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Full-chain analysis on emerging contaminants in soil: Source, migration and remediation

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HIGHLIGHTS

- The research status of ECs on soil have been reviewed.
- A possible interaction mechanism between soil and ECs has been proposed.
- The distribution and migration pathways of ECs in soil have been described.
- Promising future directions have been proposed for mitigating the contamination of soil by ECs.

GRAPHICAL ABSTRACT

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ABSTRACT

Emerging contaminants (ECs) are gaining attention due to their prevalence and potential negative impacts on the environment and human health. This paper provides a comprehensive review of the status and trends of soil pollution caused by ECs, focusing on their sources, migration pathways, and environmental implications. Significant ECs, including plastics, synthetic polymers, pharmaceuticals, personal care products, plasticizers, and flame retardants, are identified due to their widespread use and toxicity. Their presence in soil is attributed to agricultural activities, urban waste, and wastewater irrigation. The review explores both horizontal and vertical migration pathways, with factors such as soil type, organic matter content, and moisture levels influencing their distribution. Understanding the behavior of ECs in soil is critical to mitigating their long-term risks and developing effective soil remediation strategies. The paper also examines the advantages and disadvantages of in situ

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and ex situ treatment approaches for ECs, highlighting optimal physical, chemical, and biological treatment conditions. These findings provide a fundamental basis for addressing the challenges and governance of soil pollution induced by ECs.

1. Status and trends of soil contamination by ECs

Emerging contaminants (ECs) encompass substances that have a historical usage but have recently gained attention for their potential negative impacts, or are newly synthesized pollutants (Rathi et al., 2021; Tong et al., 2021). These include plastics, synthetic polymers, pharmaceuticals, personal care products, plasticizers, and fire retardants. Volatile ECs, for example, are those that can easily vaporize and enter the atmosphere, potentially leading to widespread distribution and deposition. Due to their emerging nature, Due to the emerging nature of ECs, an increasing number of trace detection methods are being utilized for their quantitative detection (Prakash, 2024). Concurrently, advancements in detection techniques have led to the introduction of more chemical substances, thereby posing significant challenges in accurately assessing ECs within environmental contexts (Cundyab et al., 2022; Sudarsan et al., 2024). Although the concentrations of ECs are relatively low, their usage is increasing, leading to a rise in their release and accumulation in the environment (Kidd et al., 2024). The continued use and environmental release of ECs pose medium- to long-term risks. Hence, it is crucial to enhance our understanding of their emission patterns and impacts on human and ecosystem health. Fig. 1 illustrates the pathways of ECs entering soil. Various sources contribute to the infiltration of pollutants into the soil environment. Once within the soil matrix, pollutants can easily propagate through accumulation, transformation, and degradation processes, driven by chemical and biological mechanisms.

Microplastics and nanoplastics are significant examples of ECs that pose potential global risks. Microplastics typically refer to plastic particles less than 5 mm in diameter, while nanoplastics are even smaller, generally ranging from 1 to 100 nm in diameter (J. Li et al., 2024a). Their small size allows them to be transported through runoff to adjacent areas and water bodies, and to undergo long-range transport and deposition in the atmosphere via dry and wet processes (Akanyange et al., 2021; Chen et al., 2020). Microplastics and nanoplastics are not only widespread in the environment but also chemically stable and persistent, making them difficult to degrade and prone to accumulating in ecosystems (Forest and Pourchez, 2023). For example, microplastics have been found in marine sediments, freshwater bodies, and even in polar regions, indicating their extensive distribution (Thompson et al., 2004). Furthermore, daily consumption of diverse food forms may

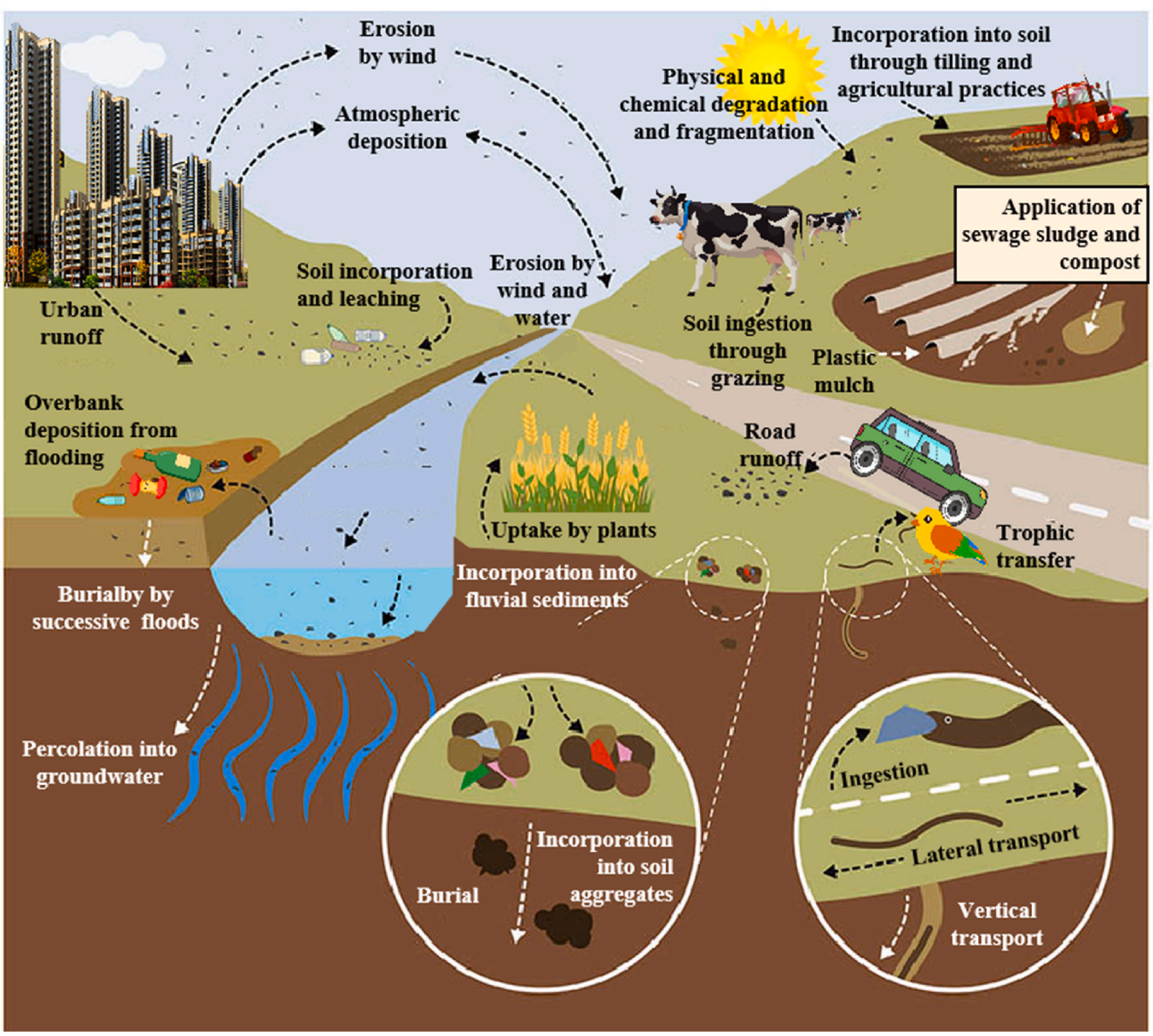

Fig. 1. The pathway the fate of ECs into soil.

expose humans to a variety of microplastics, potentially eliciting acute responses such as cytotoxicity, hemolysis, hypersensitivity reactions, and unwarranted immune responses (Mofijur et al., 2021). Although atmospheric deposition plays a significant role in plastic transport, approximately 80% of land-based plastics ultimately find their way into the oceans through rivers and coastal regions (Chen et al., 2020). Once in the ocean, microplastics are ingested by marine organisms, leading to physical blockages and chemical toxicity due to the leaching of harmful additives or the adsorption of persistent organic pollutants (Rochman et al., 2015). Studies have reported concentrations of microplastics ranging from thousands to millions of particles per square kilometer in ocean surface waters, highlighting the severity of the contamination (Eriksen et al., 2014). Fig. 2 depicts the global transport processes of ECs. Once released into the environment, volatile ECs are subject to atmospheric circulation, experiencing successive cycles of evaporation and deposition. Given the widespread use of these pollutants, the projected growth in their production, usage, and disposal, as well as their significance to human and environmental health, it is imperative to improve our understanding of their point sources and their large-scale distribution (Elliott et al., 2021).

ECs present a significant threat to both human health and ecosystems, underscoring the importance of investigating and studying these pollutants. However, current research efforts on ECs have primarily focused on water environments (Enfrin et al., 2020; Fanourakis et al., 2020; Gago-Ferrero et al., 2020; Garcia et al., 2021; Krasucka et al., 2021; Suaria et al., 2020; Wang et al., 2021a), giving relatively less attention to soil-related investigations (Surendran et al., 2023). To address this gap, it is crucial to comprehensively summarize the current status of ECs in soil and establish research directions for further experimental investigations. Transitioning from the broader impacts on human and environmental health, this section will delve into the specific dynamics of ECs in soil, including their sources, migration pathways, and environmental implications. By bridging these areas, we can develop a more integrated understanding of how ECs behave in soil environments and their potential long-term risks. The objective of this work is to enhance our understanding of the generation, global distribution, and environmental implications of ECs. The research explores various "Sources of Soil Pollution," with a specific focus on plastics and synthetic polymers from agricultural practices, electronic waste, and urban wastewater irrigation. Additionally, the study investigates the "Migration Pathways of ECs in Soil," examining both horizontal and vertical movements within the soil matrix. Factors such as soil type, organic matter content, and biological activity that influence the transport of ECs are taken into consideration, followed by an analysis of corresponding control methods. Furthermore, practical examples of ECs management are presented, along with an exploration of potential mitigation and remediation strategies to address soil pollution resulting from ECs.

2. Soil pollution sources of ECs

2.1. Sources of agricultural soils

Over the past few decades, the utilization of plastics in agriculture has experienced a significant rise (Zhang et al., 2021a). In 2019, the European Union consumed a total of 708,000 tons of non-packaging plastics, with approximately 44% allocated for applications such as greenhouse films, mulching films, small tunnels, irrigation pipes, drip emitters, and crop protection nets (Eunomia and Deloitte, 2020). Conversely, plastic items like fertilizer bags, pesticide containers, seed trays, and pots tend to be designed for single-use purposes. The absence of effective systems for retrieving plastic materials from fields has resulted in enduring challenges in achieving sustainable collection and disposal practices. Consequently, plastic has emerged as a notable source of soil pollution within the agricultural domain (Wanner, 2021). Notably, mulching films, which come into close contact with the soil, have become one of the most heavily contaminated plastic products used in agriculture. During decomposition or post-harvest management, these films can be absorbed by the soil, intensifying their potential impact (Li et al., 2021a). China stands as the largest global consumer of

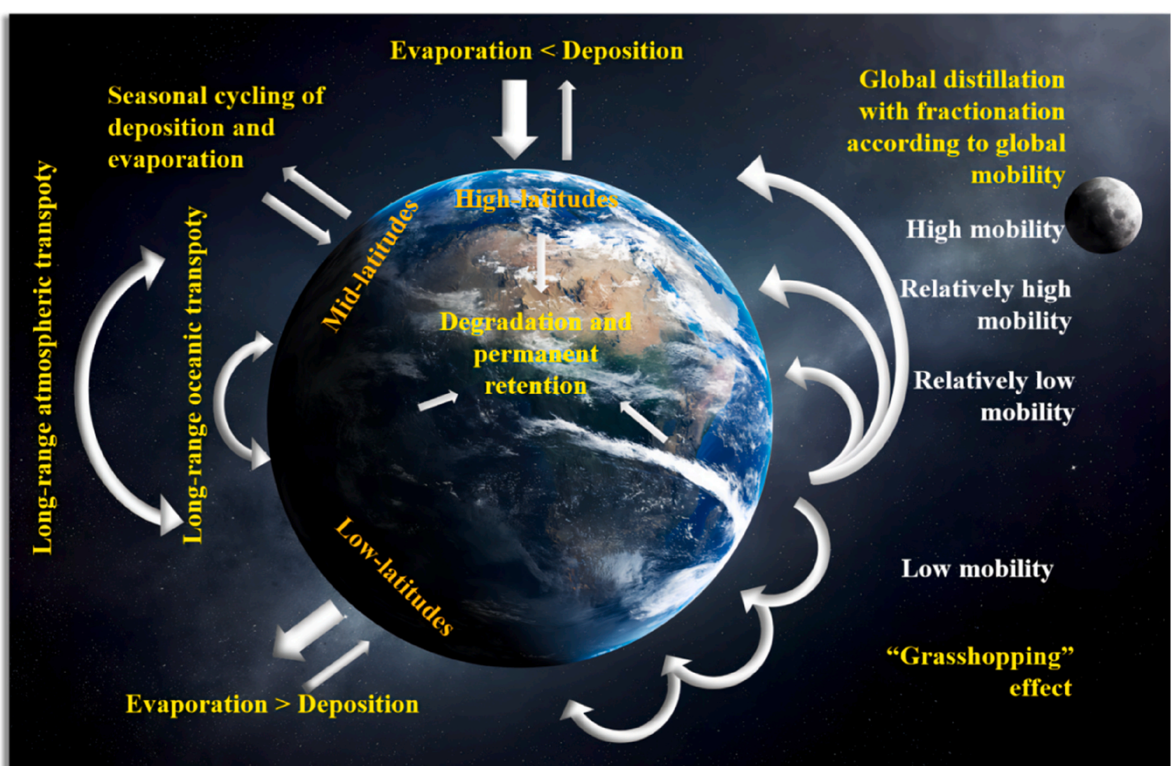

Fig. 2. The global transportation process of ECs.

plastic mulching films, with usage quadrupling between 1991 and 2011 (Luo et al., 2021). In 2011, it was estimated that approximately 427,000 ha of European farmland were covered by plastic mulching films, while China astonishingly reached a coverage area of 20 million hectares in the same year (Luo et al., 2021). Based on recent agricultural practices, there is an expectation for further expansion in the coverage area of mulching films in the future.

Polyethylene (PE) mulch films, widely used in agriculture, represent the primary source of plastic contamination in agricultural soils (Mo et al., 2021). Their appeal lies in their durability and versatility. However, these films are non-biodegradable, leading to time-consuming and expensive recovery processes in the field (Xiong et al., 2023). To address this issue, a new generation of biodegradable plastic films has been developed as a potential replacement for non-biodegradable synthetic polyethylene, which has been extensively used for several decades (Zhang et al., 2020a). Nevertheless, there exists a substantial knowledge gap regarding the complete degradation process of these films and the effects of byproducts and additives, such as plasticizers, on soil structure and soil biota (Wei et al., 2019). This knowledge gap is significant because incomplete degradation can lead to the formation of microplastics and nanoplastics, which persist in the soil and can be ingested by soil organisms, potentially entering the food chain (Astner et al., 2019). These persistent plastic particles pose long-term environmental risks, such as disrupting soil health, affecting soil microbial communities, and ultimately impacting agricultural productivity and ecosystem services. Consequently, due to concerns about the contribution of these mulch films to microplastic pollution, some European countries have implemented bans on their use (J. Qin et al., 2021a). The utilization of conventional plastic mulch has resulted in a notable accumulation of plastic residues within the soil, which can lead to diminished crop productivity due to changes in soil properties and potential toxic effects (Sintim et al., 2020). An additional concern regarding these plastic mulch residues is the occurrence of compounds that exhibit carcinogenic and endocrine-disrupting properties (Rillig et al., 2021). These residues serve as a principal source of phthalates in the soil (Xu et al., 2020), and there is potential for other hazardous organic chemicals to adsorb onto their surface and be subsequently released through transformations or contact with organisms (Khan et al., 2023). The release of such compounds into the soil poses significant risks to both ecosystems and human health, underscoring the necessity for further research to comprehensively elucidate the magnitude of this hazard.

2.2. Sources of urban areas

2.2.1. Domestic sources of pollution

Our surrounding environment is susceptible to a range of ECs, encompassing pharmaceuticals and personal care products (PPCPs), cleaning agents, indoor pesticides, and horticultural materials. Although discussions on indoor pollution often center around the impact of air pollution on human health, the pathways through which these pollutants enter the soil and their ecological ramifications receive limited attention. Antibiotics like tetracyclines, ibuprofen, and ciprofloxacin are extensively employed for treating various human and animal ailments in daily life. However, these substances are not entirely assimilated by organisms and are predominantly expelled through feces or discharged into the environment via wastewater. Plasticizers and additives, known as phthalates (PAEs), are widely utilized in numerous industrial goods, medications, and food packaging (Billings et al., 2024). Due to their high volatility, PAEs have been widely observed in diverse soil types (Kaur et al., 2024). The pervasive presence of PAEs raises concerns, particularly due to their established role as endocrine-disrupting compounds.

2.2.2. Electronic waste

The rapid pace of electronic product updates and iterations has led to the emergence of e-waste as a significant environmental challenge in contemporary society. According to global e-waste monitoring data in 2020, the global production of e-waste reached approximately 53.6 million tons in 2019, while the global collection and recycling rate of e-waste stood at a mere 17.4% (Forti et al., 2020). Projections indicate that by 2030, the global volume of e-waste will escalate to 74.7 million tons (Forti et al., 2020). Despite the implementation of regulations by many countries to appropriately manage this hazardous waste, a substantial portion of e-waste continues to end up in landfills, incinerators, is exported to developing countries, or is handled by informal sectors (Rajesh et al., 2022). Prior to the ban, China served as the largest importer of e-waste, and with the growing domestic demand, China is anticipated to become the world's largest producer of e-waste (Fu et al., 2018).

E-waste encompasses electronic devices, household appliances, and communication equipment, which contain a multitude of ECs, such as phthalates and brominated flame retardants, that pose significant risks to the environment and human health. Phthalates can act as endocrine disruptors, while brominated flame retardants are known for their persistence and bioaccumulation, potentially leading to neurodevelopmental deficits and thyroid hormone disruption (Goodes et al., 2023; Lyu et al., 2022; Rathi et al., 2021). Improper recycling methods and inadequate handling not only lead to the release of a substantial quantity of ECs from e-waste but also contribute to secondary pollution of the environment. In certain urban areas, discarded electronic devices and municipal waste are inadequately segregated and often co-disposed in landfill sites, resulting in dire consequences for the surrounding soil environment (Chakraborty et al., 2019). Additionally, particulate matter generated from incomplete incineration of e-waste settles on the soil surface, thereby becoming a source of ECs in the soil (Chakraborty et al., 2019). Operational activities in dismantling workshops can lead to the release of polybrominated diphenyl ethers (PBDEs), which are used as flame retardants in many electronic devices, into the surrounding air and soil environments (Liu et al., 2019a). Furthermore, studies have indicated severe contamination of PBDEs during the dismantling process of e-waste, particularly in waste printed circuit boards. PBDEs are persistent organic pollutants known for their potential to disrupt endocrine function, cause neurodevelopmental deficits (Płotka-Wasylka et al., 2023), and accumulate in the food chain, posing significant risks to both the environment and human health. Currently, some scholars argue that simple e-waste recycling measures in China pose a significant environmental risk. They suggest that recycling raw materials and bromine might be more effective in controlling PBDEs (Ni et al., 2013). Additionally, the latest Action Programme for the Control of New Pollutants, published by the State Council, highlights China's efforts to strengthen control of these emerging pollutants at the source and during processing. The draft list includes a system for the assessment and monitoring of related chemical substances.

2.2.3. Urban wastewater irrigation

According to data provided by the Ministry of Housing and Urban-rural Development of the People's Republic of China (2022), China's annual sewage discharge reached $6.25 \times 10\hat{}10$ m^3 in 2021, representing a year-on-year increase of 9.40% compared to 2020. As of the end of 2021, 12.37% of urban sewage treatment capacity in China still failed to meet the requirements of Class I-A standards, which are the highest level of treatment standards in China, designed to ensure that treated sewage has minimal environmental impact (Ministry of Ecology and Environment of the People's Republic of China, 2023). Challenges in urban sewage treatment in China are further underscored by issues such as delayed construction of sewage pipe networks and low levels of sewage resource utilization (Zhang et al., 2023a). With the rapid pace of urbanization and population growth, municipal sewage treatment and discharge have emerged as the primary source of pollution from ECs in urban areas.

Pharmaceutical and personal care product (PPCP) residues enter the sewage system through wastewater drains. Due to their ability to maintain their original concentration and structure in aquatic environments, PPCPs can persist for a relatively long time in water bodies and can undergo transformations into other active or inactive compounds during their lifecycle (Yang et al., 2024). Per- and polyfluoroalkyl substances (PFAS) are extensively used in industrial and consumer products such as waterproof coatings, food packaging, and household items. These substances are known for their persistence in the environment due to the strong chemical bonds, particularly the carbon-fluorine bond, which makes them extremely resistant to degradation. This chemical stability means they do not easily break down through natural processes, leading to their accumulation in the environment. Conventional wastewater treatment methods have limited efficacy in removing these substances, and as a result, wastewater treatment plants are considered a primary source of PFAS in aquatic environments. The persistence of PFAS poses significant challenges for environmental remediation and human health (Seay et al., 2023). Conventional wastewater treatment methods have limited efficacy in removing these substances. As a result, wastewater treatment plants are considered the primary source of PPCPs and PFAS in aquatic environments (Golovko et al., 2021). With the substantial discharge and utilization of wastewater, these compounds can enter the soil, leading to irreversible impacts and contamination. While the impacts of plastic usage on agricultural soil have been well-documented, it is crucial to recognize that urban sources such as wastewater irrigation and electronic waste also significantly contribute to the presence of ECs in agricultural soil. These contaminants can travel through water runoff, atmospheric deposition, and direct application of urban wastewater in agriculture, thus influencing soil health and productivity.

While the utilization of treated wastewater for irrigation is not prevalent in developed countries, it is common in developing nations due to ongoing water scarcity issues. This practice is particularly prominent in urban or peri-urban areas (Gallego et al., 2022; Pratap et al., 2021; Samarah et al., 2020; Zhang and Shen, 2019). In Israel, approximately 50% of the overall agricultural production is achieved through the use of treated wastewater for irrigation (Reznik et al., 2017). This practice not only facilitates the advancement of agriculture in urban and peri-urban areas but also mitigates the dependence on chemical fertilizers, thanks to the presence of organic pollutants (Pratap et al., 2021; Samarah et al., 2020).

3. Migration pathways of ECs in soil

The migration pathways of emerging contaminants (ECs) in soil can be categorized into horizontal and vertical movements. Horizontal migration pathways typically include surface runoff and lateral subsurface flow (Y. Qin et al., 2021b), where contaminants move across the soil surface, influenced by factors such as soil type, soil composition, and the properties of the contaminants (Yan et al., 2020; Xiang et al., 2022; Kumar et al., 2023). Vertical migration pathways involve the downward movement of contaminants through the soil profile via processes like infiltration and leaching, as well as the impact of soil organisms (Sun et al., 2022a,b; Shu et al., 2023). In soil, the extent of horizontal migration often determines the contamination spread of ECs, with wider dispersion affecting more ecosystems and water bodies. In contrast, vertical migration usually determines the severity of EC impact, making ECs more likely to contaminate groundwater and deeper soil layers, which are more challenging to remediate. A comprehensive understanding of the transport and migration of ECs in soil, and their subsequent impact on the environment, is therefore critical.

3.1. Plastics and synthetic polymers

Water and air are widely recognized as the primary pathways for the transport of surface materials in soil. When plastic products, such as plastic mulch, are exposed to the external environment over an extended period, they undergo processes of photodegradation or weathering, gradually fragmenting into smaller particles known as microplastics/nanoplastics, which can infiltrate the soil interior through pore spaces (B. Sun et al., 2020a). Some plastic particles, characterized by their low density, disperse through air or water flow and typically contaminate other environments as diffuse sources of pollution (Chen et al., 2020). In contrast, high-density plastics exhibit a tendency to persist within the soil matrix and gradually penetrate the soil interior, rendering them less prone to horizontal migration (Xiang et al., 2022). In agricultural settings, mechanical or manual tillage and soil disturbance facilitate the downward displacement or repositioning of microplastics (MPs) in the soil, thereby promoting the redistribution and potential accumulation of MPs in deeper soil layers (Windsor et al., 2019).

Furthermore, the formation of pores by plant root systems also contributes to the vertical transport of ECs in the soil. MPs and other small particles can move up and down through the pore channels, driven by their buoyancy, thereby influencing their vertical distribution within the soil matrix (Li et al., 2021b). Recent research has highlighted the impact of soil biota on the horizontal and vertical migration of ECs (Heinze et al., 2021). For example, the burrowing activity of earthworms can lead to the expansion of soil fissures, creating additional pathways and routes for MPs to infiltrate deeper soil layers (Heinze et al., 2021). This activity facilitates the downward movement of MPs from the soil surface and promotes preferential leaching of plastic particles through the formation of burrow channels (Huerta Lwanga et al., 2017). Additionally, Rillig et al. (2012) mentioned the role of soil organisms such as Mites (Hypoaspis aculeifer) and Collembola (Folsomia candida and Proisotoma minuta) in the migration and distribution of MPs within the soil. Their scraping and chewing behavior contribute to the movement of MPs and influence their distribution in the soil. These findings underscore the significance of biological pathways in the fate and transport of MPs in soil systems.

Finally, the presence of heavy metals and organic pollutants in the soil can exert influence on the migration of MPs. Yan et al. (2020) conducted a soil leaching experiment to investigate the migration behavior of clean microplastics (CMP) without soil minerals, original microplastics (RMP) with soil minerals, and RMP with humic acid (HA). The experimental findings indicated that surface charge is among the crucial factors affecting the migration of microplastics.

3.2. Pharmaceuticals and personal care products

Pharmaceuticals and personal care products (PPCPs) are released into the environment through various pathways, including urban sewage, domestic wastewater, landfill sites, industrial wastewater, hospital wastewater, and urban or residential sewage (Ramprasad and Philip, 2018; Yoo et al., 2018). Recent studies have demonstrated the high mobility of certain PPCPs, such as ibuprofen, enabling their infiltration into deep soil layers (Shu et al., 2023). These PPCPs can also leach into groundwater, leading to their accumulation in soils and sediments. Conversely, certain PPCPs, like bisphenol A, exhibit strong adsorption and can persist in soil for prolonged durations (Zi et al., 2023). Positively charged antibiotics, including tetracyclines and ciprofloxacin, exhibit minimal migration rates in natural soils due to electrostatic and hydrophobic interactions, resulting in their long-term residence in soil profiles (Dai et al., 2020). Moreover, the transfer of PPCPs in soil is influenced by soil type. Menahem et al. (2016) observed that Gd-DTPA and ROX ions displayed more dispersed migration characteristics in soils compared to sandy soil, which can be attributed to the complex pore structure, abundant adsorbents, and active microbial activity in the soil. In conclusion, the fate of PPCPs in soil is characterized by complexity and variability, necessitating further comprehensive research to enhance our understanding of their behavior in soil environments.

It is noteworthy that studies have confirmed a significant correlation between the migration of pharmaceuticals and personal care products (PPCPs) in soil and the content of soil organic matter (SOM) as well as moisture levels (He et al., 2023; Liu et al., 2023). Multiple interaction mechanisms exist between SOM and PPCPs, resulting in soils with higher organic matter content generally exhibiting a stronger affinity for PPCPs. Conversely, the impact of soil moisture on the mobility and retention of PPCPs with lower soil affinity is relatively minor. By reducing soil moisture content, the adsorption capacity of PPCPs with higher soil affinity can be substantially enhanced (Dai et al., 2020; Jiang and Dai, 2023). The findings of this conclusion carry significant academic and practical implications for further research on soil remediation techniques and the mitigation of soil pollution risks associated with PPCPs.

3.3. Plasticizers and flame retardants

Due to the inability of plasticizers to chemically bind with polymers, they can easily leach into the environment and be released during the weathering process of plastics (X. Li et al., 2024b; Luo et al., 2022). Polybrominated diphenyl ethers (PBDEs) are brominated flame retardants extensively utilized to enhance the fire resistance of polymer materials in household appliances (Dai et al., 2023; Ma et al., 2024). As pivotal constituents, these compounds are frequently encountered in electronic waste, and they exhibit wide distribution in the environment, including soil, thereby presenting potential hazards to human health and ecosystems (Li et al., 2019; Zhu et al., 2019). Therefore, understanding the fate of such substances in soil is of paramount importance.

Phthalate esters (PAEs) are commonly used as plasticizers in electronic devices to enhance flexibility and plasticity. However, they can also enter soil through agricultural activities involving the use of plastic films and fertilizers (Zhang et al., 2020d). The distribution of PAEs in soil is influenced by environmental conditions, such as pH, temperature, and pressure, as well as their physicochemical properties, including water solubility and molecular weight (Tran et al., 2022). PAEs exhibit strong adsorption in soil, which hinders their vertical migration within soil profiles. As a result, PAEs are typically found in lower concentrations in deeper soil layers and are predominantly located in the surface soil region (Tran et al., 2022). Moreover, the vertical transport capacity of PAEs in soil decreases with increasing alkyl chain length, and their distribution patterns are affected by variations in soil particle size (Tao et al., 2023). Similarly, brominated flame retardants like PBDEs show a decreasing spatial distribution in soil with decreasing depth, mainly due to their binding with organic matter and faster degradation rates.

3.4. Per- and polyfluoroalkyl substances

Per- and polyfluoroalkyl substances (PFAS) are ubiquitous in soil, entering the environment not only through direct pathways such as landfill sites and wastewater discharge but also through atmospheric deposition. During long-distance atmospheric transport, volatile neutral PFAS undergo degradation and transform into ionized forms, which are more susceptible to wet deposition in soil (Wang et al., 2023a). Extensive research has examined the behavior and fate of PFAS in soil (Lyu et al., 2019).

The distribution and transport mechanisms of PFAS in soil are influenced by various physicochemical properties, including organic matter content, mineral composition, solution ion strength, and pH (Abou-Khalil et al., 2022; Qi et al., 2022; Li et al., 2023). Electrostatic and hydrophobic interactions are considered as the primary mechanisms governing the retention and transport of PFAS in soil (Y. Li et al., 2024c). Regarding the vertical distribution within soil profiles, different chain lengths of PFAS exhibit distinct patterns, with long-chain PFAS predominantly found in shallow soil depths and short-chain PFAS more prevalent in deeper regions (Dauchy et al., 2019). Li et al. further observed that the transport of long-chain PFAS is mainly influenced by hydrophobic interactions, while short-chain PFAS are more influenced by electrostatic interactions (Li et al., 2023). These research findings offer valuable insights into the behavior and migration of PFAS in soil.

4. Specific impacts of ECs on soil

4.1. Specific impacts of ECs on soil

4.1.1. Soil physicochemical properties

The presence of ECs in soil can result in irreversible alterations in soil composition, leading to changes in soil properties (Teklu et al., 2023). Wan et al. (2019) discovered that microplastics exacerbate soil moisture loss by influencing evaporation, thereby reducing water-holding capacity and causing significant disruption to soil structure. Furthermore, the accumulation and interaction of microplastic particles, such as polyester fibers, with soil particles can potentially induce entanglement and interlocking, consequently impacting the physical properties of soil (Zhang et al., 2019). The presence of phthalate esters (PAEs) in the soil is concerning due to their chemical impacts, such as endocrine disruption and toxicity to soil organisms. However, recent studies have also shown that PAEs can influence soil physical properties (Tao et al., 2020). For example, PAEs can interact with soil particles, leading to changes in soil structure. Research indicates that PAEs can facilitate the aggregation of soil particles, which increases bulk density and decreases soil porosity. This aggregation effect is due to the interaction between PAEs and organic matter in the soil, leading to the formation of larger soil aggregates (Gao et al., 2021; Li et al., 2020; Sun et al., 2019). These structural changes can affect soil aeration, water retention, and root penetration, ultimately impacting plant growth and soil health. Moreover, soils with elevated concentrations of PFAS commonly exhibit higher pH values, which are closely associated with soil organic matter decomposition (Xu et al., 2023). It remains unknown whether these alterations further impact the behavior and fate of ECs in soil, necessitating further research and exploration.

4.1.2. Soil fertility and nutrient cycling

In agricultural research, soil fertility is a crucial parameter used to evaluate nutrient levels and the capacity of soils to supply nutrients. Nutrient cycling is a fundamental mechanism that sustains soil fertility and plays a pivotal role in enhancing agricultural productivity. The soil microbial community plays a critical role in pollutant purification and nutrient cycling, particularly nitrogen, in the soil environment (Cavicchioli et al., 2019; Li et al., 2022). Consequently, any compounds that impact microbial communities or disrupt nutrient cycling have the potential to disrupt the delicate balance of ecosystems (Rillig et al., 2019).

Elevated concentrations of microplastics have a significant impact on the nutrient content of dissolved organic matter (DOM) solutions, leading to the accumulation of dissolved organic nitrogen (DON), dissolved organic carbon (DOC), and dissolved organic phosphorus (DOP). These findings suggest that ECs can actively participate in soil nutrient cycling and exert influences on soil fertility (Allouzi et al., 2021; Chen et al., 2022). Moreover, Iqbal et al. highlighted the potential risks of microplastics to nitrogen cycling and proposed that soil nitrogen cycling may be indirectly affected through multiple concurrent mechanisms (Iqbal et al., 2020). Certain ECs have been shown to diminish the diversity of soil microbial communities, which are essential for crop growth, consequently resulting in reduced soil fertility and compromised crop development (Ren et al., 2023). In a recent study, Tao et al. (2022) investigated the impact of di-*n*-butyl phthalate (DBP) on soil microbial nitrification performance. Their findings revealed that alterations in the microbial community composition induced by DBP favored the proliferation of heterotrophic microorganisms, consequently inhibiting the activity of chemoautotrophic nitrifying bacteria responsible for ammonia oxidation. DBP stress resulted in a significant decrease in the abundance of amoA and nxrA genes, leading to impaired soil nitrification capacity (Tao et al., 2022). Additionally, DBP exposure caused changes in soil urease activity, further influencing nitrogen cycling processes.

While considerable research has been devoted to investigating nitrogen cycling in soils, our understanding of the effects of contaminants on soil carbon cycling remains limited. The carbon cycling process plays a critical role in mediating carbon flux between soils and the atmosphere, making it of significant importance for studying greenhouse gas dynamics (Zhang et al., 2022d). Building upon this knowledge gap, Wang et al. conducted a study where they observed that DBP exposure adversely impacted the microbial community in saline soils. Specifically, they found that the activity of key soil transformation enzymes and β-glucosidase was suppressed, leading to accelerated carbon metabolism and subsequent soil carbon loss (Wang et al., 2023b). These findings highlight the intricate interactions between contaminants and soil microbial processes, underscoring the need for further research to fully comprehend their implications on soil biogeochemical cycling and ecosystem functioning.

4.1.3. Accumulation and hazards in soil biota

In recent years, there has been a growing interest in the complex interactions between ECs and soil biota. Soil organisms accumulate ECs through processes such as adsorption, ingestion, metabolism, and transport, leading to their gradual transfer through the food chain. ECs enter the soil environment and are taken up by plants and crops, subsequently consumed by animals, poultry, and humans (Lehel et al., 2021). They can also enter the soil through invertebrates, which are then consumed by birds and poultry, ultimately reaching humans (Gupta et al., 2021; Santeramo et al., 2021) (see Fig. 3). Therefore, it is of paramount importance to fully comprehend the potential hazards associated with the accumulation of ECs in soil biota.

The accumulation of ECs can exert detrimental effects on soil biota through various mechanisms. For example, the main mechanisms of antimicrobial-resistance are shown in Fig. 4 (Dias et al., 2022; Zhang et al., 2022a). Secondary consequences of pharmacological exposure include acute and chronic toxicity, bioaccumulation, endocrine anomalies, and drug-resistant micro-organisms (Bradley et al., 2020; Gould et al., 2021). Huerta Lwanga et al. (2017a,b) demonstrated the negative impact of microplastics on the growth and survival of earthworms by exposing them to soil amended with microplastics. This exposure led to a reduction in earthworm body weight and an increase in mortality rate. Additionally, the detrimental effects of ECs on the soil-plant system warrant further investigation. For instance, microplastics can impede the process of photosynthesis in plants. Plants exposed to microplastics often exhibit partial closure of stomata in

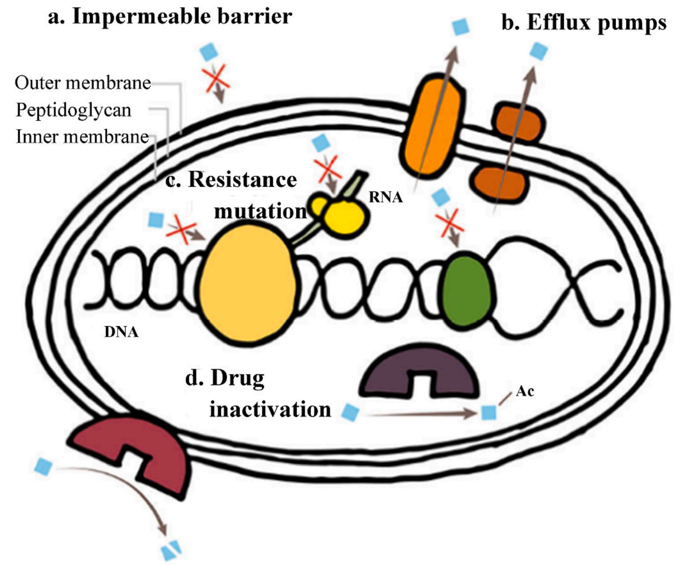

Fig. 4. Principal mechanisms of drug resistance.

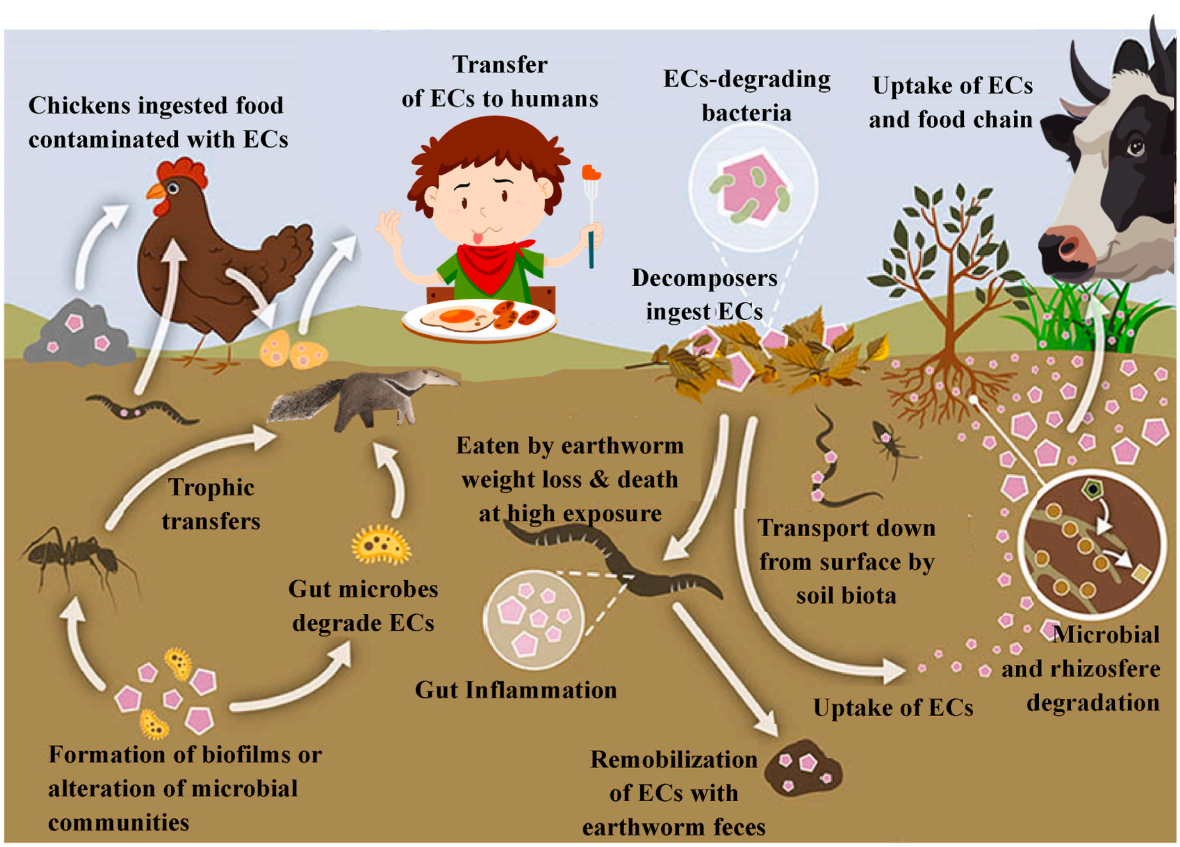

Fig. 3. ECs transfer into the terrestrial food web from the soil to pastures and crops, which are ingested by wildlife, livestock and humans, or from the soil to invertebrates, ingested by birds and poultry and ultimately transferred to humans.

response to self-protection, resulting in decreased transpiration rate and stomatal conductance, thereby limiting the dark reactions of photosynthesis and diminishing the overall photosynthetic rate (Wang et al., 2022a, 2022b; Yildiztugay et al., 2022). Moreover, experimental studies have indicated that certain ECs may adversely affect the stability of photosynthetic pigments, such as chlorophyll and carotenoids, leading to their degradation. Furthermore, some ECs can disrupt the synthesis of photosynthetic pigments within plants, inhibiting the activity of relevant enzymes or modulating the expression of genes involved in photosynthetic pigment biosynthesis. This disturbance hampers the plants' ability to produce an adequate number of photosynthetic pigments, thereby reducing photosynthetic efficiency (Sun et al., 2022a,b; Zhang et al., 2022b, 2022c). Moreover, recent studies have demonstrated the accumulation of antibiotics, including tetracyclines, in the extracellular spaces between plant cells, leading to adverse impacts on plant growth and development (Rahman et al., 2023).

In the case of lower plants, prolonged exposure to ECs can induce damage to the cell membrane and internal structures of plant cells. Phenomena such as pyrene opacification and distortion of thylakoid are attributed to the physical injury caused by ECs and the subsequent oxidative stress (Mao et al., 2018). Research has indicated that ECs can also have detrimental effects on seed germination in plants, resulting in a negative impact on germination rates (Guo et al., 2022). The precise mechanisms underlying the toxicity of ECs remain unclear, highlighting the need for comprehensive investigations to better understand their potential risks in soil ecosystems.

4.2. Feedback mechanisms of ECs by soil

The invasion of ECs has the potential to modify soil environmental indicators. However, the intricate and heterogeneous nature of the soil environment, coupled with the interplay among physical, chemical, and biological factors, can also influence the migration, transformation, and degradation processes of ECs (Zhang et al., 2023b). Therefore, alongside investigating the impacts of ECs on soil, it is crucial to consider the reciprocal relationship between soil and ECs, particularly the interaction with soil microbial communities. We postulate that the composition and function of soil microbial communities may exhibit a responsive nature in the presence of ECs. Previous studies examining the degradation and interaction of ciprofloxacin (CIP) in soil have uncovered complexities that challenge conventional understandings. It is widely accepted that the soil matrix hampers antibiotic degradation. However, under actual soil conditions, CIP degradation can surpass that in water, attributed to its resistance to non-biological degradation reactions, such as hydrolysis, and its inhibitory effect on microbial activity owing to its high toxicity (Huang et al., 2024). As the microbial inhibition caused by CIP demonstrates selectivity (Liu et al., 2018b), we speculate that while it suppresses certain microbial activities, it indirectly stimulates the growth of other active bacteria involved in CIP degradation, thereby prolonging degradation time and inflicting persistent harm to the soil. The interactions between soil and environmental pollutants encompass a comprehensive range of dynamics, extending beyond mere degradation processes. In terms of environmental pollutant migration within the soil, it is hypothesized that specific ECs, such as MPs, have the capacity to modify soil physicochemical properties (e.g., water content, particle size), thereby exerting indirect influence on other pollutants like PPCPs and PAEs. A comprehensive investigation and further research encompassing ECs as a whole are warranted.

5. Countermeasures against soil pollution

5.1. In situ

5.1.1. In situ physical treatment

Soil vapor extraction is a technique that involves creating a vacuum and pulling air through the soil in the unsaturated vadose layer with extraction pumps (Fig. 5) (Shi et al., 2020). The ECs pass from the soil and groundwater toward the air bubbles that subsequently rise further into the vadose zone. (Xu et al., 2021a,b). The air flow rates are typically high since the goal of both of these procedures is to volatilize the ECs. ECs may be degraded aerobically in the presence of high air fluxes, even if extraction is the primary goal (Liu et al., 2018a). Both techniques use a vacuum extraction system to remove air and volatile ECs, and then a treatment process to remove the volatile ECs (Das and Hageman, 2020). However, soil vapor extraction remediation efficiency is hindered by high soil moisture content. In addition, soil vapor extraction is difficult to deal with ECs with low volatility and high boiling points.

5.1.2. In situ chemical treatment

To minimize EC mobility, they are physically bound or contained inside a stabilized mass (solidification), or chemical interactions between the stabilizing substance and the ECs are generated (stabilization) (Turner et al., 2022). Organic and inorganic compounds' mobility may be lowered by a variety of precipitation, complexation, and adsorption processes (Yan et al., 2022). Some of the most commonly used inorganic stabilizing agents are soluble silicates, zeolites, lime, phosphates, and Sulphur-based binders (Budnyak et al., 2020). Organic compounds that are poorly handled by precipitation and complexation processes have been stabilized using organo-clays (Liu et al., 2022a). To bind high concentration inorganic ECs, cementitious techniques using lime or cement might be employed (Ghavami et al., 2019). In both cases, the soil is unlikely to be suitable for farming, and the location will need to be monitored over time to ensure that the ECs remain immobile.

Activated carbon and biochar may help recover contaminated soil to agricultural use as well by lowering the bioavailability of ECs (Jiang et al., 2024; Ren et al., 2023). In addition to stabilizing ECs, As a climate change mitigation method, adding carbon into the soil helps trap carbon and improves soil fertility (Dong et al., 2020). However, owing to aging causes such as the leaching of biochar alkalinity, its efficacy diminished with time (Bandara et al., 2020). Activated carbon and biochar after adsorption of ECs are difficult to recover from the soil, and the true separation of ECs from soil cannot be achieved (Palansooriya et al., 2022; Wang et al., 2021b). Chemical oxidizing agent solutions are injected directly into contaminated soil or deployed in the route of an EC plume as a permeable barrier. Peroxide, ozone, permanganate, and persulphate are the most regularly used chemical oxidizing agents (Xu et al., 2020). The ECS can be easily removed from the high permeability zones when a site has discrete zones with high and low permeability (Shao et al., 2021). After some time, ECs return from the low-permeability zones where the oxidants failed to reach them (Feng et al., 2021; Zhang et al., 2020b). Because of this phenomenon, which is

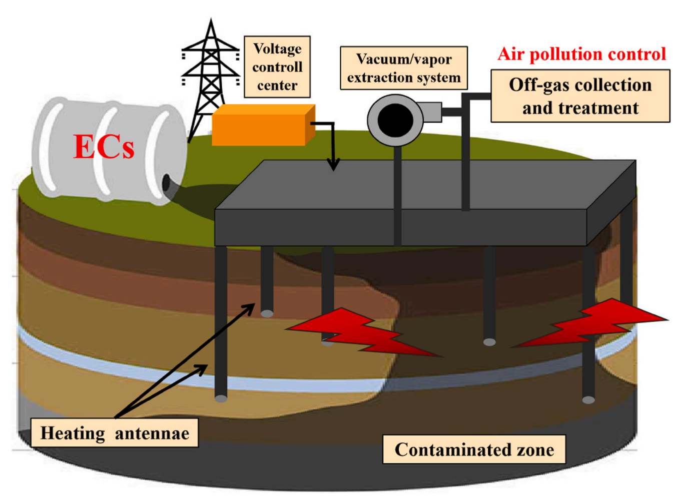

Fig. 5. The process of soil vapor extraction.

referred to as "rebound," the long-term usefulness of the technology might be severely limited, and numerous treatments may be required (Li and Jiang, 2021). A careful study of the proposed site model will aid in the identification of possible hazards associated with redevelopment (Dannecker et al., 2019). This approach can handle high quantities of organic contamination from a point source (Ostovar et al., 2021). As the method entails the use of potentially dangerous oxidizing chemicals, it is vital to assess the risks and benefits of this as well.

5.1.3. In situ biological treatment

By stimulating native microbes (bio stimulation) or by introducing specialized microbes to the local population, microbial decomposition of organic ECs in soil may be improved (bioaugmentation) (Xu et al., 2019). Aeration, the inclusion of nutrients, pH adjustment, and temperature control may all help to improve EC decomposition (Li and You, 2021). Materials having high concentrations of nutrients, labile carbon, and microorganisms, such as manure or sewage sludge, may be added to speed up biodegradation (Boonluksiri et al., 2021). Microorganisms that have evolved to breakdown certain organic ECs at another location may sometimes be used to speed up the process (Zhang et al., 2021b). Exotic microorganisms, on the other hand, might well not operate as planned when inserted into a foreign soil biome, resulting in disturbance of ecosystem functioning and autochthones biodiversity following application (Zhang et al., 2020c). The type of soil and its characteristics, as well as the site's other environmental factors, influence the speed and efficacy of in situ biological therapy.

In the case of contaminated soils, phytoremediation serves two purposes: stabilizing ECs to make them less mobile and hence less prone to cause damage, as well as eliminating them by assisting their breakdown or moving them to a new medium (see Fig. 6). As a biological treatment technology, the success of in situ bioremediation is determined by many related factors (He et al., 2020). It may take an exceptionally long time for plants to transport and concentrate ECs from the soil to the harvestable sections of the roots, based on their usage of plants to transport and concentrate ECs from the soil to the harvestable parts of the roots (Min et al., 2021). Therefore, in order to achieve a better effect of pollution removal, it is necessary to design a targeted in situ bioremediation scheme according to local conditions (for example, high-efficiency biological breeding, biological recycling).

Earthworms are tolerant of a lot of chemical contaminants, including soil trace elements and ECs, and can bioaccumulate in tissues (Zeb et al., 2020). Various earthworm species have been proven to be effective in removing pollutants from soil, including pesticides, polycyclic aromatic hydrocarbons (PAH), and other lipophilic organic contaminants. It is via their wet body wall that they are able to absorb chemicals from the soluble soil portion in the interstitial water, and they also consume them (Wang et al., 2018a). Contaminants in the soil are either bio-transformed or biodegraded as they travel through the earthworm stomach, making them non-toxic (Sanchez-Hernandez et al., 2019). Meanwhile, due to the turning action of earthworms on soil and the action of a large number of soil microorganisms, the physical, chemical and biological properties of soil are enriched (Wang et al., 2018b). However, based on earthworm toxicology and pollutant degradation rates, measuring and forecasting their impacts on soil organic pollutant elimination remains difficult (Wang et al., 2020). Therefore, consideration of earthworm susceptibility is critical for a general understanding of how earthworms affect pollutant transformation in soil.

5.1.4. In situ other measures

The categorization of in situ thermal treatment includes four technologies: heating with electrical resistance, conductive heating, radiofrequency heating, and vitrification (Shi et al., 2019). The basic purpose of these methods, with the exception of vitrification, is to heat the soil and make it simpler to remove volatile and semi-volatile ECs (Matsumoto and Liu, 2020). The chemical and biological breakdown of certain ECs may be accelerated by higher temperatures (Wei et al., 2020). Ex situ recovery or destruction of deployed ECs and their degradation products are among these strategies (Ding et al., 2019). In situ vitrification converts soil to glass at significantly greater temperatures (Kuo and Wu, 2021; Ossai et al., 2020). Non-volatile ECs, such as trace elements and radioactive compounds, are integrated into the glass, which immobilizes them when it is cooled (Shu et al., 2020; Shu et al., 2021; Yan et al., 2021). The use of reactive nanoparticles for EC transformation and detoxification is referred to as nano remediation (Ahmed et al., 2021). These nanomaterials possess characteristics that allow for both chemical and catalytic reduction of the target ECs (Baragano et al., 2020). For nano remediation in situ, no groundwater is pumped out for above-ground treatment, and no soil is transferred to other areas for

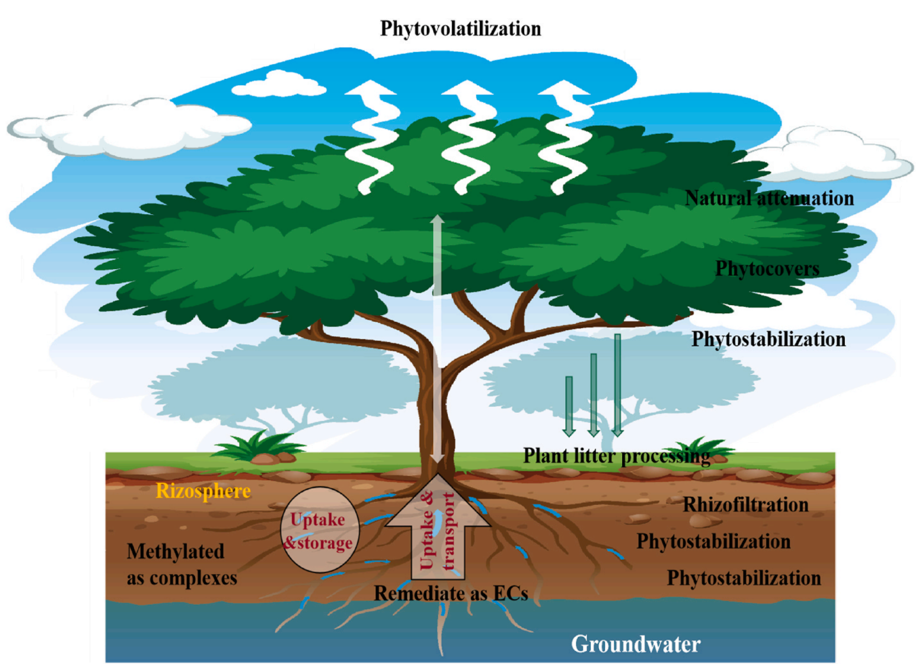

Fig. 6. The process of phytoremediation.

treatment and disposal (Rodriguez-Seijo et al., 2020). For in situ applications, nanomaterials offer a number of desirable features. Due to their tiny size and novel surface coatings, nanoparticles may be able to penetrate very small crevices in the subsurface and stay suspended in groundwater, theoretically enabling them to travel farther than bigger, macro-sized particles and achieve a broader spread (Pak et al., 2020). However, contemporary remedial nanoparticles do not travel far from their injection location (Baragano et al., 2021). Nano remediation technology is an emerging tool to mitigate the ECs pollution crisis. Nanotechnology has shown great promise in photocatalytic ECs decomposition and soil remediation.

5.2. *Ex situ*

5.2.1. *Ex situ physical treatment*

Soil washing is an ex situ water-based scrubbing procedure for removing organic and inorganic pollutants from soils (Liu et al., 2022b). The method physically eliminates ECs from soils. To distinguish those particles that carry the majority of the contamination from the bulk that is contaminant-free, variations in physical soil washing, particle grain size, settling velocity, specific gravity, surface chemical behavior, and, on rare occasions, magnetic properties are used (Ahn et al., 2021; Baek et al., 2021; Trellu et al., 2022). Scrubbing coarser particles with attrition eliminates sticky contaminant coatings (Gautam et al., 2020). Attrition washing, on the other hand, might increase the particles in the treated soils. Soil washing systems, which include the majority of the removal and separation methods, are frequently used as the initial stage in an ex-situ treatment train for EC-polluted soils. Soil washing facilitates the transfer of ECs from soil colloids into the washing solution. Additional processing techniques are required to be responsible for the efficient separation of ECs from soil particles.

5.2.2. *Ex situ chemical treatment*

Chemical extraction separates hazardous pollutants from soils, minimizing the amount of hazardous waste that has to be handled (Alvarez-Quintana et al., 2020). There are several extraction processes available, including dissolving in strong inorganic acids, the formation of complexes with chelating agents, and dissolution in organic solvents. The process is similar to soil washing, instead of water and wash-improving additives like surfactants, it employs an extracting chemical (Cao et al., 2021). Due to the high hazards and expenses associated with the use of hazardous extractants, as well as the necessity to treat secondary effluents, ex situ extraction procedures are rarely used. Chemical reduction and oxidation reactions in ex situ are equivalent to those discussed above for in situ processes (Ma et al., 2019). The benefit of ex situ reactions is that they can be done more quickly, efficiently, and controllably thanks to mechanical mixing and simple monitoring.

5.2.3. *Ex situ biological treatment*

Bio piling is a bioremediation technique widely employed by a variety of ECs for ex situ remediation of contaminated soils (Liu et al., 2019b). The procedure entails homogenizing and amending the excavated soil in order to improve the conditions for biodegradation by microorganisms. Bioleaching is one of them. It is an extractive process that involves leaching trace elements from contaminated soils using a solution infected with microorganisms (Wu et al., 2020). Landfarming is an ex situ bioremediation process in which contaminated soil is dug and spread thinly layer over biologically active land or an impermeable surface at a landfarming location (Kim et al., 2022; Macci et al., 2021). The contaminated dirt is plowed into the surface of the soil, with organic amendments applied on occasion (e.g. manure or sewage sludge) (Badzmierowski et al., 2021). This process is stimulated by inherent microbes to stimulate biodegradation and is facilitated by the addition of nutrients, and repeated tillage of the microbes can be stimulated aerobic (Lee et al., 2020; Liu et al., 2021; Y. Sun et al., 2020b; Wang et al., 2022a,b,c). Bio pile treatment is a good technique to restore residual contaminated areas by improving soil properties and reducing the toxicity of potential ECs. Regardless, bio heap treatment requires soil and biological monitoring to prevent the bioavailability of ECs from increasing over time.

Composting is an aerobic ex situ process that includes combining excavated polluted soil with straw or green waste to feed microorganisms with optimal quantities of oxygen and moisture. The mixture typically comprises 75 % contaminated soil and 25 % organic fraction, however this may be adjusted depending on the type of soil, ECs, and their concentrations (Pinzon-Nuñez et al., 2022). Mechanical agitation may be used to mix the pile well, keep air moving, and ensure sufficient bioactivity throughout the pile to speed up the composting process (Kawecki et al., 2021; Tang et al., 2020). Composting not only reduces the concentration of ECs in soil but can also be used as a soil amendment to maintain and improve functional agricultural soils. Composting is a common ex-situ biological treatment technology for ECs.

Bioreactors are open or closed systems that stir the soil while supplying nutrients to encourage microorganisms to breakdown pollutants (Srivastava et al., 2022). Based on the pollutant and microorganisms most suited for biodegradation, they may be operated aerobically or anaerobically. Typically, remediation is completed in a shorter period of time than when using a static remediation strategy. A solid phase and a liquid phase are the two most common. In most cases, a solid phase bioreactor is often a batch process carried out in a reaction chamber with internal stirrers (Sbahi et al., 2021). A filthy soil is showered with water and stirred with mixers in a liquid phase bioreactor. It's also a batch operation (Arabmarkadeh et al., 2020). After the appropriate level of contamination has been achieved, the soil solids can be filtered and returned to the site, while the water can be preserved for the next batch (Arabmarkadeh et al., 2020). Homogeneous materials and direct contact between microbes and pollutants make the biodegradation process in bioreactors extremely controlled and typically speedy (Chen et al., 2019). Bioreactors are considered an "environmentally friendly" soil cleanup technique with less impact on soil functional properties and the overall environment. Bioreactors also has some disadvantages, such as the need to excavate the soil and build and operate the bioreactor, resulting in additional costs.

5.2.4. *Ex situ other measures*

To detoxifying contaminated soil polluted with organochlorine pollutants and other ECs, ball mills have been utilized as a mechanical-chemical destruction approach (Turner et al., 2021). Mechanical chemical processing fractures the surfaces of the contaminated soil matrix, resulting in highly reactive areas. These reactive sites may combine with contaminating molecules to create contaminant radicals, which then break down into smaller 'daughter' ions and eventually neutral species (Hu et al., 2018). Ball milling mechanochemical treatment can rapidly and thoroughly mineralize organic contaminants and generate amorphous or graphitic carbon. In addition, ball milling mechanochemistry can enhance the immobilization of ECs by changing the physicochemical properties of soil minerals and metal oxides, making ECs difficult to dissolve.

For soil remediation, thermal desorption is a commercially established physical separation procedure. Rotating heaters in thermal desorption facilities are supplied with tainted soil on a continual basis. The volatile pollutants are removed from the soil as it moves through the rotating heater (Lee et al., 2021). It is necessary to run the process anaerobically in order to prevent the volatilized gases from igniting in the desorption unit. Off-gases can be condensed and collected, oxidized using a catalyst, or destroyed in a secondary combustion chamber. This all depends on the nature of the off-gases and how they are used. Regardless of whether the off-gases are condensed or destroyed, the emissions to air must be cleaned to eliminate particulates and acid gases. To reduce the danger of combustion, the majority of thermal desorption units are heated indirectly (Lee et al., 2021). The operation's temperature will be determined by the pollutants' type. Pyrolyzing of organic pollutants into reduced molecular weight molecules is feasible at high temperatures. However, thermally desorbed soils containing non-volatile ECs caused plugging, affecting soil pH, organic matter, heavy metal biotoxicity, and other physical, chemical, and biological characteristics. Therefore, thermally desorbed contaminated soil can negatively affect various soil properties and ecological functions.

5.3. Advantages and disadvantages of in situ and ex situ

Table 1 summarizes the benefits and drawbacks of in situ and ex situ technologies. The fundamental benefit of in situ treatment is that it enables soil to be treated without having to be dug and transported, thus saving money and reducing environmental impact (Shi et al., 2019). In situ treatment may also help to preserve some soil characteristics, including such organic matter, structure, and biodiversity, which may be hard to recover after more intrusive treatments. In situ treatment, on the other hand, may necessitate longer time periods due to changing environmental conditions at the site and in the soils (Aggelopoulos and Tsakiroglou, 2021). The consistency of therapy is likewise less assured, and its effectiveness is more difficult to evaluate. Ex situ methods need the excavation of the soil. Excavated soils may be treated on-site (and replaced with cleaned soil) or transported to another location for treatment. While ex situ methods are more controlled and take less time, they incur extra costs for excavation and shipment. Because ex situ treatments destroy soil structure and organic content, their restoration may be more difficult, expensive, and time-consuming.

Recent studies have highlighted the growing preference for in situ remediation due to its cost-effectiveness and minimal environmental disruption. For instance, Kwak et al. (2023) discuss an innovative bacterial delivery method for bioaugmentation, which improves remediation efficacy by optimizing ionic strength and pore-water velocity. Similarly, Feng et al. (2023) report on a novel multi-branch horizontal well technology that enhances in situ treatment efficiency for contaminated soil and groundwater, reducing exposure risks and treatment costs. In situ techniques like bioremediation and phytoremediation are also gaining traction. A study by Gong et al. (2018) provides a comprehensive overview of various field-scale applications of these methods, highlighting their potential for treating heavy metals and metalloids. These approaches leverage natural biological processes, thus preserving soil structure and biodiversity. Ex situ methods, while more controlled, involve higher costs due to excavation and transportation of contaminated soil. Paniagua-López et al. (2021) evaluated three ex situ techniques—biopiles, landfarming, and composting—in long-term contaminated soils. They found biopiles to be the most effective, significantly reducing soil toxicity and improving soil properties. This method is advantageous for managing soil contaminated with multiple pollutants, offering flexibility and control over treatment conditions. A notable advancement in ex situ methods is the use of soil washing, as discussed by Elgh-Dalgren et al. (2009). This technique effectively removes polycyclic aromatic hydrocarbons (PAHs) and arsenic from excavated soil, making it suitable for sites requiring immediate redevelopment. Despite the higher costs, the efficiency and control offered by ex situ treatments make them preferable for certain contaminants and site conditions. In situ methods are generally preferred for their lower environmental impact and cost. However, they may require longer treatment times and their effectiveness can be influenced by site-specific conditions. Ex situ methods, though costlier, provide a more controlled environment for remediation and are effective for complex contamination scenarios. The choice between in situ and ex situ methods often depends on the specific contaminants, site conditions, and available resources.

Table 1
The advantages and disadvantages of in situ and ex situ technologies.

	In situ	Ex situ
Advantages	Excavation costs avoided	Remediation processes are quicker
	Soil structure and biodiversity rehabilitates more quickly	Easier to control
		Easier to monitor
Disadvantages	Longer time to reach remediation objective	Costs of excavation
	More difficult to control remediation processes	Costs of transportation or manipulation on-site
	More difficult to monitor	Significant habitat disruption
		Soil structure and biodiversity harder to re-establish

6. Implication and future prospect

6.1. Implication from the review

The extensive review of soil pollution sources, migration pathways, specific impacts, and countermeasures against soil pollution underscores the multifaceted challenges posed by ECs in soil environments. These challenges highlight the need for comprehensive strategies that not only address the immediate threats of ECs but also consider long-term sustainability and soil health. The diversity of EC sources and their complex interactions within soil environments call for integrated management strategies. These strategies should combine in situ and ex situ remediation technologies based on site-specific conditions, pollutant characteristics, and socio-economic considerations. Incorporating preventive measures, such as reducing the use of non-biodegradable plastics in agriculture and improving e-waste recycling, is crucial alongside remediation efforts. Improved monitoring of EC concentrations and their distribution in soils is essential for assessing risks and tailoring remediation efforts. Regulatory frameworks need to be updated to include emerging contaminants not currently covered, ensuring that guidelines are based on the latest scientific knowledge about EC risks to human health and ecosystems. Continued research and development into more effective, efficient, and environmentally friendly remediation technologies are vital. This includes advancing bioremediation techniques that leverage the natural degradation capabilities of microorganisms and plants, exploring innovative materials for nano remediation, and developing more sustainable approaches to soil management. Remediation efforts should not only aim to remove contaminants but also restore soil health to support ecosystem services, including biodiversity, nutrient cycling, and water regulation. Strategies for soil health recovery should be an integral part of remediation projects, involving the restoration of soil structure, organic matter, and microbial communities.

6.2. Prospect of future

At present, more and more attention has been paid to the environmental pollution status of ECs. However, most of the current research focuses on water, and there are few studies on soil ECs. Importantly, the potential risk of ECs to ecosystems is an important issue especially before large-scale applications in agriculture and environmental governance. Therefore, it is of great importance to summarize the current status of ECs pollution in soil and summarize the management methods of ECs. This review concludes that the future directions for preventing soil contamination by ECs and remediation of soil contaminated by ECs are:

(1) Develop uniform threshold values of ECs contamination in soils and harmonize standard operating procedures for laboratory methodologies for ECs analysis in soils. Encourage the inclusion of soil ECs pollution data and information in national and global soil information systems, as well as the inclusion of soil pollution data and information in regular soil surveys. Encourage the creation of a worldwide information and monitoring system for soil

contamination. Establish and improve technological platforms for ecological governance and agricultural security.

(2) Increase investment in targeted research and innovation in ECs: detection, environmental fate, risk assessment and remediation. At the national, regional, and global levels, develop and enhance inventories and monitoring of point source and diffuse soil ECs pollution. Strengthen the identification, assessment, and monitoring of unknown damage and disease caused by soil ECs contamination, and support actions to prevent ECs contamination.

(3) Implement policies aiming at long-term agricultural soil management, with a particular emphasis on minimizing pesticide use and regulating irrigation water and ECs residues. Develop and incorporate soil pollution targets and indicators into national reporting procedures that are important to the fulfillment of the SDGs. Promote natural and environmentally sound management and remediation technologies, such as plant, microorganism, and combination bioremediation research.

7. Conclusion

This review paper focuses on the status and trends of soil pollution caused by ECs. The concentrations of ECs, such as plastics, synthetic polymers, pharmaceuticals, personal care products, plasticizers, and flame retardants, are increasing, leading to their release and accumulation in the environment. EC migration in soil occurs through horizontal pathways like wind and surface runoff and vertical pathways facilitated by soil biota, plant roots, and soil cracks. Sources of soil pollution by ECs include agricultural activities, domestic pollution, electronic waste, urban wastewater irrigation, and atmospheric deposition. Understanding the migration and environmental implications of ECs in soil is crucial for assessing their risks and developing effective control methods. Effective soil pollution control for ECs should prioritize pollution prevention and simultaneously enhance coordinated control and remediation efforts, particularly in agricultural soils and groundwater. Establishing a comprehensive governance system and sound laws and regulations is essential for the prevention, suppression, and remediation of soil pollution caused by ECs. These measures will better facilitate the control and restoration of soil pollution, ensuring safe use and promoting regional green development.

CRediT authorship contribution statement

Lu Liu: Writing – original draft, Data curation, Conceptualization. **Chunrui Liu:** Writing – review & editing, Writing – original draft. **RunZe Fu:** Writing – original draft. **Fandi Nie:** Writing – original draft. **Wei Zuo:** Writing – review & editing. **Yu Tian:** Writing – review & editing. **Jun Zhang:** Writing – review & editing, Funding acquisition, Formal analysis.

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.

Data availability

The authors do not have permission to share data.

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Appendix A. Supplementary data

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