TYPE Review
PUBLISHED 15 August 2025
DOI 10.3389/fenvs.2025.1634662

OPEN ACCESS

EDITED BY
Chamindra L. Vithana,
Southern Cross University, Australia

REVIEWED BY
Juan Sádaba,
University of the Basque Country, Spain
Fan Zhang,
Nanjing Forestry University, China
Naji Akbar,
Ajman University, United Arab Emirates

*CORRESPONDENCE
Elio Padoan,
elio.padoan@unito.it

†These authors share first authorship

RECEIVED 24 May 2025
ACCEPTED 30 July 2025
PUBLISHED 15 August 2025

CITATION
Celletti S, Poreba L, Wawer R, Padoan E, Comis S, Bartosiewicz B and Schiavon M (2025) The potential of nature-based solutions for urban soils: focus on green infrastructure and bioremediation.
Front. Environ. Sci. 13:1634662.
doi: 10.3389/fenvs.2025.1634662

The potential of nature-based solutions for urban soils: focus on green infrastructure and bioremediation

Silvia Celletti[1†], Ludwika Poreba[2†], Rafal Wawer[2], Elio Padoan[1*], Silvia Comis[1], Beata Bartosiewicz[2] and Michela Schiavon[1]

[1]Department of Agricultural, Forest and Food Sciences, University of Turin, Turin, Italy, [2]Department of Soil Science and Environmental Analyses, Institute of Soil Science and Plant Cultivation, Pulawy, Poland

Nature-based Solutions (NbS) offer a promising, sustainable framework for addressing urban environmental challenges by harnessing the intrinsic functions of natural ecosystems. Defined as economically viable strategies inspired by nature, NbS aim to protect, manage, and restore ecosystems for mutual benefits to both people and nature. In urban and peri-urban contexts, NbS, such as green infrastructure and bioremediation, provide effective approaches to mitigating climate change, enhancing soil, air and water quality, reducing urban heat, and restoring ecological balance. Green infrastructure, comprising parks, urban forests, green roofs, and wetlands, has been shown to improve soil quality by enhancing organic carbon, nitrogen, and phosphorus accumulation, as well as fostering biodiversity. Complementarily, bioremediation strategies, including microbial remediation and phytoremediation, have proven effective in decontaminating soils laden with heavy metals, petroleum hydrocarbons, microplastics, and other pollutants, while promoting soil fertility and ecosystem services. Despite their demonstrated benefits, the efficacy of NbS is influenced by environmental factors such as soil pH, temperature, oxygen availability, and pollutant diversity. In addition, while NbS continue to evolve and their integration into urban planning represents a vital step toward creating resilient, healthy, and sustainable cities, their application in urban environments remains fragmented. Therefore, further research is required to optimize NbS interventions, scale up their implementation, and evaluate long-term impacts under urban conditions characterized by anthropogenic stressors. This review examines green infrastructure and bioremediation strategies, highlighting key case studies and evaluating their effects on soil quality and overall remediation outcomes.

KEYWORDS

ecosystem services, inorganic contaminants, microplastics, organic pollutants, soil remediation, sustainability

1 Introduction

The concept and theme of Nature-based solutions (NbS) was already used at the beginning of the 21st century, and built upon earlier theories and fields of studies such as the applied ecology and the restoration of the biodiversity, which has been developed at different levels with concepts as the ecological engineering, green-blue infrastructures, ecosystem assessments and ecosystem services or the use of the natural capital, functioning as an "umbrella" concept (Nesshöver et al., 2016; Somarakis et al., 2019).

Nature-based Solutions have been only adopted later by different global institutions and their understanding and applications continue to evolve as this field develops (Bona et al., 2023). Notably, the practical application of NbS remains imprecise and fragmented, largely due to ambiguities arising from the integration of diverse scientific disciplines and the absence of well-defined standards within the NbS framework. As defined by the International Union for Conservation of Nature (IUCN) and the European Commission, NbS are strategies inspired and supported by nature, economically viable, that address societal challenges by protecting, sustainably managing, and restoring natural or modified ecosystems to benefit both people and nature (Cohen-Shacham et al., 2016).

Within urban and peri-urban areas, NbS provide sustainable approaches to address environmental challenges. By integrating nature into urban planning, they can mitigate climate change effects, improve soil, air and water quality, reduce urban heat, and restore ecological balance, which is "*a state of dynamic equilibrium within a community of organisms in which genetic, species and ecosystem diversity remains relatively stable*" (Verma, 2018). Currently, nearly half of the global population resides in cities, and projections suggest that by 2050, this proportion will increase to approximately two-thirds (United Nations Department of Economic and Social Affairs (UN DESA), (2019)). This rapid expansion of urban populations needs continuous growth in built-up areas. However, unregulated or poorly planned urban development can result in unsustainable landscapes, contributing to environmental degradation, including air and water pollution, soil contamination, and the fragmentation of natural habitats (Tayefi Nasrabadi, 2022). Such ecological disruptions have already led, and will likely continue to lead, to the loss of soil biodiversity, reduced ecosystem services, and adverse public health outcomes, highlighting the need for sustainable urban planning strategies. Without an ecological consciousness among administrators, industry and citizens on the side effects of the growth of populations, NbS alone cannot be able to help in the improvement of environmental health.

In cities, NbS particularly relevant are green infrastructures. One of the most widely cited definitions, provided by the European Commission (2013), describes green infrastructure as "a strategically planned network of natural and semi-natural areas, designed and managed to deliver a wide range of ecosystem services" (Hansen and Pauleit, 2014; Dorst et al., 2019). Key elements of green infrastructures include parks, urban forests, gardens, allotments, floral meadows, urban farming, green roofs and walls, and wetland restoration.

Complete catalogues of the NbS studied and useful in urban areas have been recently compiled to help planners depending on the factors relevant and on the governance objectives, as URBAN GreenUP (2018), or to evidence the main NbS implemented in the urban environment and their effects, as in Bona et al. (2023).

The introduction of green infrastructures in urban areas positively impacts multiple environmental aspects. They could prevent urban flooding and support water management (Bonoli et al., 2013; Cristiano et al., 2020; Valois et al., 2023), reduce the urban heat island (UHI) effect, thus contributing to climate change adaptation (Bouzouidja et al., 2021; Bochenek et al., 2022; la Cecilia et al., 2020), and promote a more sustainable urban environment (Russo et al., 2017; Silvestrini et al., 2021; Mihalakakou et al., 2023). The UHI effect is indeed an increasingly critical local climate issue associated with global urbanization, with significant implication on human health and energy consumption (Wu et al., 2024). It refers to the phenomenon where the urban areas exhibit markedly higher temperatures than their surrounding rural counterparts. This temperature disparity results from key characteristics of densely populated urban environments, characterized by elevated anthropogenic heat emissions, greater absorption of solar radiation, lower albedo, reduced heat capacity, and diminished turbulent heat exchange.

Additionally, green infrastructures could enhance urban biodiversity (Donati et al., 2022; Bretzel et al., 2024), reduce environmental contaminants (Calfapietra, 2020; Alpaidze and Pace, 2021), or soil sealing (Aimar, 2023), sequester carbon (C) (Alpaidze and Pace, 2021), support local food production (Walters and Midden, 2018; Zareba et al., 2021), and improve the quality of life for residents (Orta-Ortiz and Geneletti, 2023).

Another NbS that can be applied in urban areas is soil bioremediation. Over the past 6 decades, industrialization and technological advancements have led to the creation of large areas of abandoned, underused, and potentially contaminated land in cities worldwide. These areas, commonly known as brownfield sites (Hursthouse and Leitao, 2016; Megharaj and Naidu, 2017), require careful soil contamination monitoring and remediation. Unfortunately, many of these areas are also found in countries where environmental policies and regulations are not strict and legislation did not require (or encourage) the landowners to restore the environment. The remediation process is crucial not only for urban regeneration, but also for preserving soil quality, biodiversity, and the essential ecosystem services these sites provide (Kabisch et al., 2016; Drenning et al., 2020; Vannucchi et al., 2021).

Soil pollutants can be degraded or removed using various methods, many of which can be specifically adapted for the reclamation of brownfield sites (Hou et al., 2023). The methods range from physical (*e.g.*, heat treatment, vitrification, electro-remediation, landfilling, storage), chemical (*e.g.*, oxidation, chemical fixation, precipitation, leaching, ion exchange), biological (*e.g.*, bioremediation), and integrated approaches (Aparicio et al., 2022; Yaashikaa et al., 2022).

Bioremediation is generally more cost-effective and eco-friendlier than traditional physical and chemical methods and can enhance soil fertility along with several ecosystem services in urban areas, such as improved urban hydrology, heat mitigation, noise reduction, biodiversity, carbon dioxide (CO_2) sequestration, bioenergy production, landscape aesthetics, social cohesion, and public health (Landa-Acuña et al., 2020; Guidi Nissim et al., 2023). Bioremediation relies on non-toxic inputs and produces safe by-products throughout the degradation process. This method uses either naturally occurring microorganisms (microbial remediation) or plants (phytoremediation) to remove pollutants from soil effectively. Microorganisms such as bacteria, fungi, microalgae, and actinomycetes are particularly effective for reclaiming soils contaminated with organic pollutants, as they can use chemical contaminants as energy sources, metabolizing targeted compounds through redox reactions to support growth and energy conversion (Sharma et al., 2022). In this way, hazardous pollutants can be converted into less toxic forms. For instance, hydrocarbons can be oxidized by microorganisms, with oxygen (O_2) being reduced to form CO_2 and water (Ali et al., 2022). As pollutants are degraded, microbial populations gradually decline as the availability of harmful substances diminishes. Microorganisms can also be used to immobilize metals in soil, causing them to precipitate, adsorb, and become fixed in the soil matrix, thereby reducing their bioavailability (Cornu et al., 2017; Derakhshan Nejad et al., 2018).

Plants with high capacity to tolerate and accumulate contaminants can be used in phytoremediation technologies for the removal of both inorganic and organic pollutants from brownfield sites (Raskin et al., 1997; Petrova et al., 2022; Fernández-Braña et al., 2023). The most suitable plants for phytoremediation are those that efficiently translocate metals to their aerial parts (phytoextraction) or those that can effectively degrade organic contaminants and their by-products through phytodegradation. Combined remedial approaches using both microbes and plants have also shown promising results (Ojuederie and Babalola, 2017). Indeed, the mutualistic relationship between plant species and microbes can enhance the bioavailability of several pollutants for more effective degradation and/or removal. Additionally, root exudates stimulate the growth of rhizosphere microorganisms, further facilitating the breakdown of organic contaminants in a process known as microbial-assisted phytoremediation (Ojuederie and Babalola, 2017). Recent advances in bioremediation have led to the development of innovative strategies that enhance the efficiency, specificity, and resilience of both microbial- and plant-based approaches. Among these, the use of genetically engineered plants and microorganisms enables faster and more targeted degradation or immobilization of complex contaminants (Rafeeq et al., 2023). Similarly, the application of selected microbial consortia and fungal species (*e.g.*, white-rot fungi) has shown great promise in breaking down persistent organic pollutants through enzymatic activity (Qattan, 2025). The incorporation of functionalized amendments such as biochar, compost, and nanomaterials can further improve pollutant bioavailability, stimulate beneficial microbial activity, and stabilize soil properties (Dong et al., 2024). Moreover, hybrid systems that combine plants, microbes, and tailored soil amendments are gaining momentum as synergistic tools for sustainable soil remediation, particularly in complex and heavily contaminated environments (Xiang et al., 2022). These cutting-edge technologies not only accelerate the detoxification process but also promote long-term soil health and multifunctionality, aligning with broader goals of circular economy and green infrastructure development.

2 Green infrastructures and their impact on urban soil quality

Soils play a critical role in the urban ecosystem by supporting essential biogeochemical processes and providing key ecosystem services. However, urbanization poses a significant threat to soil integrity and health, often causing irreversible degradation.

Unsealed urban soils contribute to various ecological functions, including water filtration, C sequestration, and biodiversity support, which are essential for maintaining environmental quality and human wellbeing in cities. Therefore, the preservation and enhancement of soil quality in urban areas are crucial for sustaining ecosystem services and mitigating the adverse effects of urban expansion. Unsealed urban soils are predominantly found in green infrastructure spaces, such as urban parks and forests, allotments and under lawns and floral meadows.

The type and quality of green infrastructure significantly influences soil physical, chemical and biological properties. Despite this, research on green infrastructure has largely focused on broader environmental benefits, such as climate change mitigation, biodiversity conservation, and C sequestration, often overlooking the direct interactions between green infrastructure and soil health and quality (Xie and Bulkeley, 2020; Debele et al., 2023; Moazzem et al., 2024).

Moreover, there is a clear lack of robust results on the effects of NbS on soil quality. Looking into literature, this gap is also evident in systematic reviews on green infrastructures recently published. As an example, in the review of Mosisa et al. (2025) soil is evocated just once in all the paper, while Goodwin et al. (2024) showed that typically, NbS just look at soil erosion, while soil quality was not mentioned. Moreover, Guo et al. (2025), in a bibliometric analysis on NbS, did not mention any work studying NbS effect on soil.

The only indicator well studied in literature is soil organic carbon (SOC), which is a key indicator of soil health, and green infrastructure can enhance its accumulation. A study in Berlin (Richter et al., 2020) compared the SOC content of sealed and green areas, revealing significant differences. In the topsoil layer (0–20 cm), sealed areas stored 3.99–7.94 kg C m^{-2}, while green areas contained 2.16–9.52 kg C m^{-2}. In deeper soil layers (>20 cm), sealed areas held 10.40–58.94 kg C m^{-2}, whereas green areas stored 19.21–119.62 kg C m^{-2}. These findings indicate that green spaces can sequester up to twice as much SOC as sealed surfaces, highlighting their role in soil health and C storage (Richter et al., 2020).

Different factors such as time, the type of green infrastructure and vegetation, and green space management play a crucial role in determining soil organic carbon content in urban areas, as confirmed by multiple studies. Research conducted in Finland by Setälä et al. (2016) and Lindén et al. (2020) highlights the effect of time in urban parks. In particular, Setälä and colleagues analyzed 41 parks in Helsinki and Lahti, categorizing them into three age groups: old (more than 100 years old), intermediate (around 50 years old) and young parks 5–15 years old). Their findings revealed that younger parks had lower SOC content. Similarly, Lindén et al. (2020) studied 30 parks in Helsinki and confirmed that park age significantly influences soil C content. The impact of time on SOC content is also evident on green roofs, which are key components of green infrastructure and help regulate urban temperature, reduce stormwater runoff, improve air quality, and enhance biodiversity, while also contributing to climate resilience and the overall ecological performance of cities (Mihalakakou et al., 2023; Todeschini and Fett-Neto, 2025). A study in Hanover (Germany) examined 10 green roofs categorized as old (established between 1990 and 1994) or young (established between 1998 and 1999). Results showed that older roofs had the highest C content in their soil substrate, with an average SOC of 8.4%, compared to 3.4% on younger roofs (Schrader and Böning, 2006).

With respect to the type of green infrastructure and vegetation impacting SOC in urban areas, a study conducted in Leicester by Edmondson et al. (2014) examined variations in SOC across different green spaces. The authors analyzed areas covered with herbaceous vegetation (primarily lawns), shrubs and low trees, trees over 5 m, as well as backyard gardens with lawns, meadows, vegetable beds, or a combination of shrubs and lawns. Their findings revealed that backyard gardens with shrubs and trees had the highest SOC concentration (75.2 mg g^{-1}), whereas areas dominated by herbaceous vegetation exhibited the lowest levels. On the city scale, backyard gardens with shrubs and trees, despite covering only 11% of the total green space, contributed 16% of the total SOC stock. In contrast, areas dominated by herbaceous vegetation, which make up 46% of the green space, accounted for 40% of the total C soil content. Similarly, Kortleve et al. (2023) conducted a study in The Hague (the Netherlands), comparing SOC content across different urban green spaces, including parks, street trees, urban forests, and shrub-covered areas in both the city center and suburbs. While high SOC contents were recorded in both areas, soil C density was significantly higher in the city center (12.5 ± 6.84 kg C m^{-2}) than in the suburbs (8.45 ± 5.15 kg C m^{-2}). Notably, shrub-covered areas stored approximately 25% of the total soil C despite occupying only 13% of the total green space. Lindén et al. (2020) compared soil samples from urban parks in Helsinki and found that shrubby areas consistently stored more C than those dominated by herbaceous vegetation and lawns. This trend persisted across various park types, including regularly maintained parks with fertilization and irrigation, standard parks, and buffer strips between developed and natural areas. Another study in Helsinki compared SOC content in different urban green spaces, such as orchards, parks, urban forests, and street trees. Results showed that urban forest soils had the highest SOC content, while soils under street trees had the lowest (Karvinen et al., 2024). In the same city, tree species composition was also reported to influence SOC accumulation, which was higher in parks with evergreen trees than in parks with deciduous trees, thus reinforcing the role of vegetation type in C storage (Setälä et al., 2016).

Green space management also has a measurable impact on SOC content. Ivashchenko et al. (2023) studied a park in Grozny, analyzing soil samples from three areas: an irrigated lawn, a non-irrigated lawn, and a non-irrigated lawn with trees. Their findings showed that soil from the irrigated lawn had 35% higher SOC content than the non-irrigated areas, demonstrating the role of irrigation in enhancing soil C accumulation.

Soil health in urban areas is also influenced by nitrogen (N) and phosphorus (P) levels. Green roofs serve as a prime example of green infrastructure that facilitate N accumulation in soil, as demonstrated by studies conducted in various countries. Schrader and Böning (2006) investigated this phenomenon by analyzing ten green roofs in Hanover, constructed between 1990–1994 and 1998–1999. Their findings revealed that older roofs had a higher total N content in the soil substrate, with an average of 0.34% compared to 0.24% on newer roofs.

Like organic C, N and P accumulate in urban soils depending on time. In this respect, Bouzouidja et al. (2018) studied a green roof in Tomblaine (France), by analyzing the initial soil substrate and reassessing it after 4 years. These findings showed a significant increase in N content, rising from 0.13% to 0.54% ± 0.2%. Similar studies were conducted in Sweden, where 15 green roofs, built between 1994 and 2014 across three cities, were considered. These analyses confirmed that older roofs had higher N content in the soil substrate, with an estimated N accumulation rate of 3.8 ± 1.0 g N m^{-2} $year^{-1}$ (Mitchell et al., 2021). A similar relationship exists between time and soil P content. Setälä et al. (2017) found that total soil P content was lowest in young and intermediate parks and highest in the oldest parks, reinforcing the long-term accumulation of essential nutrients in urban soils. In another study, Setälä et al. (2016) additionally measured N contents across three different vegetation types, including lawns, deciduous trees, and evergreen trees, in various parks. They found that soil N concentrations were lowest in lawns and highest in areas with evergreen trees (Setälä et al., 2016). A similar study in Helsinki (Finland) compared organic nitrogen (N_{org}) content in soils from orchards, parks, urban forests, and areas under street trees. The highest N_{org} content was recorded in the urban forest soil (0.29% ± 0.02%), while the lowest was found under street trees (0.13% ± 0.05%) (Karvinen et al., 2024). When comparing lawns to urban meadows, soil N content was generally higher in lawns (Trémeau et al., 2024).

Beyond the effects on soil chemical properties, green infrastructure plays a significant role in enhancing soil biodiversity. A three-year study in Cambridge (United Kingdom) compared species diversity between lawns and floral meadows, revealing that species diversity was 3.6 times higher in the meadows. Additionally, total invertebrate biomass was 25 times greater in the meadows, while the species diversity of soil nematodes remained similar between the two environments (Marshall et al., 2023). Soil biodiversity was also explored in a study by Schrader and Böning (2006), who analyzed microbiological activity on green roofs of varying ages in Hanover (Germany). Their assessment of dehydrogenase activity revealed a dramatic range, from 451 μg total productivity factor (TPF) $g^{-1}d^{-1}$ on young roofs to 4,530 μg TPF $g^{-1}d^{-1}$ on older roofs. The average dehydrogenase activity was 1892 μg TPF $g^{-1}d^{-1}$ in young roofs and 2,874 μg TPF $g^{-1}d^{-1}$ in older roofs. In addition, the quantity of collembolans was measured, showing that older roofs supported a higher population of this species, with an average of 57,000 individuals m^{-2} on older roofs compared to 55,000 individuals m^{-2} on younger roofs.

Based on the findings from these studies, green infrastructure significantly influences urban soil health and indicators. However, the limited research on this topic underscores the need for further exploration into the relationship between green spaces and urban soil. A deeper understanding of this connection is crucial for developing effective strategies to enhance urban soil quality, which, in turn, can improve the overall quality of life for urban residents.

3 Bioremediation of urban soils

Urban green spaces represent a potential exposure pathway to inorganic and organic contaminants for both humans and other organisms. Soil pollution in these areas can originate from point and diffuse sources. The largest influx of contaminants into the soil from urban and commercial activities is through transport (*e.g.*, vehicle exhaust particles, tire and brake wear particles, weathered street surface particles, brake lining wear particles), industrial emissions, (*e.g.*, power plants, coal combustion, metallurgical industry, auto repair shop, chemical plant, etc.), domestic emission, weathering of building and pavement surface, atmospheric deposition, and waste (Wei and Yang, 2010). As an example, lead (Pb), which is the most concerning trace element in urban areas due to its high toxicity, predominantly originates from the use of Pb-based paints, vehicle emissions and deposition of airborne particles from Pb-emitting industries. In Europe, Pb and its compounds in paints and fuels are controlled by the European Commission under the EU REACH law (Registration, Evaluation, Authorization and Restriction of Chemicals) (European Council, 2006), which has been in effect since 2007. However, despite national and international regulations, old houses and even newly produced paints still have high concentrations of Pb and are widely commercialized mainly in low- and middle-income countries. Therefore, the high persistence and bioaccumulation of Pb in urban soils remain a significant concern (WHO, 2021).

In addition to Pb, other trace elements are also commonly found in urban soils. Among all, the most widespread are zinc (Zn), copper (Cu) and cadmium (Cd), which are primarily associated with atmospheric deposition from traffic emissions (Biasioli and Ajmone-Marsan, 2007; Binner et al., 2023), while mercury (Hg) can originate from a variety of sources, including coal and oil combustion in power stations, atmospheric deposition from industrial plant and traffic emissions, incineration of municipal solid waste. Additionally, urban garden soils may contain Hg from the use of agrochemicals applied for plant protection (Kelepertzis and Argyraki, 2015).

Organic contaminants in urban soils, such as phthalate esters (PAEs) and polycyclic aromatic hydrocarbons (PAHs) might also cause serious threats to human health and the environment. Phthalate esters raise significant concerns given their high volatility and well-documented role as endocrine disruptors (Li et al., 2023a; Li et al., 2023b). Widely used as plasticizers and additives in a range of industrial products, pharmaceuticals, personal care products, and food containers, PAEs are released during their manufacture, utilization, and waste disposal (He et al., 2015). Polycyclic aromatic hydrocarbons primarily derive from vehicle exhaust, coal combustion for cooking and heating, wood and straw burning, and high-temperature industries such as melting, cooking, and metal processing (Peng et al., 2016; Sumathi and Manian, 2023). Soil organic matter has a strong capacity to adsorb PAHs, allowing these compounds to persist in soil for decades.

Other organic contaminants in urban areas include polychlorinated biphenyls (PCBs), chlorinated solvents, volatile organic compounds (VOCs), heavy petroleum hydrocarbons, dioxins and furans, and several pesticides used in home gardens (Gao et al., 2022). These compounds are persistent, non-degradable, highly stabilized, strongly hydrophobic, and accumulate over time (Zheng et al., 2022). Their presence in soil can severely impact humans and other organisms, as they are toxic, mutagenic, teratogenic, and carcinogenic (Gao et al., 2022).

A different type of urban soil contaminant is microplastics (MPs). These polymeric particles, originating from various sources, may pose direct risks to the environment and human health, either due to the particles themselves (Kannan and Vimalkumar, 2021), or through the chemical additives they carry (Hahladakis and Iacovidou, 2018). Microplastics have been reported to cause significant changes in ecosystem properties and processes, plant growth, and soil communities, raising serious concerns about their long-term effects on the environment and human health (Leitão et al., 2023).

Bioremediation approaches can be applied either *in situ* or *ex situ* depending on the nature of contaminant, its concentration in soil, soil properties, site conditions, and logistical considerations (Megharaj and Naidu, 2017; Kuppan et al., 2024).

Figure 1 illustrates NbS-based bioremediation approaches in urban areas. *In situ* treatments are often more appealing and sustainable as they eliminate the need for excavating and transporting contaminated soils. The most common *in situ* treatments include natural attenuation, biostimulation, bioventing, biosparging, bioaugmentation, and phytoremediation (Gao et al., 2022; Kuppan et al., 2024). In contrast, *ex situ* approaches require the excavation and removal of contaminated soil for treatment, either on-site or at an off-site facility; these include land farming, composting, biopiles, and bioslurry treatments (Megharaj and Naidu, 2017; Sharma et al., 2022). Among the described bioremediation approaches, natural attenuation, biostimulation, composting, and phytoremediation are considered as NbS (Figure 1). It is noteworthy that NbS definition is not always clear-cut in the context of bioremediation. Our classification is based on the degree to which each technique aligns with core NbS principles, namely, working with natural processes and minimizing artificial or engineered interventions. The methods reported in Figure 1 prioritize the use of natural processes, intervening minimally, to enhance or support ecosystem functions. In contrast, other bioremediation approaches rely more on engineered processes, mechanical input or the introduction of non-native microorganisms. Therefore, they require active human control to achieve remediation, rather than directly harnessing natural ecosystems in their unaltered state (Kabisch et al., 2016; Song et al., 2019; Snep et al., 2020), and, for this reason, they fall outside the NbS scope.

Natural attenuation is commonly used in areas with low to moderate contamination levels and where risks to human health and the environment are minimal (Mulligan and Yong, 2004; Ayilara et al., 2023). This process relies entirely on the natural ability of soil and native microbial strains to degrade, transform, or immobilize pollutants without significant active human intervention (Mulligan and Yong, 2004). Regular monitoring is essential to ensure that the natural processes effectively manage the contamination without adverse effects.

On the other hand, biostimulation promotes the activity of soil native microorganisms and natural metabolic pathways by supplementing nutrients such as P, N, O_2 or C, or electron donors (*e.g.*, compost or other organic materials), to stimulate biodegradation (Ren et al., 2018). While it involves some human intervention, which makes its classification as NbS debatable, it amplifies natural processes and is often aligned with NbS. Thus, given this partial alignment, we included biostimulation as an NbS (Figure 1).

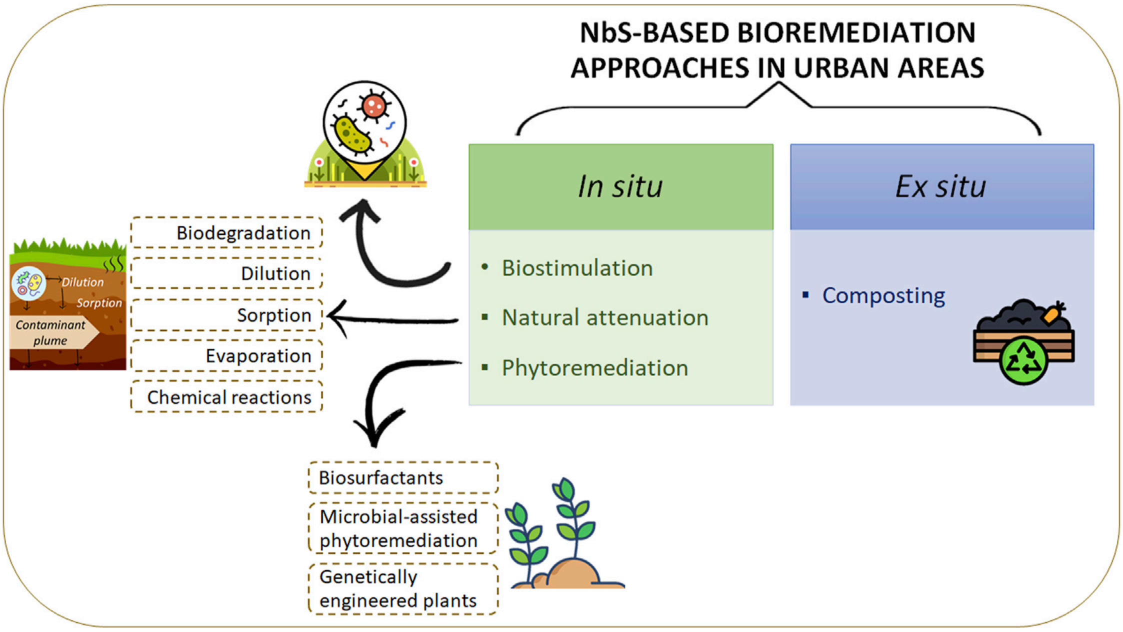

FIGURE 1
Nature-based Solutions (NbS)-based bioremediation approaches in urban areas for the removal of urban pollutants such as inorganic contaminants, organic pollutants, and microplastics. Key processes include microbial degradation, sorption, and abiotic reactions. The figure was created using Canva (www.canva.com).

Bioventing and biosparging can be applied to inject O_2 in soil to enhance the microbial activity aimed at degrading organic contaminants. However, while these methods rely on biological processes, they still require active technological intervention, which makes them less aligned with the principles of NbS.

Phytoremediation is a largely studied NbS also in urban environments, for the reclamation of soils with low levels of contamination. It must be noted that often the use of this approach for the complete remediation of urban sites could not be effective, as the estimated time for remediation exceeds the overall rate of urban development. Introducing a landscape framework within urban developments can create spaces that not only support phytoremediation processes but also enhance the spatial organization and aesthetic value of public areas. This approach transforms urban landscapes into productive spaces that serve both environmental restoration and social functions (*e.g.*, community wellbeing). This concept well aligns with the principles of NbS.

Plant species preferentially employed in brownfield sites are trees (Pulford and Watson, 2003; Capuana, 2020; Solomun et al., 2024), but there are also examples where metal hyperaccumulators have been used (Wilschut et al., 2013) or urban grassland plants (Nikolić and Stevović, 2015; Stančić et al., 2022).

Finally, composting produces nutrient-rich OM, which can be applied to soils to improve their quality, structure, and biological fertility, improving soil biodiversity. This approach finds application also in urban environments as it contributes to reducing the volume of waste that needs to be transported to landfills or incinerators, limiting municipal waste management costs and increasing the efficiency of urban waste systems. Compost can additionally support urban farming. Indeed, community composting initiatives can provide residents with nutrient-rich soil for gardening and enhancing local green spaces.

3.1 Bioremediation of inorganic contaminants

Bioremediation techniques for soils contaminated with inorganic contaminants, also defined as Potentially Toxic Elements (PTEs), have been studied for decades (Pulford and Watson, 2003; Mench et al., 2010; Wang et al., 2021). The primary focus has been on elucidating the mechanisms of metal uptake by plants (Shen et al., 1997; Dickinson et al., 2009; Deng et al., 2018), identifying plants capable of accumulating high levels of PTEs (Pollard et al., 2002; van der Ent et al., 2013; Gervais-Bergeron et al., 2023), and assessing the feasibility of remediation (Chaney et al., 2007; Song et al., 2019; Ali et al., 2022). For successful outcomes, a plant should ideally be able to explore the entire contaminated soil, but the rooting zone (typically 30–50 cm depth) is, in most cases, shallower than the depth of the contamination. As a result, only surface-level contamination can be treated *in situ*. The biomass generated during remediation could serve as a valuable resource to reduce costs, either by producing energy or being utilized in other industries. However, in many countries, biomass is classified as waste and must be landfilled, while in other countries, the lack of legislation hinders full-scale implementation (Mench et al., 2010; Ambrosini et al., 2017).

Figure 2A provides a summary of the main sources of inorganic contaminants, their effects on soil health, bioremediation techniques, and the associated ecosystem benefits.

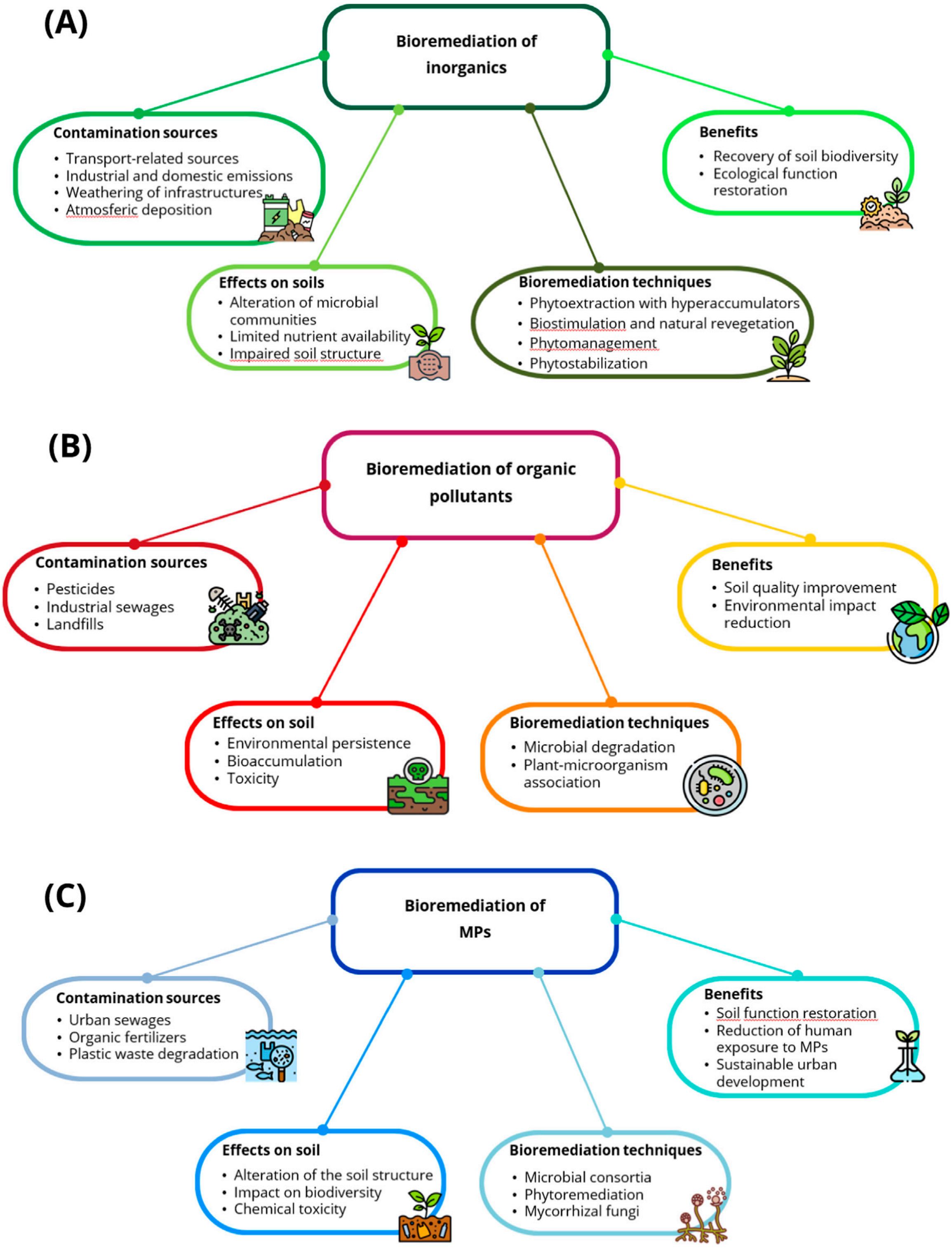

FIGURE 2
The diagrams provide a visual summary of the strategies used for bioremediation of inorganic contaminants **(A)**, organic pollutants **(B)**, and microplastics (MPs) **(C)** in soil. These diagrams illustrate the main sources, effects on soil health, bioremediation techniques, and associated ecosystem benefits. The figure was created with Lucidchart (www.lucidchart.com) and Canva (www.canva.com).

Most studies focused on bioremediation have been conducted in controlled environments or in highly polluted soils, such as industrial or mining areas, where the contaminant concentrations are higher, and the timeframes required to observe the effects of the practice are longer.

However, in this section, we focus on studies conducted in urban areas that explore bioremediation, with a focus on phytoremediation techniques, especially phytostabilization, as well as those examining soil health more broadly, particularly

TABLE 1 Selected studies addressing NbS for removal of inorganic contaminants in urban soils. EF, enrichment factor; BAF, bioaccumulation factor; TC, total concentration; SRC, short rotation coppice.

Target species	Metal	Initial concentration (mg kg^{-1})	Time (years)	Accumulation	Reference
Urban grasses	Pb	159 ± 1	2	EF 3–5 BAF 0.02–0.03	Adamo et al. (2015)
Urban grasses	Zn	333 ± 41.9	2	EF 2–40 BAF 0.05–0.16	Adamo et al. (2015)
Acer, Betula, Buddleja (high concentration)	Zn	0.5–51.4[a]	30	TC = 0.03–0.55	Fernández-Braña et al. (2023)
Acer, Betula, Buddleja (low concentration)	Zn	0.31–8[a]	30	TC = 0.08–3.68	Fernández-Braña et al. (2023)
Noccaea caerulescens	Cd	2	1	200 g ha^{-1}	Jacobs et al. (2017)
Noccaea caerulescens	Zn	333	1	47 kg ha^{-1}	Jacobs et al. (2017)
SRC: *Populus*, *Salix*, *Robinia*	Zn	413	2	*Populus* (28%–1,077 g ha^{-1}) *Salix* (36%–919 g ha^{-1}), *Robinia* (26%–997 g ha^{-1}	Padoan et al. (2020)
Urban grasses	Mn, Zn, Ni, Cu, Pb, Co	Mn: 504–589 Zn: 101–111 Ni: 25–30 Cu: 30–34 Pb: 40–46 Co: 7.13–8.09 Mo: 0.38–0.40 As: 4.14–4.61 Cd: 0.38–0.44		BAF[b] Mn (0.89), Zn (2.32), Ni (0.63), Cu (0.92), Pb (0.62), Co (0.65), Mo (4.99), As (0.62), Cd (1.19)	Petrova et al. (2022)

[a]Phytoavailable concentrations, $CaCl_2$ extract.
[b]Values for mixed stands.

the transformations induced by these techniques. Consequently, we prioritized studies from urban settings where the effectiveness of these NbS has been demonstrated, either at pilot or full scale, although their number is limited compared to the vast body of research on bioremediation, highlighting the need for further investigation before wider application in the field (Gul et al., 2021). This fact could be explained by several reasons.

In urban areas, inorganic contaminants are often the result of long-term diffuse pollution, along with site-specific acute cases. Consequently, large areas with low concentrations need remediation. Although this offers an advantage for bioremediation, as chemical or physical strategies are costly and less efficient in these conditions (*e.g.*, excavation and landfilling), the primary drawback is the time required for the process to be effective (Robinson et al., 2015; Burges et al., 2020). Indeed, although plants can accumulate metals relatively quickly, within a single growing season, the time required to reduce the total contamination to acceptable levels is typically too long, limiting their practical application in the field. Furthermore, studies at field scale (or full-scale) are limited due to the challenges of conducting a comprehensive bioremediation trial within the typical timeframe of a research project or industrial planning (*i.e.*, 3–5 years). A complete bioremediation trial requires multiple stages, including site acquisition, characterization, planting, and follow-up of the experiment (Gerhardt et al., 2017). For companies, this challenge is even more pronounced, particularly in urban areas, where allotments have a monetary value that is influenced by potential future investments. The value of a remediated area is further diminished if the prospects for changing land use in the future are scarce. Another challenge, hindering the field-scale application of phytoremediation, is that, unlike chemical or physical methods, this technology does not guarantee successful outcomes due to non-controllable factors, such as extreme weather events. Specifically, effective phytoremediation requires a thorough understanding of site-specific conditions and soil chemistry in order to select the most suitable approach. This knowledge is crucial for translating laboratory findings into practical field applications, including both remediation and ongoing monitoring efforts (Shah and Daverey, 2020). Other less prominent but still significant challenges in designing and conducting field-scale studies in urban areas include the depth of contamination and the management of biomass.

Among the field studies selected for this review (Table 1), conventional phytoextraction and phytostabilization approaches were the most commonly used, with their effectiveness in reducing heavy metal pollutants such as Cd, Zn, Cu, and Pb from urban soils being widely demonstrated. Of these approaches, phytoextraction, using accumulators or fast-growing plants, garnered the most interest due to its versatility, despite certain limitations (Jacobs et al., 2017; Padoan et al., 2020).

Studies involving hyperaccumulators primarily focused on well-known plant species, which yielded variable results at the field scale owing to differences in edaphic and pedological factors (Rosenfeld et al., 2018). For instance, Jacobs et al. (2017) explored phytoextraction in urban wastelands in Brussels (Belgium) using the hyperaccumulator *Noccaea caerulescens*. Within the first vegetative season, they achieved results that could lead to realistic remediation timeframes (<5 years) for sites with low contamination (*i.e.*, less than 2 mg kg^{-1} and 200 mg kg^{-1} of exchangeable Cd and Zn in the soil), effectively reducing the bioavailable fraction in the soil.

Fast-growing plants, particularly when using the short rotation coppice (SRC) technique, have also been studied to enhance plant uptake through increased biomass production in industrial and mining areas (Courchesne et al., 2017; Thomas et al., 2022). However, this approach has not been explored in urban, low-contaminated soils. In a 2-year field study, Padoan et al. (2020) employed the SRC technique with poplar, robinia, and willow spp. These authors found that the bioavailable concentrations of Cu and Zn in urban soil decreased by 26%–36%, while the total metal pool in the soil remained unchanged. This suggests that the process was limited by the bioavailability of inorganic contaminants. The authors also noted that, while the available fraction decreased significantly, the total metal pool remained unaffected, indicating that these plants could be beneficial in a phytostabilization approach, a concept traditionally discussed in the literature (Eriksson and Ledin, 1999).

Phytostabilization techniques, which focus on soil recovery and biodiversity restoration rather than complete remediation, aim to restore ecosystem services while mitigating potential risks. Among these techniques, those using plants already adapted to urban environments or naturally occurring in these areas have been the most widely studied on a field scale. Adamo et al. (2015) investigated the potential of spontaneous natural revegetation and phytostabilization with over 50 plant species on a soil contaminated with inorganic and organic compounds. Among the species considered in the study, *Bituminaria bituminosa* and *Dactylis glomerata* effectively immobilized metals in their roots, making them suitable for phytostabilization. In contrast, *Daucus carota* exhibited higher metal translocation to aerial parts but low biomass and metal uptake, limiting its potential for phytoremediation. The study demonstrated good adaptability of these species to the urban settings, offering a promising approach for recovering soil health in degraded areas. Similarly, Petrova et al. (2022) used perennial grasses (such as ryegrass, *Lolium perenne* L.), crested wheatgrass (*Agropyron cristatum* L.), tall fescue (*Festuca arundinacea* Schreb.), and bird's foot trefoil (*Lotus corniculatus* L.) for bioremediation of polluted urban soils in Bulgaria, identifying certain species with favorable bioaccumulation and translocation properties as suitable candidates for phytomanagement studies. The accumulation factors for all examined PTEs were significantly higher in the roots than in the aboveground parts, except for molybdenum, which exhibited the highest BAF (Bioaccumulation Factor) (2.16–15.69) and TF (Transfer Factor) (0.69–1.42) values. Additionally, their research showed improvements in soil microbiota, with a notable recovery of soil functionality, as indicated by a higher presence of bacteria and fungi and enhanced microbial physiological profiles at the community level. Other studies, such as Fernández-Braña et al. (2023) have explored the use of spontaneous plant species, including *Acer pseudoplatanus* L. and *Betula celtiberica* Rothm. & Vasc., and the shrub *Buddleja davidii* Franch. For the recovery of urban brownfields. Their results indicated that these species adapted their phytoremediation strategies based on soil contamination levels, as they were effective candidates for phytostabilization in highly contaminated areas, while their phytoextraction capacity was more suitable for soils with low contamination.

Based on the findings from these studies, effective phytoremediation requires a tailored approach based on the contamination levels and specific environmental conditions of the site. For moderate pollution, hyperaccumulators are ideal for metal removal, while stabilizing species such as *Salix* and *Acer* are better suited for high-pollution areas, where they help immobilize contaminants. Combining these methods in soils with multiple contaminants offers a balanced and efficient solution, promoting a sustainable and long-term remediation strategy. However, newer and more innovative techniques, such as biostimulation (which induces modifications to the microbiota to increase the availability of inorganic contaminants) or alterations to soil characteristics (*e.g.*, pH, Eh, and nutrient availability), have yet to be explored at the field level in urban areas. This gap in research should be a priority for future studies.

3.2 Bioremediation of organic pollutants

Persistent Organic Pollutants (POPs) pose a significant challenge in soil contamination. Figure 2B provides a summary of the main sources of organic pollutants, their effects on soil health, bioremediation techniques, and the associated ecosystem benefits.

These pollutants primarily result from human activities, including the rupture of underground storage tanks, pesticide application, percolation of contaminated surface water into subsurface layers, oil and fuel disposal, leaching of landfill waste, and the direct discharge of industrial waste into the soil (United Nations Environment Programme (UNEP), 2001). The persistence of POPs in the environment due to their resistance to the environmental degradation (*i.e.*, chemical, biological, and photolytic), exacerbates their impact, making them a long-term threat to soil health and ecosystems (Gupta and Ali, 2012; Alharbi et al., 2018).

This class of compounds includes pesticides, industrial chemicals such as PCBs (polychlorinated biphenyls), PBDEs (polybrominated diphenyl ethers), and PFOS (perfluorooctane sulfonate), as well as by-products of industrial processes like dioxins and furans. These substances have the potential to bioaccumulate in the food chain, affecting animals and humans, which can lead to significant health risks and ecotoxicological impacts. As a result, numerous studies have been conducted to better understand the behavior and risks of these priority pollutants (Smaranda and Gavrilescu, 2008; Crinnion, 2011).

Most POPs are highly lipophilic and semi-volatile. Additionally, they can associate with atmospheric aerosols, facilitating their long-range transport and redistribution across different ecosystems. However, some POPs, such as PFOS, exhibit water solubility. Halogenated POPs, in particular, demonstrate greater resistance to degradation due to their stable molecular structures.

Biodegradation is the natural process by which living organisms, primarily microorganisms, break down synthetic chemicals through biologically catalyzed reactions, decreasing their environmental impact (Anderson et al., 2022). This process involves degradation of POPs into smaller, less harmful components (Marinescu et al., 2009). Certain bacteria and fungi have evolved specialized metabolic and enzymatic pathways that enable them to utilize POPs as C and energy sources (Ayilara and Babalola, 2023). However, the effectiveness of biodegradation depends on the bioavailability of these pollutants, which is influenced by both their chemical properties and soil characteristics that affect microbial access (Sivaperumal et al., 2017; Kumar et al., 2020a; b).

Bioremediation of urban soils can exploit the process of biodegradation and be further enhanced through the use of plants in combination with microorganisms. This approach, known as plant-assisted bioremediation, utilizes plants to stimulate bacterial degradation of POPs. Plant-microorganism interactions in the rhizosphere can enhance PCB degradation by fostering synergistic relationships between plant roots and the native soil microbial community, as many plant species not only thrive in PCB-contaminated soils but also stimulate indigenous microbial populations with biodegradative capabilities (Leigh et al., 2006; Mackova et al., 2009 Glick, 2010; Xu et al., 2010; Slater et al., 2011; Sylvestre and Toussaint, 2011; Meggo and Schnoor, 2013; Uhlik et al., 2013).

The application of a plant-assisted bioremediation strategy in a historically contaminated area near Taranto (Southern Italy), demonstrated the effectiveness of the Monviso poplar clone (*Populus generosa* x *Populus nigra*) in promoting pollutant degradation (Ancona et al., 2017). Significant reductions in PCBs and heavy metals were observed in soil where poplars were planted, whereas no reduction occurred in non-vegetated areas. Additionally, microbial analyses indicated an overall improvement in soil quality and microbial activity, particularly in the rhizosphere, with effects detected up to 1 m from the trunk and at depths of 40 cm. Within just over a year of planting, the Monviso clone successfully enhanced microbial abundance and facilitated contaminant degradation, highlighting its potential as a NbS for remediating chronically polluted urban environments. In the same area, Barra Caracciolo et al. (2020) demonstrated that the Monviso poplar clone promoted a significant shift in the structure and predicted function of the belowground microbial community, revealing an increase in Proteobacteria genera known for PCB degradation and heavy metal resistance. Functional profiling using PICRUSt2 bioinformatics tool identified key genes associated with PCB transformation (bphAa, bphAb, bphB, bphC), oxidative stress response (catalase, superoxide reductase, peroxidase), and heavy metal uptake and expulsion (ABC transporters). Gómez-Sagasti et al. (2021) conducted a 2-year study on a peri-urban vacant site contaminated with petroleum hydrocarbons and PCBs to evaluate the effects of two phytoremediation strategies on soil chemical, physical, and biological properties. The study compared intercropping of alfalfa (*Medicago sativa* L.) with hybrid black poplar (*Populus canadensis*) vs. monocultures of each species. Additionally, the impact of inoculation with a commercial arbuscular mycorrhizal and ectomycorrhizal inoculum was assessed in comparison to non-inoculated conditions. A key finding was that, although none of the phytoremediation treatments led to a significant reduction in organic contaminant concentrations in 2 years, they effectively improved soil health, as indicated by increased microbial biomass, activity, and biodiversity. These results highlight the positive effects of alfalfa sowing on soil microbial biomass and the benefits of intercropping in stimulating enzyme activity in soils contaminated with organic pollutants.

In the study by Meggo and Schnoor (2013), a different approach was explored to promote PCB degradation, based on sequential cycles of dechlorination followed by aerobic bio-oxidation. Soil was artificially contaminated with PCBs both as a mixture and as single congeners, then aged and planted with two different plant species. To enhance degradation, alternating redox cycles were established in the plant root zone through periodic flooding and draining of the soil. Over 32 weeks, planted systems containing switchgrass (*Panicum virgatum*) and poplar (*Populus deltoids* x *nigra* DN34), that were exposed to alternate cycles of flooding, demonstrated superior PCB reduction compared to non-cycled planted systems. Additionally, cycled systems accumulated a higher mass of PCB transformation products, indicating more effective degradation. Notably, multiple redox cycles were required to achieve significant differences between cycled and non-cycled treatments, underscoring the importance of controlled redox fluctuations in optimizing PCB bioremediation.

Table 2 illustrates selected studies concerning the removal of organic pollutants by NbS-bioremediation approaches.

3.3 Bioremediation of microplastics

The problem of MP pollution is undoubtedly one of the most emerging threats of environmental contamination and represents one of the greatest environmental challenges to be faced in recent decades. In the context of urban environments, one major source of MP pollution derives from individuals who improperly dispose of plastic waste. This behaviour contributes significantly to environmental pollution, leading to the accumulation of microplastics in soil.

The relentless and widespread use of plastics requires a regulamentation and some government already provided for some laws (Directive (EU), 2019), but the efforts to limit plastics should be more. Given its importance, this phenomenon is attracting huge attention from the scientific community and policymakers globally (Bergmann et al., 2015; Li et al., 2018). In the context of plastics, bioplastics represent an emerging category of materials designed to address some of the main environmental issues associated with conventional plastics, such as reliance on fossil fuels and the accumulation of plastic waste. The term “bioplastic” can refer to plastics made from renewable materials, plastics that undergo natural degradation, or both characteristics simultaneously. However, it is important to note that, like plastic materials, bioplastics may not always be biodegradable, even if they are made from renewable materials. Therefore, if bioplastics do not fully degrade, they can break down into MPs, which persist in the environment and can be taken up by different organisms (Celletti et al., 2023).

In general, MPs are plastic polymer particles less than 5 mm in size and resulting from the fragmentation of larger plastic products or from the direct use of industrial microgranules (e.g., in cosmetics and synthetic textiles) (Ghosh et al., 2023). These dimensions are essentially at the core of the pollution and environmental hazard problem of MPs, as they are difficult to detect, collect and are easily dispersed over long distances in the environment, and eventually be found in a wide range of ecosystems, ranging from terrestrial to aquatic (Pandey et al., 2023). However, originally, early scientific studies mainly focused on the impact of MPs in marine environments, where their presence has been thoroughly documented (Law and Thompson, 2014); instead, more recent

TABLE 2 Selected studies addressing NbS for removal of organic pollutants in urban soils. PAHs: polycyclic aromatic hydrocarbons; PCBs: polychlorinated biphenyls.

Target species	Contaminant	Initial concentration	Time (months)	Accumulation	Reference
Alfalfa	Hydrocarbons, PAHs, PCBs	70–200 mg kg^{-1} (Hydrocarbons), 0.02–0.14 mg kg^{-1} (PAHs), 0–22 mg kg^{-1} (PCBs)	16	51% (w w^{-1}) (hydrocarbons) 22%–33% (w w^{-1}) (PAHs)	Gómez-Sagasti et al. (2021)
Poplar (Rhizosphere)	PCBs	246 ng g^{-1}	14	90% (w w^{-1})	Ancona et al. (2017)
Alfalfa + *Rhizobium*	PCBs	414–498 μg kg^{-1}	3	36%–43% (w w^{-1})	Xu et al. (2010)
Switchgrass, poplar + flood cycling	PCBs	499–1,120 ng g^{-1}	8	44%–56% (w w^{-1}) (switchgrass) 42%–57% (w w^{-1}) (poplar)	Meggo and Schnoor (2013)
Plant-PSM combinations	PCBs	30 μg g^{-1}	2	29%–50% (w w^{-1})	Uhlik et al. (2013)
Rhizoremediation (willow, spruce)	PCBs	38 mg kg^{-1}	6	40% (w w^{-1}) (willow)	Slater et al. (2011)

studies have begun to highlight an equally worrying problem: the spread of these particles in agroecosystems (Ng et al., 2018; Hoang et al., 2024). Indeed, global studies, such as the one conducted by Bouwmeester et al. (2015), have highlighted how MPs have silently invaded different soil types, from agricultural to natural areas, with varying but often alarming concentrations.

Figure 2C provides a summary of the main sources of MPs, their effects on soil health, bioremediation techniques, and the associated ecosystem benefits.

Most of the MPs present in agricultural soils often have an urban origin, deriving from various activities related to city life (Sajjad et al., 2022). The main sources include: (i) MPs originating from synthetic fibers and cosmetic products that are not completely removed by sewage treatment plants and end up in sludge, which is used as organic fertilizer in agricultural fields; (ii) rainwater that carries MPs from roads and construction sites to agricultural areas; (iii) mismanaged urban plastic wastes, which degrade and release MPs, which enter the agricultural chain through the use of soil organic amendments (*i.e.*, compost) derived from urban wastes. In turn, MPs of agricultural origin can reach urban soils when contaminated organic fertilizers are used in city parks and gardens. Furthermore, one of the main sources of MPs in urban soils is due to the degradation by weathering and UV radiation of abandoned or mismanaged plastic wastes (such as bags, bottles, packaging, etc.), which fragment into increasingly smaller particles (Büks and Kaupenjohann, 2020).

Despite the evidence of a direct and reciprocal link between urban and agricultural pollution, which highlights the urgency of improving urban waste management to limit the spread of MPs, the existing literature on MPs in urban soils is scarce, as it remains an emerging field compared to studies on agricultural environments and especially compared to marine environments (Ihenetu et al., 2024). In general, recent research on MPs in urban soils mainly focuses on the sources of contamination, their distribution, ecotoxicological effects, and detection methodologies. However, urban soils are highly vulnerable to pollution by MPs because, on the one hand, concentrations of these contaminants are significantly high due to intensive anthropogenic activities and, on the other hand, because the problem of MPs is added to the other pollutants resulting from urbanization, exacerbating soil degradation, reducing fertility and the capacity to support vegetation and thus threatening food security and environmental sustainability (Sajjad et al., 2022).

Once in the soil, MPs can persist for a long time and resist chemical degradation by contributing to soil accumulation, interacting with the soil matrix and altering fundamental physical, chemical, and biological properties of the soil. Specifically, the presence of MPs in soils can affect fundamental processes such as water retention, soil aeration, soil structure and nutrient cycling (de Souza Machado et al., 2018). As an example, Chen et al. (2024) examined the distribution of MPs in agricultural soil microaggregates, highlighting how their presence can alter soil structure, affecting porosity, and water retention capacity. Microplastics also have an impact on soil biodiversity; in particular, MPs can alter the composition and functionality of the microbial community and can be ingested by soil invertebrate organisms (such as earthworms), causing physical damage to their digestive tracts, reducing their activity and ability to improve soil structure. Furthermore, MPs may contain a certain chemical toxicity, due to the presence of chemical additives (*e.g.*, plasticizers as phthalates, dyes, stabilizers as organotin compounds), which can then be released into the soil and interfere with the physiological and metabolic processes of plants (Teuten et al., 2009; Bouwmeester et al., 2015). Alternatively, MPs can act as vectors for pathogenic microorganisms, facilitating their survival and spread in the soil and, consequently, leading to significant reductions in crop yields. Therefore, the presence of MPs can lead, in general, to potentially negative effects on plants, microorganisms, and edaphic invertebrates, compromising soil quality and plant health, potentially affecting agricultural production and food safety (Hoang et al., 2024). Recent findings further highlight the role of plants in interacting with environmental MPs not only through root uptake but also via aerial parts. A study by Li et al. (2025) has shown that the absorption and accumulation of airborne MPs by plant leaves occurs widely in the environment, representing an underestimated pathway of exposure. This mechanism should

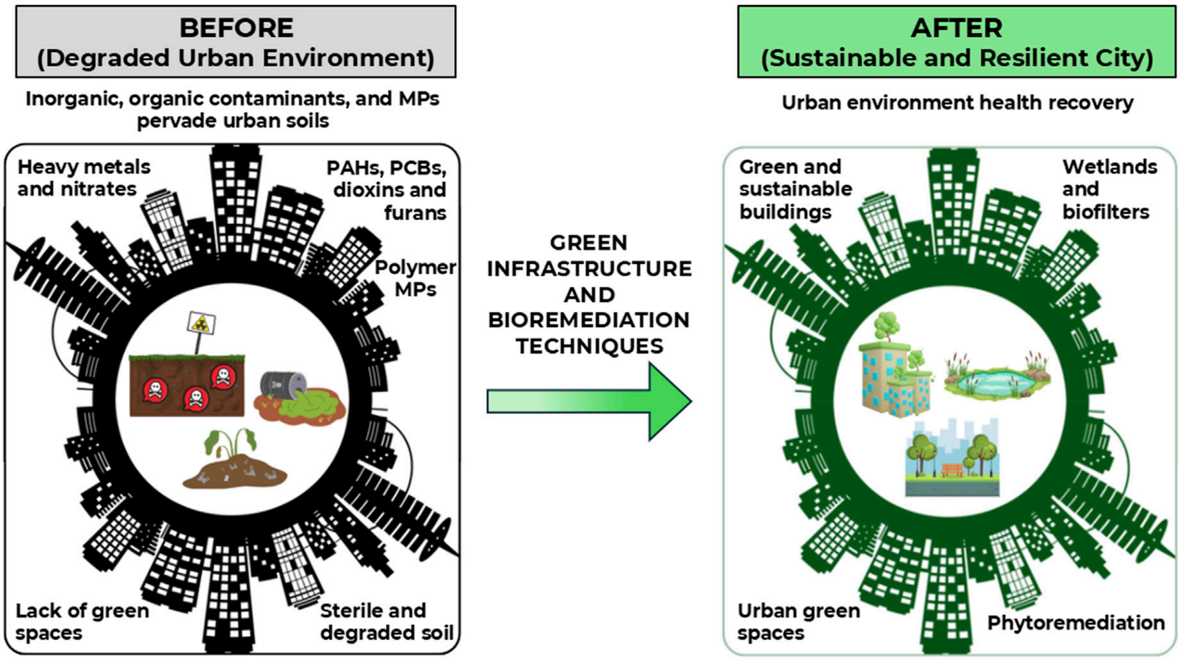

FIGURE 3
Conceptual framework illustrating the transformation of degraded urban soil through green infrastructure and bioremediation strategies to mitigate contaminants. The figure was created using Canva (www.canva.com).

not be overlooked when assessing the risk of MPs exposure for humans and other organisms through the food chain.

In light of the expanding concern, bioremediation techniques to degrade, remove or transform contaminants in soil, water or air into less toxic or harmless compounds are also proposed as an environmentally friendly and sustainable solution to mitigate MPs pollution in soils (Thapliyal et al., 2024).

Microorganisms, in particular bacteria and fungi, play a central role in the degradation, stabilization and removal of MPs in soil, thanks to their ability to metabolize complex compounds and adapt to different environmental conditions (Harshvardhan and Jha, 2013; Vaksmaa et al., 2023). Several studies have explored different potentials offered by specific bacterial genera, such as *Pseudomonas*, *Bacillus* and *Rhodococcus*, which are capable of degrading various types of polymers, including polyethylene (PE), polypropylene (PP) and polystyrene (PS), contributing to the reduction of MPs accumulation in soil (Yuan et al., 2020; Thapliyal et al., 2024). These bacteria produce hydrolytic enzymes that attack polymeric bonds, facilitating the fragmentation of plastic particles and transforming them into simpler, biodegradable compounds (Restrepo-Flórez et al., 2014). Similarly, fungi, especially ligninolytic fungi such as *Aspergillus* and *Penicillium*, have shown promising abilities to degrade plastic polymers, thanks to the production of extracellular enzymes such as laccase and peroxidase, which both catalyze the breaking of polymer chains (Temporiti et al., 2022). These enzymes are particularly effective in degrading complex polymers, reducing their mass and modifying their chemical and physical properties. However, the microbial degradation of MPs can be a slow process, influenced by environmental factors such as nutrient availability, temperature and soil pH (Sutkar Pankaj et al., 2023).

A promising strategy to optimize the effectiveness of bioremediation is the use of microbial consortia, *i.e.*, communities composed of different bacterial and fungal species working in synergy. These consortia can simultaneously attack different polymeric components, accelerating the degradation of MPs (Lokesh et al., 2023). For instance, combinations of *Bacillus subtilis* and *Pseudomonas putida* have been shown to degrade PE more efficiently than single strains, thanks to a wider range of degrading enzymes produced (Yuan et al., 2020). Furthermore, the use of phytodepurative plants, such as *Phragmites australis* and *Typha latifolia*, is proving to be an innovative option. These plants can absorb plastic particles through their roots, limiting their dispersion and helping to reduce soil contamination (Fletcher et al., 2020). The extensive root system of phytodepurators facilitates the contact of MPs with the degrading microorganisms in the soil, thus enhancing the biodegradation process. It has been demonstrated that plants can not only affect the physical degradation of MPs, through interaction with microorganisms in the rhizosphere, but can also act as bioindicators, signaling the presence and concentration of MPs through changes in root growth and development (Jia et al., 2023). For example, it has been observed that high concentrations of MPs in soil can reduce root growth and water and nutrient uptake, negatively affecting plant health and agricultural productivity.

A further enhancer of bioremediation is arbuscular mycorrhizal fungi (AMF), which establish mutualistic symbioses with plant roots, improving nutrient uptake and tolerance to environmental stresses. The AMF have been shown to interact with MPs in the soil, contributing to their degradation through the production of extracellular enzymes and promoting polymer mineralization (Thapliyal et al., 2024). Applying organic soil amendments, such as biochar, could be a valuable aid in improving the efficiency of these enzymatic processes. Biochar, which is derived from the pyrolysis of organic residues, not only increases the stability and activity of microbial enzymes, but also adsorbs MPs, reducing their mobility and prolonging their contact time with the degrading microorganisms (Li et al., 2024). In addition, using compost enriched with biochar could create a favourable environment for microbial growth, improving the effectiveness of MPs degradation.

Another very innovative approach is the use of algae, in particular microalgae, in the bioremediation of MPs, as discussed by Chia et al. (2020). Microalgae, usually used to treat contaminated water, can also be applied to soils to bioaccumulate MPs. Indeed, thanks to their ability to form biofilms, algae can capture and immobilize plastic particles, thus facilitating their removal from the environment. Although this technique is still in the experimental phase for soil applications, it represents a promising area of research for the future.

4 Concluding remarks

Nature-based Solutions represent a promising and sustainable approach to addressing contemporary environmental and social challenges in urban areas. By exploiting the intrinsic properties of natural ecosystems, NbS provide multiple benefits, including climate change mitigation, biodiversity conservation, and enhanced human wellbeing. The integration of NbS into urban planning, particularly through green infrastructure and bioremediation practices, offers a pathway toward more sustainable and resilient cities while also providing economic and social benefits (see Figure 3). Among the social benefits, civic engagement is arguably the most important output: without shared responsibilities and collective knowledge, it would be impossible to develop and sustain NbS. Also, before implementing an NbS, it is also important to consider the social aspects, including how the stakeholders and citizens could be actively involved in the project.

Beyond their role in urban greening, NbS contribute to the bioremediation of contaminated urban soils, offering a sustainable and ecosystem-centered approach in improving soil fertility and restoring ecosystem services. Generic recommendations for bioremediation within the NbS framework include the use of native or climate-resilient plant species, consortia of beneficial microorganisms adapted to local conditions, and the application of organic amendments to support soil microbial activity and structure. Such integrated approaches have shown significant potential in reducing pollutants like heavy metals, hydrocarbons, and MPs, restoring soil functionality and promoting ecological balance. Innovative NbS-aligned proposals might involve the design of multifunctional landscapes that combine remediation approaches with biodiversity enhancement, including constructed wetlands, urban green spaces, or green corridors that incorporate plant species employed in phytoremediation. The use of nature-compatible technologies and monitoring systems can further improve the efficiency of bioremediation.

Despite these advantages, the effectiveness of bioremediation still presents numerous challenges, as it depends on various environmental factors, including pH, temperature, O_2 availability, nutrient content, and the chemical-physical diversity of contaminants. In some cases, additional interventions such as biostimulation (*e.g.*, the addition of nutrients to enhance microbial activity) or bioaugmentation (*e.g.*, the introduction of specific microorganisms) may be required to accelerate the degradation process. Furthermore, a deeper understanding of the interactions between contaminants, plants, and microorganisms is needed within the urban context, which is characterized by anthropogenic stresses and variable environmental conditions. Future research should focus on optimizing NbS strategies in urban environments, improving their scalability, and assessing long-term impacts to maximize their effectiveness.

Author contributions

SCe: Visualization, Writing – original draft, Investigation, Writing – review and editing, Conceptualization. LP: Writing – review and editing, Methodology, Writing – original draft, Investigation. RW: Funding acquisition, Conceptualization, Writing – review and editing. EP: Resources, Writing – review and editing, Funding acquisition, Writing – original draft, Project administration, Conceptualization. SCo: Investigation, Writing – original draft. BB: Writing – review and editing, Methodology. MS: Investigation, Conceptualization, Visualization, Writing – review and editing, Methodology, Writing – original draft.

Funding

The author(s) declare that financial support was received for the research and/or publication of this article. The work was funded from the Horizon Europe scheme under the project “Nature Based Solutions for Soil Management”, Grant agreement ID: 101091246.

Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

The author(s) declared that they were an editorial board member of Frontiers, at the time of submission. This had no impact on the peer review process and the final decision.

Generative AI statement

The author(s) declare that no Generative AI was used in the creation of this manuscript.

References

Adamo, P., Mingo, A., Coppola, I., Motti, R., Stinca, A., and Agrelli, D. (2015). Plant colonization of brownfield soil and post-washing sludge: effect of organic amendment and environmental conditions. *Int. J. Environ. Sci. Technol.* 12, 1811–1824. doi:10.1007/s13762-014-0597-9

Aimar, F. (2023). "Limiting soil sealing and depaving: local actions for regenerating public spaces to build green infrastructures," in *Urban book series*. doi:10.1007/978-3-031-28772-5_11

Alharbi, O. M. L., Basheer, A. A., Khattab, R. A., and Ali, I. (2018). Health and environmental effects of persistent organic pollutants. *J. Mol. Liq.* 263, 442–453. doi:10.1016/j.molliq.2018.05.029

Ali, M., Song, X., Ding, D., Wang, Q., Zhang, Z., and Tang, Z. (2022). Bioremediation of PAHs and heavy metals co-contaminated soils: challenges and enhancement strategies. *Environ. Pollut.* 295, 118686. doi:10.1016/j.envpol.2021.118686

Alpaidze, L., and Pace, R. (2021). Ecosystem services provided by urban forests in the southern caucasus region: a modeling study in tbilisi, Georgia. *Climate* 9 (11), 157. doi:10.3390/cli9110157

Ambrosini, P., Cazzaniga, B., Comba, S., De Angelis, P., Diligenti, A., Doni, S., et al. (2017). *Tecniche di fitorimedio nella bonifica dei siti contaminati*. Roma: Consiglio Nazionale delle Ricerche.

Ancona, V., Barra Caracciolo, A., Grenni, P., Di Lenola, M., Campanale, C., Calabrese, A., et al. (2017). Plant-assisted bioremediation of a historically PCB and heavy metal-contaminated area in southern Italy. *New Biotechnol.* 38 (Pt B), 65–73. doi:10.1016/j.nbt.2016.09.006

Anderson, A., Anbarasu, A., Pasupuleti, R. R., Manigandan, S., Praveenkumar, T. R., and Aravind Kumar, J. (2022). Treatment of heavy metals containing wastewater using biodegradable adsorbents: a review of mechanism and future trends. *Chemosphere* 295, 133724. doi:10.1016/j.chemosphere.2022.133724

Aparicio, J. D., Raimondo, E. E., Saez, J. M., Costa-Gutierrez, S. B., Alvarez, A., Benimeli, C. S., et al. (2022). The current approach to soil remediation: a review of physicochemical and biological technologies, and the potential of their strategic combination. *J. Environ. Chem. Eng.* 10, 107141. doi:10.1016/j.jece.2022.107141

Ayilara, M. S., and Babalola, O. O. (2023). Bioremediation of environmental wastes: the role of microorganisms. *Front. Agron.* 5, 1183691. doi:10.3389/fagro.2023.1183691

Ayilara, M. S., Adeleke, B. S., Adebajo, M. T., Akinola, S. A., Fayose, C. A., Adeyemi, U. T., et al. (2023). Remediation by enhanced natural attenuation; an environment-friendly remediation approach. *Front. Environ. Sci.* 11, 1182586. doi:10.3389/fenvs.2023.1182586

Barra Caracciolo, A., Grenni, P., Garbini, G. L., Rolando, L., Campanale, C., Aimola, G., et al. (2020). Characterization of the belowground microbial community in a poplar-phytoremediation strategy of a multi-contaminated soil. *Front. Microbiol.* 11, 2073. doi:10.3389/fmicb.2020.02073

Bergmann, M., Gutow, L., and Klages, M. (2015). *Marine anthropogenic litter* (Springer). doi:10.1007/978-3-319-16510-3

Biasioli, M., and Ajmone-Marsan, F. (2007). Organic and inorganic diffuse contamination in urban soils: the case of Torino (Italy). *JEM* 9 (8), 862–868. doi:10.1039/b705285e

Binner, H., Sullivan, T., Jansen, M. A. K., and McNamara, M. E. (2023). Metals in urban soils of Europe: a systematic review. *Sci. Total Environ.* 854, 158734. doi:10.1016/j.scitotenv.2022.158734

Bochenek, A. D., Klemm, K., and Woźna, M. (2022). Assessment of effectiveness of selected adaptation actions to climate change. The example of the new centre of lodz. *Bud. Archit.* 21 (4), 025–042. doi:10.35784/bud-arch.3150

Bona, S., Silva-Afonso, A., Gomes, R., Matos, R., and Rodrigues, F. (2023). Nature-based solutions in urban areas: a european analysis. *Appl. Sci.* 13 (1), 168. doi:10.3390/app13010168

Bonoli, A., Conte, A., Maglionico, M., and Stojkov, I. (2013). Green roofs for sustainable water management in urban areas. *Environ. Eng. Manag. J.* 12 (S11), 153–156.

Bouwmeester, H., Hollman, P. C. H., and Peters, R. J. B. (2015). Potential health impact of environmentally released micro- and nanoplastics in the human food production chain: experiences from nanotoxicology. *Environ. Sci. Technol.* 49 (15), 8932–8947. doi:10.1021/acs.est.5b01090

Bouzouidja, R., Rousseau, G., Galzin, V., Claverie, R., Lacroix, D., and Séré, G. (2018). Green roof ageing or isolatic Technosol's pedogenesis? *J. Soils Sediments* 18 (2), 418–425. doi:10.1007/s11368-016-1513-3

Bouzouidja, R., Leconte, F., Kiss, M., Pierret, M., Pruvot, C., Détriché, S., et al. (2021). Experimental comparative study between conventional and green parking lots: analysis of subsurface thermal behavior under warm and dry summer conditions. *Atmosphere* 12 (8), 994. doi:10.3390/atmos12080994

Bretzel, F., Vannucchi, F., Pezzarossa, B., Paraskevopoulou, A., and Romano, D. (2024). Establishing wildflower meadows in anthropogenic landscapes. *Front. Hortic.* 2, 1248785. doi:10.3389/fhort.2023.1248785

Büks, F., and Kaupenjohann, M. (2020). Global concentrations of microplastics in soils – a review. *Soil* 6, 649–662. doi:10.5194/soil-6-649-2020

Burges, A., Fievet, V., Oustriere, N., Epelde, L., Garbisu, C., Becerril, J. M., et al. (2020). Long-term phytomanagement with compost and a sunflower –Tobacco rotation influences the structural microbial diversity of a Cu-contaminated soil. *Sci. Total Environ.* 700, 134529. doi:10.1016/j.scitotenv.2019.134529

Calfapietra, C. (2020). "Nature-based solutions for microclimate regulation and air quality," in *Nature-based solutions: state of the art in EU-funded projects*. doi:10.2777/383904

Capuana, M. (2020). A review of the performance of woody and herbaceous ornamental plants for phytoremediation in urban areas. *iForest* 13, 139–151. doi:10.3832/ifor3242-013

Celletti, S., Fedeli, R., Ghorbani, M., and Loppi, S. (2023). Impact of starch-based bioplastic on growth and biochemical parameters of basil plants. *Sci. Total Environ.* 856 (Part 2), 159163. doi:10.1016/j.scitotenv.2022.159163

Chaney, R. L., Angle, J. S., Broadhurst, C. L., Peters, C. A., Tappero, R. V., and Sparks, D. L. (2007). Improved understanding of hyperaccumulation yields commercial phytoextraction and phytomining technologies. *J. Environ. Qual.* 36, 1429–1443. doi:10.2134/jeq2006.0514

Chen, Z., Carter, L. J., Banwart, S. A., Das Pramanik, D., and Kay, P. (2024). Multifaceted effects of microplastics on soil-plant systems: exploring the role of particle type and plant species. *Sci.Total Environ.* 954, 176641. doi:10.1016/j.scitotenv.2024.176641

Chia, W. Y., Ying Tang, D. Y., Khoo, K. S., Kay Lup, A. N., and Chew, K. W. (2020). Nature's fight against plastic pollution: algae for plastic biodegradation and bioplastics production. *Environ. Sci. Ecotechnology* 5 (4), 100065. doi:10.1016/j.ese.2020.100065

Cohen-Shacham, E., Walters, G., Janzen, C., and Maginnis, S. (2016). *Nature-based solutions to address global societal challenges* (IUCN). Switzerland: Gland

Cornu, J. Y., Huguenot, D., Jézéquel, K., Lollier, M., and Lebeau, T. (2017). Bioremediation of copper-contaminated soils by bacteria. *World J. Microb. Biot.* 33 (2), 26. doi:10.1007/s11274-016-2191-4

Courchesne, F., Turmel, M.-C., Cloutier-Hurteau, B., Constantineau, S., Munro, L., and Labrecque, M. (2017). Phytoextraction of soil trace elements by willow during a phytoremediation trial in southern Québec, Canada. *Int. J. Phytoremediation* 19 (6), 545–554. doi:10.1080/15226514.2016.1267700

Crinnion, W. J. (2011). Polychlorinated biphenyls: persistent pollutants with immunological, neurological, and endocrinological consequences. *Altern. Med. Rev.* 16 (1), 5–13.

Cristiano, E., Urru, S., Farris, S., Ruggiu, D., Deidda, R., and Viola, F. (2020). Analysis of potential benefits on flood mitigation of a CAM green roof in mediterranean urban areas. *Build. Environ.* 183, 107179. doi:10.1016/j.buildenv.2020.107179

de Souza Machado, A. A., Lau, C. W., Till, J., Kloas, W., Lehmann, A., Becker, R., et al. (2018). Impacts of microplastics on the soil biophysical environment. *Environ. Sci. Technol.* 52 (17), 9656–9665. doi:10.1021/acs.est.8b02212

Debele, S. E., Leo, L. S., Kumar, P., Sahani, J., Ommer, J., Bucchignani, E., et al. (2023). Nature-based solutions can help reduce the impact of natural hazards: a global analysis of NBS case studies. *Sci. Total Environ.* 902, 165824. doi:10.1016/j.scitotenv.2023.165824

Deng, T.-H.-B., van der Ent, A., Tang, Y.-T., Sterckeman, T., Echevarria, G., Morel, J.-L., et al. (2018). Nickel hyperaccumulation mechanisms: a review on the current state of knowledge. *Plant Soil* 423 (1-2), 1–11. doi:10.1007/s11104-017-3539-8

Derakhshan Nejad, Z., Jung, M. C., and Kim, K. H. (2018). Remediation of soils contaminated with heavy metals with an emphasis on immobilization technology. *Environ. Geochem. Health* 40 (3), 927–953. doi:10.1007/s10653-017-9964-z

Dickinson, N. M., Baker, A. J. M., Doronila, A., Laidlaw, S., and Reeves, R. D. (2009). Phytoremediation of inorganics: realism and synergies. *Int. J. Phytoremediation* 11, 97–114. doi:10.1080/15226510802378368

Directive (EU) (2019). 2019/904 of the european parliament and of the council of 5 June 2019 on the reduction of the impact of certain plastic products on the environment. Available online at: https://eur-lex.europa.eu/eli/dir/2019/904/oj/eng?utm_source=chatgpt.com.

Donati, G. F. A., Bolliger, J., Psomas, A., Maurer, M., and Bach, P. M. (2022). Reconciling cities with nature: identifying local blue-green infrastructure interventions for regional biodiversity enhancement. *J. Environ. Manag.* 316, 115254. doi:10.1016/j.jenvman.2022.115254

Dong, X., Chu, Y., Tong, Z., Sun, M., Meng, D., Yi, X., et al. (2024). Mechanisms of adsorption and functionalization of biochar for pesticides: a review. *Ecotoxicol. Environ. Saf.* 272, 116019. doi:10.1016/j.ecoenv.2024.116019

Dorst, H., van der Jagt, A., Raven, R., and Runhaar, H. (2019). Urban greening through nature-based solutions – key characteristics of an emerging concept. *Sustain. Cities Soc.* 49, 101620. doi:10.1016/j.scs.2019.101620

Drenning, P., Norrman, J., Chowdhury, S., Rosén, L., Volchko, Y., and Andersson-Sköld, Y. (2020). Enhancing ecosystem services at urban brownfield sites - what value does contaminated soil have in the built environment? *IOP Conf. Ser. Earth Environ. Sci.* 588, 052008. doi:10.1088/1755-1315/588/5/052008

Edmondson, J. L., Davies, Z. G., McCormack, S. A., Gaston, K. J., and Leake, J. R. (2014). Land-cover effects on soil organic carbon stocks in a European city. *Sci. Total Environ.* 472, 444–453. doi:10.1016/j.scitotenv.2013.11.025

Eriksson, J., and Ledin, S. (1999). Changes in phytoavailability and concentration of cadmium in soil following long term salix cropping. *Water Air Soil Pollut.* 114 (1/2), 171–184. doi:10.1023/A:1005050005459

European Commission (2013). "Green infrastructure (GI) - enhancing Europe's natural capital," in *Communication from the commission to the european parliament, the european economic council, the european economic and social committee and the committee of the regions*. doi:10.1017/CBO9781107415324.004

European Council (2006). Regulation (EC) No 1907/2006 of the european parliament and of the council of 18 December 2006 concerning the registration, evaluation, authorisation and restriction of chemicals (REACH). *Off. J. Eur. Union.* Available online at: http://data.europa.eu/eli/reg/2006/1907/oj.

Fernández-Braña, A., Salgado, L., Gallego, J. L. R., Afif, E., Boente, C., and Forján, R. (2023). Phytoremediation potential depends on the degree of soil pollution: a case study in an urban brownfield. *Environ. Sci. Pollut. Res.* 30 (25), 67708–67719. doi:10.1007/s11356-023-26968-5

Fletcher, J., Willby, N., Oliver, D. M., and Quilliam, R. S. (2020). "Phytoremediation using aquatic plants," in *Phytoremediation. Concepts and strategies in plant sciences*. Editor B. Shmaefsky (Cham: Springer). doi:10.1007/978-3-030-00099-8_7

Gao, D., Zhao, H., Wang, L., Li, Y., Tang, T., Bai, Y., et al. (2022). Current and emerging trends in bioaugmentation of organic contaminated soils: a review. *J. Environ. Manag.* 320, 115799. doi:10.1016/j.jenvman.2022.115799

Gerhardt, K. E., Gerwing, P. D., and Greenberg, B. M. (2017). Opinion: taking phytoremediation from proven technology to accepted practice. *Plant Scie* 256, 170–185. doi:10.1016/j.plantsci.2016.11.016

Gervais-Bergeron, B., Paul, A. L., Chagnon, P. L., Baker, A. J., van der Ent, A., Faucon, M. P., et al. (2023). Trace element hyperaccumulator plant traits: a call for trait data collection. *Plant Soil* 488 (1), 187–196. doi:10.1007/s11104-023-05996-7

Ghosh, S., Sinha, J. K., Ghosh, S., Vashisth, K., Han, S., and Bhaskar, R. (2023). Microplastics as an emerging threat to the global environment and human health. *Sustainability* 15, 10821. doi:10.3390/su151410821

Glick, B. R. (2010). Using soil bacteria to facilitate phytoremediation. *Biotechnol. Adv.* 28 (3), 367–374. doi:10.1016/j.biotechadv.2010.02.001

Gómez-Sagasti, M. T., Garbisu, C., Urra, J., Míguez, F., Artetxe, U., Hernández, A., et al. (2021). Mycorrhizal-assisted phytoremediation and intercropping strategies improved the health of contaminated soil in a peri-urban area. *Front. Plant Sci.* 12, 693044. doi:10.3389/fpls.2021.693044

Goodwin, S., Olazabal, M., Castro, A. J., and Pascual, U. (2024). Measuring the contribution of nature-based solutions beyond climate adaptation in cities. *Glob. Environ. Change* 89, 102939. doi:10.1016/j.gloenvcha.2024.102939

Guidi Nissim, W., Castiglione, S., Guarino, F., Pastore, M. C., and andl Labra, M. (2023). Beyond cleansing: ecosystem services related to phytoremediation. *Plants* 12, 1031. doi:10.3390/plants12051031

Gul, I., Manzoor, M., Hashim, N., Shah, G. M., Waani, S. P. T., Shahid, M., et al. (2021). Challenges in microbially and chelate-assisted phytoextraction of cadmium and lead-a review. *Environ. Pollut.* 287, 117667. doi:10.1016/j.envpol.2021.117667

Guo, Y., Liu, S., Dong, Y., Wu, G., Liu, J., Wang, W., et al. (2025). Research progress and prospects of nature-based solutions in green infrastructure: a bibliometric analysis. *Ecol. Front.* 45, 561–571. doi:10.1016/j.ecofro.2024.11.007

Gupta, V. K., and Ali, I. (2012). *Environmental water: advances in treatment, remediation and recycling*. Netherlands: Elsevier.

Hahladakis, J. N., and Iacovidou, E. (2018). Closing the loop on plastic packaging materials: what is quality and how does it affect their circularity? *Sci. Total Environ.* 630, 1394–1400. doi:10.1016/j.scitotenv.2018.02.330

Hansen, R., and Pauleit, S. (2014). From multifunctionality to multiple ecosystem services? A conceptual framework for multifunctionality in green infrastructure planning for urban areas. *Ambio* 43 (4), 516–529. doi:10.1007/s13280-014-0510-2

Harshvardhan, K., and Jha, B. (2013). Biodegradation of low-density polyethylene by marine bacteria from pelagic waters, Arabian Sea, India. *Mar. Pollut. Bull.* 77 (1-2), 100–106. doi:10.1016/j.marpolbul.2013.10.025

He, L., Gielen, G., Bolan, N. S., Xiaokai, Z., Qin, H., Huang, H., et al. (2015). Contamination and remediation of phthalic acid esters in agricultural soils in China: a review. *Agron. Sustain. Dev.* 35, 519–534. doi:10.1007/s13593-014-0270-1

Hoang, V. H., Nguyen, M. K., Hoang, T. D., Ha, M. C., Huyen, N. T. T., Bui, V. K. H., et al. (2024). Sources, environmental fate, and impacts of microplastic contamination in agricultural soils: a comprehensive review. *Sci. Total Environ.* 10 (950), 175276. doi:10.1016/j.scitotenv.2024.175276

Hou, D., Al-Tabbaa, A., O'Connor, D., Hu, Q., Zhu, Y. G., Wang, L., et al. (2023). Sustainable remediation and redevelopment of brownfield sites. *Nat. Rev. Earth Environ.* 4, 4. doi:10.1038/s43017-023-00404-1

Hursthouse, A., and Leitao, T. (2016). "Environmental pressures on and status of urban allotments," in *Urban allotment gardens in Europe*. Editors S. Bell, R. Fox-Kämper, N. Keshavarz, M. Benson, S. Caputo, and S. Noori (Abingdon: Routledge), 142–164.

Ihenetu, S. C., Enyoh, C. E., Wang, C., and Li, G. (2024). Sustainable urbanization and microplastic management: implications for human health and the environment. *Urban Sci.* 8, 252. doi:10.3390/urbansci8040252

Ivashchenko, K., Bekmurzaeva, R., Iriskhanova, Z., Vasenev, V., Sarzhanov, D., Korneykova, M., et al. (2023). "Carbon stocks and fluxes in soils of the urban park in grozny city,". Springer Geography, 159–167. doi:10.1007/978-3-031-37216-2_13*Part F1411*

Jacobs, A., Drouet, T., Sterckeman, T., and Noret, N. (2017). Phytoremediation of urban soils contaminated with trace metals using noccaea caerulescens: comparing non-metallicolous populations to the metallicolous 'ganges' in field trials. *Environ. Sci. Pollut. Res.* 24, 8176–8188. doi:10.1007/s11356-017-8504-9

Jia, L., Liu, L., Zhang, Y., Fu, W., Liu, X., Wang, Q., et al. (2023). Microplastic stress in plants: effects on plant growth and their remediations. *Front. Plant Sci.* 11 (14), 1226484. doi:10.3389/fpls.2023.1226484

Kabisch, N., Frantzeskaki, N., Pauleit, S., Naumann, S., Davis, M., Artmann, M., et al. (2016). Nature-based solutions to climate change mitigation and adaptation in urban areas: perspectives on indicators, knowledge gaps, barriers, and opportunities for action. *Ecol. Soc.* 21 (2), 39. doi:10.5751/ES-08373-210239

Kannan, K., and Vimalkumar, K. (2021). A review of human exposure to microplastics and insights into microplastics as obesogens. *Front. Endocrinol.* 12, 724989. doi:10.3389/fendo.2021.724989

Karvinen, E., Backman, L., Järvi, L., and Kulmala, L. (2024). Soil respiration across a variety of tree-covered urban green spaces in helsinki, Finland. *Soil* 10 (1), 381–406. doi:10.5194/soil-10-381-2024

Kelepertzis, E., and Argyraki, A. (2015). Mercury in the urban topsoil of Athens, Greece. *Sustainability* 7, 4049–4062. doi:10.3390/su7044049

Kortleve, A. J., Mogollón, J. M., Heimovaara, T. J., and Gebert, J. (2023). Topsoil carbon stocks in urban greenspaces of the hague, the Netherlands. *Urban Ecosyst.* 26 (3), 725–742. doi:10.1007/s11252-022-01315-7

Kumar, A., Kumar, A., Cabral-Pinto, M. M. S., Chaturvedi, A. K., Shabnam, A. A., Subrahmanyam, G., et al. (2020a). Lead toxicity: health hazards, influence on food chain, and sustainable remediation approaches. *Int. J. Environ. Res. Public Health* 17 (7), 2179. doi:10.3390/ijerph17072179

Kumar, R. S., Singh, D., Bose, S. K., and Trivedi, P. K. (2020b). "Biodegradation of environmental pollutant through pathways engineering and genetically modified organisms approaches," in *Microorganisms for sustainable environment and health*. Editors P. Chowdhary, A. Raj, D. Verma, and Y. Akhter (Amsterdam: Elsevier), 137–165. doi:10.1016/B978-0-12-819001-2.00007-3

Kuppan, N., Padman, M., Mahadeva, M., Srinivasan, S., and Devarajan, R. (2024). A comprehensive review of sustainable bioremediation techniques: eco-Friendly solutions for waste and pollution management. *Waste manage. Bull.* 2 (3), 154–171. doi:10.1016/j.wmb.2024.07.005

la Cecilia, D., Malvolti, G., Despini, F., Teggi, S., Bigi, A., and Bach, P. (2020). *Int. Conf. Environ. Sci. Technol. Proc.* 18. Urban Heat Island mitigation scenarios by Blue-Green Infrastructures modelling. doi:10.30955/gnc2023.00287

Landa-Acuña, D., Acosta, R. A. S., Hualpa Cutipa, E., Vargas de la Cruz, C., and Luis Alaya, B. (2020). "Bioremediation: a low-cost and clean-green technology for environmental management," in *Microbial bioremediation & biodegradation*. Editor M. Shah (Singapore: Springer). doi:10.1007/978-981-15-1812-6_7

Law, K. L., and Thompson, R. C. (2014). Microplastics in the seas. *Science* 345 (6193), 144–145. doi:10.1126/science.1254065

Leigh, M. B., Prouzová, P., Macková, M., Macek, T., Nagle, D. P., and Fletcher, J. S. (2006). Polychlorinated biphenyl (PCB)-Degrading bacteria associated with trees in a PCB-contaminated site. *Appl. Environ. Microbiol.* 72 (4), 2331–2342. doi:10.1128/AEM.72.4.2331-2342.2006

Leitão, I. A., van Schaik, L., Ferreira, A. J. D., Alexandre, N., and Geissen, V. (2023). The spatial distribution of microplastics in topsoils of an urban environment - coimbra city case-study. *Environ. Res.* 218, 114961. doi:10.1016/j.envres.2022.114961

Li, J., Liu, H., and Chen, J. P. (2018). Microplastics in freshwater systems: a review on occurrence, environmental effects, and methods for microplastics detection. *Water Res.* 137, 362–374. doi:10.1016/j.watres.2017.12.056

Li, S., Yang, Y., Yang, S., Zheng, H., Zheng, Y. M. J., Nagarajan, D., et al. (2023a). Recent advances in biodegradation of emerging contaminants - microplastics (MPs): feasibility, mechanism, and future prospects. *Chemosphere* 331, 138776. doi:10.1016/j.chemosphere.2023.138776

Li, Y., Yang, H., He, W., and Li, Y. (2023b). Human endocrine-disrupting effects of phthalate esters through adverse outcome pathways: a comprehensive mechanism analysis. *Int. J. Mol. Sci.* 24 (17), 13548. doi:10.3390/ijms241713548

Li, W., Xing, Y., Guo, Y., Zhang, D., Tang, Y., Chen, J., et al. (2024). The removal and mitigation effects of biochar on microplastics in water and soils: application and mechanism analysis. *Sustainability* 16, 9749. doi:10.3390/su16229749

Li, Y., Zhang, J., Xu, L., Li, R., Zhang, R., Li, M., et al. (2025). Leaf absorption contributes to accumulation of microplastics in plants. *Nature* 641, 666–673. doi:10.1038/s41586-025-08831-4

Lindén, L., Riikonen, A., Setälä, H., and Yli-Pelkonen, V. (2020). Quantifying carbon stocks in urban parks under cold climate conditions. *Urban For. Urban Green.* 49, 126633. doi:10.1016/j.ufug.2020.126633

Lokesh, P., Shobika, R., SoghraNashath, O., Madhavi, R., Panchamoorthy, S., Rajeshkannan, R., et al. (2023). Bioremediation of plastics by the help of microbial tool: a way for control of plastic pollution. *Sustain. Chem. Environ.* 3, 100027. doi:10.1016/j.scenv.2023.100027

Mackova, M., Prouzova, P., Stursa, P., Ryslava, E., Uhlik, O., Beranova, K., et al. (2009). Phyto/rhizoremediation studies using long-term PCB-contaminated soil. *Environ. Sci. Pollut. Res. Int.* 16 (7), 817–829. doi:10.1007/s11356-009-0240-3

Marinescu, M., Dumitru, M., and Lacatusu, A. (2009). Biodegradation of petroleum hydrocarbons in an artificial polluted soil. *Res. J. Agr. Sci.* 41, 157–162.

Marshall, C. A. M., Wilkinson, M. T., Hadfield, P. M., Rogers, S. M., Shanklin, D., Eversham, B C.., et al. (2023). Urban wildflower meadow planting for biodiversity, climate and society: an evaluation at King's college Cambridge. *Ecol. Sol. Evid.* 4 (1), e12243. doi:10.1002/2688-8319.12243

Meggo, R. E., and Schnoor, J. L. (2013). Rhizospere Redox cycling and implications for rhizosphere biotransformation of selected polychlorinated biphenyl (PCB) congeners. *Ecol. Eng.* 57, 285–292. doi:10.1016/j.ecoleng.2013.04.052

Megharaj, M., and Naidu, R. (2017). Soil and brownfield bioremediation. *Microb. Biotechnol.* 10 (5), 1244–1249. doi:10.1111/1751-7915.12840

Mench, M., Lepp, N., Bert, V., Schwitzguébel, J. P., Gawronski, S. W., Schröder, P., et al. (2010). Successes and limitations of phytotechnologies at field scale: outcomes, assessment and outlook from COST action 859. *J. Soils Sediments* 10, 1039–1070. doi:10.1007/s11368-010-0190-x

Mihalakakou, G., Souliotis, M., Papadaki, M., Menounou, P., Dimopoulos, P., Kolokotsa, D., et al. (2023). Green roofs as a nature-based solution for improving urban sustainability: progress and perspectives. *Renew. Sustain. Energy Rev.* 180, 113306. doi:10.1016/j.rser.2023.113306

Mitchell, M. E., Emilsson, T., and Buffam, I. (2021). Carbon, nitrogen, and phosphorus variation along a green roof chronosequence: implications for green roof ecosystem development. *Ecol. Eng.* 164, 106211. doi:10.1016/j.ecoleng.2021.106211

Moazzem, S., Bhuiyan, M., Muthukumaran, S., Fagan, J., and Jegatheesan, V. (2024). A critical review of nature-based systems (NbS) to treat stormwater in response to climate change and urbanization. *Curr. Pollut. Rep.* 10 (2), 286–311. doi:10.1007/s40726-024-00297-8

Mosisa, G. B., Bedadi, B., Dalle, G., and Tassie, N. (2025). Nature-based solutions for urban climate resilience: implementation, contribution, and effectiveness. *Nature-Based Solutions* 8, 100245. doi:10.1016/j.nbsj.2025.100245

Mulligan, C. N., and Yong, R. N. (2004). Natural attenuation of contaminated soils. *Environ. Int.* 30 (4), 587–601. doi:10.1016/j.envint.2003.11.001

Nesshöver, C., Assmuth, T., Irvine, K. N., Rusch, G. M., Waylen, K. A., Delbaere, B., et al. (2016). The science, policy and practice of nature-based solutions: an interdisciplinary perspective. *Sci. Total Environ.* 579, 1215–1227. doi:10.1016/j.scitotenv.2016.11.106

Ng, E.-L., Huerta Lwanga, E., Eldridge, S. M., Johnston, P., Hu, H.-W., Geissen, V., et al. (2018). An overview of microplastic and nanoplastic pollution in agroecosystems. *Sci. Total Environ.* 627, 1377–1388. doi:10.1016/j.scitotenv.2018.01.341

Nikolić, M., and Stevović, S. (2015). Family asteraceae as a sustainable planning tool in phytoremediation and its relevance in urban areas. *Urban For. Urban Green.* 14 (4), 782–789. doi:10.1016/j.ufug.2015.08.002

Ojuederie, O. B., and Babalola, O. O. (2017). Microbial and plant-assisted bioremediation of heavy metal polluted environments: a review. *Int. J. Environ. Res. Public Health* 14 (12), 1504. doi:10.3390/ijerph14121504

Orta-Ortiz, M. S., and Geneletti, D. (2023). Prioritizing urban nature-based solutions to support scaling-out strategies: a case study in Las Palmas de Gran Canaria. *Environ. Impact Assess. Rev.* 102, 107158. doi:10.1016/j.eiar.2023.107158

Padoan, E., Passarella, I., Prati, M., Bergante, S., Facciotto, G., and Ajmone-Marsan, F. (2020). The suitability of short rotation coppice crops for phytoremediation of urban soils. *Appl. Sci.* 10, 307. doi:10.3390/app10010307

Pandey, B., Pathak, J., Singh, P., Kumar, R., Kumar, A., Kaushik, S., et al. (2023). Microplastics in the ecosystem: an overview on detection, removal, toxicity assessment, and control release. *Water* 15, 51. doi:10.3390/w15010051

Peng, C., Wang, M., and Chen, W. (2016). Spatial analysis of PAHs in soils along an urban–suburban–rural gradient: scale effect, distribution patterns, diffusion and influencing factors. *Sci. Rep.* 6, 37185. doi:10.1038/srep37185

Petrova, S., Velcheva, I., Nikolov, B., Angelov, N., Hristozova, G., Zaprjanova, P., et al. (2022). Nature-based solutions for the sustainable management of urban soils and quality of life improvements. *Land* 11 (4), 569. doi:10.3390/land11040569

Pollard, A. J., Powell, K. D., Harper, F. A., and Smith, J. A. C. (2002). The genetic basis of metal hyperaccumulation in plants. *Crit. Rev. Plant. Sci.* 21, 539–566. doi:10.1080/0735-260291044359

Pulford, I. D., and Watson, C. (2003). Phytoremediation of heavy metal-contaminated land by trees--a review. *Environ. Int.* 29 (4), 529–540. doi:10.1016/S0160-4120(02)00152-6

Qattan, S. Y. A. (2025). Harnessing bacterial consortia for effective bioremediation: targeted removal of heavy metals, hydrocarbons, and persistent pollutants. *Environ. Sci. Eur.* 37, 85. doi:10.1186/s12302-025-01103-y

Rafeeq, H., Afsheen, N., Rafique, S., Arshad, A., Intisar, M., Hussain, A., et al. (2023). Genetically engineered microorganisms for environmental remediation. *Chemosphere* 310, 136751. doi:10.1016/j.chemosphere.2022.136751

Raskin, I., Smith, R. D., and Salt, D. E. (1997). Phytoremediation of metals: using plants to remove pollutants from the environment. *Curr. Opin. Biotechnol.* 8 (2), 221–226. doi:10.1016/S0958-1669(97)80106-1

Ren, X., Zeng, G., Tang, L., Wang, J., Wan, J., Wang, J., et al. (2018). The potential impact on the biodegradation of organic pollutants from composting technology for soil remediation. *Waste Manage* 72, 138–149. doi:10.1016/j.wasman.2017.11.032

Restrepo-Flórez, J. M., Bassi, A., and Thompson, M. R. (2014). Microbial degradation and deterioration of polyethylene–a review. Int. Biodeterior. *Biodegradation* 88, 83–90. doi:10.1016/j.ibiod.2013.12.014

Richter, S., Haase, D., Thestorf, K., and Makki, M. (2020). Carbon pools of Berlin, Germany: organic carbon in soils and aboveground in trees. *Urban For. Urban Green.* 54, 126777. doi:10.1016/j.ufug.2020.126777

Robinson, B. H., Anderson, C. W. N., and Dickinson, N. M. (2015). Phytoextraction: where's the action? *J. Geochem. Explor.* 151 (2015), 34–40. doi:10.1016/j.gexplo.2015.01.001

Rosenfeld, C. E., Chaney, R. L., and Martínez, C. E. (2018). Soil geochemical factors regulate Cd accumulation by metal hyperaccumulating noccaea caerulescen FK Mey in field-contaminated soils. *Sci. Total Environ.* 616, 279–287. doi:10.1016/j.scitotenv.2017.11.016

Russo, A., Escobedo, F. J., Cirella, G. T., and Zerbe, S. (2017). Edible green infrastructure: an approach and review of provisioning ecosystem services and disservices in urban environments. *Agric. Ecosyst. Environ.* 242, 53–66. doi:10.1016/j.agee.2017.03.026

Sajjad, M., Huang, Q., Khan, S., Khan, M. A., Liu, Y., Wang, J., et al. (2022). Microplastics in the soil environment: a critical review. *Environ. Technol. Innov.* 27, 102408. doi:10.1016/j.eti.2022.102408

Schrader, S., and Böning, M. (2006). Soil formation on green roofs and its contribution to urban biodiversity with emphasis on collembolans. *Pedobiologia* 50 (4), 347–356. doi:10.1016/j.pedobi.2006.06.003

Setälä, H. M., Francini, G., Allen, J. A., Hui, N., Jumpponen, A., and Kotze, D. J. (2016). Vegetation type and age drive changes in soil properties, nitrogen, and carbon sequestration in urban parks under cold climate. *Front. Ecol. Evol.* 4. doi:10.3389/fevo.2016.00093

Setälä, H., Francini, G., Allen, J. A., Jumpponen, A., Hui, N., and Kotze, D. J. (2017). Urban parks provide ecosystem services by retaining metals and nutrients in soils. *Environ. Pollut.* 231, 451–461. doi:10.1016/j.envpol.2017.08.010

Shah, V., and Daverey, A. (2020). Phytoremediation: a multidisciplinary approach to clean up heavy metal contaminated soil. *Environ. Technol. Innov.* 18, 100774. doi:10.1016/j.eti.2020.100774

Sharma, P., Bano, A., Singh, S. P., Dubey, N. K., Chandra, R., and Iqbal, H. M. N. (2022). Recent advancements in microbial-assisted remediation strategies for toxic contaminants. *CLCE* 2, 100020. doi:10.1016/j.clce.2022.100020

Shen, Z. G., Zhao, F. J., and McGrath, S. P. (1997). Uptake and transport of zinc in the hyperaccumulator Thlaspi caerulescens and the non-hyperaccumulator Thlaspi ochroleucum. *Plant Cell Environ.* 20, 898–906. doi:10.1046/j.1365-3040.1997.d01-134.x

Silvestrini, F., Fratini, F., and Magini, R. (2021). "Planning with nature: sustainable urban prototypes for portuense district in rome,", 253. Italy. doi:10.2495/SC210021*WIT Trans. Ecol. Environ.*

Sivaperumal, P., Kamala, K., and Rajaram, R. (2017). Bioremediation of industrial waste through enzyme producing marine microorganisms. *Adv. Food Nutr. Res.* 80, 165–179. doi:10.1016/bs.afnr.2016.10.006

Slater, H., Gouin, T., and Leigh, M. B. (2011). Assessing the potential for rhizoremediation of PCB contaminated soils in northern regions using native tree species. *Chemosphere* 84 (2), 199–206. doi:10.1016/j.chemosphere.2011.04.058

Smaranda, C., and Gavrilescu, M. (2008). Migration and fate of persistent organic pollutants in the atmosphere - a modelling approach. *Environ. Eng. Manag. J.* 7, 743–761. doi:10.30638/eemj.2008.100

Snep, R. P. H., Voeten, J. G. W. F., Mol, G., and Van Hattum, T. (2020). Nature based solutions for urban resilience: a distinction between no-tech, low-tech and high-tech solutions. *Front. Environ. Sci.* 8, 599060. doi:10.3389/fenvs.2020.599060

Solomun, M. K., Ilić, Z. H., Kalantari, Z., Eremija, S., Čigoja, I., Ferreira, C., et al. (2024). Phytoremediation by trees as a nature-based solution for mitigating metal contamination in urban soils. *Environ. Sci. Pollut. Res.* 31 (17), 24936–24950. doi:10.1007/s11356-024-32773-5

Somarakis, G., Stagakis, S., and Chrysoulakis, N. (2019). *ThinkNature nature-based solutions handbook. ThinkNature project funded by the EU horizon 2020 research and innovation programme under grant agreement No. 730338.* doi:10.26225/jerv-w202

Song, Y., Kirkwood, N., Maksimović, Č., Zheng, X., O'Connor, D., Jin, Y., et al. (2019). Nature based solutions for contaminated land remediation and brownfield redevelopment in cities: a review. *Sci. Total Environ.* 663, 568–579. doi:10.1016/j.scitotenv.2019.01.347

Stančić, Z., Fiket, Ž., and Vujević, D. (2022). Can urban grassland plants contribute to the phytoremediation of soils contaminated with heavy metals. *Molecules* 27, 6558. doi:10.3390/molecules27196558

Sumathi, K., and Manian, R. (2023). Bioremediation of polycyclic aromatic hydrocarbons contaminated soils: recent progress, perspectives and challenges. *Environ. Monit. Assess.* 195, 1441. doi:10.1007/s10661-023-12042-7

Sutkar Pankaj, R., Gadewar, R. D., and Dhulap Vinayak, P. (2023). Recent trends in degradation of microplastics in the environment: a state-of-the-art review. *J. Hazard. Mater. Adv.* 11, 100343. doi:10.1016/j.hazadv.2023.100343

Sylvestre, M., and Toussaint, J. P. (2011). "Engineering microbial enzymes and plants to promote PCB degradation in soil: current state of knowledge," in *Microbial bioremediation of nonmetals – current research*. Editor A. I. Koukkou (Norfolk (UK): Caister Academic), 177–196.

Tayefi Nasrabadi, M. (2022). How do nature-based solutions contribute to urban landscape sustainability? *Environ. Dev. Sustain.* 24 (1), 576–591. doi:10.1007/s10668-021-01456-3

Temporiti, M. E. E., Nicola, L., Nielsen, E., and Tosi, S. (2022). Fungal enzymes involved in plastics biodegradation. *Microorganisms* 8 (6), 1180. doi:10.3390/microorganisms10061180

Teuten, E. L., Saquing, J. M., Knappe, D. R. U., Barlaz, M. A., Jonsson, S., Bjorn, A., et al. (2009). Transport and release of chemicals from plastics to the environment and to wildlife. *Philos. Trans. R. Soc. Lond. B Biol. Sci.* 364 (1526), 2027–2045. doi:10.1098/rstb.2008.0284

Thapliyal, C., Priya, A., Singh, S. B., Bahuguna, V., and Daverey, A. (2024). Potential strategies for bioremediation of microplastic contaminated soil. *J. Environ. Chem. Ecotoxicol.* 6, 117–131. doi:10.1016/j.enceco.2024.05.001

Thomas, G., Sheridan, C., and Holm, P. E. (2022). A critical review of phytoremediation for acid mine drainage-impacted environments. *Sci. Total Environ.* 811, 152230. doi:10.1016/j.scitotenv.2021.152230

Todeschini, C. C., and Fett-Neto, A. G. (2025). Life at the top: extensive green roof plant species and their traits for urban use. Plants *(Basel)* 14 (5), 735. doi:10.3390/plants14050735

Trémeau, J., Olascoaga, B., Backman, L., Karvinen, E., Vekuri, H., and Kulmala, L. (2024). Lawns and meadows in urban green space - a comparison from perspectives of greenhouse gases, drought resilience and plant functional types. *Biogeosciences* 21 (4), 949–972. doi:10.5194/bg-21-949-2024

Uhlik, O., Musilova, L., Ridl, J., Hroudova, M., Vlcek, C., Koubek, J., et al. (2013). Plant secondary metabolite-induced shifts in bacterial community structure and degradative ability in contaminated soil. *Appl. Microbiol. Biotechnol.* 97 (20), 9245–9256. doi:10.1007/s00253-012-4627-6

United Nations Department of Economic and Social Affairs (UN DESA) (2019). *World urbanization prospects: the 2018 revision*. United Nations: World Economic Situation and Prospects. Available online at: https://population.un.org/wup/.

United Nations Environment Programme (UNEP) (2001). The stockholm convention on persistent organic pollutants (POPs).

URBAN GreenUP (2018). Deliverable 1.1:NBS catalogue. *New Strategy Re-Naturing Cities through Nature-Based Solutions*, 730426. doi:10.3030/730426

Vaksmaa, A., Guerrero-Cruz, S., Ghosh, P., Zeghal, E., Hernando-Morales, V., and Niemann, H. (2023). Role of fungi in bioremediation of emerging pollutants. *Front. Mar. Sci.* 10, 1070905. doi:10.3389/fmars.2023.1070905

Verma, A. K. (2018). Unsustainable agriculture, environmental ethics and ecological balance. *HortFlora Res. Spectr*. 7 (3), 239–241.

Valois, L., Brachet, A., Schiopu, N., and Barot, S. (2023). Performance assessment of the ecosystem services provided by urban Nature-based solutions: focus on rainwater management. *IOP Conf. Ser. Earth Environ. Sci.* 1196 (1), 012028. doi:10.1088/1755-1315/1196/1/012028

van der Ent, A., Baker, A. J. M., Reeves, R. D., Pollard, A. J., and Schat, H. (2013). Hyperaccumulators of metal and metalloid trace elements: facts and fiction. *Plant Soil* 362, 319–334. doi:10.1007/s11104-012-1287-3

Vannucchi, F., Bretzel, F., Pini, R., and Rumble, H. (2021). "Less is more: soil and substrate quality as an opportunity for urban greening and biodiversity conservation," in *Urban services to ecosystems. Future city*. Editors C. Catalano, M. B. Andreucci, R. Guarino, F. Bretzel, M. Leone, and S. Pasta (Cham: Springer), 17, 207–224. doi:10.1007/978-3-030-75929-2_11

Walters, S. A., and Midden, K. S. (2018). Sustainability of urban agriculture: vegetable production on green roofs. *Agriculture* 8 (11), 168. doi:10.3390/agriculture8110168

Wang, L., Rinklebe, J., Tack, F. M. G., and Hou, D. (2021). A review of green remediation strategies for heavy metal contaminated soil. *Soil Use Manag.* 37, 936–963. doi:10.1111/sum.12717

Wei, B., and Yang, L. (2010). A review of heavy metal contaminations in urban soils, urban road dusts and agricultural soils from China. *Microchem. J.* 94 (2), 99–107. doi:10.1016/j.microc.2009.09.014

WHO (2021). *Exposure to lead: a major public health concern*. 2nd edition. Available online at: https://www.who.int/publications/i/item/9789240037656.

Wilschut, M., Theuws, P. A., and Duchhart, I. (2013). Phytoremediative urban design: transforming a derelict and polluted harbour area into a green and productive neighbourhood. *Environ. Pollut.* 183, 81–88. doi:10.1016/j.envpol.2013.01.033

Wu, Q., Huang, Y., Irga, P., Kumar, P., Li, W., Wei, W., et al. (2024). Synergistic control of urban heat island and urban pollution island effects using green infrastructure. *J. Environ. Manage.* 370, 122985. doi:10.1016/j.jenvman.2024.122985

Xiang, L., Harindintwali, J. D., Wang, F., Redmile-Gordon, M., Chang, S. X., Fu, Y., et al. (2022). Integrating biochar, bacteria, and plants for sustainable remediation of soils contaminated with organic pollutants. *Environ. Sci. Technol.* 56 (23), 16546–16566. doi:10.1021/acs.est.2c02976

Xie, L., and Bulkeley, H. (2020). Nature-based solutions for urban biodiversity governance. *Environ. Sci. Policy* 110, 77–87. doi:10.1016/j.envsci.2020.04.002

Xu, L., Teng, Y., Li, Z. G., Norton, J. M., and Luo, Y. M. (2010). Enhanced removal of polychlorinated biphenyls from alfalfa rhizosphere soil in a field study: the impact of a rhizobial inoculum. *Sci. Total Environ.* 408 (5), 1007–1013. doi:10.1016/j.scitotenv.2009.11.031

Yaashikaa, P. R., Senthil Kumar, P., Jeevanantham, S., and Saravanan, R. (2022). A review on bioremediation approach for heavy metal detoxification and accumulation in plants. *Environ. Pollut.* 301, 119035. doi:10.1016/j.envpol.2022.119035

Yuan, J., Ma, J., Sun, Y., Zhou, T., Zhao, Y., and Yu, F. (2020). Microbial degradation and other environmental aspects of microplastics/plastics. *Sci. Total Environ.* 715, 136968. doi:10.1016/j.scitotenv.2020.136968

Zareba, A., Krzeminska, A., and Kozik, R. (2021). Urban vertical farming as an example of nature-based solutions supporting a healthy society living in the urban environment. *Resources* 10 (11), 109. doi:10.3390/resources10110109

Zheng, W., Cui, T., and Li, H. (2022). Combined technologies for the remediation of soils contaminated by organic pollutants A review. *Environ. Chem. Lett.* 20, 2043–2062. doi:10.1007/s10311-022-01407-y