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MICROBE BASED REMEDIATION OF PROBLEMATIC AND POLLUTED SOILS AND ITS CONSEQUENCES IN NUTRIENT CYCLING

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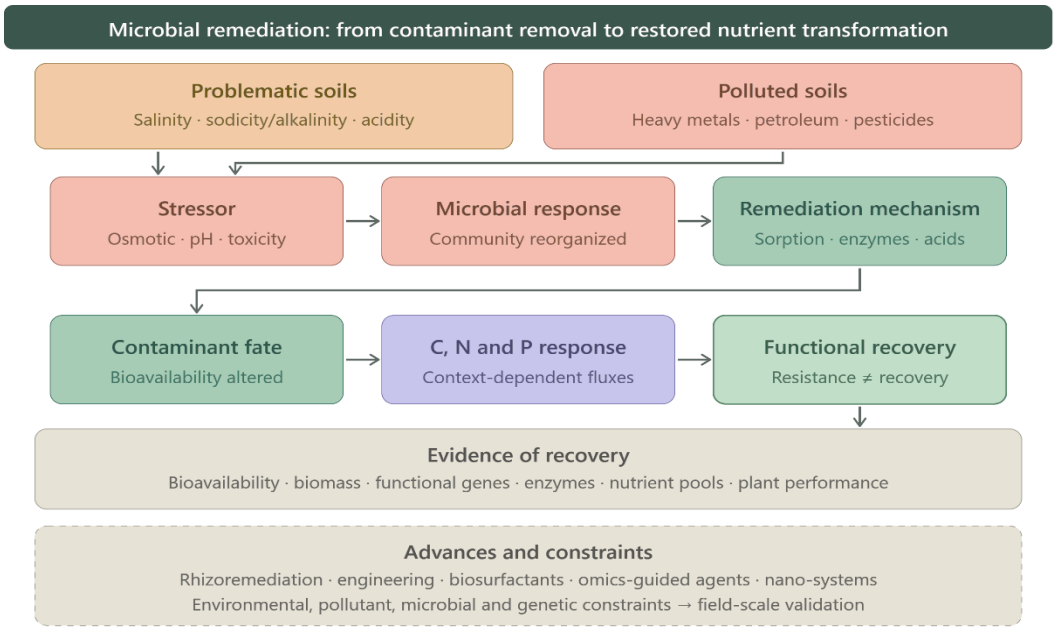
ABSTRACT

Microbe-mediated remediation offers a sustainable route for managing problematic and polluted soils, but removing the contaminant does not, by itself, restore soil function. An integrated framework linking microbial stress responses and contaminant transformation to carbon (C), nitrogen (N), and phosphorus (P) cycling in soils affected by heavy metals, petroleum hydrocarbons, pesticides, salinity, alkalinity, and acidity is developed in this review. Key microbial mechanisms—biosorption, bioaccumulation, biomineralization, enzymatic degradation, redox transformation, organic-acid production, siderophore activity, and extracellular polymeric substance formation—are evaluated according to how they alter contaminant bioavailability and the microbial processes that govern nutrient fluxes. It is shown by the comparative synthesis that the link between remediation and nutrient recovery is context dependent: microbial C demand can be raised, and N and P transiently immobilized, by hydrocarbon degradation; N- and P-cycling functions can be reorganized by metal stress; and nutrient availability is improved by correcting pH or salinity only when microbial functional capacity is also restored. Microbial resistance is distinguished from functional recovery, and a complementary set of indicators is proposed: microbial biomass, functional genes, enzyme activities, nutrient pools, contaminant bioavailability, and plant performance. Recent advances—rhizoremediation, microbial consortia, biosurfactants, genome engineering, omics-guided selection, and nano-enabled systems—are assessed for their potential and limitations in restoring soil function. Finally, key knowledge gaps in connecting contaminant fate to process-level C, N, and P recovery are identified, and the need for field-scale validation under realistic soil conditions is stressed.

KEYWORDS: Bioremediation; microbial nutrient transformation; problematic soil; petroleum hydrocarbons; heavy metals; functional recovery; nutrient cycling.

Graphical abstract



1. INTRODUCTION

Soil degradation and pollution are critical constraints on global food security and environmental sustainability. Approximately 33% of the world's soils are moderately to highly degraded by erosion, salinization, compaction, acidification, and chemical pollution (FAO 2015). Two broad categories of affected soils are considered here (Fig. 1). Problematic soils are characterized mainly by physicochemical constraints—salinity, sodicity, acidity, and alkalinity—by which plant growth and soil functioning are restricted, whereas polluted soils are contaminated with heavy metals, petroleum hydrocarbons, pesticides, and other chemicals at concentrations at which ecological or health risks may be posed. Although these categories differ in origin and characteristics, they often overlap in practice, and soil microbial communities can be substantially altered, and key biogeochemical processes disrupted, by both. For instance, microbial community composition, enzyme activities, soil organic carbon dynamics, and carbon and nitrogen cycling are affected by soil salinity (Haj-Amor et al. 2022). Similarly, microbial carbon fixation and functional pathways associated with nutrient cycling, including nitrogen and sulfur cycling, can be altered by long-term heavy-metal contamination (Ma et al. 2022).

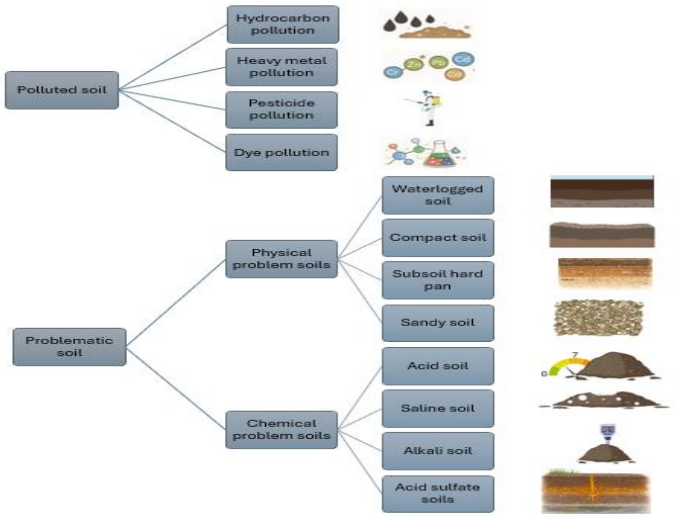
A functional link between remediation and nutrient transformation is provided by soil microorganisms. Organic matter decomposition, nutrient mineralization and immobilization, nitrogen transformations, and phosphorus mobilization are regulated by them, and nutrient availability and soil fertility are thereby influenced. In heavy metal contaminated soils, microbial nutrient acquisition and carbon decomposition are altered by metal stress, as reflected in changes in carbon, nitrogen, and phosphorus acquiring enzyme activities and in microbial nutrient limitation (Duan et al. 2022). In petroleum contaminated soils, nutrient availability is closely tied to biodegradation because nitrogen and phosphorus are required for microbial growth and hydrocarbon metabolism, bioremediation efficiency can therefore be influenced by nutrient amendments (Kundu et al. 2022). Together, these findings show that not only microbial pollutant transformation but also the processes that sustain nutrient cycling and soil fertility are affected by soil constraints and contaminant stress.

Microbe mediated mitigation should therefore not be judged solely by contaminant removal or relief of physicochemical stress. Its effectiveness should also be assessed through microbial biomass, nutrient transforming microbial groups, functional genes, soil enzyme activities, nutrient availability, and broader biogeochemical functions. Contaminants are detoxified by microorganisms through biosorption, bioaccumulation, enzymatic degradation, biomineralization, and precipitation, while nutrient cycling and soil recovery are influenced by the same metabolic activities (Gadd 2010; Chicca et al. 2022). A more complete framework for determining whether contaminant concentrations are merely lowered by microbial interventions, or soil fertility and ecosystem functioning are also restored, is therefore provided by integrating contaminant remediation with microbial carbon (C), nitrogen (N), phosphorus (P), and other nutrient transformations.

1.1. Scope and distinctive perspective

Microbial remediation is evaluated in most studies through contaminant disappearance, detoxification, or plant-growth responses. These endpoints are important, but recovery of the microbial processes that sustain soil fertility is not necessarily demonstrated by them. The distinctive focus of this review is therefore the coupling between remediation mechanisms and nutrient transformation. For example, a strong microbial demand for N and P is created by hydrocarbon degradation; N and P transforming functions can be suppressed or reorganized by metal stress; and pH, contaminant mobility, and P availability can be simultaneously changed by microbial acidification or organic-acid production. Resistance, in which microorganisms remain active under stress, is also distinguished by the framework from recovery, in which community structure and process-level nutrient functions are restored once stress is reduced. Microbial strategies can be compared using common functional endpoints when the same analytical logic is applied to every soil system.

Fig. 1 Conceptual classification of polluted and problematic soils



2. Problematic soil

Problematic soils are those in which plant growth, nutrient availability, and microbial functioning are restricted by physicochemical constraints and for which corrective management is therefore required. Constraints with direct consequences for microbial processes—salinity, sodicity/alkalinity, and acidity—are emphasized in this review (Weil and Brady 2016). Osmotic conditions, pH, nutrient solubility, contaminant mobility, and microbial community structure are altered by these constraints, and nutrient transformation and soil fertility are thereby affected.

2.1. Saline soil

High concentrations of soluble salts, primarily sodium chloride (NaCl), are present in saline soils, and plant growth and soil health are severely affected by them. The dominant cations are sodium (Na^+), magnesium (Mg^{2+}), and calcium (Ca^{2+}), with potassium (K^+) less common, and the dominant anions are chloride (Cl^-), bicarbonate (HCO_3^-), and sulfate (SO_4^{2-}). These soils are characterized by an electrical conductivity (EC) greater than 4 dS m^{-1}, an exchangeable sodium percentage (ESP) below 15, a pH below 8.5, and white salt crusts that are often visible at the surface (Weil and Brady 2016); for this reason, they were formerly called "white alkali soils". Salts accumulate through natural processes such as mineral weathering and evaporation, or through human activities such as improper irrigation and poor drainage (Haj-Amor et al. 2022; FAO 2015).

2.2. Alkaline and sodic soils

Soil alkalinity arises from the natural weathering of minerals (e.g., silicates, aluminosilicates, carbonates) and from human activities such as irrigation and liming. OH^- ions are released through hydrolysis during weathering of minerals containing Na^+, Mg^{2+}, K^+, and Ca^{2+}, and soil pH is thereby raised, while alkalinity is further increased by irrigation with bicarbonate-rich water. Alkaline soils are rich in carbonates (CO_3^{2-}) and bicarbonates (HCO_3^-), by which acidity is buffered and pH is elevated.

Alkalization occurs when exchangeable sodium accumulates on soil particles through cation exchange. Cations are adsorbed by soil colloids, with the exchange complex normally dominated by calcium and magnesium. Under saline conditions, the sodium concentration in the soil solution is raised by evaporation and the precipitation of calcium and magnesium salts; although calcium and magnesium are preferentially adsorbed, they are replaced by sodium on exchange sites once it makes up a major proportion of the soluble cations, leading to sodicity (Tóth 2022).

Saline–alkali soils result from the combined processes of salinization and alkalization and are characterized by an electrical conductivity of the saturation extract above 4 dS m^{-1} at 25 °C and an ESP above 15. While excess soluble salts remain, they resemble saline soils in appearance and behaviour, with pH values usually below 8.5 and flocculated soil particles. When the soluble salts are leached, however, their properties change substantially and they come to resemble nonsaline alkali soils. Hydrolysis of exchangeable sodium is promoted by the lower salt concentration, with sodium hydroxide and then sodium carbonate formed, by which soil pH is raised above 8.5.

Nonsaline–alkali soils have an ESP above 15, an EC below 4 dS m^{-1}, and a pH of 8.5–10. Commonly known as "black alkali" soils, they occur mainly in arid and semi-arid regions, often in irregular patches, and frequently develop when soluble salts are leached from saline–alkali soils and soil pH is raised by sodium hydrolysis. Clay dispersion, low permeability, and poor tilth are caused by high exchangeable sodium, making these soils difficult to manage. The soil may be darkened by organic matter deposited at the surface, while calcium and magnesium are precipitated, and soil physical and chemical properties further degraded, by sodium dominance in the soil solution (Rezapour et al. 2017).

2.3. Acid soil

Acid soils have a pH below 5.5 and are commonly associated with metal toxicities, particularly aluminium, and with nutrient deficiencies such as molybdenum, by both of which plant growth is restricted. Acid soils are widespread in humid tropical regions, and nearly 30% of the world's ice-free land area is affected by them (von Uexküll and Mutert 1995).

Soil acidification is driven by both natural and anthropogenic processes. In humid regions, base cations such as Ca^{2+}, Mg^{2+}, and K^+ are leached by heavy rainfall and replaced by H^+ and Al^{3+}, while acidity is further increased by mineral weathering and aluminium hydrolysis. Strong acids, by which soil pH is sharply lowered, are produced by oxidation of organic matter and of sulfide minerals, particularly pyrite, and H^+ can also be released into the soil through plant uptake. These processes are intensified by human activities through acid rain from fossil fuel combustion, ammonium-based fertilizers that generate H^+ during nitrification, and intensive agriculture or industrial operations that deplete base cations.

Acidification has widespread effects on soil properties, plant health, microbial communities, and water quality. Leaching of base cations such as calcium and magnesium is enhanced by a decline in pH, while the solubility of toxic metals such as aluminium and manganese is increased, resulting in poor soil structure, reduced infiltration, and root damage. Nutrient availability is disrupted, with deficiencies of major nutrients and, in some cases, toxic accumulation of micronutrients, leading to impaired plant growth, reduced photosynthesis, and yield losses. Beneficial soil microorganisms are also suppressed by acidic conditions, disrupting nutrient cycling and soil biological balance. In addition, greater metal mobility at low pH contributes to soil and water pollution, and risks are posed to ecosystems and human health (Weil and Brady 2016).

3. Polluted soil

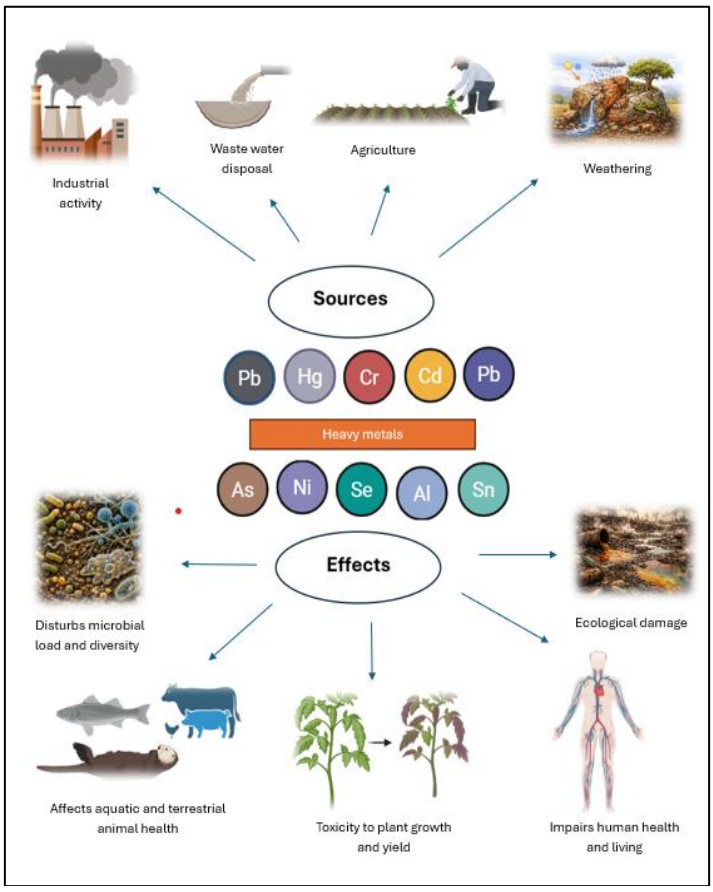
Soil pollution is a form of chemical degradation by which soil fertility is progressively diminished and serious risks are posed to global food security and human health. It occurs when natural background levels are exceeded by contaminant concentrations, so that soil quality is degraded and human health and ecosystems are threatened—particularly for people and wildlife living near former industrial sites and reclaimed lands where remediation has been insufficient or poorly implemented. Contaminants arise from both anthropogenic and natural sources (Fig. 1), including unsustainable agricultural practices, poor irrigation management, improper disposal of hazardous chemicals and nuclear wastes, and industrial, military, and extractive operations. In agricultural soils, contaminants accumulate in the surface layer and move downward into the subsurface. Surface soils are mainly affected by industrial effluents and solid wastes, by which the natural soil profile can be disrupted, whereas contaminant build-up in topsoil and infiltration into deeper layers result from urban activities (Weil and Brady 2016).

3.1. Heavy metal pollution

Ecological and biological risks are posed at elevated concentrations by potentially toxic elements such as chromium (Cr), cadmium (Cd), lead (Pb), mercury (Hg), nickel (Ni), copper (Cu), and zinc (Zn), together with metalloids such as arsenic (As) (Fig. 2). Cu, Zn, and Ni are essential micronutrients at appropriate levels but become toxic when their bioavailable concentrations are exceeded. Metal toxicity cause disruption of protein and enzyme function, oxidative stress, membrane damage, and interference with nutrient acquisition and cellular metabolism; its magnitude depends on metal speciation, bioavailability, concentration, and soil properties (Popoola et al. 2024).

Unlike organic pollutants, heavy metals (HMs) are non-biodegradable and persist in the environment for long periods. They bioaccumulate in living organisms and, for some metals (e.g., methylmercury), biomagnify through the food chain, with serious risks thereby being posed to ecosystems and human health (Gadd 2010; Popoola et al. 2024). Microbial enzymes involved in nutrient cycling, including dehydrogenase, urease, and phosphatase, are inhibited by metals such as cadmium and lead. Microbial survival is reduced and community structure is altered by prolonged exposure, so that microbial abundance, biomass, and diversity are decreased. Microbial respiration and enzymatic activity are suppressed by toxic levels of Pb, Cd, or Cr, so that mineralization rates are reduced, whereas mineralization may be transiently enhanced by low concentrations of some metals through stimulation of microbial stress responses or enzyme production. Ammonification and nitrification are particularly sensitive to HM toxicity, by which microbial community structure and enzyme function are altered(Popoola et al. 2024)

Fig. 2 Origin and ecological consequences of heavy metal contamination



3.2. Petroleum hydrocarbons

Petroleum hydrocarbons (PHCs) are a chemically diverse group of organic pollutant compounds composed mainly of carbon and hydrogen and derived from crude oil and its refined products. They include alkanes, aromatics, resins, and asphaltenes, together with volatile fractions and polycyclic aromatic hydrocarbons (PAHs), which differ in volatility, solubility, and toxicity. Soil can be infiltrated by oil spills to depths of 10–30 cm or more, so that soil quality and fertility are degraded (Zabbey et al. 2017). Once in soil, PHCs are subjected to volatilization, photo-oxidation, dissolution, leaching, adsorption, and biodegradation (Fig. 3). Soil health, microbial diversity, and human well-being are threatened by their persistence (Kakde and Sharma 2024).

Petroleum products are essential energy sources and raw materials in industrial, agricultural, and domestic sectors, but significant soil and water contamination has been caused by their widespread use through accidental spills, leakage, and improper disposal (Kakde and Sharma 2024). Soil structure is disrupted, fertility is reduced, and carcinogenic compounds such as PAHs are introduced into the tissues of organisms by PHC contamination. Soil enzyme activities and microbial biomass carbon are among the properties most strongly affected. In subantarctic soils, for example, species richness, evenness, and phylogenetic diversity were decreased by diesel exposure, and dominance by *Pseudomonas* spp. was favoured (van Dorst et al. 2014).

3.3. Pesticides

Pesticides are a broad class of chemicals used to eliminate or suppress pests, including insects, rodents, fungi, and weeds. They can be classified by chemical group (organochlorines, organophosphates, carbamates, and pyrethroids) or by target class (insecticides, herbicides, fungicides, nematicides, molluscicides, and rodenticides) (Gavrilescu 2005; Kumar et al. 2026). Although crop protection is supported by their widespread use, residues may persist depending on compound properties and soil conditions. Pesticide fate in soil is governed by adsorption, desorption, chemical and microbial transformation, and movement into water and other environmental compartments. Microbial remediation therefore depends strongly on the chemical structure of the pesticide, the metabolic capacity of the microbial community, and the persistence and toxicity of transformation products (Chia et al. 2024; Kumar et al. 2026).

Microbial community composition, enzyme activities, and nutrient cycling can be altered by pesticide contamination, while parent compounds can be transformed by microbial degradation through hydrolysis, oxidation–reduction, dehalogenation, or extracellular enzymatic reactions. Remediation should therefore be assessed by both the removal of the parent pesticide and the persistence or toxicity of its transformation products, rather than by parent-compound disappearance alone (Liu et al. 2025)

Table 1 Soil stressors and nutrient transformation

The major problematic and polluted soil types, their key characteristics, causes, and specific impacts on microbial nutrient transformation processes such as N-cycling, P-solubilization, and organic matter decomposition are summarized in Table 1.

Soil type	Causes	Impact on nutrient transformation	References
Acid soil • Low pH (<5.5), high solubility of toxic metals (Al^{3+}, Fe^{3+}), low base saturation.	Heavy rainfall, leaching, acid rain, continuous use of ammonium fertilizers.	• Phosphate becomes fixed (Al–P, Fe–P); P-solubilizing microbes decline. • Nitrifiers and N-fixers suppressed due to acidic stress. • Reduced microbial biomass and enzyme activity.	(Weil and Brady 2016)

Saline soil • High electrical conductivity (EC >4 dS/m), excess soluble salts (NaCl, Na_2SO_4).	Poor drainage, seawater intrusion, excessive irrigation, arid climates.	• Reduced nitrogen mineralization due to osmotic stress. • Salt-sensitive phosphate solubilizers become inactive. • Decline in overall microbial activity and enzyme expression.	(Egamberdieva et al. 2019)
Sodic soil • High pH (>8.5), high exchangeable sodium percentage (ESP),	Weathering of Na-rich parent materials, poor-quality irrigation water, industrial effluents.	• P becomes unavailable as Ca–P complexes. • Reduced microbial respiration and biomass. • Lower decomposition and organic matter turnover.	(Qadir and Oster 2004)
Heavy metal contamination	Industrial discharge, mining, electroplating, tannery waste.	• Inhibition of N-fixation, nitrification, and P-solubilization. • Microbial enzymes are suppressed by metal toxicity. • Reduced soil fertility and organic matter breakdown.	(Popoola et al. 2024)
Hydrocarbon contamination	Oil spills, petrochemical waste, leakage from storage tanks.	• Nitrification is restricted by anaerobic pockets. • Soil particles are coated by hydrocarbons, reducing microbial access to nutrients. • Disrupted N and C cycling.	(Labud et al. 2007)
Pesticide contamination	Agricultural application of agrochemicals; persistence of residues and transformation products.	• Altered microbial abundance, community composition and C , N and P cycling enzyme activities • sensitive microbial groups suppressed, affecting decomposition, N mineralization and P mobilization.	(Chia et al. 2024; Kumar et al. 2026)

4. Microbiome function and mechanistic coupling between remediation and nutrient transformation

The soil microbiome is altered by pollution and physicochemical constraints through changes in osmotic pressure, pH, redox potential, substrate availability, metal bioavailability, and toxicity. The response is not simply a loss of microorganisms but community composition, metabolic allocation, and functional redundancy are reorganized. Stress-tolerant taxa may increase in relative abundance while sensitive taxa and functions decline, so taxonomic resistance should not be equated with recovery of soil function (Zhao et al. 2024). Microbial community structure should therefore be connected, in a functional assessment, with process-level measurements of C turnover, N transformation, P mobilization, enzyme activity, and nutrient availability.

Four mechanistic links are particularly important. First, microbial exposure is controlled by contaminant bioavailability: exposure to toxic elements can be reduced by biosorption, extracellular polymeric substances (EPS), mineral precipitation, complexation, and pH changes, whereas contaminant solubility and mobility can be increased by microbial acidification or redox reactions. Second, contaminant transformation competes with, or interacts with, central metabolism: C, reducing power, and nutrients may be redirected by microorganisms toward stress defence or contaminant degradation, so that investment in nutrient acquisition is potentially lowered. Third, contaminant transformation is coupled to Fe, Mn, S, and N cycling by microbial redox reactions, and the mineral phases that retain both nutrients and contaminants can be altered by them. Fourth, carbon substrates are supplied to rhizosphere microorganisms through plant–microbe interactions, while plant nutrient acquisition and stress tolerance are improved by microbial metabolites (Narayanan and Ma 2023; Fetsiukh et al. 2024).

The effects on C cycling differ among contaminants. Abundant organic C but little N or P is supplied by petroleum hydrocarbons, creating a high substrate C:N:P ratio and hence a strong microbial demand for N and P, by which C turnover can be increased while N and P limitation is induced. In contrast, decomposition and extracellular enzyme activity can be suppressed by metal or pesticide stress, although selected pathways may be sustained by tolerant or specialized populations. An increase in respiration or hydrocarbon degradation is therefore not equivalent to restoration of balanced C cycling. Measurements of microbial biomass C, soil organic C, respiration, and C-acquiring enzymes are most informative when interpreted together.

N fixation can be suppressed by salinity while some denitrification pathways persist, indicating functional reorganization rather than uniform inhibition (Li et al. 2024). During petroleum biodegradation, degradation may be stimulated by added N where N is limiting, but microbial allocation can be altered and efficiency reduced by excessive amendment; isotope-tracing evidence shows that both hydrocarbon removal and microbial N utilization are changed by N supply (Liu et al. 2023). N recovery should therefore be assessed using functional genes or process rates together with available inorganic N, rather than inferred from contaminant removal alone.

P cycling is strongly controlled by pH, mineral interactions, and microbial phosphatase and solubilization activity. P fixation by Fe and Al minerals can be increased in acidic soils, whereas P availability can be reduced by alkaline and saline–sodic conditions through precipitation and sorption. These constraints can be relaxed by microbial organic acids, phosphatases, and mineral-solubilizing activity. The same principle applies to metal-contaminated soils, where P-solubilizing populations may be allowed to recover as toxicity is reduced, but P sorption and mineral equilibria may also be altered.

Why contaminant removal and nutrient-functional recovery can follow different trajectories is explained by these interactions. Contaminant concentration may be rapidly reduced by a treatment while functional genes, microbial biomass, or nutrient pools remain depressed; conversely, nutrient transformation may be maintained by microbial adaptation before substantial contaminant removal occurs. Evidence for recovery is strongest when contaminant bioavailability, microbial biomass or community function, process-level nutrient measurements, and plant or soil-performance indicators change consistently in the same direction.

Table 2 Microbial indicators of soil health in polluted and problematic soils

Key microbial indicators of soil health, their diagnostic significance under pollution stress, and their implications for microbially mediated nutrient cycling in contaminated soils are summarized in Table 2.

Soil indicators	Indications	References
Microbial biomass carbon (MBC)	• MBC often decreases in heavy-metal-polluted soils and at high hydrocarbon concentrations because of microbial toxicity, although moderate hydrocarbon loads can transiently raise MBC as degraders proliferate. • Reduced microbial activity, impaired carbon turnover, and suppressed soil functioning are indicated by low MBC.	(Anderson 2003)
Enzyme activities	• Dehydrogenase activity often declines in heavily petroleum-polluted soils because of metabolic inhibition, but responses depend on contamination level and soil type. • Urease and phosphatase activity is reduced in metal-polluted soils because active sites are bound by metals.	(Daunoras et al. 2024; Labud et al. 2007)

Microbial diversity	• Microbial diversity responses are context dependent. Richness, evenness and phylogenetic diversity can be reduced, and tolerant taxa selected for, by diesel contamination, with consequences for nitrification.	(van Dorst et al. 2014)
Functional gene abundance (*amoA, nirK, phoD, alkB*, etc.)	• Functional-gene responses are context dependent. Hydrocarbon degradation genes such as alkB may increase under hydrocarbon exposure, whereas genes associated with nitrification or other nutrient processes may decline under toxic metal or salinity stress. Gene abundance should be interpreted with process rates and nutrient pools.	(Yang et al. 2018; Bai et al. 2023; Li et al. 2024)
Stress-responsive microbial groups	• Contaminant pressure and natural attenuation potential can be indicated by an increase in metal-tolerant or hydrocarbon-degrading bacteria. • Examples include genera like *Acinetobacter*, *Sphingomonas*, *Rhodococcus*, *Burkholderia*.	(Gadd 2010; Mekonnen et al. 2024)
Microbial respiration / basal CO_2 evolution	• Metabolic inhibition due to pollutants is indicated by a reduction in respiration. • Stress-induced over-respiration or depletion of soil organic matter may be suggested by an abnormal increase.	(Doran and Zeiss 2000)

5. Remediation

5.1. Salinity

Physical management of saline soils is aimed at reducing salt concentration in the root zone and improving soil permeability. Infiltration is enhanced, and downward movement of salts promoted, by scraping, subsoiling, and deep ploughing. Uniform water distribution is ensured, and uneven salt accumulation prevented, by land levelling, and percolation is improved by sand mixing in heavy clay soils. Leaching is the primary reclamation method, by which soluble salts are removed below the root zone, while evaporation and upward salt movement are reduced by mulching, thereby supporting better soil conditions and crop productivity (Qadir and Oster 2004).

Chemical management mainly involves gypsum application and nutrient supplementation. Gypsum is particularly effective in saline–sodic soils, where exchangeable sodium is replaced by calcium, improving soil structure and allowing sodium to be leached. Salt stress is alleviated by nutrient management using NPK, magnesium, and growth regulators through correction of ionic imbalances. Together, plant tolerance, soil condition, and crop productivity in saline environments are improved by these practices (Qadir and Oster 2004; Kumar et al. 2020).

5.1.1. Microbial intervention

Microbial mitigation of salinity relies primarily on halotolerant plant growth-promoting rhizobacteria (PGPR), arbuscular mycorrhizal fungi (AMF), and beneficial endophytic fungi that remain active under osmotic and ionic stress. Their mechanisms include EPS production, 1-aminocyclopropane-1-carboxylate (ACC) deaminase activity, phytohormone production, phosphate and potassium solubilization, improved root colonization, and modulation of plant ion homeostasis. In a controlled wheat study, several biological constraints imposed by salinity were alleviated by native PGPR, AMF, and *Serendipita indica* (formerly *Piriformospora indica*) with particularly strong improvements in microbial biomass and phosphatase activity produced by PGPR (Rouydel et al. 2021). Salt stress can thus be buffered by inoculants, but their contribution to soil functional recovery depends on their persistence and compatibility with the resident microbiome.

5.1.2. Nutrient transformation and soil-functional implications

Microbial nutrient transformation is impaired by soil salinity through the imposition of osmotic and ionic stress, by which microbial biomass, metabolic activity, and the functional pathways involved in C, N, and P cycling are altered. It has been shown by metagenomic analysis that several microbial processes, including C degradation, N fixation, and P uptake and transport, can be suppressed by increasing salinity, while other pathways, such as denitrification and organic P dissolution, may be maintained or even enhanced (Li et al. 2024). A uniform decline in nutrient cycling is therefore not caused by salinity; rather, a functional reorganization of the soil microbiome, dependent on the degree of salinity and on nutrient status, is driven by it. Salinity indicators (EC and exchangeable Na^+) should therefore be paired with microbial biomass, phosphatase activity, and N- and P-cycling genes as evidence of recovery in saline soils, rather than plant growth alone being relied upon.

5.2. Alkalinity and sodicity

Sodic and saline–sodic soils are managed primarily by gypsum-mediated replacement of exchangeable Na^+ by Ca^{2+}, followed by leaching.

5.2.1. Microbial intervention

A mechanistically distinct approach to alkaline and saline–alkali soils is offered by sulfur-oxidizing bacteria. When reduced sulfur compounds are oxidized to sulfuric acid, acidity is generated by which pH can be lowered and the solubility of poorly available nutrient forms increased. Improved soil quality after sulfur-oxidizer treatment was reported by Bao et al. (2016). The effect is inherently site dependent, because the magnitude and persistence of acidification are determined by sulfur availability, buffering capacity, moisture, and microbial establishment.

5.2.2. Nutrient transformation and soil-functional implications

The availability of phosphorus and several micronutrients can be reduced by high soil pH through the promotion of precipitation and adsorption reactions, and microbial activity and nutrient transformation can also be restricted by alkaline and saline–alkali conditions. These constraints can be partly alleviated by microbial sulfur oxidation through the production of acidity and modification of the soil chemical environment, whereby the availability of nutrients that are poorly soluble under strongly alkaline conditions is increased. Nutrients can also be mobilized by plant-associated microorganisms through organic acid production, mineral solubilization, and decomposition of soil organic matter. Potential recovery of microbial processes involved in C, N, and P cycling is further suggested by the enrichment of beneficial microbial taxa observed during plant-assisted remediation (Cui et al. 2025). Soil pH and the availability of P and micronutrients should therefore be combined with microbial biomass, enzyme activities, and C, N and P cycling functions as evidence of recovery in alkaline and saline–alkali soils, rather than pH correction alone being relied upon.

5.3. Acid soil

5.3.1. Microbial intervention

In acidic soils, microbial remediation can act through local alkalinization, organic matter transformation, mineral solubilization, and interactions with metal bearing phases. Alkaline metabolites associated with acid neutralization were produced by indigenous consortia containing *Serratia grimesii*, *Citrobacter youngae*, *Pseudescherichia vulneris*, and *Pseudomonas protegens*, and individual isolates were outperformed by the consortia under the tested conditions (Cho et al. 2022), note that these isolates came from abandoned mine areas, so extrapolation to agricultural acid soils requires caution. Exploitation of complementary metabolic functions, rather than reliance on a single isolate, is favoured by this result, but field-scale persistence and effects on metal mobility still need to be validated.

5.3.2. Nutrient transformation and soil-functional implications

Nutrient transformation is strongly influenced by low soil pH through effects on microbial growth, enzyme activity, nutrient solubility, and the composition of nutrient-transforming communities. N cycling is particularly sensitive to acidity because nitrifying microorganisms can be suppressed by low pH and the balance among mineralization, nitrification, and other N transforming processes shifted. Conversely, local pH can be raised and N availability increased by microbial production of ammonium during urea hydrolysis and amino acid decarboxylation, providing a direct link between microbial acid tolerance and nutrient transformation (Cho et al. 2022). P availability is also affected by soil acidity through increased P fixation by iron and aluminium minerals, by which plant-available P is limited despite relatively high total soil P; these constraints may be partly offset, and P mobilization improved, by microbial organic acid production and mineral solubilization. pH and Al^{3+} availability should likewise be combined with available P, nitrifier and other N-cycling functions, and microbial biomass as evidence of recovery in acid soils, rather than acid neutralization alone being relied upon.

5.4. Heavy metals

The ecosystem is disturbed and natural habitats are destroyed by conventional techniques for heavy metal remediation soil washing, encapsulation, solidification, excavation, and landfilling (Raffa et al. 2021). Biochar amendment is a potentially more convenient and economical alternative: metals are adsorbed by it, and soil structure, nutrient retention, and microbial activity can be improved, although its effectiveness varies with feedstock, pyrolysis conditions, soil type, and crop species (Wang et al. 2022b).

5.4.1. Microbial intervention

Microbial intervention in heavy metal contaminated soils operates mainly through biosorption, bioaccumulation, biomineralization, redox transformation, and changes in metal speciation or bioavailability; siderophores and organic acids further modify speciation by chelating metals and Fe-associated nutrients (Gadd 2010). Metal ions are bound through complexation and ion exchange by cell-surface functional groups, extracellular polymeric substances, and mineral-associated microbial products. Biosorption is rapid and largely independent of metabolism, whereas intracellular uptake and biotransformation are metabolically regulated. The practical consequence is important: immediate metal exposure is reduced by immobilization, and nutrient transforming microorganisms can be protected, whereas bioleaching or acidification, which deliberately increase metal solubility, are better suited to metal recovery than to in situ immobilization (Gadd 2010; Raffa et al. 2021).

In bioleaching, mineral phases are dissolved and metals mobilized for recovery through microbial oxidation of iron or sulfur and the production of organic acids (Gadd 2010). Microbial remediation is thus not synonymous with immobilization: the preferred mechanism depends on whether the objective is risk reduction through stabilization or contaminant removal through mobilization and recovery.

A metal's oxidation state or chemical speciation is changed by biotransformation, and its toxicity and bioavailability can therefore be altered. Chromium toxicity, for example, is decreased by reduction of Cr(VI) to Cr(III), whereas metal mobility may be increased by other redox reactions depending on the element and soil conditions (Abo-Alkasem et al. 2023). This context dependence is central to evaluating nutrient recovery, because metal exposure and the availability of Fe, P, or S associated nutrient pools can be simultaneously modified by the same change in redox state.

5.4.2. Nutrient transformation and soil-functional implications

C, N, and P cycling can be disrupted by heavy metal contamination through alteration of microbial community composition, enzyme activity, nutrient limitation, and the chemical availability of nutrient pools. Chromium

stress, for example, has been associated with changes in *nifH, amoA, narG,* and *nirK*, linking metal exposure directly to reorganization of the N cycle (Bai et al. 2023). Across metal contaminated soils, the magnitude and direction of these responses depend on metal speciation, concentration, soil pH, organic matter, texture, and fertilization regime (Duan et al. 2022; Shi et al. 2024; Tian et al. 2025). The key point is that the balance among microbial growth, stress tolerance, contaminant transformation, and nutrient acquisition is shifted by metal stress rather than all microbial functions being simply suppressed.

Functional recovery can be supported by microbial remediation when the bioavailable metal fraction is lowered, and nutrient transforming populations are allowed to re-establish. Immobilization and removal, however, have different implications: toxicity can be rapidly reduced by immobilization without the metal being extracted, whereas the total soil metal concentration can be lowered by bioleaching while dissolved-metal exposure is temporarily increased. C:N:P stoichiometry and metal resistance may be further modified by fertilization, so outcomes depend on treatment context (Tian et al. 2025). Metal bioavailability should therefore be combined with microbial biomass, N and P cycling genes, enzyme activities, and available nutrients as evidence of successful remediation, rather than total-metal concentration alone being relied upon.

5.5. Petroleum hydrocarbons

Physical and chemical approaches are the most frequently used for petroleum hydrocarbon removal and focus on stabilizing the soil and oxidizing the contaminant (Kakde and Sharma 2024). The processes governing hydrocarbon fate in soil are summarized in Fig. 3. Plants, microorganisms, and their synergistic interactions are used in biological remediation to degrade or immobilize PHCs in an environmentally sustainable way. Phytoremediation operates through phytodegradation, phytostabilization, phytoextraction, and rhizosphere stimulation, although its effectiveness is limited by plant tolerance, contaminant bioavailability, and soil properties.

5.5.1. Microbial intervention

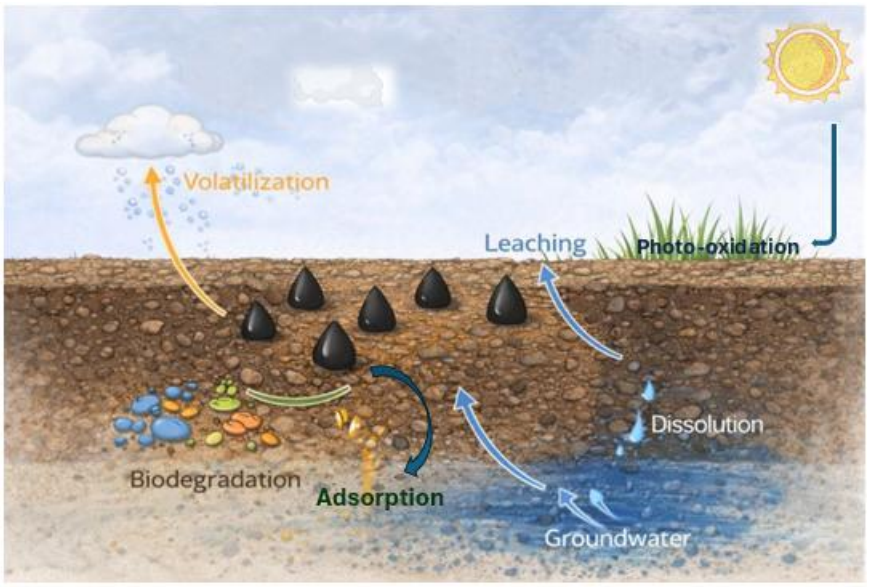
Petroleum contaminated soils commonly develop a high C:N:P imbalance because abundant carbon and energy, but little bioavailable N or P, are supplied by hydrocarbons. Alkanes and aromatic compounds are oxidized by hydrocarbonoclastic bacteria and fungi through oxygenase- and dehydrogenase mediated pathways, while alternative electron acceptors are used by anaerobic communities under oxygen limitation (Kakde and Sharma 2024; Mekonnen et al. 2024). The key constraint is therefore not merely the presence of degraders but the coupling among hydrocarbon flux, nutrient supply, oxygen or other electron acceptors, and microbial biomass production. Nutrient limitation can be relieved by biostimulation, and complementary degradative functions can be introduced by bioaugmentation, but both remain strongly site dependent.

5.5.2. Nutrient transformation and soil functional implications

Soil nutrient dynamics are strongly altered by petroleum hydrocarbon contamination because abundant carbon and energy are provided by hydrocarbons, which are poor sources of essential nutrients such as N and P. The demand for N and P is therefore increased by microbial hydrocarbon degradation, and nutrient limitations that restrict biodegradation can be created. Recent studies show that hydrocarbon degradation is strongly influenced by nutrient availability and C:N:P balance, although the response depends on contamination level and soil characteristics (Mekonnen et al. 2024; Schryer and Siciliano 2024). Isotope tracing studies further indicate that both hydrocarbon removal and microbial N utilization are modified by N addition during bioremediation, demonstrating close coupling between hydrocarbon degradation and N cycling (Liu et al. 2023).

Biostimulation can therefore have both beneficial and context-dependent effects on nutrient transformation. The response to P supplementation in heavily contaminated soils depends on soil properties and contamination level (Schryer and Siciliano 2024), and degradation may be inhibited, or microbial metabolic pathways altered, by excessive N addition (Liu et al. 2023); nutrient amendment must therefore be optimized rather than applied universally. Microbial community composition and the pathways involved in hydrocarbon transformation can also be changed by nutrient supplementation. Full restoration of soil nutrient cycling is thus not necessarily indicated by enhanced hydrocarbon removal after nutrient addition.

Fig. 3 Fate of hydrocarbon pollutants in soil



5.6. Pesticides

Microbial pesticide remediation depends on the chemical structure of the pesticide and the metabolic capacity of the resident community. Some pesticides or their transformation products can be used by bacteria as carbon, energy, or nutrient sources, whereas recalcitrant compounds can be attacked by fungi through extracellular oxidative enzymes. The relevant mechanisms include hydrolysis, oxidation–reduction, dehalogenation, and conjugation, and degradation rates are controlled by pH, moisture, temperature, nutrient status, concentration, and exposure history (Chia et al. 2024; Kumar et al. 2026). The principal limitation is that disappearance of the parent pesticide does not necessarily mean detoxification, because transformation products can persist or remain biologically active.

5.6.1. Microbial interventions

Bacterial degradation is often associated with *Pseudomonas*, *Arthrobacter*, *Flavobacterium*, *Stenotrophomonas*, and related taxa, but performance is pesticide specific rather than genus specific. Pesticide residues may additionally be reduced by endophytic bacteria within plant tissues or in the rhizosphere (Murthy 2025). Remediation is supported by microalgae through the production of oxygen, the use of pollutants as energy sources, and the removal of contaminants through bioaccumulation and biosorption; pesticides can be accumulated and transformed by them, with efficiency depending on pesticide properties. Under stress, resilience and degradation capacity are enhanced by their mixotrophic metabolism (Suresh et al. 2013).

5.6.2. Nutrient transformation and soil-functional implications

Soil nutrient transformation can be influenced by pesticide contamination through alteration of microbial abundance, community composition, and the activity of enzymes involved in C, N, and P cycling. Pesticides or their transformation products are used by some microorganisms as carbon or energy sources, but sensitive microbial groups may also be suppressed, and nutrient-transforming processes disrupted, by exposure. Decomposition, N mineralization, and P mobilization can in turn be affected by changes in community structure and functional activity, with the magnitude of the response depending on pesticide type, concentration, application history, and soil properties (Chia et al. 2024; Liu et al. 2025). Degradation of a pesticide by a specific microbial population should therefore not automatically be taken as evidence that overall soil function has recovered. The microbial groups, mechanisms and nutrient-transformation outcomes across the stressors are compared in Table 3.

Table 3 Microbial groups, remediation mechanisms, and nutrient-transformation outcomes across major soil stressors

Soil issue	Microbial group involved	Key mechanisms	Nutrient transformation and soil-functional outcomes	References
Saline soil	Halotolerant PGPR, AMF, endophytic fungi (e.g., *Serendipita indica*)	• EPS production • ACC deaminase activity • P and K solubilization • Improved root colonization and modulation of plant ion homeostasis	• Salt stress is buffered; microbial biomass and phosphatase activity were strongly improved by PGPR in wheat • Salinity reorganizes rather than uniformly suppresses nutrient cycling: C degradation, N fixation and P uptake/transport decline, while denitrification and organic P dissolution may be maintained or enhanced • Recovery depends on inoculant persistence and compatibility with the resident microbiome	Rouyd el et al. 2021; Li et al. 2024
Alkaline / saline–alkali soil	Sulfur-oxidizing bacteria; plant-associated microorganisms	• Oxidation of reduced sulfur to sulfuric acid, lowering pH • Organic acid production and mineral solubilization	• Greater availability of P and micronutrients that are poorly soluble at high pH • Improved soil quality after sulfur-oxidizer treatment • Enrichment of beneficial taxa suggests recovery of C, N and P cycling	Bao et al. 2016; Cui et al. 2025
Acid soil	Indigenous consortia (*Serratia grimesii*, *Citrobacter youngae*, *Pseudescherichia vulneris*, *Pseudomonas protegens*)	• Local alkalinization, acid neutralization • Local pH is raised by ammonium production via urea hydrolysis and amino acid decarboxylation • Organic acid production and mineral solubilization	• Single isolates were outperformed by consortia, favouring complementary functions • Higher N availability and partial relief of nitrifier suppression • Microbial organic acids and mineral solubilization may partly offset P fixation by Fe/Al minerals (not tested in the cited study) • Field-scale persistence and effects on metal mobility need validation	Cho et al. 2022
Sodic soil	EPS-producing PGPR and fungi (proposed; direct evidence limited)	• Possible solubilization of Ca-bound phosphates at high pH • Reduction of exchangeable Na^+ is achieved mainly by gypsum	• Better water infiltration and microbial activity, expected mainly from amendment • Greater availability of P, Fe and micronutrients	Qadir and Oster 2004
Heavy-metal contamination	Metal-tolerant bacteria and fungi	• Biosorption (cell-surface groups and EPS: complexation, ion exchange) • Bioaccumulation, biomineralization • Redox transformation, e.g., Cr(VI) → Cr(III) • Siderophore-mediated chelation • Bioleaching (Fe/S oxidation, organic acids) for metal recovery, which mobilizes rather than immobilizes	• Bioavailable metal is rapidly lowered by immobilization, allowing N-fixers and P-solubilizers to re-establish • Total metal is lowered by bioleaching, but dissolved-metal exposure is temporarily raised • Metal stress alters N-cycle genes (*nifH*, *amoA*, *narG*, *nirK*) and shifts the balance among growth, stress tolerance and nutrient acquisition • Success should be judged by bioavailability, biomass, N/P-cycling genes, enzymes and available nutrients, not total metal alone	Gadd 2010; Raffa et al. 2021; Abo-Alkasem et al. 2023; Bai et al. 2023; Tian et al. 2025; Popoola et al. 2024
Petroleum hydrocarbons	Hydrocarbonoclastic bacteria and fungi (*Pseudomonas*, *Rhodococcus*, *Acinetobacter*); anaerobic communities	• Oxygenase- and dehydrogenase-mediated oxidation of alkanes and aromatics • Biosurfactants increase bioavailability • Biostimulation (nutrients) and bioaugmentation (added degraders)	• C but little N and P are supplied by hydrocarbons, creating a C:N:P imbalance and nutrient limitation • Response to P supplementation is soil dependent and excess N may inhibit degradation, so amendments must be optimized • Degradation and N cycling are closely coupled (isotope tracing) • Faster hydrocarbon removal does not necessarily mean full restoration of nutrient cycling	Kakde and Sharma 2024; Mekonnen et al. 2024;

				Schryer and Siciliano 2024; Liu et al. 2023
Pesticide contamination	*Pseudomonas*, *Arthrobacter*, *Flavobacterium*, *Stenotrophomonas*; fungi; endophytic bacteria; microalgae	• Hydrolysis, oxidation–reduction, dehalogenation, conjugation • Pesticides or their products used as C, energy or nutrient sources • Recalcitrant compounds are attacked by fungal extracellular oxidative enzymes	• Decomposition, N mineralization and P mobilization are affected by changes in microbial abundance and enzyme activity • Response depends on pesticide type, concentration, application history and soil properties	Chia et al. 2024; Kumar et al. 2026; Murthy 2025; Gavrilescu 2005

6. Advances in remediation

6.1. Rhizoremediation

In rhizoremediation, plant-mediated contaminant uptake or stabilization is combined with microbial transformation in the rhizosphere. Root development, nutrient acquisition, metal tolerance, and contaminant bioavailability can be improved by plant growth-promoting bacteria (PGPB) and mycorrhizal fungi through siderophores, organic acids, ACC deaminase, and other metabolites (Glick 2012; Narayanan and Ma 2023). The key advantage over microbial inoculation alone is that carbon substrates are supplied and spatially structured niches are created by roots, while nutrients are mobilized and contaminant speciation is modified by microorganisms. However, restoration of soil nutrient cycling is not proved by enhanced plant growth or metal uptake; rhizoremediation should therefore be evaluated using rhizosphere microbial biomass, functional genes, nutrient pools, and contaminant bioavailability.

6.2. Genetic and metabolic engineering

Defined remediation traits, including catabolic pathways, metal resistance, transport, sequestration, and stress tolerance, can be improved by genetic and metabolic engineering. Precise genome editing and regulatory control are provided by CRISPR–Cas systems (Wang et al. 2022a). A major limitation of conventional bioaugmentation—the gap between laboratory performance and field conditions is partly addressed by these technologies, but ecological constraints are not removed by them. Metabolic costs may be imposed by engineered traits, which may also prove unstable under field conditions or interact unpredictably with indigenous communities. Engineered strains should therefore be assessed not only for target pollutant transformation but also for persistence, gene stability, effects on the resident community, and nutrient-functional performance.

Release of engineered strains also raises biosafety and regulatory questions, including containment, horizontal transfer of engineered genes to native microorganisms, and compliance with national regulations on genetically modified organisms; these issues currently limit field deployment and should be addressed alongside functional evaluation.

6.3. Biosurfactants and bioavailability management

The apparent bioavailability of hydrophobic hydrocarbons can be increased by biosurfactants through the reduction of interfacial tension and enhancement of emulsification, whereby microbial access to otherwise poorly accessible substrates is improved. Enhanced hydrocarbon removal and biodegradation in soil have been shown in studies of green surfactants (da Silva et al. 2021), and chemotaxis, substrate access, and degradation can be combined by biosurfactant-producing consortia (Ibrar et al. 2022). The trade-off is that transient exposure and toxicity may also be increased by greater contaminant mobilization. Biosurfactant application should therefore be optimized against contaminant bioavailability, microbial activity, and nutrient status rather than judged solely by short-term hydrocarbon removal.

6.4. Next-generation microbial agents and omics-guided selection

Stress-tolerant taxa, degradative pathways, and nutrient-cycling functions that are difficult to resolve in culture-based studies can be identified by metagenomics, metatranscriptomics, metabolomics, and genome-resolved approaches. Consortia can be selected, based on these datasets, for complementary functions for example, pairing a hydrocarbon degrader with organisms that maintain N or P transformation rather than strains being selected solely for maximum contaminant removal. The major gap is validation: functional genes or predicted pathways do not necessarily translate into process rates under field conditions. Omics-guided remediation therefore needs to be integrated with enzyme assays, nutrient flux measurements, contaminant bioavailability, and long-term field data. (Sharma et al. 2022)

6.5. Nano-enabled and immobilized microbial systems

Microbial retention can be improved, cells protected from toxic conditions, or contaminant accessibility modified by nanomaterials and immobilization supports. Nano-enhanced microbial remediation has been investigated for persistent organic contaminants, including PAHs (Rajput et al. 2023). Such systems may be useful where inoculants fail because of poor survival or limited substrate access, but nanoparticle toxicity, changes in microbial community structure, soil transport, and regulatory uncertainty remain important limitations. Nano-enabled systems should therefore be evaluated as integrated microbial–material systems, with nutrient-functional endpoints measured alongside contaminant transformation.

7. Constraints in microbial remediation

7.1. Environmental constraints

Bioremediation is inherently slower than physicochemical approaches and is highly site specific, and the maintenance of optimal conditions across the treatment area is made difficult by environmental heterogeneity (Vidali 2001). Microbial physiology can be disrupted by extreme pH, temperature fluctuations, moisture imbalance, and limited oxygen availability through alteration of enzyme conformation and metabolic pathways. In acidic soils, microbial activity is inhibited by the increased solubility of toxic metals such as Al^{3+} and Mn^{2+}, whereas nutrient availability is reduced by alkaline conditions, and microbial growth is thereby limited. Enzymatic kinetics are slowed by temperature variability, and substrate diffusion is restricted by low moisture. Conversely, anaerobic conditions, by which aerobic degraders are suppressed and less efficient metabolic pathways favoured, are created by excess moisture (Vidali 2001; Glick 2010).

7.2. Pollutant-related constraints

Microbial enzymes can be denatured and cell membrane integrity disrupted by high contaminant toxicity, leading to inhibition or cell death. In soils with mixed pollutants, interactions among compounds may be synergistic or antagonistic, complicating degradation pathways and reducing overall efficiency. Many contaminants are also strongly adsorbed onto clay minerals or soil organic matter, by which their bioavailability is markedly lowered. Recalcitrant compounds such as high-molecular-weight PAHs and other persistent organic structures are particularly resistant to degradation; specialized enzymatic systems and extended adaptation periods are required for their breakdown, by which the speed and effectiveness of bioremediation are limited (Semple et al. 2003; Mekonnen et al. 2024).

7.3. Microbial constraints

Introduced microbial inoculants often fail under field conditions because they face strong competition from well-adapted native communities and have limited ecological fitness. Their survival and persistence are reduced by low tolerance to stresses such as salinity and heavy metals, poor colonization ability, and the inability to form stable biofilms. Degradative genes may also be poorly expressed under environmental stress, lowering metabolic activity and remediation efficiency. In addition, degradation is frequently limited by inadequate nitrogen, phosphorus, and micronutrients, and microbial growth, biomass production, and enzyme activity can be restricted by an imbalanced C:N:P ratio. Additional substrates are often required to initiate degradation by co-metabolism, increasing dependence on external amendments such as fertilizers, organic matter, or biochar (Chaudhry et al. 2005).

7.4. Genetic and metabolic constraints

The efficiency of microbial remediation is also limited by genetic constraints. Many degradative pathways are encoded on plasmids or other mobile genetic elements, which are often unstable and may be lost during cell division, particularly under stress, reducing degradation capacity. Expression of catabolic genes is strongly environment dependent: expression can be suppressed under field conditions by pH, temperature, nutrient availability, and the absence of specific inducers, resulting in poor biodegradation. Metabolic versatility is also limited in many microorganisms, and complex or mixed contaminants cannot be degraded by them without synergistic interactions, which are not always stable in situ. Finally, the long-term effectiveness of microbial populations may be reduced by mutation or loss of functional genes over time (Top and Springael 2003).

8. CONCLUSION

A sustainable approach to restoring problematic and polluted soils is offered by microbe-mediated mitigation through the alleviation of physicochemical constraints and the transformation or immobilization of contaminants. Microorganisms contribute not only to pollutant degradation and detoxification but also to the carbon, nitrogen, and phosphorus transformations that support soil fertility and ecosystem functioning. However, complete soil recovery is not necessarily indicated by contaminant removal or stress alleviation alone, because microbial community structure, functional genes, enzyme activities, and nutrient availability may respond differently. Microbial remediation should therefore be evaluated using both contaminant-related and nutrient-functional indicators. Multi-omics-guided selection of stable microbial consortia should be combined, in future research, with genetic and metabolic engineering, rhizoremediation, and field-scale validation under realistic soil conditions. Overall, a more comprehensive pathway toward sustainable soil restoration and long-term soil health is offered by linking contaminant mitigation to recovery of microbial nutrient transformation.

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REFERENCES

1. Abo-Alkasem MI, Hassan NH, Abo Elsoud MM (2023) Microbial bioremediation as a tool for the removal of heavy metals. Bull Natl Res Cent 47:31. https://doi.org/10.1186/s42269-023-01006-z
2. Anderson TH (2003) Microbial eco-physiological indicators to assess soil quality. Agric Ecosyst Environ 98(1–3):285–293. https://doi.org/10.1016/S0167-8809(03)00088-4
3. Bai X, Li Y, Jing X, Zhao X, Zhao P (2023) Response mechanisms of bacterial communities and nitrogen cycle functional genes in millet rhizosphere soil to chromium stress. Front Microbiol 14:1116535. https://doi.org/10.3389/fmicb.2023.1116535
4. Bao S, Wang Q, Bao X, Li M, Wang Z (2016) Biological treatment of saline-alkali soil by sulfur-oxidizing bacteria. Bioengineered 7(5):372–375. https://doi.org/10.1080/21655979.2016.1226664
5. Chaudhry Q, Blom-Zandstra M, Gupta SK, Joner E (2005) Utilising the synergy between plants and rhizosphere microorganisms to enhance breakdown of organic pollutants in the environment. Environ Sci Pollut Res 12(1):34–48. https://doi.org/10.1065/espr2004.08.213
6. Chia XK, Hadibarata T, Kristanti RA, Jusoh MNH, Tan IS, Foo HCY (2024) The function of microbial enzymes in breaking down soil contaminated with pesticides: a review. Bioprocess Biosyst Eng 47(5):597–620. https://doi.org/10.1007/s00449-024-02978-6
7. Chicca I, Becarelli S, Di Gregorio S (2022) Microbial involvement in the bioremediation of total petroleum hydrocarbon polluted soils: challenges and perspectives. Environments 9(4):52. https://doi.org/10.3390/environments9040052

8. Cho MJ, Park SY, Kim CG (2022) Acidic neutralization by indigenous bacteria isolated from abandoned mine areas. Appl Sci 12(7):3324. https://doi.org/10.3390/app12073324
9. Cui Y, Ning Z, Li M, Qin X, Yue X, Chen X, Zhu C, Sun H, Huang Y (2025) Microbial network-driven remediation of saline-alkali soils by salt-tolerant plants. Front Microbiol 16:1565399. https://doi.org/10.3389/fmicb.2025.1565399
10. da Silva IGS, de Almeida FCG, da Rocha e Silva NMP, de Oliveira JTR, Converti A, Sarubbo LA (2021) Application of green surfactants in the remediation of soils contaminated by hydrocarbons. Processes 9(9):1666. https://doi.org/10.3390/pr9091666
11. Daunoras J, Kačergius A, Gudiukaitė R (2024) Role of soil microbiota enzymes in soil health and activity changes depending on climate change and the type of soil ecosystem. Biology 13(2):85. https://doi.org/10.3390/biology13020085
12. Doran JW, Zeiss MR (2000) Soil health and sustainability: managing the biotic component of soil quality. Appl Soil Ecol 15(1):3–11. https://doi.org/10.1016/S0929-1393(00)00067-6
13. Duan C, Wang Y, Wang Q, Ju W, Zhang Z, Cui Y, Beiyuan J, Fan Q, Wei S, Li S, Fang L (2022) Microbial metabolic limitation of rhizosphere under heavy metal stress: Evidence from soil ecoenzymatic stoichiometry. Environ Pollut 300:118978. https://doi.org/10.1016/j.envpol.2022.118978
14. Egamberdieva D, Wirth S, Bellingrath-Kimura SD, Mishra J, Arora NK (2019) Salt-tolerant plant growth promoting rhizobacteria for enhancing crop productivity of saline soils. Front Microbiol 10:2791. https://doi.org/10.3389/fmicb.2019.02791
15. FAO (2015) Status of the world's soil resources (SWSR)–main report. Food and Agriculture Organization of the United Nations, Rome.
16. Fetsiukh A, Pall T, Timmusk S (2024) Decrease due to pollution in the rhizosphere microbial diversity can be amended by supplementation from adapted plants of another species. Sci Rep 14(1):18806. https://doi.org/10.1038/s41598-024-68123-1
17. Gadd GM (2010) Metals, minerals and microbes: geomicrobiology and bioremediation. Microbiology 156(3):609–643. https://doi.org/10.1099/mic.0.037143-0
18. Gavrilescu M (2005) Fate of pesticides in the environment and its bioremediation. Eng Life Sci 5(6):497–526. https://doi.org/10.1002/elsc.200520098
19. Glick BR (2010) Using soil bacteria to facilitate phytoremediation. Biotechnol Adv 28(3):367–374. https://doi.org/10.1016/j.biotechadv.2010.02.001
20. Glick BR (2012) Plant growth-promoting bacteria: mechanisms and applications. Scientifica 2012:963401. https://doi.org/10.6064/2012/963401
21. Haj-Amor Z, Araya T, Kim DG, Bouri S, Lee J, Ghiloufi W, Yang Y, Kang H, Jhariya MK, Banerjee A, Lal R (2022) Soil salinity and its associated effects on soil microorganisms, greenhouse gas emissions, crop yield, biodiversity and desertification: A review. Sci Total Environ 843:156946. https://doi.org/10.1016/j.scitotenv.2022.156946
22. Ibrar M, Khan S, Hasan F, Yang X (2022) Biosurfactants and chemotaxis interplay in microbial consortium-based hydrocarbons degradation. Environ Sci Pollut Res 29(17):24391–24410. https://doi.org/10.1007/s11356-022-18492-9
23. Kakde P, Sharma J (2024) Microbial bioremediation of petroleum contaminated soil: structural complexity, degradation dynamics and advanced remediation techniques. J Pure Appl Microbiol 18:2244–2261. https://doi.org/10.22207/JPAM.18.4.28
24. Kumar A, Singh S, Gaurav AK, Srivastava S, Verma JP (2020) Plant growth-promoting bacteria: biological tools for the mitigation of salinity stress in plants. Front Microbiol 11:1216. https://doi.org/10.3389/fmicb.2020.01216
25. Kumar A, Solanki MK, Kumar M, Kaushik A, Arya A, Saikia M, Gaur VK, Singh RP, Singh SK, Singh VK (2026) The microbial strategies for the management of chemical pesticides: a comprehensive review. Curr Res Microb Sci 10:100519. https://doi.org/10.1016/j.crmicr.2025.100519
26. Kundu A, Harrisson O, Ghoshal S (2022) Evaluating the impact of nutrient doses on biostimulation of petroleum hydrocarbon biodegradation in cold region soils. ACS ES&T Engg 2(12):2287–2300. https://doi.org/10.1021/acsestengg.2c00220
27. Labud V, Garcia C, Hernandez T (2007) Effect of hydrocarbon pollution on the microbial properties of a sandy and a clay soil. Chemosphere 66(10):1863–1871. https://doi.org/10.1016/j.chemosphere.2006.08.021
28. Li Y, Li W, Jiang L, Li E, Yang X, Yang J (2024) Salinity affects microbial function genes related to nutrient cycling in arid regions. Front Microbiol 15:1407760. https://doi.org/10.3389/fmicb.2024.1407760
29. Liu H, Wu M, Zhang M, Gao H, Yan Z, Yang Z (2023) New insight into the effect of nitrogen on hydrocarbon degradation in petroleum-contaminated soil revealed through 15N tracing and flow cytometry. Sci Total Environ 891:164409. https://doi.org/10.1016/j.scitotenv.2023.164409
30. Liu P, Wen S, Zhu S, Hu X, Wang Y (2025) Microbial degradation of soil organic pollutants: mechanisms, challenges and advances in forest ecosystem management. Processes 13(3):916. https://doi.org/10.3390/pr13030916
31. Ma S, Qiao L, Liu X, Zhang S, Zhang L, Qiu Z, Yu C (2022) Microbial community succession in soils under long-term heavy metal stress from community diversity-structure to KEGG function pathways. Environ Res 214(Pt 2):113822. https://doi.org/10.1016/j.envres.2022.113822

32. Mekonnen BA, Aragaw TA, Genet MB (2024) Bioremediation of petroleum hydrocarbon contaminated soil: a review on principles, degradation mechanisms and advancements. Front Environ Sci 12:1354422. https://doi.org/10.3389/fenvs.2024.1354422
33. Murthy MK (2025) Environmental dynamics of pesticides: sources, impacts on amphibians, nanoparticles and endophytic microorganism remediation. Environ Sci Pollut Res 32(13):7860–7893. https://doi.org/10.1007/s11356-025-36216-7
34. Narayanan M, Ma Y (2023) Mitigation of heavy metal stress in the soil through optimized interaction between plants and microbes. J Environ Manage 345:118732. https://doi.org/10.1016/j.jenvman.2023.118732
35. Popoola OJ, Ogundele OD, Ladapo EA, Senbore S (2024) The impact of heavy metal contamination in soils on soil microbial communities and its potential health risks for humans. In: Soil microbiome in green technology sustainability. Springer, pp 351–375. https://doi.org/10.1007/978-3-031-71844-1_15
36. Qadir M, Oster J (2004) Crop and irrigation management strategies for saline-sodic soils and waters aimed at environmentally sustainable agriculture. Sci Total Environ 323(1-3):1–19. https://doi.org/10.1016/j.scitotenv.2003.10.012
37. Raffa CM, Chiampo F, Shanthakumar S (2021) Remediation of metal/metalloid-polluted soils: a short review. Appl Sci 11(9):4134. https://doi.org/10.3390/app11094134
38. Rajput VD, Kumari S, Minkina T, Sushkova S, Mandzhieva S (2023) Nano-enhanced microbial remediation of PAHs contaminated soil. Air Soil Water Res 16:11786221231170099. https://doi.org/10.1177/11786221231170099
39. Rezapour S, Kalashypour E, Asadzadeh F (2017) Assessment of the quality of salt-affected soils after irrigation and cultivation in semi-arid condition. Int J Environ Res 11(3):301–313. https://doi.org/10.1007/s41742-017-0028-0
40. Rouydel Z, Barin M, Rasouli-Sadaghiani MH, Khezri M, Vetukuri RR, Kushwaha S (2021) Harnessing the potential of symbiotic endophytic fungi and plant growth-promoting rhizobacteria to enhance soil quality in saline soils. Processes 9(10):1810. https://doi.org/10.3390/pr9101810
41. Schryer AD, Siciliano SD (2024) Do phosphorus amendments enhance biodegradation activity in stalled petroleum hydrocarbon-contaminated soil? J Environ Qual 53(5):669–683. https://doi.org/10.1002/jeq2.20594
42. Semple KT, Morriss A, Paton GI (2003) Bioavailability of hydrophobic organic contaminants in soils: fundamental concepts and techniques for analysis. Eur J Soil Sci 54(4):809–818. https://doi.org/10.1046/j.1351-0754.2003.0564.x
43. Shi J, Du Y, Zou J, Ma S, Mao S, Li W, Yu C (2024) Mechanisms of microbial-driven changes in soil ecological stoichiometry around gold mines. J Hazard Mater 465:133239. https://doi.org/10.1016/j.jhazmat.2023.133239
44. Pooja Sharma, Surendra Pratap Singh, Hafiz M.N. Iqbal, Yen Wah Tong, Omics approaches in bioremediation of environmental contaminants: An integrated approach for environmental safety and sustainability, Environmental Research, Volume 211, 2022, 113102, ISSN 0013-9351, https://doi.org/10.1016/j.envres.2022.113102.
45. Suresh R. Subashchandrabose, Balasubramanian Ramakrishnan, Mallavarapu Megharaj, Kadiyala Venkateswarlu, Ravi Naidu, Mixotrophic cyanobacteria and microalgae as distinctive biological agents for organic pollutant degradation, Environment International, Volume 51, 2013, Pages 59-72, ISSN 0160-4120, https://doi.org/10.1016/j.envint.2012.10.007.
46. Tian J, Du Y, Yu C, Liu W, Zou R, Zhao Y, Zhang T, Jiang Y, Tian Z (2025) The influences of heavy metals on soil microbial C, N, P cycling and heavy metal resistance under different fertilization regimes. Environ Pollut 370:125915. https://doi.org/10.1016/j.envpol.2025.125915
47. Top EM, Springael D (2003) The role of mobile genetic elements in bacterial adaptation to xenobiotic organic compounds. Curr Opin Biotechnol 14(3):262–269. https://doi.org/10.1016/S0958-1669(03)00066-1
48. Tóth T (2022) Review of sodic soil reclamation with a snapshot of current research activity. Chin Geogr Sci 32(6):1099–1109. https://doi.org/10.1007/s11769-022-1310-4
49. van Dorst J, Siciliano SD, Winsley T, Snape I, Ferrari BC (2014) Bacterial targets as potential indicators of diesel fuel toxicity in subantarctic soils. Appl Environ Microbiol 80(13):4021–4033. https://doi.org/10.1128/AEM.03939-13
50. Vidali M (2001) Bioremediation: an overview. Pure Appl Chem 73(7):1163–1172. https://doi.org/10.1351/pac200173071163
51. von Uexküll HR, Mutert E (1995) Global extent, development and economic impact of acid soils. Plant Soil 171:1–15. https://doi.org/10.1007/BF00009558
52. Wang JY, Pausch P, Doudna JA (2022a) Structural biology of CRISPR–Cas immunity and genome editing enzymes. Nat Rev Microbiol 20:641–656. https://doi.org/10.1038/s41579-022-00739-4
53. Wang Y, Wang HS, Tang CS, Gu K, Shi B (2022b) Remediation of heavy-metal-contaminated soils by biochar: a review. Environ Geotech 9(3):135–148. https://doi.org/10.1680/jenge.18.00091
54. Weil R, Brady N (2016) The nature and properties of soils, 15th edn. Pearson, Columbus, Ohio.
55. Yang J, Li G, Qian Y, Yang Y, Zhang F (2018) Microbial functional gene patterns related to soil greenhouse gas emissions in oil contaminated areas. Sci Total Environ 628:94–102. https://doi.org/10.1016/j.scitotenv.2018.02.007

56. Zabbey N, Sam K, Onyebuchi AT (2017) Remediation of contaminated lands in the Niger Delta, Nigeria: prospects and challenges. Sci Total Environ 586:952–965. https://doi.org/10.1016/j.scitotenv.2017.02.075
57. Zhao S, Yuan X, Wang X, Ai Y, Li F (2024) Research progress and hotspots in microbial remediation for polluted soils. Sustainability 16(17):7458. https://doi.org/10.3390/su16177458

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Data Availability
Data sharing is not applicable to this article because no datasets were generated or analysed during the current study.