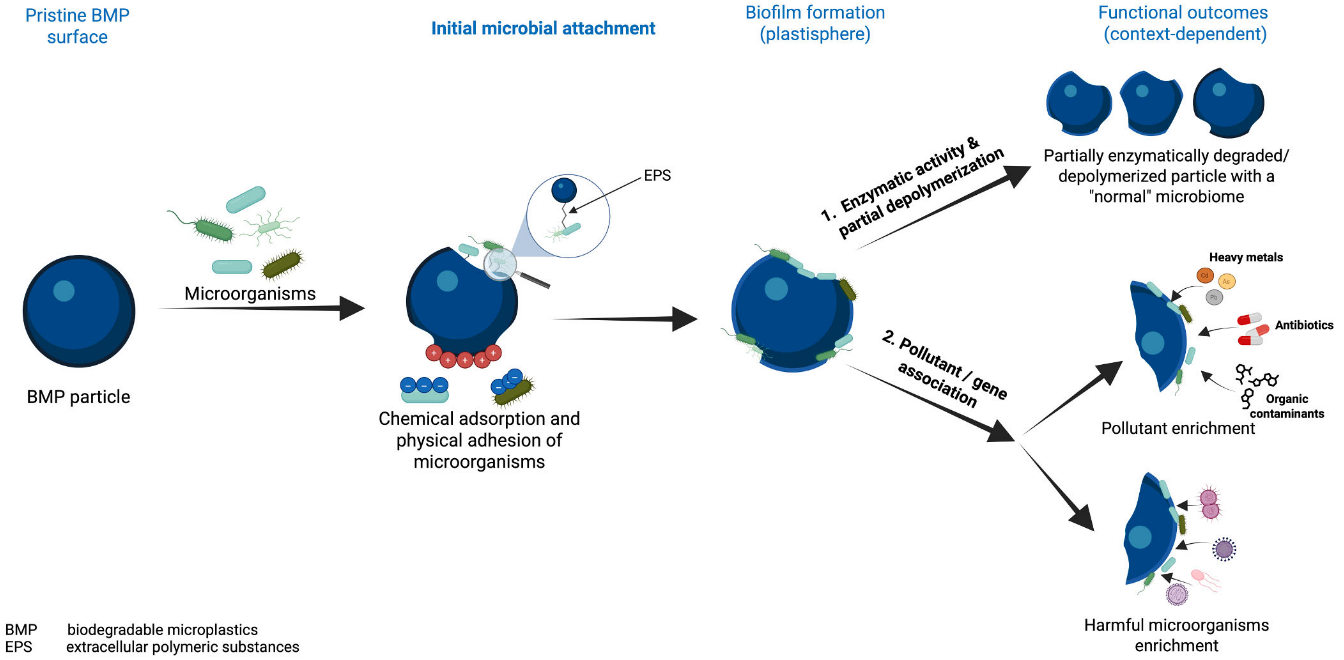


Fig. 2. Conceptual representation of microbial colonization and biofilm formation on biodegradable microplastic (BMP) surfaces (please note that the spherical shape was chosen for schematic purposes and is representative of all different shapes of microplastics). Following initial microbial attachment, BMPs support the development of biofilms ("plastisphere"), which modify particle surface properties and create functionally distinct microhabitats. These biofilms may facilitate enzymatic activity and partial polymer depolymerization, as well as the association of microorganisms, pollutants, and genetic material. The relative importance of these processes is context-dependent and governed by polymer type, environmental conditions, and microbial community composition.

with PLA and PBS showed enhanced microbial potential for exogenous carbohydrate and amino-acid uptake, whereas PE and polystyrene (PS) had little effect on functional diversity [106]. Together, these findings suggest that BMPs do not simply alter rhizosphere communities by their presence alone; rather, their effects depend on whether they act primarily as metabolized carbon-rich microsites or as physicochemical disturbances within the soil matrix.

Across studies, however, responses of bacterial alpha diversity to BMPs remain clearly context-dependent rather than unidirectional (Fig. 3). Although the most common outcome is no significant change, decreases are reported more frequently than increases overall, and negative responses become especially pronounced at higher concentrations (more than 1% w/w) and for certain particle-size classes (less than 50 µm). This trend is consistent with individual studies reporting diversity declines under relatively high BMP loads, including 2% PLA or PBAT [98], 5% PBAT [107], 5–10% PLA [108], 7% PLA [71], and 10% PHBV [39]. By contrast, increases in diversity were reported only in specific exposure scenarios, such as 1% PLA or PBS in surface soil [75], 0.02% PBAT in loessial soil [109], 1% PBAT in loamy soil [110], and 0.8% PBAT in standard natural soil [111]. At the same time, neutral or even positive responses occur under other conditions, indicating that alpha diversity alone does not capture the full ecological significance of BMP exposure. Rather, shifts in diversity should be interpreted alongside changes in community composition, substrate availability, and functional potential.

At the taxonomic level, some repeatedly responsive groups nevertheless emerge (Table 1). A formal meta-analysis of taxon-level rhizosphere responses was not feasible because the available studies differ widely in reported endpoints, taxonomic resolution, sequencing and bioinformatic workflows, and often lack the quantitative summary statistics required for effect-size calculation. We therefore used a structured qualitative synthesis and semi-quantitative comparison of response direction across polymer types, concentrations, and particle-size classes. Several studies reported increases in *Pseudomonadota* under PLA, PBS, PBAT, or mixed biodegradable polymers, suggesting that at least under some conditions BMP exposure may favor fast-growing taxa able to exploit newly available carbon substrates [75,98,109,113]. However, responses were not uniform across systems, as opposite trends were also reported for some polymer–soil combinations. Declines were more consistently observed for groups such as *Actinomycetota*, *Acidobacteriota*, *Nitrospirota*, and taxa affiliated with *Mycobacterium* and the family *Acidobacteriaceae* [49,102,109]. Fungal responses were likewise variable: some studies reported enrichment of *Ascomycota* or *Mortierellomycota*, whereas others observed declines in *Mortierellomycota*, *Mucoromycota*, or *Basidiomycota* [109]. These patterns indicate that BMPs do not simply increase or suppress rhizosphere microbial diversity, but rather restructure community composition by favoring some decomposer- or copiotroph-associated groups while disadvantaging others, depending on polymer chemistry, concentration, soil properties, and degradation stage.

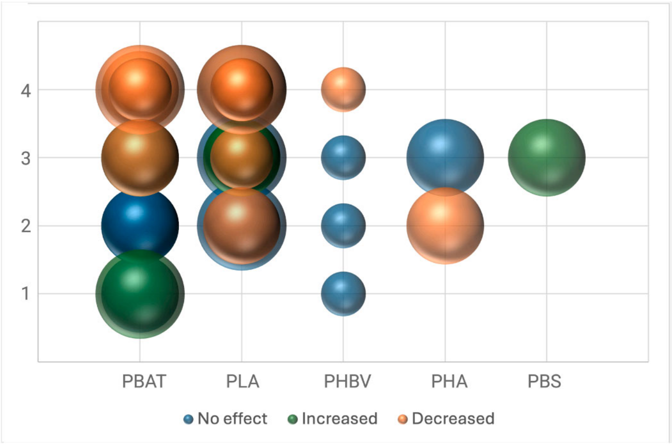


Fig. 3. Bubble graph summarizing reported bacterial alpha-diversity responses to biodegradable microplastics (BMPs) in rhizosphere-associated soils. Each bubble represents one literature observation and is classified by polymer type (x-axis), concentration class (y-axis), particle-size class (bubble size), and response direction (color). Numbers on the y-axis correspond to BMP concentration classes (% w/w): 1 = <0.05%, 2 = 0.05–0.5%, 3 = 0.5–1%, and 4 = >1%. Bubble size represents particle-size class, from smallest (<25 µm) to largest (>200 µm), with intermediate classes of 26–50 µm and 51–200 µm.

Microbial metabolic responses to BMPs are closely linked to polymer chemistry and environmental conditions. In general, higher BMP concentrations induce stronger shifts in bacterial community composition, as they represent more accessible microbial substrates. In addition, smaller particles may exacerbate toxicity due to increased surface area and enhanced interactions with soil constituents, further influencing microbial diversity [39,98,107]. BMPs such as PLA, PBAT, and PBS have been shown to reshape microbial community structure by altering carbon and nitrogen cycling pathways [63,93].

Functional consequences for nutrient cycling and plant–microbe interactions have also been reported. For example, PLA amendments increased soil urease activity by up to 104% while reducing NO_3^-–N content by 97%, accompanied by shifts in nitrogen-fixing bacterial communities and enrichment of symbiotic *Rhizobiales* [104]. This enrichment enhanced root nodulation in legumes such as peanut, potentially mitigating nitrogen limitation under MP exposure [104]. Conversely, other studies indicate that BMPs can hinder PGPR colonization and nutrient bioavailability, thereby disrupting beneficial symbioses and reducing overall microbial functionality [104,114].

BMPs may also alter microbial competition by favoring plastic-degrading taxa over native soil microorganisms, with potential consequences for both biodegradation efficiency and ecosystem stability [115]. Such shifts may have cascading effects on microbial network structure and resilience, particularly under long-term or repeated BMP inputs.

Fungal communities, particularly AMF, are critical for soil structure and nutrient acquisition—especially phosphorus—but are sensitive to environmental stressors [101]. BMPs, alone or in combination with co-contaminants such as cadmium (Cd) or nanoscale ZnO, have been shown to alter AMF community composition and diversity [116,117]. Responses depend strongly on polymer type and concentration, with moderate PLA inputs stimulating organic acid secretion and phosphorus mobilization, whereas higher BMP loads suppress fungal growth and reduce AMF abundance [116,118]. AMF may also play a protective role by limiting MP translocation within plant tissues and buffering nutrient imbalances induced by MPs [15]. Recent evidence suggests that BMPs such as PBAT and PLA can enhance fungal diversity and network stability relative to CPs, indicating more complex and nuanced interactions than previously assumed [119].

Because root exudates supply low-molecular-weight carbon compounds and organic acids, they can either amplify or buffer BMP effects depending on polymer type, degradation state, and rhizosphere conditions. In rhizosphere soils, these exudates may stimulate microbial activity and favor taxa able to exploit both plant-derived and polymer-derived substrates, while also intensifying nutrient competition and reshaping microbial assembly on BMP surfaces. This may help explain why microbial and enzymatic responses to BMPs are often stronger, and sometimes differ in direction, in rhizosphere soil than in bulk soil. As a result, some taxa and functions may be promoted whereas others are suppressed, leading to shifts in community composition, nutrient cycling, and plant–microbe interactions. The long-term ecological consequences of these changes remain uncertain and require integrative approaches combining high-resolution molecular, biochemical, and functional analyses under environmentally realistic conditions [120].

Table 1
Taxon-level shifts in rhizosphere microbial communities reported under biodegradable microplastic (BMP) exposure.

BMPs type	Concentration (% w/w)	Size	Soil type[a]	Exposure time (days)	Affected taxa[b]	Observations[c]	Reference
PLA and PBAT	2	approx. 50 µm	Agricultural soil	70	*Pseudomonadota*	↑	[98]
					Verrucomicrobiota	↓	
					Acidobacteriota	↓	
					Bacteroidota	↓	
PBS	1	150–180 µm	Surface soil	60	*Actinomycetota*	↓	[75]
					Pseudomonadota	↑	
PLA	1	150–180 µm	Surface soil	60	*Actinomycetota*	↓	[75]
					Pseudomonadota	↑	
PLA	10	approx. 67 µm	Loamy soil	100	*Neobacillus vireti*	↑	[112]
					Bacillota	↑	
					Nitrospirota	↓	
					Nitrospira	↓	
PHA	0.1	approx. 100 µm	N.A.	60	*Mycobacterium*	↓	[102]
					Acidobacteriaceae	↓	
PLA	1	approx. 100 µm	N.A.	60	*Mycobacterium*	↓	[102]
					Acidobacteriaceae	↓	
PBSA	1	approx. 100 µm	N.A.	60	*Mycobacterium*	↓	[102]
					Acidobacteriaceae	↓	
PBAT	1	approx. 100 µm	N.A.	60	*Mycobacterium*	↓	[102]
					Acidobacteriaceae	↓	
PLA	0.1; 1; 5; 10	approx. 40 µm	Farmland soil	30	*Ascomycota*	↑	[108]
					Mortierellomycota	↓	
					Mucoromycota	↓	
					Basidiomycota	↓	
PBAT	0.02; 0.2; 2; 5	<1000 µm	Loessial soil	120	*Gemmatimonadota*	↑	[109]
					Pseudomonadota	↑	
					Bdellovibrionota	↑	
					Myxococcota	↑	
					Patescibacteria	↑	
					Nitrospirota	↓	
PBAT	0.025; 0.05; 0.2; 0.8	approx. 131 µm	Loamy soil	77	*Planctomycetota*	↑	[111]
					Bacillota	↓	
					Planctomycetota	↑	
					Ascomycota	↑	
					Mortierellomycota	↑	
PHBV	0.01; 0.1; 1; 10	1–15 µm	N.A.	56	*Bacillota*	↓	[39]
					Gemmatimonadota	↓	
					Verrucomicrobiota	↑	
					Acidobacteriota	↑	
PLA	2	20–50 µm	Rice paddy field soil	70	*Proteobacteria*	↓	[57]
PBAT	0.1; 1; 5	approx. 100 µm	Farmland soil	120	*Pseudomonadota*	↑	[107]
					Actinomycetota	↓	
PLA	1	35–48 µm	Farmland soil	28	*Bacillota*	↑	[49]
					Actinomycetota	↓	
					Acidobacteriota	↓	
					Desulfobacterota	↓	
PLA	5	<20 µm	Agricultural soil	60	*Pseudomonadota*	↑	[113]
					Bacillota	↑	
PBAT	0.1; 1	<1200 µm	Loamy soil	28	*Caulobacteraceae*	↑	[110]
PLA	2; 7	150–250 µm	Agricultural soil	90	*Actinomycetota*	↑	[71]
					Myxococcota	↓	
					Bacteroidota	↓	
					Chloroflexota	↓	
					Gemmatimonadota	↓	

[a] N.A. = no available data in the study

[b] Phylum names were updated to current nomenclature where applicable; lower-rank taxa are given as reported in the original studies.

[c] increase (green), decrease (red); significant difference between treatments with and without BMPs at $p < 0.05$.

4.3. Plastic-degrading microorganisms and associated enzymatic mechanisms

Numerous bacterial and fungal taxa have been identified as capable of degrading BPs under soil or laboratory conditions (Table 2). Among bacteria, strains belonging to *Celeribacter*, *Bacillus*, *Paenibacillus*, *Streptomyces*, *Terribacillus*, and *Bacillus megaterium* have demonstrated the ability to depolymerize polymers such as PBAT, PCL, PLA, and PBS [121–125]. Fungal degraders are similarly diverse and include genera such as *Penicillium*, *Aspergillus*, *Thermomyces*, *Fusarium*, *Cladosporium*, *Mortierella*, and *Doratomyces*, many of which are common soil saprotrophs with broad enzymatic capabilities [122,126–128].

Beyond taxonomic identification, several studies have focused on the characterization of enzymes responsible for polymer depolymerization. For example, intracellular depolymerases from *Rhodospirillum rubrum* play a key role in polyhydroxybutyrate (PHB) degradation [136], while a PCL-specific depolymerase has been isolated from *Streptomyces thermoviolaceus* subsp. *thermoviolaceus* 76T-2 [134]. Additional enzymes implicated in biodegradable plastic degradation include lipases (*Alcaligenes faecalis*), esterases (*Comamonas acidivorans*, *Bacillus subtilis*), proteinase K (*Tritirachium album*), proteases (*Bacillus licheniformis*), and serine hydrolases from *Pestalotiopsis microspora* [137]. These enzymes differ in substrate specificity, efficiency, and environmental stability, highlighting the biochemical diversity underpinning BMP degradation.

The degradative capacity of microorganisms is commonly assessed using clear-zone assays, in which the formation of halos around colonies on polymer-containing agar indicates extracellular enzymatic activity [138]. Although semi-quantitative, these assays provide a rapid screening tool for identifying potential degraders. However, degradation efficiency is strongly influenced by environmental conditions, microbial community composition, and polymer properties, including crystallinity and additive content [136].

These findings indicate that BMP degradation is mediated by diverse microbial taxa and enzymatic pathways rather than by a limited set of specialist organisms. Linking taxonomic identity with functional gene expression and in situ degradation rates remains a key challenge. Future research should therefore integrate metagenomic, transcriptomic, and biochemical approaches to better resolve the ecological relevance and environmental efficiency of BMP-degrading microorganisms under realistic soil conditions.

Table 2
Microorganisms reported to degrade biodegradable plastics (BPs) under soil or soil-derived experimental conditions.

Microorganisms	BPs type	Shape of BPs[a]	Reference
Bacteria			
Celeribacter flavus	PBS	Film	[125]
Actinomadura keratinilytica	PLA	Pellet	[129]
Bacillus sp.	PLA	N.A.	[124]
Bacillus licheniformis, *Laceyella sacchari,* *Thermoactinomyces vulgaris,* *Thermophilic actinomycete,* *Nonomuraea fastidiosa,* *Nonomuraea terrinata,* *Micromonospora viridifaciens,* *Micromonospora echinospora,* *Actinomadura keratinilytica*	PLA	Pellet	[130]
Bacillus sp.	PBAT	Film	[122]
Terribacillus goriensis	PBS	Pellets	[121]
Bordetella petrii	PLA	Pellet	[131]
Amycolatopsis sp.	PLA, PCL	Pellets and films	[132]
Laceyella sacchari	Starch-based	Pellet	[133]
Pseudomonas aeruginosa, *Bacillus subtilis*	PHA	Film	[127]
Streptomyces thermoviolaceus	PCL	Pellet	[134]
Streptomyces coelicoflavu, *Variovorax*	PHB/PBAT	Film	[128]
Peribacillus frigoritolerans	PBAT	Film	[135]
Bacillus megaterium, *Alcaligenes aquatilis,* *Shewanella haliotis*	PCL	Pellets	[123]
Fungi			
Candida albicans, *Fusarium oxysporum*	PHA	Film	[127]
Fusarium solani, *Purpureocillium lilacinu,* *Aspergillus insuetu,* *Clonostachys* sp.	PHB/PBAT	Film	[128]
Apophysomyces thailandensis	PLA, PBS	Pellets and films	[132]
Cladosporium sp. *Purpureocillium* sp.	PLA, PBS, PCL	Pellets and films	[132]
Fennellomyces linderi, *Fusarium solani,* *Doratomyces microspores,* *Mortierella sp.,* *Aspergillus fumigatus,* *Thermomyces lanuginosus*	PLA	Filaments	[126]
Filobasidium uniguttalatum	PCL	Pellets	[123]
Geomyces sp.	PCL	Film	[122]
Fusarium sp., *Sclerotinia* sp.	PBSA, PCL	Film	[122]
Alternaria sp.	PBS	Film	[122]

[a] N.A. - no available data in the study

4.4. Biodegradable microplastics as reservoirs and vectors for soil–plant pathogens

BMPs can also function as carriers for pathogenic microorganisms, facilitating their persistence, dissemination, and colonization within soil–plant systems. Pathogenic fungal taxa, including *Candida*, *Fusarium*, and *Rhodotorula*, have frequently been detected on plastic surfaces used in medical, industrial, and household contexts, indicating that plastics can provide suitable substrates for pathogen attachment and survival [139]. In soils exposed to elevated plastic inputs, BMPs may therefore contribute to the accumulation and spread of plant-associated fungal pathogens, with potential consequences for rhizosphere stability and crop productivity [74].

Beyond acting as passive carriers, BMPs may also influence pathogen dynamics indirectly through their interactions with plant roots and rhizosphere processes. Pathogen-induced root damage and associated physiological changes can modify BMP uptake and distribution, while alterations in root exudation patterns may affect BMP retention and availability within the soil–plant continuum [140]. In parallel, pathogen-driven shifts in rhizosphere microbial communities can further modulate BMP behavior and persistence in soil.

As discussed previously, the relatively hydrophilic surfaces of BMPs and their capacity to release bioavailable carbon promote microbial colonization and biofilm formation [93]. Once coated with biofilms, BMPs exhibit increased adsorption capacity for both contaminants and microorganisms, including pathogens, which may enhance their mobility within soil and facilitate pathogen transport toward plant roots [93,95]. In this way, BMPs can act as artificial microbial habitats that concentrate and protect pathogenic taxa, potentially amplifying plant disease risk and posing broader ecological and human health concerns [39,139].

It is important to note, however, that much of the available evidence is derived from short-term experiments using freshly prepared BMPs under controlled conditions. The extent to which these pathogen-related effects persist under long-term field exposure, aging, and repeated BMP inputs remains largely unresolved and represents a critical knowledge gap [120]. In addition to their potential to associate with pathogenic organisms under certain environmental conditions, BMP-associated biofilms may also create microenvironments that facilitate genetic exchange among co-occurring microbes, including the transfer of virulence traits and resistance determinants, a topic addressed in the following section.

4.5. Genetic exchange and resistome dynamics within biodegradable microplastic- associated microbiomes

BMPs have been increasingly implicated in shaping genetic exchange processes within soil microbial communities, particularly with respect to antibiotic resistance genes (ARGs) and virulence factors (VFs). Several studies report higher abundances of ARGs and VFs on BMP-associated microbiomes compared with bulk soil or CMP controls, suggesting that BMP surfaces can act as localized hotspots for gene accumulation under certain conditions [141]. For example, PBS-associated plastispheres showed enrichment of ARGs and VFs, with genera such as *Rhizobium* and members of *Amoebozoa* identified as key hosts [141].

Metagenomic analyses further indicate that BMPs can restructure microbial communities and promote gene exchange by providing rough, heterogeneous surfaces that enhance microbial attachment and close cell–cell proximity within biofilms—conditions known to favor horizontal gene transfer (HGT) [73,141]. Given that soils already represent major reservoirs of antibiotic resistance [142], BMP-associated biofilms may intensify local gene exchange processes, raising concerns about the potential emergence and persistence of highly resistant or virulent strains under favorable environmental conditions [142].

Specific host–gene associations have been documented across different polymer types. *Bacillus cereus* has been identified as a carrier of ARGs and metal resistance genes (MRGs) on PHA and PHB surfaces [143], while PLA/PBAT plastispheres were enriched in genes linked to carbon and nitrogen cycling, primarily within *Proteobacteria* [8]. Experimental studies further demonstrate that PLA and PHA nanoparticles can enhance ARG transfer in *Colibacillus* [144], and that PLA and PBS selectively alter ARG and VF profiles in *Bradyrhizobium* and *Pseudomonas* communities [141,145]. Comparative analyses also show polymer-specific resistome signatures, with PHA-associated *Pseudomonas* exhibiting distinct multidrug resistance patterns relative to polyethylene terephthalate (PET) [146], and PCL particles facilitating elevated HGT potential in mangrove soils [78]. PLA and PBAT surfaces have likewise been reported to host dense microbial biomass enriched in ARGs, particularly among *Actinomycetota*, *Flavobacteriaceae*, and *Pseudomonas* [37,112].

Beyond microbial composition, ARG dynamics on BMPs are strongly modulated by environmental co-stressors. Heavy metals, antibiotics, and pesticides can adsorb to BMP surfaces, increasing selective pressure and favoring the proliferation of ARG-bearing taxa [147]. In addition, differences in BMP degradation rates influence carbon release patterns, with downstream effects on microbial metabolism and gene regulation [145]. Particle size further modulates these processes, as smaller MPs and nanoplastics (NPs) may induce oxidative stress and increase membrane permeability, thereby facilitating gene uptake and transfer [148].

Across the processes discussed in this section, BMPs emerge as structurally and chemically distinct microhabitats that reorganize microbial life in soils. By providing new surfaces, releasing labile carbon during degradation, and supporting biofilm formation, BMPs influence microbial community assembly, functional activity, and genetic exchange in ways that differ from both bulk soil and CPs. The direction and magnitude of these effects vary with polymer chemistry, particle size, aging state, soil physicochemical properties, and co-occurring stressors, and can range from stimulation of specific microbial functions to disruption of established microbial interactions. Importantly, the microbial niches associated with BMPs represent dynamic interfaces where biogeochemical cycling, degradation processes, and potential ecological risks converge. How these microbially mediated changes propagate beyond the microbial compartment—particularly toward plant performance and stress responses—forms the basis of the following section.

5. Plant responses to biodegradable microplastics in soil systems

BMPs influence plant performance primarily through indirect pathways operating at the soil–microbe–root interface. As outlined in Sections 2 and 3, BMPs alter soil physical structure, water dynamics, nutrient availability, and microbial community composition. These changes propagate into the rhizosphere, where plant roots interact closely with microorganisms and root-derived metabolites. Consequently, plant responses to BMP exposure are largely mediated by integrated soil–microbial processes rather than by direct plastic–plant contact (Fig. 4).

5.1. Rhizosphere-mediated pathways linking biodegradable microplastics and plant performance

Root exudates are key regulators of rhizosphere interactions and play a pivotal role in plant responses to environmental stressors, including MP contamination [149]. These exudates, which are highly dynamic in composition, shape rhizosphere microbial communities by selectively recruiting beneficial or antagonistic taxa and by modulating microbial metabolic activity [150]. Through these feedbacks, plants indirectly influence nutrient cycling, pathogen suppression, and stress resilience.

The composition of rhizosphere microbial communities strongly affects plant health through mechanisms such as enhanced nutrient mobilization, plant growth promotion, and disease resistance. Beneficial taxa, including *Bacillus*, *Pseudomonas*, and *Lysobacter*, are commonly associated with healthy rhizospheres and contribute to nitrogen, phosphorus, and potassium availability [151,152]. In contrast, shifts toward pathogenic or functionally less beneficial microbial groups can reduce nutrient availability and negatively impact plant growth and productivity [153].

BMP-induced changes in soil structure, moisture regime, and microbial composition can therefore indirectly reshape rhizosphere functioning. By modifying microbial recruitment patterns and altering competitive interactions within the rhizosphere, BMPs influence plant performance through pathways that integrate physical, chemical, and biological processes. Such indirect pathways are expected to dominate because plant roots are primarily exposed to BMP-modified soil environments and microbially processed metabolites, rather than to the polymer particles themselves. Plant responses to BMPs may also involve shifts in nutrient-foraging strategies, as recent work showed that BPs can promote the invasion of *Trifolium repens* by altering plant acquisition of nitrogen and phosphorus, highlighting that BMP effects may differ strongly among species depending on nutrient-use strategy and competitive context [101].

5.2. Effects on plant growth and biomass allocation

A consistent outcome across studies is the sensitivity of plant growth and biomass allocation to BMP exposure. Suppressed shoot and root growth have been reported for multiple species exposed to PBAT-, PHB-, and PLA-based MPs, with inhibitory effects frequently observed between 0.25% and 4.5% w/w [92,154,155].

Pronounced growth inhibition has been observed in horticultural crops. Cherry tomato exhibited up to a 90% reduction in shoot biomass when exposed to PHB-based mulch films, while PBAT–starch blends significantly reduced biomass and leaf area in lettuce and maize [92,154]. Root systems appear particularly vulnerable: reductions in radicle length and root biomass have been documented in lettuce exposed to PBAT across all tested concentrations [156,157] as well as in cucumber and maize following PLA exposure [108,158].

Similar trends were reported by Pelko et al. [36], who observed reduced shoot and root biomass in sunflower grown in soil amended with PBAT-based MPs, whereas PE MPs had comparatively minor effects under the same conditions. These findings align with broader evidence that biomass allocation patterns are sensitive to BMP type and concentration, likely reflecting indirect effects mediated by changes in soil structure, microbial activity, and nutrient availability rather than direct phytotoxicity alone.

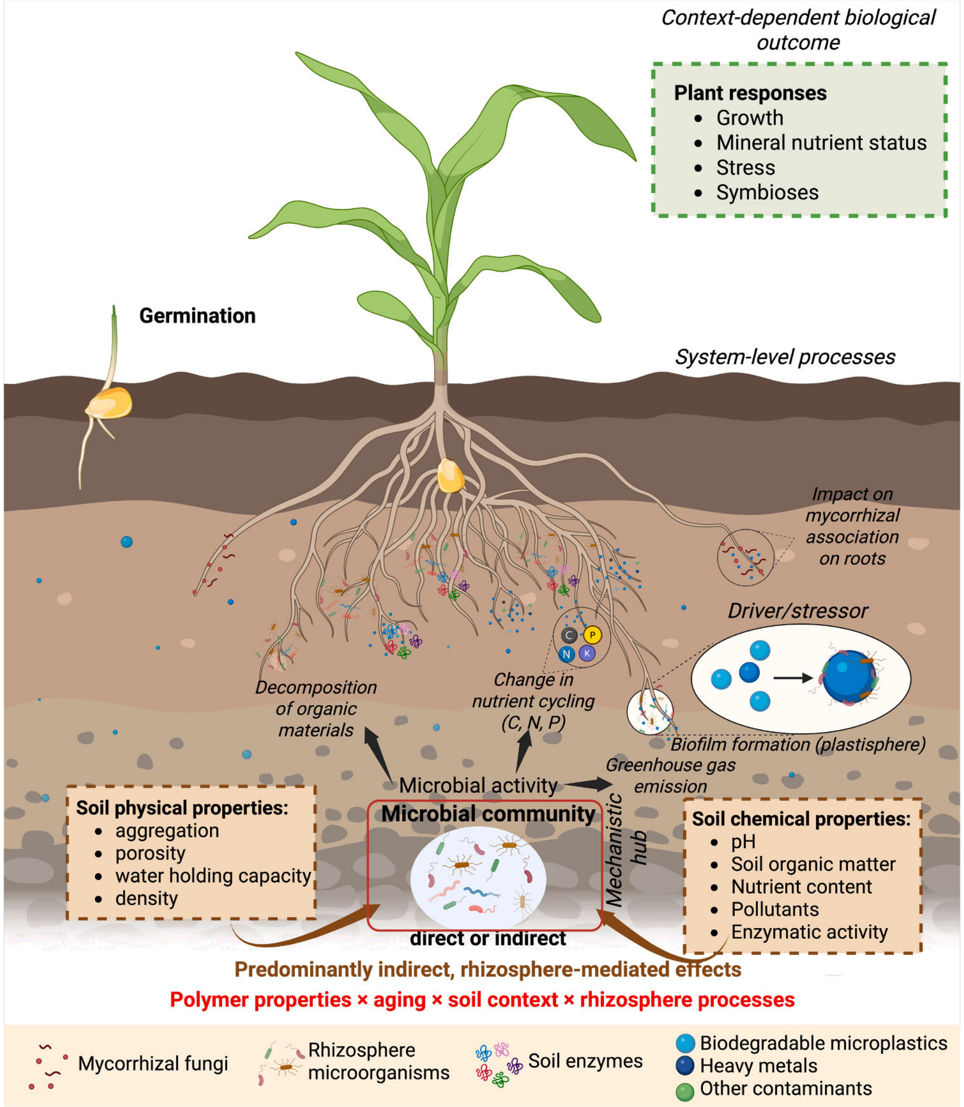


Fig. 4. Conceptual framework of biodegradable microplastic (BMP) effects on the plant–soil–microbiome system. BMPs act as context-dependent drivers of soil–plant–microbe interactions, modifying soil physical and chemical properties and reshaping microbial communities and activity in the rhizosphere. These changes influence key processes, including organic matter decomposition, nutrient cycling (C, N, P), enzyme activity, greenhouse gas emissions, and contaminant dynamics. Plant responses are predominantly indirect and mediated through rhizosphere processes and microbial interactions, rather than direct contact with plastic particles. The magnitude and direction of these effects are governed by polymer properties, aging, soil conditions, and plant–microbe interactions.

5.3. *Physiological and biochemical stress responses in plants*

BMP-induced changes in the rhizosphere can also trigger physiological and biochemical stress responses in plants. Altered microbial communities and soil conditions influence the accumulation of reactive oxygen species (ROS), which function both as signaling molecules and indicators of cellular stress [36,154]. Elevated ROS levels have been reported in cucumber, thale cress, rice, and maize exposed to PBAT or PLA MPs [135,154,158,159].

In parallel, malondialdehyde (MDA), a marker of lipid peroxidation and membrane damage, increased in response to PBAT exposure in several species [135,159]. Antioxidant enzyme responses are variable and speciesdependent. Superoxide dismutase (SOD) activity often remains unchanged, although increases have been observed at higher BMP concentrations [100,160]. Peroxidase (POD) activity is generally stable but increased in maize exposed to high PLA levels [108]. In contrast, CAT activity declined in cucumber at elevated PLA concentrations, suggesting partial disruption of oxidative defense systems [158].

Osmotic stress markers show less consistent responses. Proline accumulation did not change significantly in some studies involving PBAT- and PHB-based materials [92], whereas moderate increases were reported under PBAT exposure in sunflower [36], suggesting that osmotic stress responses vary with polymer type, exposure level, and plant species. Overall, these physiological responses are most consistently linked to BMP-induced changes in rhizosphere conditions and microbial activity, rather than to direct interactions between plant tissues and plastic particles.

5.4. *Nutrient assimilation and metabolic constraints*

Beyond growth and stress signaling, BMP exposure can interfere with nutrient assimilation and metabolic balance. Reductions in leaf nitrogen and carbon content were observed in maize exposed to PLA MPs, suggesting impaired nutrient uptake or altered metabolic allocation [108]. Similar disruptions were reported in rice exposed to PBAT, accompanied by reduced lignin content and increased oxidative stress [159].

Pelko et al. [36] likewise reported selective changes in mineral nutrient profiles in sunflower, including altered manganese accumulation in roots under PBAT exposure, while most macronutrients remained relatively stable. These findings suggest that BMPs influence nutrient availability through changes in soil structure, sorption processes, and microbial and enzymatic regulation of nutrient transformations, resulting in variable nutrient responses rather than uniformly negative effects on plant nutrition.

Current evidence suggests that plant responses to BMPs are driven predominantly by indirect, rhizosphere-mediated pathways. By altering soil structure, microbial community composition, and nutrient fluxes, BMPs reshape the biological and physicochemical environment experienced by plant roots. Direct phytotoxic effects cannot be excluded, particularly at high concentrations or for small particles, but available evidence indicates that they are usually secondary to rhizosphere-mediated mechanisms. The relative importance of these pathways is likely to vary with polymer properties, exposure regime, and plant species. Taken together, the diversity of reported responses highlights the need to assess BMP effects within integrated soil–microbiome–plant systems rather than through plant endpoints alone.

6. Synthesis, knowledge gaps, limitations of current evidence, and future research perspectives

The increasing use of BPs as alternatives to conventional polymers has shifted environmental concern from persistence alone to the consequences of degradation within soil systems. Evidence reviewed here shows that once BPs fragment into BMPs, they become active components of the soil environment rather than transient residues. Their presence influences soil structure, microbial habitats, and

Table 3

Impact of biodegradable microplastics (BMPs) on growth, physiological, and biochemical parameters of different plant species.

Polymer type	Exposure level (%, w/w)	Plant species	Exposure time (days)	Effects on growth parameters	Effects on physiological parameters	Effect on biochemical parameters	Reference
PBAT	0.01, 0.1, 1	Barley, *Hordeum vulgare*	14	Decrease of fresh shoot biomass at 0.1% and 1% w/w, no effect on specific leaf area	No effect on chlorophyll content	N.A.	[157]
PBAT	0.01, 0.1, 1	Lettuce, *Lactuca sativa*	21	Decrease in root length at all concentrations; decrease in fresh biomass at 1% w/w, reduced number of leaves at 0.01% and 0.1%, no effect on specific leaf area	Increased chlorophyll content at 1%	N.A.	[157]
PBAT	0.05, 0.5, 1	Sunflower, *Helianthus annuus*	56	Reduced shoot and root biomass at 0.5% and 1.0%	No effect on transpiration and photochemical efficiency; no changes on water capacity	Decreased proline content at 0.05% and 1.0%, different changes in mineral composition	[36]
PHB	4.5	Cherry tomato, *Lycopersicon esculentum* cv. *cerasiforme*	49	Slower shoot growth; 90% decrease in shoot biomass	Decreased chlorophyll content	No changes in proline accumulation	[92]
PBAT-potato starch blend	4.5	Cherry tomato, *Lycopersicon esculentum* cv. *cerasiforme*	49	Lower leaf canopy; 30% decrease in shoot biomass	Decreased chlorophyll content	No changes in proline accumulation	[92]
PBAT-potato starch blend	4.5	Lettuce, *Lactuca sativa*	35	70% decrease in shoot biomass	Decreased chlorophyll content	No changes in proline accumulation	[92]
PBAT-PLA blend	4.5	Cherry tomato, *Lycopersicon esculentum* cv. *cerasiforme*	49	No effect	No effect on chlorophyll content	No changes in proline accumulation	[92]
PBAT-PLA blend	4.5	Lettuce, *Lactuca sativa*	35	40% decrease in shoot biomass	No effect on chlorophyll content	No changes in proline accumulation	[92]
PLA	0.02, 0.2, 2	Cucumber, *Cucumis sativus*	21	Decrease in dry root biomass at 0.02% and 0.2%; decrease in shoot biomass at 2%; reduction of leaf area at 0.2% and 2%; decreased width of the plant basal stem at 0.2%	Decreased chlorophyll content at 2%	Increased ROS at 0.02% and 0.2%; no effect on superoxide dismutase (SOD); no effect on peroxidase (POD); decrease catalase (CAT) activity at 2%	[158]
PBAT	0.25, 0.5, 1, 2	Thale cress, *Arabidopsis thaliana*	49	Decrease in dry shoot weight at 0.25%, 0.5% and 1%; reduction of leaf area at 0.25%, 0.5% and 1%; all plants died at 2.0%	N.A.	Increased ROS; increased MDA; no effect on POD; no effect on SOD	[135]
PLA	0.1, 1, 5, 10	Maize, *Zea mays*	30	Decreased root and shoot biomass at 1%, 5% and 10%	Decreased chlorophyll content at 1%, 5% and 10%	Decreased leaf C at 5% and 10%; decreased leaf N at 1%, 5% and 10%; increased POD at 10%; no effect on SOD; no effect on MDA	[108]
PBAT	1	Rice cv. Ruanhuayou 1179	60	Decreased root and shoot dry weight; decreased plant height	Decreased chlorophyll content	Increased ROS; increased MDA; decreased lignin content	[159]
PBAT-PLA blend	0.1, 1, 10	Maize, *Zea mays*	30	Reduced leaf area at all concentrations, inhibitory effect on leaf growth	Decreased chlorophyll content at all concentrations	Increased ROS at all concentrations; SOD increased at 10%	[154]

N.A. = no available data in the study

biogeochemical processes in ways that depend strongly on polymer properties, environmental conditions, and biological context. Compared with CPs, BMP effects are more strongly linked to degradation-related changes in surface chemistry, carbon release, microbial colonization, and biofilm formation. These processes can stimulate or suppress microbial activity, alter nutrient transformations, and, under certain conditions, create localized microenvironments that facilitate pathogen persistence or genetic exchange. Plant responses therefore often emerge indirectly through modified soil and rhizosphere conditions rather than through direct effects alone.

Despite the rapid growth of this research field, important uncertainties remain. Most available studies are short-term laboratory experiments using pristine materials and simplified substrates, which limits inference about the long-term ecological significance of aged BMPs under realistic field conditions. A further limitation is the low comparability among existing studies. Current studies vary substantially in polymer type and formulation, particle size and shape, aging status, exposure metrics, and measured endpoints. This heterogeneity also limits the feasibility of formal meta-analysis across soil, microbial, and plant-response endpoints, as many studies are not designed around comparable controls, response variables, or reporting formats needed for robust effect-size calculation. In addition, the literature remains strongly biased toward a few polymers, particularly PLA and PBAT, while many other biodegradable materials are still poorly studied. Existing evidence also tends to report shifts in microbial diversity or enzyme activity without consistently linking these responses to ecosystem-level functions such as nutrient retention, disease suppression, greenhouse gas emissions, or long-term plant productivity.

Addressing these gaps will require long-term, field-relevant studies using weathered materials, repeated exposure scenarios, and standardized reporting of polymer chemistry, additive composition, particle properties, aging status, soil characteristics, and exposure units. Such standardization will be essential for improving comparability across studies, supporting future quantitative synthesis, and distinguishing polymer-specific effects from broader experimental noise. Future research should also move beyond descriptive observations toward stronger functional integration by directly linking plastisphere and rhizosphere responses with measurable ecosystem processes, including carbon, nitrogen, and phosphorus cycling, plant nutrient acquisition, disease dynamics, and greenhouse-gas fluxes. Greater attention is also needed for underrepresented polymers, the formation of NPs during degradation, co-exposure with other stressors, and the role of BMPs in pathogen persistence and horizontal gene transfer. Co-exposure scenarios, including the coexistence of BMPs with soil amendments such as biochar, also require further attention because these interactions may substantially modify SOM decomposition and carbon dynamics [161].

Overall, BPs should not be assumed to represent a uniformly low-risk alternative in soil environments. Their ecological significance depends on how material design, degradation behavior, environmental conditions, and soil biological processes interact over time. Assessing these risks will require integrated, environmentally realistic approaches that treat BMPs as active components of soil–microbe–plant systems rather than as transient by-products of degradation.

Environmental implication

Biodegradable microplastics (BMPs) are increasingly promoted as safer alternatives to conventional plastics, yet evidence remains fragmented across soil physicochemical, microbial, and plant-focused studies, potentially underestimating environmental risks. By integrating these perspectives, this review shows that BMPs act as system-level modifiers of soil–microbe–plant interactions. They reshape soil microhabitats, redirect microbial processes involved in carbon and nutrient cycling, and alter rhizosphere function, ultimately affecting plant performance. This synthesis reframes biodegradability from a descriptor of persistence to a functionally defined hazard and highlights key pathways by which BMPs may affect soil ecosystem services, while identifying critical gaps in field-relevant, long-term risk assessment.

CRediT authorship contribution statement

Anita Jemec Kokalj: Writing – review & editing, Writing – original draft, Visualization, Methodology, Funding acquisition, Conceptualization. **Marjana Regvar:** Writing – review & editing, Conceptualization. **Teja Pelko:** Writing – review & editing, Writing – original draft, Visualization, Methodology, Conceptualization. **Marina Dermastia:** Writing – review & editing. **Katarina Vogel-Mikuš:** Writing – review & editing, Visualization, Methodology, Funding acquisition, Conceptualization.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper

Acknowledgments

This work was co-funded by the Slovenian Research and Innovation Agency (ARIS) by Research program P1–0212 (Plant biology), Research program P1–0184 (Integrative zoology and speleobiology), and project MicroBioPlast J1–50014. Teja Pelko receives the Young researcher grant from ARIS.

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.jhazmat.2026.142138.

Data Availability

Data will be made available on request.

References

[1] Thompson, R.C., Courtene-Jones, W., Boucher, J., Pahl, S., Raubenheimer, K., Koelmans, A.A., 2024. Twenty years of microplastic pollution research-what have we learned? Science 386, eadl2746. https://doi.org/10.1126/science.adl2746.
[2] ISO/DIS 24187(en), Principles for the analysis of plastics and microplastics present in the environment. ⟨https://www.iso.org/obp/ui#iso:std:iso:24187:dis:ed-1:v1:en⟩. Accessed 21 Mar 2025.
[3] Cui, G., Wu, X., Bhat, S.A., Li, F., He, P., Xu, Q., 2024. Dynamics of Biodegradable Plastics in the Process of Food Waste Biotreatment and Environmental Risks of Residual Plastic Fragments. Management of Micro and Nano-plastics in Soil and Biosolids. Springer Nature Switzerland, Cham, pp. 315–324.
[4] Büks, F., Kaupenjohann, M., 2020. Global concentrations of microplastics in soils – a review. SOIL 6, 649–662. https://doi.org/10.5194/soil-6-649-2020.
[5] Johansen, J.L., Magid, J., Vestergård, M., Palmqvist, A., 2024. Extent and effects of microplastic pollution in soil with focus on recycling of sewage sludge and composted household waste and experiences from the long-term field experiment CRUCIAL. TrAC Trends Anal Chem 171, 117474. https://doi.org/10.1016/j.trac.2023.117474.
[6] Zhou, Y., Du, Y., Sheng, J., Liu, Y., Wan, C., Dong, H., Gu, J., Long, H., Zhang, H., 2025. Assessment of microplastic ecological risk and environmental carrying capacity of agricultural soils based on integrated characterization: a case study. Sci Total Environ 960, 178375. https://doi.org/10.1016/j.scitotenv.2025.178375.
[7] Togliatti, E., Pugliese, D., Giubilini, A., Messori, M., Milanese, D., Sciancalepore, C., 2022. Novel PBAT-based biocomposites reinforced with bioresorbable phosphate glass microparticles. Macromol Symp 405. https://doi.org/10.1002/masy.202100238.
[8] Li, K., Jia, W., Xu, L., Zhang, M., Huang, Y., 2023. The plastisphere of biodegradable and conventional microplastics from residues exhibit distinct microbial structure, network and function in plastic-mulching farmland. J Hazard Mater 442, 130011. https://doi.org/10.1016/J.JHAZMAT.2022.130011.
[9] Agarwal, S., 2020. Biodegradable polymers: present opportunities and challenges in providing a microplastic-free environment. Macromol Chem Phys 221. https://doi.org/10.1002/macp.202000017.
[10] de Souza Machado, A.A., Kloas, W., Zarfl, C., Hempel, S., Rillig, M.C., 2018. Microplastics as an emerging threat to terrestrial ecosystems. Glob Chang Biol 24, 1405–1416. https://doi.org/10.1111/gcb.14020.

[11] Hartman, K., Tringe, S.G., 2019. Interactions between plants and soil shaping the root microbiome under abiotic stress. Biochem J 476, 2705–2724. https://doi.org/10.1042/BCJ20180615.

[12] Hahladakis, J.N., Velis, C.A., Weber, R., Iacovidou, E., Purnell, P., 2018. An overview of chemical additives present in plastics: Migration, release, fate and environmental impact during their use, disposal and recycling. J Hazard Mater 344, 179–199. https://doi.org/10.1016/J.JHAZMAT.2017.10.014.

[13] Li, Z., Feng, C., Lei, J., He, X., Wang, Q., Zhao, Y., Qian, Y., Zhan, X., Shen, Z., 2024. Farmland microhabitat mediated by a residual microplastic film: microbial communities and function. Environ Sci Technol 58, 3654–3664. https://doi.org/10.1021/acs.est.3c07717.

[14] Lozano, Y.M., Aguilar-Trigueros, C.A., Onandia, G., Maaß, S., Zhao, T., Rillig, M. C., 2021. Effects of microplastics and drought on soil ecosystem functions and multifunctionality. J Appl Ecol 58, 988–996. https://doi.org/10.1111/1365-2664.13839.

[15] Leifheit, E.F., Lehmann, A., Rillig, M.C., 2021. Potential effects of microplastic on arbuscular mycorrhizal fungi. Front Plant Sci 12. https://doi.org/10.3389/fpls.2021.626709.

[16] Yu, P., Hochholdinger, F., 2018. The role of host genetic signatures on root–microbe interactions in the rhizosphere and endosphere. Front Plant Sci 9. https://doi.org/10.3389/fpls.2018.01896.

[17] Andreote, F.D., Gumiere, T., Durrer, A., 2014. Exploring interactions of plant microbiomes. Sci Agric 71, 528–539. https://doi.org/10.1590/0103-9016-2014-0195.

[18] Tkacz, A., Poole, P., 2021. The plant microbiome: the dark and dirty secrets of plant growth. PLANTS PEOPLE PLANET 3, 124–129. https://doi.org/10.1002/ppp3.10167.

[19] Aziz, U., Rehmani, M.S., Wang, L., Luo, X., Xian, B., Wei, S., Wang, G., Shu, K., 2021. Toward a molecular understanding of rhizosphere, phyllosphere, and spermosphere interactions in plant growth and stress response. CRC Crit Rev Plant Sci 40, 479–500. https://doi.org/10.1080/07352689.2022.2031728.

[20] Uzma Jalil, S., Ansari, M.I., 2018. Plant microbiome and its functional mechanism in response to environmental stress. Int J Green Pharm 12, 81–92. https://doi.org/10.22377/ijgp.v12i01.1603.

[21] Jia, K., Tian, M., Liu, J., Li, X., Xue, K., Song, Y., Liang, Q., Zhao, C., Li, C., 2026. The impact of biodegradable microplastics on soil aggregate stability across different afforestation ages and soil depths in poplar shelterbelts. Waste Manag 209, 115205. https://doi.org/10.1016/j.wasman.2025.115205.

[22] Zhang, G.S., Liu, Y.F., 2018. The distribution of microplastics in soil aggregate fractions in southwestern China. Sci Total Environ 642, 12–20. https://doi.org/10.1016/j.scitotenv.2018.06.004.

[23] Han, L., Chen, L., Feng, Y., Kuzyakov, Y., Chen, Q., Zhang, S., Chao, L., Cai, Y., Ma, C., Sun, K., Rillig, M.C., 2024. Microplastics alter soil structure and microbial community composition. Environ Int 185, 108508. https://doi.org/10.1016/j.envint.2024.108508.

[24] Li, Y., Yan, Q., Zou, C., Li, X., Wang, J., Shao, M., Jia, H., 2025. Microplastic-induced alterations in soil aggregate-associated carbon stabilization pathways: evidence from δ^{13}C Signature Analysis. Environ Sci Technol 59, 5545–5555. https://doi.org/10.1021/acs.est.4c09242.

[25] Zhang, Q., Ma, J., Zhao, Q., Ma, Y., Ma, L., Jiang, X., Xin, Z., Huang, T., Wang, Y., 2025. Aging of biodegradable plastics alters soil aggregate stability and organic carbon through shifts in microbial community composition. J Environ Manag 396, 128122. https://doi.org/10.1016/j.jenvman.2025.128122.

[26] Sepehrnia, N., Gubry-Rangin, C., Tanino, Y., Hallett, P.D., 2024. Microplastics alter soil structural stability as quantified by high-energy moisture characteristics. J Hazard Mater 476, 134940. https://doi.org/10.1016/j.jhazmat.2024.134940.

[27] Gong, K., Peng, C., Chen, X., Cai, L., Zhang, W., 2026. Rhizosphere accelerates breakdown of large biodegradable microplastics in soil. Environ Sci Ecotechnol. 30, 100678. https://doi.org/10.1016/j.ese.2026.100678.

[28] Wang, D., Xiong, F., Wu, L., Liu, Z., Xu, K., Huang, J., Liu, J., Ding, Q., Zhang, J., Pu, Y., Sun, R., 2024. A progress update on the biological effects of biodegradable microplastics on soil and ocean environment: A perfect substitute or new threat? Environ Res 252, 118960. https://doi.org/10.1016/j.envres.2024.118960.

[29] Kravchenko, A.N., Guber, A.K., 2017. Soil pores and their contributions to soil carbon processes. Geoderma 287, 31–39. https://doi.org/10.1016/j.geoderma.2016.06.027.

[30] Lara-Topete, G.O., Castanier-Rivas, J.D., Bahena-Osorio, M.F., Krause, S., Larsen, J.R., Loge, F.J., Mahlknecht, J., Gradilla-Hernández, M.S., González-López, M.E., 2024. Compounding one problem with another? A look at biodegradable microplastics. Sci Total Environ 944.

[31] Fojt, J., Denková, P., Brtnický, M., Holátko, J., Řezáčová, V., Pecina, V., Kučerík, J., 2022. Influence of poly-3-hydroxybutyrate micro-bioplastics and polyethylene terephthalate microplastics on the soil organic matter structure and soil water properties. Environ Sci Technol 56, 10732–10742. https://doi.org/10.1021/acs.est.2c01970.

[32] Zhang, G.S., Zhang, F.X., Li, X.T., 2019. Effects of polyester microfibers on soil physical properties: Perception from a field and a pot experiment. Sci Total Environ 670, 1–7. https://doi.org/10.1016/j.scitotenv.2019.03.149.

[33] Guo, Z., Li, P., Yang, X., Wang, Z., Lu, B., Chen, W., Wu, Y., Li, G., Zhao, Z., Liu, G., Ritsema, C., Geissen, V., Xue, S., 2022. Soil texture is an important factor determining how microplastics affect soil hydraulic characteristics. Environ Int 165, 107293. https://doi.org/10.1016/j.envint.2022.107293.

[34] Yu, Y., Battu, A.K., Varga, T., Denny, A.C., Zahid, T.Md, Chowdhury, I., Flury, M., 2023. Minimal impacts of microplastics on soil physical properties under environmentally relevant concentrations. Environ Sci Technol 57, 5296–5304. https://doi.org/10.1021/acs.est.2c09822.

[35] Zhao, T., Lozano, Y.M., Rillig, M.C., 2021. Microplastics increase soil ph and decrease microbial activities as a function of microplastic shape, polymer type, and exposure time. Front Environ Sci 9. https://doi.org/10.3389/fenvs.2021.675803.

[36] Pelko, T., Mravlje, J., Kalčíková, G., Skalar, T., Kržan, A., Regvar, M., Starin, M., Jemec Kokalj, A., Vogel-Mikuš, K., 2026. Microplastics from biodegradable plastic bags alter soil properties and trigger stress response in sunflowers. Environ Pollut 391, 127618. https://doi.org/10.1016/j.envpol.2025.127618.

[37] Zhou, J., Gui, H., Banfield, C.C., Wen, Y., Zang, H., Dippold, M.A., Charlton, A., Jones, D.L., 2021. The microplastisphere: Biodegradable microplastics addition alters soil microbial community structure and function. Soil Biol Biochem 156, 108211. https://doi.org/10.1016/j.soilbio.2021.108211.

[38] Zamanian, K., Zarebanadkouki, M., Kuzyakov, Y., 2018. Nitrogen fertilization raises CO_2 efflux from inorganic carbon: A global assessment. Glob Chang Biol 24, 2810–2817. https://doi.org/10.1111/gcb.14148.

[39] Brown, R.W., Chadwick, D.R., Zang, H., Graf, M., Liu, X., Wang, K., Greenfield, L. M., Jones, D.L., 2023. Bioplastic (PHBV) addition to soil alters microbial community structure and negatively affects plant-microbial metabolic functioning in maize. J Hazard Mater 441, 129959. https://doi.org/10.1016/j.jhazmat.2022.129959.

[40] Liu, S., Chen, B., Wang, K., Wang, J., Wang, K., Suo, Y., Yang, X., Zhu, Y., Zhang, J., Lu, M., Liu, Y., 2025. Unveiling the impact of biodegradable polylactic acid microplastics on meadow soil health. Environ Geochem Health 47, 45. https://doi.org/10.1007/s10653-025-02358-3.

[41] Zhang, Z., Wang, W., Liu, J., Wu, H., 2024. Discrepant responses of bacterial community and enzyme activities to conventional and biodegradable microplastics in paddy soil. Sci Total Environ 909, 168513. https://doi.org/10.1016/j.scitotenv.2023.168513.

[42] Peng, H., Dang, L., Lin, Z., Yu, B., Li, H., Yao, J., 2025. Responses of soil physical and chemical properties to degradable and non-degradable microplastics. J Environ Chem Eng 13, 116898. https://doi.org/10.1016/j.jece.2025.116898.

[43] Reay, M.K., Graf, M., Murphy, M., Li, G., Yan, C., Bhattacharya, M., Osbahr, H., Ma, J., Chengtao, W., Shi, X., Ren, S., Cui, J., Collins, C., Chadwick, D., Jones, D. L., Evershed, R.P., Lloyd, C.E.M., 2025. Higher potential leaching of inorganic and organic additives from biodegradable compared to conventional agricultural plastic mulch film. J Hazard Mater 488, 137147. https://doi.org/10.1016/j.jhazmat.2025.137147.

[44] Rukshana, F., Butterly, C.R., Baldock, J.A., Xu, J.M., Tang, C., 2012. Model organic compounds differ in priming effects on alkalinity release in soils through carbon and nitrogen mineralisation. Soil Biol Biochem 51, 35–43. https://doi.org/10.1016/j.soilbio.2012.03.022.

[45] She, Y., Qi, X., Sun, S., Li, Z., 2024. Biodegradable microplastics boost dissimilatory nitrate reduction to ammonium (DNRA) process contributing to ammonium nitrogen retention in farmland soils. J Clean Prod 438, 140835. https://doi.org/10.1016/j.jclepro.2024.140835.

[46] Allison S.D., Weintraub M.N., Gartner T.B., Waldrop M.P., 2010. Evolutionary-Economic Principles as Regulators of Soil Enzyme Production and Ecosystem Function. 229–243. https://doi.org/10.1007/978-3-642-14225-3_12.

[47] Zuccarini, P., Sardans, J., Asensio, L., Peñuelas, J., 2023. Altered activities of extracellular soil enzymes by the interacting global environmental changes. Glob Chang Biol 29, 2067–2091. https://doi.org/10.1111/GCB.16604.

[48] Luo, L., Meng, H., Gu, J.D., 2017. Microbial extracellular enzymes in biogeochemical cycling of ecosystems. J Environ Manag 197, 539–549. https://doi.org/10.1016/J.JENVMAN.2017.04.023.

[49] Lai, S., Fan, C., Yang, P., Fang, Y., Zhang, L., Jian, M., Dai, G., Liu, J., Yang, H., Shen, L., 2024. Effects of different microplastics on the physicochemical properties and microbial diversity of rice rhizosphere soil. Front Microbiol 15, 1513890. https://doi.org/10.3389/FMICB.2024.1513890/BIBTEX.

[50] Khan, T.F., Sikder, A.H.F., 2024. Microplastic Can Decrease Enzyme Activities and Microbes in Soil. Open J Soil Sci 14, 1–12. https://doi.org/10.4236/ojss.2024.141001.

[51] Duan, C., Fang, L., Yang, C., Chen, W., Cui, Y., Li, S., 2018. Reveal the response of enzyme activities to heavy metals through in situ zymography. Ecotoxicol Environ Saf 156, 106–115. https://doi.org/10.1016/J.ECOENV.2018.03.015.

[52] Chen, X., Dai, Y., Zhao, M., Löffler, F.E., Zhuang, J., 2022. Hydrobiological Mechanism Controlling the Synergistic Effects of Unsaturated Flow and Soil Organic Matter on the Degradation of Emerging Organic Contaminants in Soils. Environ Sci Technol 56, 11409–11417. https://doi.org/10.1021/acs.est.2c03013.

[53] Hoffland, E., Kuyper, T.W., Comans, R.N.J., Creamer, R.E., 2020. Eco-functionality of organic matter in soils. Plant Soil 455, 1–22. https://doi.org/10.1007/s11104-020-04651-9.

[54] Cotrufo, M.F., Haddix, M.L., Kroeger, M.E., Stewart, C.E., 2022. The role of plant input physical-chemical properties, and microbial and soil chemical diversity on the formation of particulate and mineral-associated organic matter. Soil Biol Biochem 168, 108648. https://doi.org/10.1016/j.soilbio.2022.108648.

[55] Ouyang, Z., Li, S., Xue, J., Liao, J., Xiao, C., Zhang, H., Li, X., Liu, P., Hu, S., Guo, X., Zhu, L., 2023. Dissolved organic matter derived from biodegradable microplastic promotes photo-aging of coexisting microplastics and alters microbial metabolism. J Hazard Mater 445, 130564. https://doi.org/10.1016/j.jhazmat.2022.130564.

[56] Zumstein, M.T., Schintlmeister, A., Nelson, T.F., Baumgartner, R., Woebken, D., Wagner, M., Kohler, H.P.E., McNeill, K., Sander, M., 2018. Biodegradation of synthetic polymers in soils: Tracking carbon into CO2 and microbial biomass. Sci Adv 4. https://doi.org/10.1126/SCIADV.AAS9024/SUPPL_FILE/AAS9024_SM.PDF.

[57] Chen, H., Wang, Y., Sun, X., Peng, Y., Xiao, L., 2020. Mixing effect of polylactic acid microplastic and straw residue on soil property and ecological function. Chemosphere 243, 125271. https://doi.org/10.1016/j.chemosphere.2019.125271.

[58] Chen, M., Cao, M., Zhang, W., Chen, X., Liu, H., Ning, Z., Peng, L., Fan, C., Wu, D., Zhang, M., Li, Q., 2024. Effect of biodegradable PBAT microplastics on the C and N accumulation of functional organic pools in tropical latosol. Environ Int 183, 108393. https://doi.org/10.1016/j.envint.2023.108393.

[59] Huang, F., Zhang, Q., Wang, L., Zhang, C., Zhang, Y., 2023. Are biodegradable mulch films a sustainable solution to microplastic mulch film pollution? A biogeochemical perspective. J Hazard Mater 459, 132024. https://doi.org/10.1016/J.JHAZMAT.2023.132024.

[60] Sun, Y., Li, X., Li, X., Wang, J., 2022. Deciphering the Fingerprint of Dissolved Organic Matter in the Soil Amended with Biodegradable and Conventional Microplastics Based on Optical and Molecular Signatures. Environ Sci Technol 56, 15746–15759. https://doi.org/10.1021/acs.est.2c06258.

[61] Chen, L., Han, L., Wang, F., Chen, Q., Huang, H., Wang, J., Ma, C., Sun, K., Rillig, M.C., Kuzyakov, Y., Yang, Z., 2025. Polylactic acid microplastics induced negative priming and improved carbon sequestration via microbial processes in different paddy soils. Soil Biol Biochem 201, 109653. https://doi.org/10.1016/j.soilbio.2024.109653.

[62] Nelson, T.F., Baumgartner, R., Jaggi, M., Bernasconi, S.M., Battagliarin, G., Sinkel, C., Künkel, A., Kohler, H.P.E., McNeill, K., Sander, M., 2022. Biodegradation of poly(butylene succinate) in soil laboratory incubations assessed by stable carbon isotope labelling, 13 Nat Commun 2022 13 (1), 1–16. https://doi.org/10.1038/s41467-022-33064-8.

[63] Zhang, F., Chen, X., Yao, S., Ye, Y., Zhang, B., 2022. Responses of soil mineral-associated and particulate organic carbon to carbon input: A meta-analysis. Sci Total Environ 829, 154626. https://doi.org/10.1016/J.SCITOTENV.2022.154626.

[64] Ding, F., Flury, M., Schaeffer, S.M., Xu, Y., Wang, J., 2021. Does long-term use of biodegradable plastic mulch affect soil carbon stock? Resour Conserv Recycl 175, 105895. https://doi.org/10.1016/j.resconrec.2021.105895.

[65] Nicoloso, R.S., Rice, C.W., Amado, T.J.C., Costa, C.N., Akley, E.K., 2018. Carbon saturation and translocation in a no-till soil under organic amendments. Agric Ecosyst Environ 264, 73–84. https://doi.org/10.1016/j.agee.2018.05.016.

[66] Lin, J., Chen, B., Dong, H., Zhang, W., Kumar, A., Hui, D., Zhang, C., Shan, S., Zhu, B., 2025. Effects of soil moisture fluctuation and microplastics types on soil organic matter decomposition and carbon dynamics. Soil Biol Biochem 205, 109781. https://doi.org/10.1016/j.soilbio.2025.109781.

[67] Domeignoz-Horta, L.A., Pold, G., Liu, X.-J.A., Frey, S.D., Melillo, J.M., DeAngelis, K.M., 2020. Microbial diversity drives carbon use efficiency in a model soil. Nat Commun 11, 3684. https://doi.org/10.1038/s41467-020-17502-z.

[68] Jia, K., Nie, S., Tian, M., Sun, W., Gao, Y., Zhang, Y., Xie, X., Xu, Z., Zhao, C., Li, C., 2024. Biodegradable microplastics can cause more serious loss of soil organic carbon by priming effect than conventional microplastics in farmland shelterbelts. Funct Ecol 38, 2447–2458. https://doi.org/10.1111/1365-2435.14662.

[69] Liu, X.X., Zhang, H.H., Zhu, Q.Y., Ye, J.Y., Zhu, Y.X., Jing, X.T., Du, W.X., Zhou, M., Lin, X.Y., Zheng, S.J., Jin, C.W., 2022. Phloem iron remodels root development in response to ammonium as the major nitrogen source. Nat Commun 13, 561. https://doi.org/10.1038/s41467-022-28261-4.

[70] Xiang, Y., Peñuelas, J., Sardans, J., Liu, Y., Yao, B., Li, Y., 2023. Effects of microplastics exposure on soil inorganic nitrogen: A comprehensive synthesis. J Hazard Mater 460, 132514. https://doi.org/10.1016/j.jhazmat.2023.132514.

[71] Rong, L., Wang, Y., Meidl, P., Baqar, M., Li, A., Wang, L., Sun, H., 2025. Insights into soil microbial assemblages and nitrogen cycling function responses to conventional and biodegradable microplastics. J Hazard Mater 491, 137889. https://doi.org/10.1016/j.jhazmat.2025.137889.

[72] Zhang, H., Zhu, W., Zhang, J., Müller, C., Wang, L., Jiang, R., 2024. Enhancing soil gross nitrogen transformation through regulation of microbial nitrogen-cycling genes by biodegradable microplastics. J Hazard Mater 478, 135528. https://doi.org/10.1016/j.jhazmat.2024.135528.

[73] Hu, X., Gu, H., Sun, X., Wang, Y., Liu, J., Yu, Z., Li, Y., Jin, J., Wang, G., 2023. Distinct influence of conventional and biodegradable microplastics on microbe-driving nitrogen cycling processes in soils and plastispheres as evaluated by metagenomic analysis. J Hazard Mater 451, 131097. https://doi.org/10.1016/j.jhazmat.2023.131097.

[74] Zhou, J., Wen, Y., Marshall, M.R., Zhao, J., Gui, H., Yang, Y., Zeng, Z., Jones, D.L., Zang, H., 2021. Microplastics as an emerging threat to plant and soil health in agroecosystems. Sci Total Environ 787, 147444. https://doi.org/10.1016/j.scitotenv.2021.147444.

[75] Sun, Y., Li, X., Cao, N., Duan, C., Ding, C., Huang, Y., Wang, J., 2022. Biodegradable microplastics enhance soil microbial network complexity and ecological stochasticity. J Hazard Mater 439, 129610. https://doi.org/10.1016/j.jhazmat.2022.129610.

[76] Liu, Z., Cai, L., Dong, Q., Zhao, X., Han, J., 2022. Effects of microplastics on water infiltration in agricultural soil on the Loess Plateau, China. Agric Water Manag 271, 107818. https://doi.org/10.1016/j.agwat.2022.107818.

[77] Xing, X., Yu, M., Xia, T., Ma, L., 2021. Interactions between water flow and microplastics in silt loam and loamy sand. Soil Sci Soc Am J 85, 1956–1962. https://doi.org/10.1002/SAJ2.20337.

[78] Sun, Y., Duan, C., Cao, N., Ding, C., Huang, Y., Wang, J., 2022. Biodegradable and conventional microplastics exhibit distinct microbiome, functionality, and metabolome changes in soil. J Hazard Mater 424, 127282. https://doi.org/10.1016/j.jhazmat.2021.127282.

[79] Chang, S., Chen, C., Fu, Q.-L., Zhou, A., Hua, Z., Zhu, F., Li, S., He, H., 2024. PBAT biodegradable microplastics enhanced organic matter decomposition capacity and CO2 emission in soils with and without straw residue. J Hazard Mater 480, 135872. https://doi.org/10.1016/j.jhazmat.2024.135872.

[80] Yan, Z., Lin, S., Hu, R., Cheng, H., Xiang, R., Xu, H., Zhao, J., 2024. Effects of biodegradable microplastics and straw addition on soil greenhouse gas emissions. Environ Pollut 356, 124315. https://doi.org/10.1016/j.envpol.2024.124315.

[81] George, T.S., Hinsinger, P., Turner, B.L., 2016. Phosphorus in soils and plants – facing phosphorus scarcity. Plant Soil 401, 1–6. https://doi.org/10.1007/s11104-016-2846-9.

[82] Song, X., Ding, J., Tian, W., Xu, H., Zou, H., Wang, Z., 2023. Effects of plastisphere on phosphorus availability in freshwater system: Critical roles of polymer type and colonizing habitat. Sci Total Environ 870, 161990. https://doi.org/10.1016/j.scitotenv.2023.161990.

[83] Zhou, J., Xu, H., Xiang, Y., Wu, J., 2024. Effects of microplastics pollution on plant and soil phosphorus: A meta-analysis. J Hazard Mater 461, 132705. https://doi.org/10.1016/j.jhazmat.2023.132705.

[84] Feng, X., Wang, Q., Sun, Y., Zhang, S., Wang, F., 2022. Microplastics change soil properties, heavy metal availability and bacterial community in a Pb-Zn-contaminated soil. J Hazard Mater 424, 127364. https://doi.org/10.1016/j.jhazmat.2021.127364.

[85] Wang, Y., Qin, J., Zhou, S., Lin, X., Ye, L., Song, C., Yan, Y., 2015. Identification of the function of extracellular polymeric substances (EPS) in denitrifying phosphorus removal sludge in the presence of copper ion. Water Res 73, 252–264. https://doi.org/10.1016/j.watres.2015.01.034.

[86] Shen, L., Wang, Y., Liu, R., Yang, Y., Liu, Y., Xing, B., 2023. Aging characteristics of degradable and non-biodegradable microplastics and their adsorption mechanism for sulfonamides. Sci Total Environ 901, 166452. https://doi.org/10.1016/j.scitotenv.2023.166452.

[87] Guo, T., Zhu, C., Cai, Y., Gustave, W., Ouyang, D., Li, M., Zhang, H., 2025. Discrepant effects of microplastics on soil phosphorus availability under different phosphorus fertilizer applications. J Environ Manag 385, 125656. https://doi.org/10.1016/j.jenvman.2025.125656.

[88] Gewert, B., Plassmann, M.M., MacLeod, M., 2015. Pathways for degradation of plastic polymers floating in the marine environment. Environ Sci Process Impacts 17, 1513–1521. https://doi.org/10.1039/C5EM00207A.

[89] Artru, M., Lecerf, A., 2019. Slow degradation of compostable plastic carrier bags in a stream and its riparian area. Ann De Limnol - Int J Limnol 55, 18. https://doi.org/10.1051/limn/2019017.

[90] Cao, Z., Kim, C., Li, Z., Jung, J., 2024. Comparing environmental fate and ecotoxicity of conventional and biodegradable plastics: A critical review. Sci Total Environ 951, 175735. https://doi.org/10.1016/j.scitotenv.2024.175735.

[91] Sciscione, F., Hailes, H.C., Miodownik, M., 2023. The performance and environmental impact of pro-oxidant additive containing plastics in the open unmanaged environment—a review of the evidence. R Soc Open Sci 10. https://doi.org/10.1098/rsos.230089.

[92] Serrano-Ruiz, H., Martin-Closas, L., Pelacho, A.M., 2023. Impact of buried debris from agricultural biodegradable plastic mulches on two horticultural crop plants: Tomato and lettuce. Sci Total Environ 856, 159167. https://doi.org/10.1016/J.SCITOTENV.2022.159167.

[93] Sun, Y., Wang, X., Xia, S., Zhao, J., 2022. Cu(II) adsorption on Poly(Lactic Acid) Microplastics: Significance of microbial colonization and degradation. Chem Eng J 429, 132306. https://doi.org/10.1016/j.cej.2021.132306.

[94] Klun, B., Starin, M., Novak, J., Putar, U., Čelan Korošin, N., Binda, G., Kalčíková, G., 2025. Biofilm formation on polyethylene and polylactic acid microplastics in freshwater: Influence of environmental factors. J Environ Chem Eng 13, 118689. https://doi.org/10.1016/j.jece.2025.118689.

[95] Wang, J., Peng, C., Dai, Y., Li, Y., Jiao, S., Ma, X., Liu, X., Wang, L., 2022. Slower antibiotics degradation and higher resistance genes enrichment in plastisphere. Water Res 222, 118920. https://doi.org/10.1016/j.watres.2022.118920.

[96] Allemann, M.N., Tessman, M., Reindel, J., Scofield, G.B., Evans, P., Pomeroy, R. S., Burkart, M.D., Mayfield, S.P., Simkovsky, R., 2024. Rapid biodegradation of microplastics generated from bio-based thermoplastic polyurethane. Sci Rep 14. https://doi.org/10.1038/s41598-024-56492-6.

[97] Gross N., Muhvich J., Ching C., Gomez B., Horvath E., Nahum Y., Zaman M.H., 2024. Microplastics as a novel facilitator for antimicrobial resistance: Effects of concentration, composition, and size on *Escherichia coli* multidrug resistance.

[98] Song, T., Liu, J., Han, S., Li, Y., Xu, T., Xi, J., Hou, L., Lin, Y., 2024. Effect of conventional and biodegradable microplastics on the soil-soybean system: A perspective on rhizosphere microbial community and soil element cycling. Environ Int 190, 108781. https://doi.org/10.1016/j.envint.2024.108781.

[99] Wang, J., Liu, W., Zeb, A., Wang, Q., Mo, F., Shi, R., Sun, Y., Wang, F., 2024. Biodegradable Microplastic-Driven Change in Soil pH Affects Soybean Rhizosphere Microbial N Transformation Processes. J Agric Food Chem 72, 16674–16686. https://doi.org/10.1021/acs.jafc.4c04206.

[100] Carneiro, B., Marques, P., Lopes, T., Figueira, E., 2025. Biodegradable Microplastics from Agricultural Mulch Films: Implications for Plant Growth-Promoting Bacteria and Plant's Oxidative Stress. Antioxidants 14, 230. https://doi.org/10.3390/antiox14020230.

[101] Tian, M., Qiao, B., Xu, Z., Liang, Q., Xie, X., Chen, K., Zhang, Y., Zhao, C., Li, C., 2024. Invasion of Trifolium repens L. aggravated by biodegradable plastics: adjustable strategy for foraging N and P. J Hazard Mater 474, 134777. https://doi.org/10.1016/j.jhazmat.2024.134777.

[102] Fan, H., Hong, X., Wang, H., Gao, F., Su, Z., Yao, H., 2025. Biodegradable microplastics affect tomato (Solanum lycopersicum L.) growth by interfering

rhizosphere key phylotypes. J Hazard Mater 487, 137208. https://doi.org/10.1016/j.jhazmat.2025.137208.

[103] Jin, J., Wang, X., Sha, Y., Wang, F., Huang, X., Zong, H., Liu, J., Song, N., 2024. Changes in soil properties and microbial activity unveil the distinct impact of polyethylene and biodegradable microplastics on chromium uptake by peanuts. Environ Sci Pollut Res 31, 53369–53380. https://doi.org/10.1007/s11356-024-34743-3.

[104] Wang, Q., Feng, X., Liu, Y., Li, W., Cui, W., Sun, Y., Zhang, S., Wang, F., Xing, B., 2023. Response of peanut plant and soil N-fixing bacterial communities to conventional and biodegradable microplastics. J Hazard Mater 459, 132142. https://doi.org/10.1016/j.jhazmat.2023.132142.

[105] Qi, Y., Ossowicki, A., Yergeau, É., Vigani, G., Geissen, V., Garbeva, P., 2022. Plastic mulch film residues in agriculture: impact on soil suppressiveness, plant growth, and microbial communities. FEMS Microbiol Ecol 98. https://doi.org/10.1093/femsec/fiac017.

[106] Sun, M., Yang, Y., Huang, M., Fu, S., Hao, Y., Hu, S., Lai, D., Zhao, L., 2022. Adsorption behaviors and mechanisms of antibiotic norfloxacin on degradable and nondegradable microplastics. Sci Total Environ 807, 151042. https://doi.org/10.1016/J.SCITOTENV.2021.151042.

[107] Fang, Y., Lin, C., Zhao, J., Gao, Y., Jia, X., 2025. Dosages of Biodegradable Poly (butylene adipate-co-terephthalate) Microplastics Affect Soil Microbial Community, Function, and Metabolome in Plant–Soil System. Agronomy 15, 990. https://doi.org/10.3390/agronomy15040990.

[108] Liu, R., Liang, J., Yang, Y., Jiang, H., Tian, X., 2023. Effect of polylactic acid microplastics on soil properties, soil microbials and plant growth. Chemosphere 329, 138504. https://doi.org/10.1016/j.chemosphere.2023.138504.

[109] Li, C., Cui, Q., Li, Y., Zhang, K., Lu, X., Zhang, Y., 2022. Effect of LDPE and biodegradable PBAT primary microplastics on bacterial community after four months of soil incubation. J Hazard Mater 429, 128353. https://doi.org/10.1016/J.JHAZMAT.2022.128353.

[110] Rauscher, A., Meyer, N., Jakobs, A., Bartnick, R., Lueders, T., Lehndorff, E., 2023. Biodegradable microplastic increases CO2 emission and alters microbial biomass and bacterial community composition in different soil types. Appl Soil Ecol 182, 104714. https://doi.org/10.1016/J.APSOIL.2022.104714.

[111] Kim, S.W., Šmídová, K., van Loon, S., van Gestel, C.A.M., Rillig, M.C., Fritze, H., Velmala, S., 2025. Effects of biodegradable microplastics on soil microbial communities and activities: Insight from an ecological mesocosm experiment. Sci Total Environ 975, 179288. https://doi.org/10.1016/J.SCITOTENV.2025.179288.

[112] Wang, Q., Feng, X., Liu, Y., Cui, W., Sun, Y., Zhang, S., Wang, F., 2022. Effects of microplastics and carbon nanotubes on soil geochemical properties and bacterial communities. J Hazard Mater 433, 128826. https://doi.org/10.1016/j.jhazmat.2022.128826.

[113] Liu, M., Wang, C., Zhu, B., 2024. Independent and combined effects of microplastics pollution and drought on soil bacterial community. Sci Total Environ 913, 169749. https://doi.org/10.1016/j.scitotenv.2023.169749.

[114] Azeem, I., Adeel, M., Ahmad, M.A., Shakoor, N., Jiangcuo, G.D., Azeem, K., Ishfaq, M., Shakoor, A., Ayaz, M., Xu, M., Rui, Y., 2021. Uptake and Accumulation of Nano/Microplastics in Plants: A Critical Review. Nanomaterials 11, 2935. https://doi.org/10.3390/nano11112935.

[115] Li, C., Li, Y., Tang, L., Ikenaga, M., Liu, R., Xu, G., 2023. Soil microbial community shifts explain habitat heterogeneity in two *Haloxylon* species from a nutrient perspective. Ecol Evol 13. https://doi.org/10.1002/ece3.9727.

[116] Wang, F., Zhang, X., Zhang, S., Zhang, S., Sun, Y., 2020. Interactions of microplastics and cadmium on plant growth and arbuscular mycorrhizal fungal communities in an agricultural soil. Chemosphere 254, 126791. https://doi.org/10.1016/J.CHEMOSPHERE.2020.126791.

[117] Yang, W., Cheng, P., Adams, C.A., Zhang, S., Sun, Y., Yu, H., Wang, F., 2021. Effects of microplastics on plant growth and arbuscular mycorrhizal fungal communities in a soil spiked with ZnO nanoparticles. Soil Biol Biochem 155, 108179. https://doi.org/10.1016/j.soilbio.2021.108179.

[118] Moreno-Jiménez, E., Leifheit, E.F., Plaza, C., Feng, L., Bergmann, J., Wulf, A., Lehmann, A., Rillig, M.C., 2022. Effects of microplastics on crop nutrition in fertile soils and interaction with arbuscular mycorrhizal fungi. J Sustain Agric Environ 1, 66–72. https://doi.org/10.1002/sae2.12006.

[119] Liu, L., Sun, Y., Du, S., Li, Y., Wang, J., 2024. Nanoplastics promote the dissemination of antibiotic resistance genes and diversify their bacterial hosts in soil. Eco-Environ & Health 3, 1–10. https://doi.org/10.1016/j.eehl.2023.09.005.

[120] Fei, J., Bai, X., Jiang, C., Yin, X., Ni, B.-J., 2024. A state-of-the-art review of environmental behavior and potential risks of biodegradable microplastics in soil ecosystems: Comparison with conventional microplastics. Sci Total Environ 954, 176342. https://doi.org/10.1016/j.scitotenv.2024.176342.

[121] Kim, S.H., Cho, J.Y., Cho, D.H., Jung, H.J., Kim, B.C., Bhatia, S.K., Park, S.H., Park, K., Yang, Y.H., 2022. Acceleration of Polybutylene Succinate Biodegradation by Terribacillus sp. JY49 Isolated from a Marine Environment. Polym (Basel) 14, 3978. https://doi.org/10.3390/POLYM14193978/S1.

[122] Urbanek, A.K., Strzelecki, M.C., Mirończuk, A.M., 2021. The potential of cold-adapted microorganisms for biodegradation of bioplastics. Waste Manag 119, 72–81. https://doi.org/10.1016/J.WASMAN.2020.09.031.

[123] Nchedo Ariole, C., George-West, O., 2020. Bioplastic degradation potential of microorganisms isolated from the soil. Am J Chem Biochem Eng 4 (1). https://doi.org/10.11648/j.ajcbe.20200401.11.

[124] Malik, S., Maurya, A., Khare, S.K., Srivastava, K.R., 2023. Computational exploration of bio-degradation patterns of various plastic types. Polymers) 15, 1540. https://doi.org/10.3390/polym15061540.

[125] Kitamoto, H.K., Shinozaki, Y., Cao, X., Morita, T., Konishi, M., Tago, K., Kajiwara, H., Koitabashi, M., Yoshida, S., Watanabe, T., Sameshima-Yamashita, Y., Nakajima-Kambe, T., Tsushima, S., 2011. Phyllosphere yeasts rapidly break down biodegradable plastics. AMB Express 1, 44. https://doi.org/10.1186/2191-0855-1-44.

[126] Karamanlioglu, M., Houlden, A., Robson, G.D., 2014. Isolation and characterisation of fungal communities associated with degradation and growth on the surface of poly(lactic) acid (PLA) in soil and compost. Int Biodeterior Biodegrad 95, 301–310. https://doi.org/10.1016/J.IBIOD.2014.09.006.

[127] Phukon, P., Saikia, J.P., Konwar, B.K., 2012. Bio-plastic (P-3HB-co-3HV) from Bacillus circulans (MTCC 8167) and its biodegradation. Colloids Surf B Biointerfaces 92, 30–34. https://doi.org/10.1016/J.COLSURFB.2011.11.011.

[128] Fernandes, M., Salvador, A.F., Vicente, A.A., 2024. Biodegradation of PHB/PBAT films and isolation of novel PBAT biodegraders from soil microbiomes. Chemosphere 362, 142696. https://doi.org/10.1016/J.CHEMOSPHERE.2024.142696.

[129] Panyachanakul, T., Kitpreechavanich, V., Tokuyama, S., Krajangsang, S., 2017. Poly(dl-lactide)-degrading enzyme production by immobilized Actinomadura keratinilytica strain T16-1 in a 5-L fermenter under various fermentation processes. Electron J Biotechnol 30, 71–76. https://doi.org/10.1016/j.ejbt.2017.09.001.

[130] Sukkhum, S., Tokuyama, S., Tamura, T., Kitpreechavanich, V., 2009. A novel poly (L-lactide) degrading actinomycetes isolated from Thai forest soil, phylogenic relationship and the enzyme characterization. J Gen Appl Microbiol 55, 459–467. https://doi.org/10.2323/JGAM.55.459.

[131] Kim, M.N., Park, S.T., 2010. Degradation of poly(L-lactide) by a mesophilic bacterium. J Appl Polym Sci 117, 67–74. https://doi.org/10.1002/APP.31950.

[132] Penkhrue, W., Khanongnuch, C., Masaki, K., Pathom-aree, W., Punyodom, W., Lumyong, S., 2015. Isolation and screening of biopolymer-degrading microorganisms from northern Thailand. World J Microbiol Biotechnol 31, 1431–1442. https://doi.org/10.1007/S11274-015-1895-1/TABLES/4.

[133] Lomthong, T., Chotineeranat, S., Kitpreechavanich, V., 2015. Production and characterization of raw starch degrading enzyme from a newly isolated thermophilic filamentous bacterium, Laceyella sacchari. Starch - Stärke 67 (255–266), LP175. https://doi.org/10.1002/STAR.201400150.

[134] Chua, T.-K., Tseng, M., Yang, M.-K., 2013. Degradation of Poly(ε-caprolactone) by thermophilic Streptomyces thermoviolaceus subsp. thermoviolaceus 76T-2, 3 AMB Express 2013 3 (1), 1–7. https://doi.org/10.1186/2191-0855-3-8.

[135] Liu, J., Wang, P., Wang, Y., Zhang, Y., Xu, T., Zhang, Y., Xi, J., Hou, L., Li, L., Zhang, Z., Lin, Y., 2022. Negative effects of poly(butylene adipate-co-terephthalate) microplastics on Arabidopsis and its root-associated microbiome. J Hazard Mater 437, 129294. https://doi.org/10.1016/J.JHAZMAT.2022.129294.

[136] Emadian, S.M., Onay, T.T., Demirel, B., 2017. Biodegradation of bioplastics in natural environments. Waste Manag 59, 526–536. https://doi.org/10.1016/J.WASMAN.2016.10.006.

[137] Richert, A., Dąbrowska, G.B., 2021. Enzymatic degradation and biofilm formation during biodegradation of polylactide and polycaprolactone polymers in various environments. Int J Biol Macromol 176, 226–232. https://doi.org/10.1016/J.IJBIOMAC.2021.01.202.

[138] Ishii, N., Inoue, Y., Tagaya, T., Mitomo, H., Nagai, D., Kasuya, K. ichi, 2008. Isolation and characterization of poly(butylene succinate)-degrading fungi. Polym Degrad Stab 93, 883–888. https://doi.org/10.1016/J.POLYMDEGRADSTAB.2008.02.005.

[139] Gkoutselis, G., Rohrbach, S., Harjes, J., Obst, M., Brachmann, A., Horn, M.A., Rambold, G., 2021. Microplastics accumulate fungal pathogens in terrestrial ecosystems. Sci Rep 11, 13214. https://doi.org/10.1038/s41598-021-92405-7.

[140] Yu, Z., Song, S., Xu, X., Ma, Q., Lu, Y., 2021. Sources, migration, accumulation and influence of microplastics in terrestrial plant communities. Environ Exp Bot 192, 104635. https://doi.org/10.1016/j.envexpbot.2021.104635.

[141] Song, R., Sun, Y., Li, X., Ding, C., Huang, Y., Du, X., Wang, J., 2022. Biodegradable microplastics induced the dissemination of antibiotic resistance genes and virulence factors in soil: A metagenomic perspective. Sci Total Environ 828, 154596. https://doi.org/10.1016/J.SCITOTENV.2022.154596.

[142] Liu, Y., Liu, W., Yang, X., Wang, J., Lin, H., Yang, Y., 2021. Microplastics are a hotspot for antibiotic resistance genes: Progress and perspective. Sci Total Environ 773, 145643. https://doi.org/10.1016/j.scitotenv.2021.145643.

[143] Di Cesare, A., Pinnell, L.J., Brambilla, D., Elli, G., Sabatino, R., Sathicq, M.B., Corno, G., O'Donnell, C., Turner, J.W., 2021. Bioplastic accumulates antibiotic and metal resistance genes in coastal marine sediments. Environ Pollut 291, 118161. https://doi.org/10.1016/J.ENVPOL.2021.118161.

[144] Liu, Y., Zhang, J., Cao, W., Hu, Y., Shen, W., 2022. The influence of Pb(II) adsorption on (Non) biodegradable microplastics by UV/O3 oxidation treatment. J Environ Chem Eng 10, 108615. https://doi.org/10.1016/J.JECE.2022.108615.

[145] Zhang, X., Wu, D., Jiang, X., Xu, J., Liu, J., 2024. Source, Environmental behavior and ecological impact of biodegradable microplastics in soil ecosystems: a review. Rev Environ Contam Toxicol 262.

[146] Sun, Y., Wang, X., Xia, S., Zhao, J., 2021. New insights into oxytetracycline (OTC) adsorption behavior on polylactic acid microplastics undergoing microbial adhesion and degradation. Chem Eng J 416, 129085. https://doi.org/10.1016/J.CEJ.2021.129085.

[147] Fan, X., Zou, Y., Geng, N., Liu, J., Hou, J., Li, D., Yang, C., Li, Y., 2021. Investigation on the adsorption and desorption behaviors of antibiotics by degradable MPs with or without UV ageing process. J Hazard Mater 401, 123363. https://doi.org/10.1016/J.JHAZMAT.2020.123363.

[148] Mirsoleimani Azizi, S.M., Zakaria, B.S., Haffiez, N., Dhar, B.R., 2023. Sludge Thermal Hydrolysis for Mitigating Oxidative Stress of Polystyrene Nanoplastics in Anaerobic Digestion: Significance of the Solids Content. ACS Sustain Chem Eng 11, 7253–7262. https://doi.org/10.1021/acssuschemeng.3c01349.

[149] Lei, X., Shen, Y., Zhao, J., Huang, J., Wang, H., Yu, Y., Xiao, C., 2023. Root Exudates Mediate the Processes of Soil Organic Carbon Input and Efflux, 12, 630 Plants 2023 12, 630. https://doi.org/10.3390/PLANTS12030630.

[150] Santoyo, G., 2022. How plants recruit their microbiome? New insights into beneficial interactions. J Adv Res 40, 45–58. https://doi.org/10.1016/J.JARE.2021.11.020.

[151] Zhao, J., Liu, J., Liang, H., Huang, J., Chen, Z., Nie, Y., Wang, C., Wang, Y., 2018. Manipulation of the rhizosphere microbial community through application of a new bio-organic fertilizer improves watermelon quality and health. PLoS One 13. https://doi.org/10.1371/JOURNAL.PONE.0192967.

[152] Zhalnina, K., Louie, K.B., Hao, Z., Mansoori, N., Da Rocha, U.N., Shi, S., Cho, H., Karaoz, U., Loqué, D., Bowen, B.P., Firestone, M.K., Northen, T.R., Brodie, E.L., 2018. Dynamic root exudate chemistry and microbial substrate preferences drive patterns in rhizosphere microbial community assembly, 3 Nat Microbiol 2018 3 (4), 470–480. https://doi.org/10.1038/s41564-018-0129-3.

[153] Zhang, Y., Wang, W., Shen, Z., Wang, J., Chen, Y., Wang, D., Liu, G., Han, M., 2021. Comparison and interpretation of characteristics of Rhizosphere microbiomes of three blueberry varieties. BMC Microbiol 21, 1–13. https://doi.org/10.1186/S12866-021-02092-7/FIGURES/6.

[154] Sun, H., Shi, Y., Zhao, P., Long, G., Li, C., Wang, J., Qiu, D., Lu, C., Ding, Y., Liu, L., He, S., 2023. Effects of polyethylene and biodegradable microplastics on photosynthesis, antioxidant defense systems, and arsenic accumulation in maize (Zea mays L.) seedlings grown in arsenic-contaminated soils. Sci Total Environ 868, 161557. https://doi.org/10.1016/J.SCITOTENV.2023.161557.

[155] Fu Y., Oduor A.M.O., Jiang M., Liu Y., 2023. Live soil ameliorated the negative effects of biodegradable but not non-biodegradable microplastics on the growth of plant communities. https://doi.org/10.1101/2023.10.12.562149.

[156] Souza, P.M.S., Sommaggio, L.R.D., Marin-Morales, M.A., Morales, A.R., 2020. PBAT biodegradable mulch films: Study of ecotoxicological impacts using Allium cepa, Lactuca sativa and HepG2/C3A cell culture. Chemosphere 256, 126985. https://doi.org/10.1016/J.CHEMOSPHERE.2020.126985.

[157] Zantis, L.J., Adamczyk, S., Velmala, S.M., Adamczyk, B., Vijver, M.G., Peijnenburg, W., Bosker, T., 2024. Comparing the impact of microplastics derived from a biodegradable and a conventional plastic mulch on plant performance. Sci Total Environ 935, 173265. https://doi.org/10.1016/J.SCITOTENV.2024.173265.

[158] Zhang, Y.F., Huang, Z.Y., Li, Y.F., Lu, X.L., Li, G.R., Qi, S.S., Khan, I.U., Li, G.L., Dai, Z.C., Du, D.L., 2024. The Degradability of Microplastics May Not Necessarily Equate to Environmental Friendliness: A Case Study of Cucumber Seedlings with Disturbed Photosynthesis. Agric (Switz) 14, 53. https://doi.org/10.3390/AGRICULTURE14010053/S1.

[159] Yang, C., Gao, X., 2022. Impact of microplastics from polyethylene and biodegradable mulch films on rice (Oryza sativa L.). Sci Total Environ 828, 154579. https://doi.org/10.1016/J.SCITOTENV.2022.154579.

[160] Adamczyk, S., Zantis, L.J., van Loon, S., van Gestel, C.A.M., Bosker, T., Hurley, R., Nizzetto, L., Adamczyk, B., Velmala, S., 2024. Biodegradable microplastics induce profound changes in lettuce (Lactuca sativa) defense mechanisms and to some extent deteriorate growth traits. Environ Pollut 363, 125307. https://doi.org/10.1016/j.envpol.2024.125307.

[161] Lin, J., Cheng, Q., Kumar, A., Zhang, W., Yu, Z., Hui, D., Zhang, C., Shan, S., 2025. Effect of degradable microplastics, biochar and their coexistence on soil organic matter decomposition: A critical review. TrAC Trends Anal Chem 183, 118082. https://doi.org/10.1016/j.trac.2024.118082.