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The impacts of remediation techniques on soil functions: a review

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Abstract. Soil contamination is a major driver of soil degradation in Europe, and remediation practices are increasingly expected to support not only risk reduction but also the recovery of soil health and ecosystem services. In line with the European Soil Monitoring and Resilience Directive, this review evaluates how commonly applied remediation techniques impact key soil functions that underpin soil health. A structured narrative review of the literature was conducted to assess the impacts of conventional (physical and chemical) and low-impact (biological and nature-based) remediation techniques on six aggregated soil functions: organic matter storage and transformation, water regulation, nutrient cycling, contaminant retention and degradation, physical stability, and habitat provision. The analysis shows that conventional remediation techniques such as thermal treatment, soil washing, in-situ chemical oxidation/reduction, and stabilization/solidification frequently achieve effective contaminant removal but often cause substantial and sometimes irreversible degradation of soil structure, organic matter, microbial activity, and habitat quality. In contrast, low-impact remediation approaches, including bioremediation, phytoremediation, monitored natural attenuation, and immobilization with organic or selected inorganic amendments tend to preserve or enhance soil functions, although typically at slower contaminant removal rates and with strong site specificity. The review highlights critical knowledge gaps, particularly regarding limited information on specific remediation techniques and impacts on soil fauna, the limited availability of quantitative thresholds for when a soil can be considered healthy or unhealthy for performing various functions, and limited soil health assessments in urban and industrial context. Hybrid and adaptive remediation strategies that combine risk reduction with rehabilitation measures are identified as key to balancing risk reduction and soil functionality while also planning for refunctionalization at the onset as part of the soil health trajectory. Overall, embedding soil health as a central management objective is essential for sustainable contaminated land management aligned with European policy goals.

Keywords: Soil remediation; Soil functions; Soil health; Contaminated land; Soil properties; Low-impact remediation techniques (LIRT); Ecosystem services

1 Introduction

The European Soil Monitoring and Resilience Directive (Soil Monitoring Law, SML), adopted in 2025, establishes a target of achieving healthy soils across the European Union by 2050 and identifies contaminated land as a priority area for action. Healthy soils are defined as those in good chemical, biological, and physical condition that can continuously provide ecosystem services (EC, 2025). However, an estimated 60-70% of the

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EU soils are currently considered healthy due to degradation (Veerman et al., 2020), with soil contamination being a major driver. Soil health assessment focuses on evaluating a soil's capacity to deliver multiple ecosystem services (i.e., multifunctionality) and how this capacity responds to different management practices. Numerous frameworks have been proposed, incorporating a wide range of soil quality indicators (SQIs), measurement approaches, and interpretations of soil properties, processes, and functions (Guo, 2021; Rinot et al., 2019; Zwaan et al., 2026). In general, assessments rely on measurable soil descriptors related to physical, chemical, and biological characteristics that collectively support key soil functions and ecosystem services (Bünemann et al., 2018; Rinot et al., 2019). Following Bünemann et al. (2018), soil functions are defined as *(bundles of) soil processes that underpin the delivery of ecosystem services*, including carbon cycling and storage, nutrient cycling, water regulation, habitat for biodiversity, and physical stability. Despite growing interest in soil health, its assessment remains only weakly integrated into contaminated land management. Greater consideration of soil functions and health indicators could support more informed decision-making and help prioritize areas for protection, restoration, or development according to their condition and functional capacity (Blanchart et al., 2018; Séré et al., 2024; Volchko et al., 2014).

In general, contaminated soil has long been viewed as waste to be disposed of rather than as a valuable resource to be treated and reused (Gerhardt et al., 2017; Mench et al., 2010). For contaminated land managers and practitioners, the new emphasis on soil health requires a shift from a primarily risk-based paradigm to determine fitness-for use, which focuses primarily on chemical measurements of contaminant concentrations for managing human exposure and spreading to environmental receptors, towards accounting for additional physical, chemical and biological properties and more comprehensive soil health assessment. However, selecting relevant indicators and interpreting results to classify soil as 'healthy' remains a challenge, particularly in the context of contaminated land. Furthermore, it will become increasingly important to consider that the remediation techniques deployed can have detrimental impacts on soil functions while achieving effective risk reduction (O'Brien et al., 2017; Volchko et al., 2013). From the soil protection point of view, the remediation process should be as soil-protective as feasible (Meuser, 2013). Remediation, however, can be completely destructive to the soil ecosystem when using conventional, excavation-based methods (Breure et al., 2018; O'Brien et al., 2017). Indeed, in some cases, remediation may result in a greater negative impact on ecological receptors than leaving the limited contamination in place (Burger et al., 2016; FAO et al., 2020). Broadly, biological approaches (e.g., phytoremediation) generally promote soil health and microbial recovery, albeit often at a slow pace, while conventional physical and chemical methods (e.g., thermal treatment, soil washing) are often preferred for rapid and efficient contaminant removal but frequently cause significant and sometimes irreversible damage to soil ecosystems (O'Brien et al., 2017; Volchko et al., 2013). For instance, several recent studies have shown using quantitative soil quality indices that phytoremediation (Burges et al., 2016, 2017; Epelde et al., 2014; Mench et al., 2023) and soil amendments like biochar (Bera et al., 2016; Drenning et al., 2024) can have positive impacts on soil health. Few studies, however, have systematically investigated the impacts of the various physical, chemical and biological soil remediation techniques on soil functions. There is also a notable lack of discussion about the implications for soil functionality and the potentially extended recovery times following remediation (O'Brien et al., 2017a). Accounting for the impacts of remediation techniques on soil functions in decision-making is an integral part of considering soil health in contaminated land management that is often neglected during remediation option appraisal, design, and verification monitoring, but is well supported

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by recent trends in policy and legislation such as the SML. There is thus a need for additional knowledge to support integrated and carefully monitored remediation strategies that prioritize not only contaminant reduction but also the long-term restoration and preservation of soil's functionality.

Accordingly, the overall aim of this review is to evaluate how commonly applied physical, chemical and biological remediation techniques impact six aggregated soil functions and associated indicators as well as discuss the implications for practice. The specific objectives of this study are to i) synthesize evidence regarding the expected impacts from conventional or low-impact remediation techniques on soil indicators and generalize as qualitative trends to improve or diminish soil functions; ii) highlight where evidence is robust or knowledge gaps remain; iii) discuss the practical implications of considering soil functionality to minimize the impacts from remediation techniques, consider recovery and refunctionalization of soil functionality following remediation, and balancing risk reduction and soil functionality in selection and application of remediation techniques. This study presents the results from the Europe Horizon Mission Soil project, ISLANDR (see ISLANDR D2.3 (Couto et al., 2026) for more information), and provides practitioners with a pragmatic basis to anticipate trade-offs between risk reduction and soil functionality to support decision-making and integration of soil health into contaminated land management processes.

2 Methodology

The remediation techniques included in this review were selected based on Annex IV of the SML (EC, 2025) and grouped into conventional remediation techniques (primarily physical and chemical methods) and low-impact remediation techniques (LIRT; primarily biological methods, see (Bardos et al., 2026)) to facilitate comparison between the two approaches. A distinction was also made between in-situ and ex-situ applications, as excavation and ex-situ treatment are generally considered detrimental to soil functions. Because no direct measure of overall soil functionality exists, the assessment relied on proxy indicators commonly used in soil health frameworks. Remediation impacts were evaluated across six overarching soil functions and their associated sub-functions. Effects on each function were inferred from expected changes in relevant soil quality indicators and descriptors linked to those functions. The indicators/descriptors considered and grouping within functions/sub-functions were harmonized with the list of descriptors suggested in the *Soil Directive* as well as the recently published ISO standards (ISO, 2026a, b): **i) Organic matter storage, transformation and recycling** – consists of the sub-functions decomposition, resource reallocation and biochemical transformation and the primary indicators used to assess these sub-functions include soil organic carbon (SOC) or organic matter (SOM) as well as microbial biomass and microbial activity such as organic matter mineralization and catabolic activities (e.g., enzyme activity); **ii) Water regulation, retention and release** – consists of the sub-functions biological retention by plants, water retention, and infiltration and percolation and the primary indicators used to assess these sub-functions include soil texture, soil organic carbon (SOC), water holding capacity, bulk density, and effects on macrofauna (earthworms); **iii) Nutrient cycling –** consists of the sub-functions nutrient transformation, nutrient reallocation, and nutrient assimilation by plants and the primary indicators used to assess these functions include total and available nutrient (e.g., N and P) content, pH, microbial catabolic activities, soil microbial biomass, cation exchange capacity (CEC), and functional diversity and genes; **iv) Contaminants retention, transformation and degradation –** consists of the sub-functions retention and transformation & degradation and the primary indicators used to assess these functions include soil

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organic carbon (SOC), pH, soil texture (clay content), changes in bioavailable concentrations of metals and organics, and microbial activities (this function is considered here as impacts on the soil's inherent capacity to retain, transform and degrade contaminants and manage residual contaminants); **v) Physical stability** – consists of the sub-function of inherent soil stability and evolution and the primary indicators used to assess this function include soil texture, soil microbial biomass, change in pore networks (compaction), aggregation, soil organic carbon (SOC), effects on macrofauna (earthworms), salinity and bulk density. The geotechnical properties of the soil are also included in this category (see e.g., Rehman et al., 2023); and **vi) Habitat provision** – consists of the sub-functions habitat quality and harboring biodiversity and the primary indicators used to assess these functions include soil texture, pH, soil organic carbon (SOC), soil microbial biomass, plant biomass production, ecotoxicological quality, and microbial and invertebrate diversity.

A structured, narrative literature review (Snyder, 2019) was performed by searching the Scopus database for scientific studies that have measured the impacts of remediation techniques on specific soil functions or indicators. First, a broad, simple search for combinations of "remediation techniques", "impacts", "soil function", "soil health", and "soil quality" were tested, which resulted in many hits (945) but few relevant papers after a cursory review of titles and abstracts (6). Following this search, combinations of primary and secondary keywords were used to collect a representative (i.e., non-exhaustive) sample of the literature to make the assessment, which followed the same general search strategy for all remediation techniques: 1) primary keywords – conventional and low-impact remediation techniques considered in the review including related terms and abbreviations, e.g., "chemical remediation" OR "ISCO" OR "ISCR"; AND 2) search terms related to soil functions and specific indicators, e.g., "soil function", "soil functionality", "soil health", "soil quality", "soil properties", "organic matter", "enzyme activity", "biodiversity", etc. Titles and abstract were screened for relevance, followed by a full-text review of shortlisted articles (131). Studies were included if they reported quantifiable changes in soil properties or indicators associated with soil functions following remediation, but were excluded if they focused solely on contaminant removal effectiveness without reporting changes in other soil parameters. The search was conducted for studies published up to 2026, without restriction on publication year. The full list of primary and secondary keywords is provided in Table S1 in the Supplementary Material.

The expected impacts of each remediation technique on soil functions were assessed by reviewing the relevant literature and assigning a semi-quantitative score based on the aggregated evidence (Table 1). For each indicator, the direction of change reported across studies was evaluated, and where multiple studies were available, the predominant trend was identified qualitatively based on the consistency of findings. Given the heterogeneity of the reviewed studies, expert judgement was used to account for differences in site conditions (e.g., soil type and climate) and treatment parameters when deriving generalized conclusions. Because remediation outcomes are highly context dependent, the results should be interpreted as broad trends rather than predictive outcomes. Where evidence was conflicting, a mixed (+/−) score was assigned and identified as an area of uncertainty.

Results are presented both as the average expected impact on each soil function, assuming equal weighting of indicators, and at the level of individual indicators (see Table S2 and S3 in SM).

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157 **Table 1. Scoring the impacts on soil functions.**

++ : clear or highly positive impact
+ : somewhat positive impact
+/- : mixed or no impact
— : somewhat negative impact
— — : clear or highly negative impact
Unclear impact or no information available

3 Results

3.1 Impacts of conventional remediation techniques on soil functions

Table 2 presents the aggregated, average expected impacts of conventional remediation techniques on the six main soil functions. The following sections will provide a summary of the impacts of specific remediation techniques on the six soil functions based on changes in individual indicators (see Table S2 in the SM and ISLANDR D2.3 for a more complete overview).

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Table 2. Impacts of conventional remediation techniques on soil functions (color coding according to Table 1).

Soil functions \ Remediation techniques	Thermal treatment (in-situ/ex-situ*)	Soil washing (ex-situ*) or flushing (in-situ)	ISCR/ISCO (in-situ)	Electrokinetic remediation (in-situ)	Multi-phase extraction (in-situ)	Air sparging (in-situ)	Containment/ capping/barriers (in-situ)	Stabilisation/ solidification (S/S, in-situ/ex-situ*)
Organic matter storage, transformation and recycling	— —	— or +/-	— —	— or +/-		+/-		— —
Water regulation, retention and release	— —	— or +/-	—		+/-		— or +/-	— —
Nutrient cycling	— or +/-	—	—	+/-			+/-	— —
Contaminants retention, transformation and degradation	—	+/-	—	—	+/-	+ or +/-	+ or +/-	+
Physical stability	—	— or +/-	—	+/-				+
Habitat provision	—	— or +/-	— or +/-	—		+/-	—	— —

++: clear or highly positive impact; +: somewhat positive impact; +/-: mixed or no impact; —: somewhat negative impact; — —: clear or highly negative impact; grey indicates there is no information available.

*Excavation is generally considered to have a deleterious effect on soil functions. Ex-situ treatment and replacement can restore some soil functionality, but most functions will be lost and additional treatment could further reduce functionality.

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3.1.1 Thermal treatment

Thermal treatment remediates contaminated soil by applying heat to volatilize, transform, or destroy contaminants, depending on the temperature applied. In general, temperatures below 250°C primarily increase contaminant mobility, temperatures between 250 and 500°C promote thermal desorption and transformation, and temperatures above 500°C result in contaminant destruction through combustion (O'Brien et al., 2017, 2018). While effective for contaminant removal, thermal remediation can substantially alter soil functions. The extent of these impacts is largely controlled by treatment temperature, heating duration, and soil characteristics. Key effects include losses of soil organic matter (SOM), mineralogical alteration of clay fractions, changes in soil pH, and reductions in biological communities (Biabani et al., 2023; Liao et al., 2025; O'Brien et al., 2018). Low-temperature, short-duration treatments generally cause limited disruption and may allow partial or complete recovery of soil functions, whereas higher temperatures often result in severe and potentially irreversible degradation.

Thermal treatment reduces SOM through volatilization, transformation, and/or combustion with cascading effects on all soil functions (Liao et al., 2025; O'Brien et al., 2017, 2018). However, the severity of the impact varies depending on the temperature and specific composition of SOM (Biabani et al., 2023). For instance, mild heating at short duration (e.g., <1hour) and lower temperatures (<300°C) may lead to small reductions in SOM and microbial enzyme activities (Yi et al., 2016), or even negligible changes at <250°C while forming a biochar-like pyrolytic carbon (Ren et al., 2020), and the soil may be expected to recover within a few (2–4) years following treatment (Biabani et al., 2023; O'Brien et al., 2018; Pape et al., 2015; Ren et al., 2020; Yi et al., 2016; Zhao et al., 2026). In contrast, SOM decreases rapidly at temperatures exceeding ca. 300°C (e.g., heating at 350°C for longer than 20 min reduced SOM by more than 50% in most cases (Biabani et al., 2023)) with reductions in microbial activities resulting in significant impacts to soil function and inhibiting recovery both in the short- and long-term (O'Brien et al., 2018; Pape et al., 2015). Soil pH responses are also dependent on temperature and heating duration, with minimal changes or slight acidification at low temperatures, but pronounced alkalinization at higher temperatures due to combustion of SOM and enrichment of alkali cations, further constraining nutrient availability and biological activity (Biabani et al., 2023; O'Brien et al., 2018; Pape et al., 2015; Ren et al., 2020; Yi et al., 2016).

Water regulation and physical stability are commonly impaired following thermal remediation. Loss of SOM and clay mineral alteration reduce water-holding capacity, cation exchange capacity, aggregation, and pore connectivity, while inducing hydrophobicity at moderate temperatures (Biabani et al., 2023; Lee et al., 2021; Liao et al., 2025; O'Brien et al., 2017, 2018). Heating at higher temperatures (>450°C) can markedly shift particle size distributions toward sand-dominated textures (Lee et al., 2021; Liao et al., 2025; O'Brien et al., 2018), with a recent meta-analysis reporting a 54.2% reduction in soil clay content and increase in soil sand content by 15.2% after thermal treatment with corresponding declines in water retention and soil fertility (Liao et al., 2025). Soil electrical conductivity can also increase by as much as 69.5% due to the loss of structural hydroxyl groups and release of ions during heating (Liao et al., 2025). Thermal treatment can result in significant changes to soil texture and mineralogy because mineral clay lattice structures can become dehydrated and break down under excessive heating resulting in shifts in the particle size distribution (Lee et al., 2021; Liao et al., 2025; O'Brien et al., 2018). These changes alongside reductions in SOM can greatly impact inherent soil

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stability and evolution and affect pore networks, aggregation and bulk density. However, from a geotechnical standpoint, thermal treatment can improve geotechnical properties by solidifying soil and improving mechanical characteristics significantly when heating at high temperatures, such as in vitrification where temperatures can exceed 1500°C (Rehman et al., 2023).

Thermal remediation also modifies contaminant retention and transformation processes. Reductions in clay content and SOM reduce sorption capacity, potentially increasing contaminant mobility, though metal responses are soil specific. Some studies report that metals can potentially be mobilized and more genotoxic to organisms following thermal treatment (Bonnard et al., 2010), while others show no remobilization of metals following heating at high temperatures and several metals had increased residual fractionation indicating that they will be less mobile and less bioavailable (O'Brien et al., 2018). Since microorganisms are effectively destroyed, or at least significantly impaired, following thermal treatment at higher temperatures (O'Brien et al., 2018), the inherent capacity of soils to transform or degrade organic contaminants may be greatly reduced and not easily recoverable.

Habitat provision and biomass production are consequently reduced due to physical, chemical and biological constraints induced by thermal remediation. Microorganisms can persist and recover rapidly after heating at lower temperatures (< 300°C), but higher temperature heating will inhibit soil function and significantly delay recovery, which may require rehabilitation measures such as organic amendments (Biabani et al., 2023; O'Brien et al., 2018; Pape et al., 2015; Yi et al., 2016; Zhao et al., 2026). Few studies have directly evaluated how thermal remediation affects soil invertebrates following treatment, but evidence suggests it may increase the ecotoxicity of residual metals to organisms like earthworms (*Eisenia fetida*) by enhancing metal bioavailability (Bonnard et al., 2010). Generally, biomass production declines with increasing heating temperatures (O'Brien et al., 2017, 2018; Pape et al., 2015; Yi et al., 2016). A recent meta-analysis showed that thermal treatment leads to the reduction of plant germination rate, length, and biomass by 19–45% relative to untreated soils, which was attributed to residual contaminants and the loss of soil fertility during the heating process (Liao et al., 2025).

3.1.2 Soil washing or flushing

Soil washing, include surfactant and chelate extraction, is a remediation technique used to separate contaminants from soil particles through dissolution by applying chemicals at low concentrations to enhance bioremediation (increasing contaminant bioavailability) or at high concentrations to extract contaminants in either a washing (ex-situ) or flushing (in-situ) process (O'Brien et al., 2017). The chelating agent EDTA has been most widely studied due to its high metal removal efficiency; however, its persistence and toxicity raise substantial concerns for post-remediation soil function (Epelde et al., 2008a; Jelusic and Lestan, 2014; Lee et al., 2021; Wang et al., 2018). Increasing attention has therefore shifted toward biodegradable chelators, and more efficient washing processes, which show variable but often adequate removal efficiencies and generally reduced impacts on soil processes (Kaurin et al., 2020; Wang et al., 2018).

In general, soil washing is expected to lead to a loss of SOM and nutrients through dissolution, enhanced mineralization and loss of fine particles, which may only recover slowly after several crop cycles (Gluhar et al., 2021b; Jelusic and Lestan, 2014; Lee et al., 2021). This typically leads to a decrease in water and nutrient holding capacity due to the loss of SOM as well as a loss of soil structure resulting from the stringent conditions during the washing process (Biabani et al., 2023; Gluhar et al., 2021a; Jelusic and Lestan, 2014; Lee et al., 2021;

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O'Brien et al., 2017). Using gentler, biodegradable chelators (e.g., GLDA), however, can improve nutrient retention thereby maintaining soil fertility (Wang et al., 2016, 2018). Most studies did not report a significant change in soil pH as a result of soil washing, though significant decreases have been noted (Biabani et al., 2023; Yi and Sung, 2015). Soil washing is also generally shown to decrease microbial activities governing nutrient cycling, e.g., enzyme activities and respiration, across most studies (Epelde et al., 2008a; Jelusic and Lestan, 2014; Kaurin et al., 2021; Kaurin and Lestan, 2018; Wang et al., 2016). However, these appeared to be short-term impacts that evened out over time (Kaurin et al., 2021). Also, biodegradable chelates have been shown to result in higher soil enzyme activity than that of EDTA treatment (Wang et al., 2018).

Soil washing can have negative or mixed impacts on indicators for physical stability where mechanical agitation and physical separation during ex-situ soil washing can remove much of the fine particles thereby changing the soil texture and inherent stability, which may increase bulk density, decrease CEC, and decrease finer pores and the formation of macropores and soil aggregates (Gluhar, Kaurin, Vodnik, et al. 2021; Jelusic and Lestan 2014; Lee et al. 2021; O'Brien et al. 2017b). These changes may not support geotechnical properties either due to deterioration of mechanical characteristics, not being well-suited for fine-grained soils, and that it may cause cavities within ground due to infiltration of water (Rehman et al., 2023). The initial loss of natural soil aggregation, which cannot be avoided during soil washing, can result in a lasting reduction in soil stability that also negatively impacts biomass production (Gluhar et al., 2021a). Indeed, several studies have shown reduced germination and growth rates of crops following soil washing (Jelusic and Lestan, 2014; Kaurin and Lestan, 2018; Lee et al., 2021). However, other studies have shown that wheat germination rates increased by 13–40% after soil washing with biodegradable chelators, which was higher than both the original soil and soil washed with EDTA (Wang et al., 2018).

While soil washing reduces total contaminant concentrations, it may also lower the soil's inherent retention capacity to manage residual contaminants by reducing SOM and CEC. EDTA is also poorly biodegradable and remains in the environment, which has raised concerns about the leaching of toxic chelates from remediated soil and the risk of groundwater contamination due to residual metal-EDTA complexes (Biabani et al., 2023; Gluhar et al., 2021b; Lee et al., 2021; Wang et al., 2018). Some studies have, however, shown that bioavailable metal fractions and leaching decrease over a period of time after washing with various chemicals and shift to more chemically stable forms that are less toxic and available for uptake into plants thereby having positive impacts on biomass production (Gluhar et al., 2021a; Jelusic and Lestan, 2014; Wang et al., 2018).

Habitat provision is commonly impaired following soil washing. Many studies have shown reductions in microbial biomass, especially with more toxic chemicals such as EDTA, but others show mixed effects across studies depending on the type of microbial biomass measured (C, N or P), soil type and chemicals used (Jelusic and Lestan, 2014; Kaurin et al., 2020; Wang et al., 2018). Washing with EDTA is expected to have larger negative impacts than washing with biodegradable chelators (Kaurin et al., 2020; Wang et al., 2018). Mycorrhizal fungi colonization may also greatly be reduced following soil washing (Jelusic and Lestan, 2014; Kaurin et al., 2021), but it can be recovered after reinoculation with commercial and native fungal inoculum (Kaurin et al., 2021). Studies have reported changes in DNA content and microbial population structure as a result of soil washing (Lee et al., 2021). For instance, DNA fingerprinting showed changes in the microbial community composition seven weeks after soil washing that varied depending on the type of soil, acidic or calcareous (Kaurin and Lestan, 2018). Adverse effects have also been observed in earthworms after soil

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washing, which were related to changes in soil pH, EC, TN, and TP (Kwak et al., 2019). Despite the removal of metal contaminants, recently washed soil was shown to not be healthy for the earthworms due to changes in soil physical and chemical properties, and remediated soil needs time to recover from the remediation before habitat quality and function are restored (Kwak et al., 2019). There may also be a potential toxicity that impacts such species from residual extractants remaining after washing (Lee et al., 2021), particularly for persistent, non-biodegradable chemicals like EDTA.

3.1.3 In-situ chemical oxidation (ISCO) or reduction (ISCR)

Chemical remediation uses reactive agents to degrade or stabilize contaminants in soil or groundwater. It is typically divided into in-situ chemical oxidation (ISCO), employing oxidants such as Fenton's reagent, hydrogen peroxide, and permanganate, and in-situ chemical reduction (ISCR), which relies on reducing agents such as zero-valent iron (ZVI) and other iron minerals (Kuppusamy et al., 2017; Lim et al., 2016; Lima et al., 2017; O'Brien et al., 2017). The impacts on soil functions vary according to the reagent used, as different chemical treatments can significantly influence soil properties and processes.

Chemical remediation commonly alters or degrades soil organic matter (SOM) due to the non-selective nature of oxidation and reduction processes (O'Brien et al., 2017). Oxidants such as Fenton's reagent (H_2O_2) can reduce SOM by up to 80% (O'Brien et al., 2017; Sirguey et al., 2008), although some studies have reported little or no change in SOM content or distribution following treatment (Romero et al., 2011). Comparable reductions have also been observed following ISCR using ferrous sulphate (Liao et al., 2014). SOM degradation often increases dissolved organic carbon (DOC) and nutrient availability, which may temporarily stimulate microbial respiration but can also divert microbial activity away from target contaminants, potentially slowing biodegradation (O'Brien et al., 2017; Sutton et al., 2014). In contrast, ferrous sulphate treatment may reduce available phosphorus through immobilization by iron compounds (Liao et al., 2014). Chemical remediation can also cause substantial shifts in soil pH, although the magnitude and direction depend on the reagent applied. Fenton's reagent and persulfate typically acidify soils, in some cases to approximately pH 3 (Liu et al., 2019; O'Brien et al., 2017; Usman et al., 2022) whereas permanganate and iron-based treatments can increase pH to around 10 (O'Brien et al., 2017; Sahl and Munakata-Marr, 2006; Usman et al., 2022; Wu and Lu, 2021). Such pH changes frequently suppress pH-sensitive microbial communities, resulting in reduced respiration and enzyme activity immediately after treatment. However, microbial recovery is often observed once reactive agents are depleted, particularly following lower or repeated doses of H_2O_2, with similar trends reported for ferrous sulphate treatments (Liao et al., 2014; O'Brien et al., 2017; Polli et al., 2018; Venny and Ng, 2014). Chemical remediation does not seem to significantly impact EC (Venny and Ng, 2014).

Loss of SOM and aggregation generally decrease water retention and physical stability by altering soil porosity and permeability with potential clogging by residual precipitates such as permanganate in fine soils (O'Brien et al., 2017; Romero et al., 2011; Usman et al., 2022). In some studies, however, Fenton oxidation has been reported to improve water retention capacity (Sirguey et al., 2008). Impacts on geotechnical properties are mixed and depend on the application, e.g., the injection or thrusting methods used could deteriorate the mechanical response of treated soil alongside SOM loss without compaction (Liu et al., 2021; Rehman et al., 2023) but some chemicals such as persulfate could lead to localized gains in the undrained shear strength (Liu et al., 2019).

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While target contaminants are degraded, the large changes in pH, SOM loss and increased DOC can enhance metal solubility and leaching (e.g., Zn, Cu, Mn) thus impacting retention and water quality (O'Brien et al., 2017). The negative impacts on the microbial community can also be a major hurdle to coupling ISCO with in-situ bioremediation (Sutton et al., 2014), though the recovery of microbial biomass and/or bioremediation activity can be expected over time and hastened with the addition of amendments and nutrients necessary for biodegradation (Sahl and Munakata-Marr, 2006; Sutton et al., 2014).

Chemical remediation methods generally adversely impact habitat provision with reduced microbial diversity and abundance due to a combination of direct chemical toxicity and strong changes in soil pH and SOM (Huang et al., 2024; Polli et al., 2018; Usman et al., 2022). Fenton-like treatment causes a dramatic decrease in bacterial density, diversity, and functionality immediately after treatment, with hydrogen peroxide functioning as a biocide, but recovery over time is frequently observed (Polli et al., 2018; Usman et al., 2022). In general, shifts in community composition can be expected due to changing conditions and species sensitivity (Huang et al., 2024; Wu and Lu, 2021), some studies suggest milder chemicals like permanganate may not significantly alter the site's microbial community structure (Azadpour-Keeley et al., 2004). Plant responses are mixed with many studies showing a generally diminished capacity for biomass production following chemical remediation due to residual toxicity, especially at high doses, and lower soil fertility that delays or reduces germination as well as decreasing root and shoot biomass (O'Brien et al., 2017; Usman et al., 2022), though some studies have showed a positive impact on seed germination and ryegrass growth following Fenton oxidation (Sirguey et al., 2008).

3.1.4 Electrokinetic remediation

Electrokinetic remediation applies low-intensity, direct electric current between the cathode and anode electrodes installed in the contaminated soil through which dissolved metal ions in pore water are transported via electroosmosis, electromigration, and electrophoresis enabling dissolution and removal at the electrodes (Hamdi et al., 2025; Kuppusamy et al., 2017; Lima et al., 2017; O'Brien et al., 2017). The imposed electric field can significantly alter soil properties through redistribution of ions and water and that can create significant spatial heterogeneity in soil function that generally intensifies with higher voltages (O'Brien et al., 2017).

No significant direct impacts to SOM from electrokinetic remediation were found across the reviewed studies. However, microbial responses are mixed and highly spatially heterogeneous depending on electric field intensity, electrolyte composition, and pH gradients. Indeed, a ubiquitous consequence of electrokinetic remediation is the creation of a steep pH gradient between the anode (acidic) and cathode (alkaline) in the soil ranging from pH 3.5-3.8 near the anode to pH 7.9-10.8 near the cathode (Kim et al., 2010; Lear et al., 2004, 2007; O'Brien et al., 2017), although alternating current direction and combining on/off cycles can prevent the formation of strong pH gradients (Ferreira et al., 2020; Johansson et al., 2025). Studies report both stimulation and inhibition of respiration, enzyme activity and carbon substrate utilization (i.e., functional diversity), with the most pronounced adverse effects typically occurring in strongly acidified zones near the anode and at higher voltages (Cang et al., 2007, 2012; Kim et al., 2010; Lear et al., 2004, 2007; Pazos et al., 2012). In general, the spatial distribution of microbial activity is highly uneven that may lead to hotspots of nutrient cycling and SOM degradation (O'Brien et al., 2017). Regarding specific electrolytes, EDTA reduce microbial activity, particularly near the cathode, whereas enhanced dehydrogenase activity and increases in some microbial populations have been observed near the anode (Kim et al., 2010), and the elevated pH combined with NaClO markedly reduced

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microbial functional diversity in a Cr-contaminated soil (Cang et al., 2007). Combining electrokinetic remediation with phytoremediation may partially offset electrokinetic-induced stress (Cang et al., 2012). The electric field also increases the mobility of nutrients and redistributes them towards the anode or cathode depending on charge (e.g., N and P towards the anode; K towards the cathode) (O'Brien et al., 2017; Pazos et al., 2012). One study showed that after 60h of electrokinetic remediation, the available N, P, and K in soil were increased by an average of 0.44-, 3.31-, and 1.25-fold, respectively (Chen et al., 2006).

There are limited studies evaluating the impacts of electrokinetic remediation on water regulation and retention. One study suggests that electrokinetic treatment can redistribute moisture through electroosmosis causing drying near the anode and corresponding increase in moisture near the cathode (Kim et al., 2010). Regarding physical stability, such shifts may increase risks of compaction and reduced permeability due to temperature increases and moisture changes, but there is little published information about the impacts of electrokinetic remediation on soil structure. Electrokinetic remediation can, however, have positive impacts on the geotechnical properties of soils. For example, it can improve the permeability of textured soils and stimulate chemical processes and precipitation in the pores to stabilize the mechanical characteristics of soil (Rehman et al., 2023). Electrokinetic remediation has also been shown to improve the unconfined compression strength by 30-100% in saline soils, thereby improving soil structure and other mechanical properties such as the stress-strain characteristics (Jayasekera and Hall, 2007).

Electrokinetic remediation effectively mobilizes and removes metals and other contaminants, particularly for low-permeability soils and at high voltages but may have mixed impacts on the soil's inherent contaminant retention and degradation. The increased solubility and mobility can potentially lead to leaching or increased ecotoxicity of metals due to pH changes and contaminant degradation can vary across the electric gradient (O'Brien et al., 2017). Cell architecture also significantly influences the fate of contaminants, with undivided (1-compartment) designs favoring electrodegradation within the soil matrix rather than mobilization to electrolyte compartments (Ferreira et al., 2017).

Regarding habitat provision, the impacts of electrokinetic remediation on soil microorganisms can vary considerably, but many studies show that the only significant change in the microbial community occurred in soil immediately next to the anode where the strong acidification is generally expected to reduce microbial cell number and microbial diversity (Kim et al., 2010; Lear et al., 2007). In uncontaminated soil, the electric current alone (3.14 A/m^2) did not significantly harm soil microbes (Lear 2004), whereas in PCP-contaminated soil, culturable bacteria and fungi decreased by 17%, and 30%, respectively alongside reduced microbial respiration and carbon substrate utilization, particularly near the acidic anode where PCP accumulated (Lear 2007). Despite the increased availability of nutrients in some areas, electrokinetic remediation can have direct adverse impacts on biomass production, e.g., decreased root elongation by up to 25% in *Brassica juncea* when exposed to a charge potential difference above $2Vm^{-1}$ (O'Brien et al., 2017). Other studies, however, have shown that lower-intensity electrical current combined with phytoremediation can improve remediation effectiveness and plant growth while mitigating some of the negative impacts on soil properties (Cang et al., 2012; Hamdi et al., 2025).

3.1.5 Multi-phase extraction

Multi-phase extraction (MPE), also referred to as bioslurping, vacuum-enhanced extraction, or dual-phase extraction (DPE), combines vacuum-assisted recovery of free-phase product and/or groundwater with bioventing

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and soil vapor extraction to remediate contaminants in the vadose zone, capillary fringe, and shallow saturated zone (Technology Screening Matrix | Federal Remediation Technologies Roundtable (FRTR), 2025; Kuppusamy et al., 2016). Evidence regarding the effects of MPE on soil functions is scarce, and the literature review identified little to no information on its impacts on soil organic matter storage and cycling, nutrient cycling, physical stability, or habitat provision. Regarding water regulation, the impacts of MPE are not directly addressed in the literature yet vacuum extraction may alter soil moisture conditions and cause compaction, reducing permeability and infiltration. Lowering the water table can increase oxygen availability in previously saturated soils, potentially enhancing aerobic degradation of organic contaminants such as PAHs, while also promoting contaminant volatilization.

3.1.6 Air sparging

Air sparging, also referred to as soil venting or vacuum extraction, is an in-situ remediation technique that injects atmospheric air into contaminated soil to volatilize organic contaminants and stimulate aerobic biodegradation through oxygen addition (bioventing) (Bass et al., 2000; Benner et al., 2002; Lim et al., 2016). Air may be injected into either the saturated or unsaturated zone using horizontal or vertical wells, depending on site conditions. Soil vapor extraction (SVE) wells are typically used to recover injected air and volatilized contaminants for treatment. Remediation performance is influenced by factors such as soil moisture, organic matter content, air permeability, and temperature, with coarse-grained soils such as sand and gravel generally being most suitable (Albergaria et al., 2008, 2012; Benner et al., 2002; Clayton et al., 1996; Lee et al., 2002; Lim et al., 2016). Although evidence on the effects of air sparging on soil functions remains limited, available studies suggest mixed impacts on several soil quality indicators. No information was identified regarding effects on physical stability.

The addition of oxygen to soil enhances aerobic biodegradation rates of contaminants as well as organic matter degradation (Bass et al., 2000; Lee et al., 2002; Malina et al., 2002; Nguyen et al., 2023). For instance, since nitrification is an aerobic process air sparging can enhance the rate of nitrogen fixation in the soil, but it also depends on other factors (e.g., available nutrients for microbes, water content, pH) (Pan et al., 2022). Denitrification occurs mainly in anaerobic conditions, so air injection decreases nitrogen loss in gaseous form. Nitrogenase enzymes are sensitive to oxygen as oxygen inactivates the enzyme, meaning increased oxygen concentrations may decrease the rate on nitrogen loss for the duration of the remediation (Pan et al., 2022). However, no quantitative research on the impacts of air sparging on nutrient cycling was found.

Injecting oxygen into the saturated zone may lead to precipitation of dissolved iron and manganese if they are present, which may form oxides and cause blocking of pores. This has implications for contaminant retention as it may in turn lead to slower groundwater flow and decrease contaminant transport. The impacts on contaminant retention and transformation are mixed. For instance, air sparging can greatly enhance contaminant degradation and transformation through the addition of oxygen present in the injected air boosting microbial activity and may also enable its transition from aqueous phase to gaseous phase leading to its volatilisation from the soil (Benner et al., 2002). However, the increased degradation of SOM may decrease the soil's retention capacity.

There is little information regarding the impacts of air sparging on habitat provision although a short-term increase in microbial activity in the soils leading to increased biodegradation can be expected (Lim et al., 2016). As the air is injected with some pressure, water is displaced and porosity increased thereby making more

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contaminants available for consumption as well as offering more pore space as habitat for microbes (Bass et al., 2000). This effect would last only the duration of air injection, after which the water table returns to previous level.

3.1.7 Containment, capping or barrier techniques

Containment, capping, and barrier techniques are in-situ remediation methods used to prevent contaminant migration and minimize exposure risks when other remediation options are not feasible or cost-effective. Physical barriers such as slurry walls, geosynthetic liners, and capping systems isolate contaminated soil by limiting water infiltration and contaminant transport, while also reducing dispersal through wind and water erosion (Bhattacharyya et al., 2010; Meuser, 2013; Padhye et al., 2023; Roy et al., 2023). Their impacts on soil functions vary with system design: localized measures such as vertical barriers are likely to have limited effects, whereas complete soil sealing can result in the loss of most soil functions. Available evidence is scarce and suggests mixed impacts overall, with no relevant information identified for organic matter storage and cycling or habitat provision.

Vertical permeable reactive barriers generally have limited effects on water regulation because they permit continued infiltration and groundwater flow. Geomembranes, however, may become less permeable over time due to clogging or biofouling, thereby reducing infiltration (Meuser, 2013). Similarly, soil capping is often intended to restrict infiltration and may lead to drying of the underlying soil, creating less favorable conditions for plants and microorganisms and fragmenting soil habitats (Tobias et al., 2018). Permeable containment systems are unlikely to substantially affect nutrient cycling, whereas impermeable caps can effectively eliminate habitat for soil biota. However, sealing with concrete or asphalt may be justified where future land use prioritizes physical stability, such as roads or other heavily loaded infrastructure (Meuser, 2013).

3.1.8 Stabilization/Solidification (S/S)

Soil stabilization/solidification (S/S) aims to both decrease contaminant leaching and improve the mechanical properties of weak or water-sensitive soils for engineering applications (Dermatas and Meng, 2003; Paria and Yuet, 2006; Zhou et al., 2024). S/S typically combines contaminant-stabilizing amendments with cementitious binders such as cement, lime, fly ash, or industrial by-products. Through physical encapsulation, sorption, and cementation, contaminants become strongly immobilized within the soil matrix while soil strength increases (Dermatas and Meng, 2003; Paria and Yuet, 2006; Zhou et al., 2024). Consequently, contaminant retention is greatly enhanced, although most other soil functions are adversely affected by the resulting soil cementation.

S/S can increase the apparent stability of soil organic matter (SOM), but this should not be interpreted as enhanced carbon sequestration. During stabilization, organic matter becomes physically encapsulated within cement hydration products (e.g., C–S–H gels and ettringite), reducing its accessibility to microorganisms and enzymes and disconnecting it from natural soil processes (Zhao and Wong, 2025). S/S also decreases soil permeability and pore connectivity (Paria and Yuet, 2006), restricting oxygen diffusion and substrate transport and thereby limiting microbial activity. The addition of cement can raise soil pH to values exceeding 12 (Zhou et al., 2024), with elevated pH associated with lower nutrient availability and adverse effects on soil microorganisms (Mao et al., 2023). High cement contents may further reduce soil fertility by suppressing nitrogen mineralization and release, ultimately limiting plant growth and nutrient cycling (Caravaca et al., 2017).

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By reducing permeability, S/S generally impairs the water regulation function. Water infiltration is often substantially reduced, which is frequently an intended outcome of the treatment (Zhou et al., 2024). Cementitious binders such as Portland cement, slag, and fly ash undergo hydration and pozzolanic reactions that fill pore spaces, particularly larger pores, and alter soil structure (Zhou et al., 2024). The resulting reductions in water and substrate availability further suppress microbial activity, causing organic matter turnover to shift from biologically mediated processes toward slower abiotic ageing and carbonation pathways. From a geotechnical perspective, however, S/S is a widely used ground improvement technique that enhances soil strength and physical stability (Zhao & Wong 2025).

Overall, S/S severely impairs soil ecosystem functioning. Cement addition often causes a sharp decline in microbial biomass and long-term inhibition of microbial recolonization due to high pH and cementitious mineralogy. Reduced pore space limits or completely blocks water movement, while both contaminants and organic matter become physically bound within the stabilized matrix, making them inaccessible to soil organisms. Plant establishment is also strongly restricted because roots are generally unable to penetrate cemented soil unless cracks are present. As a result, habitat provision, biological activity, and biomass production are substantially diminished.

3.2 Impacts of low-impact remediation techniques (LIRT) on soil functions

Table 3 presents the aggregated, average expected impacts of conventional remediation techniques on the six main soil functions. The following sections will provide a summary of the impacts of specific remediation techniques on the six soil functions based on changes in individual indicators (see Table S3 in the SM and ISLANDR D2.3 for a more complete overview).

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Table 3. Impacts of low-input remediation techniques (LIRT) on soil functions (color coding according to Table 1).

Soil functions \ Remediation techniques	Bioremediation (in-situ/ex-situ*)	Phytoremediation (all, in-situ)	Monitored natural attenuation/NSZD (in-situ)	Immobilisation with amendments (organic, in-situ)	Immobilisation with amendments (inorganic, in-situ)
Organic matter storage, transformation and recycling	+ or +/-	++	+	++	+/-
Water regulation, retention and release	+	++	+/-	+	+/-
Nutrient cycling	+/-	+	+/-	+	+/-
Contaminants retention, transformation and degradation	+/-	+	+/-	+	+
Physical stability	+ or +/-	+	+/-	+	+
Habitat provision	+ or +/-	+	+	+	+/-

++ : clear or highly positive impact; + : somewhat positive impact; +/- : mixed or no impact; — : somewhat negative impact; — — : clear or highly negative impact
*Excavation is generally considered to have a deleterious effect on soil functions. Ex-situ treatment and replacement can restore some soil functionality, but most functions will be lost and additional treatment could further reduce functionality.

3.2.1 Bioremediation

Bioremediation relies on indigenous (biostimulation) or introduced (bioaugmentation) microorganisms to reduce contaminant concentrations through degradation, transformation, detoxification, or stabilization. It includes both in situ techniques, such as biostimulation and bioaugmentation, and ex situ methods, such as landfarming and biopiles (Kuppusamy et al., 2016, 2017; O'Brien et al., 2017). By optimizing conditions that support microbial activity, bioremediation can improve contaminant breakdown, although its effects on soil functions vary depending on the specific approach and any amendments applied. Emerging techniques such as mycoremediation and vermiremediation are not considered further in this review.

Biostimulation commonly involves the addition of organic amendments such as compost or manure, which increase soil organic matter (SOM) and support microbial activity (Lacalle et al., 2020; O'Brien et al., 2017). Although bioremediation often causes an initial increase in total organic carbon (TOC), concentrations typically decline over time as contaminants and added organic matter are degraded (Polyak et al., 2018). By supplying organic matter and nutrients, bioremediation stimulates microbial biomass, activity, and the transformation of organic matter while enhancing the degradation of contaminants such as petroleum hydrocarbons (O'Brien et al., 2017). Consequently, microbial respiration and enzyme activities associated with organic matter mineralization generally increase (Lacalle et al., 2020). For instance, aeration can stimulate microbial processes involved in biodegradation (Militon et al., 2010), while biostimulation and bioaugmentation have been shown to increase the activity of enzymes involved in carbon cycling, including lipase and fluorescein diacetate hydrolase (Cui et al., 2020). Similar increases have been reported for enzymes linked to nutrient cycling, such as urease, catalase, and dehydrogenase (Cui et al., 2020; Polyak et al., 2018; Shen et al., 2016). However, these effects are often most pronounced during the early stages of treatment and may diminish over time as soils recover. Responses also vary between biostimulation and bioaugmentation approaches, with one comparative study identifying biostimulation as the more effective strategy (Polyak et al., 2018). Bioremediation can further enhance nutrient cycling through increased microbial activity and amendment addition (O'Brien et al., 2017). Because nutrient availability frequently limits biodegradation, nutrient supplementation may be as effective as bioaugmentation for promoting contaminant degradation. Available nutrients such as N, P, and K typically increase following amendment application but are gradually consumed during microbial growth and biodegradation, leading to declining concentrations over time (Polyak et al., 2018). In a nine-year field study, substantial increases in available phosphorus and inorganic nitrogen were observed initially under both biostimulation and bioaugmentation treatments, but concentrations later declined to levels comparable to the control soil (Polyak et al., 2018). As a result, nutrient depletion during biodegradation may ultimately reduce nutrient availability for plant growth.

Addition of organic soil amendments in biostimulation usually increases SOM leading to improvements in soil structure through increased water holding capacity, porosity, and soil aggregate stability (O'Brien et al., 2017). Bioremediation may enhance soil structure and physical stability, leading to improved geotechnical properties. For instance, bioaugmentation of an oil-contaminated clay soil increased shear strength and cohesion while reducing maximum dry density, effects linked to the formation of porous and quasi-fibrous soil structures (Salimnezhad et al., 2021).

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Although bioremediation generally promotes microbial activity and improves soil physicochemical properties, thereby enhancing the biodegradation potential of indigenous microorganisms (Polyak et al., 2018), incomplete contaminant degradation may generate toxic intermediate metabolites (O'Brien et al., 2017). These compounds can contribute to residual ecotoxicity and adversely affect soil biota. For example, biodegradation of petroleum hydrocarbons has been associated with the formation of bioavailable toxic metabolites that negatively impacted both plant growth and microbial communities (Shen et al., 2016). Nevertheless, when remediation is successful, reductions in ecotoxicity can be achieved; bioremediation using an *Actinobacteria* consortium significantly improved the survival and growth of *Eisenia fetida* in soils co-contaminated with Cr(VI) and lindane (Lacalle et al., 2020).

When effective, bioremediation practices such as compost addition, aeration, irrigation, and fertilization generally increase microbial abundance and diversity, thereby enhancing habitat provision (O'Brien et al., 2017). However, these interventions may also selectively favor contaminant-degrading microorganisms and associated functional genes involved in the breakdown of compounds such as PAHs and lindane, potentially reducing overall microbial diversity (Cui et al., 2020; Lacalle et al., 2020; Wolf et al., 2019). For example, oxygen biostimulation has been shown to shift community composition toward a limited number of well-adapted phylotypes, including a transition from *Gammaproteobacteria* to *Actinobacteria* dominance (Militon et al., 2010). Similarly, in petroleum hydrocarbon-contaminated soils, both biostimulation and bioaugmentation increased bacterial alpha diversity, with the combined approach producing the greatest diversity while enriching hydrocarbon-degrading groups such as *Proteobacteria* (Cui et al., 2020). Overall, bioremediation often promotes specialist degraders and stress-tolerant microorganisms, although the broader implications for soil functioning remain uncertain (O'Brien et al., 2017).

3.2.2 Phytoremediation

Phytoremediation employs plants and their associated microorganisms to reduce, stabilize, degrade, or extract contaminants through processes such as phytoextraction, phytostabilization, rhizodegradation, phytovolatilization, and rhizofiltration (Couto et al., 2011, 2012; Cundy et al., 2016; Drenning et al., 2022). As a widely recognized gentle remediation option (GRO), it supports sustainable contaminated land management by improving soil functions and generating broader ecosystem benefits (Burges et al., 2018; Cundy et al., 2016). In particular, phytoremediation combined with organic amendments can increase SOM, improve soil structure and nutrient retention, and enhance carbon storage, leading to improved soil fertility and ecosystem functioning (Fiorentino et al., 2018; Lacalle et al., 2018a). Notably, improvements in soil health are often observed even when reductions in total contaminant concentrations are limited (Anza et al., 2019).

Phytoremediation, particularly when combined with organic amendments such as compost, manure, or biochar, generally increases soil organic matter (SOM) and nutrient availability (Adl, 2008; Ciadamidaro et al., 2014; O'Brien et al., 2017). For example, the application of compost and dolomitic limestone during phytomanagement of Cu-contaminated soils improved both nutrient availability and plant growth (Burges et al., 2021; Mench et al., 2023). Perennial vegetation, commonly used in phytomanagement, further promotes carbon sequestration by minimizing soil disturbance and incorporating plant residues and root-derived carbon into SOM (Adl, 2008). Soil pH often trends toward neutral conditions during phytoremediation, although amendments can modify this response, either lowering pH through increased organic matter inputs or raising it through liming

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(O'Brien et al., 2017). Phytoremediation also commonly enhances microbial biomass and activity, including respiration and enzyme activities involved in nutrient and carbon cycling (Ciadamidaro et al., 2014; Epelde et al., 2008b, 2009; Gómez-Sagasti et al., 2012; Kumpiene et al., 2009; Lacalle et al., 2018a; Míguez et al., 2020; Touceda-González et al., 2017b, a). Reported increases in enzymes associated with C, N, P, and S cycling have reached 164% during phytoextraction with *Thlaspi caerulescens* (Epelde et al., 2008b) and 675% during aided phytostabilization with *Lolium perenne* (Epelde et al., 2009). Similar improvements have been observed across numerous studies, particularly where phytoremediation is combined with organic amendments (Burges et al., 2016, 2017; Garaiyurrebaso et al., 2017; Gómez-Sagasti et al., 2012; Kumpiene et al., 2009; Macci et al., 2013, 2016; Touceda-González et al., 2017a). Additional strategies, including mycorrhizal-assisted phytoremediation, intercropping with legumes, and inoculation with plant growth-promoting rhizobacteria (PGPR), can further enhance nutrient cycling and nutrient availability through stimulation of microbial processes and enzyme activity (Gómez-Sagasti et al., 2021; Ju et al., 2019; Marchand et al., 2016). Some studies have also reported increased carbon substrate utilization based on community-level physiological profiles, indicating enhanced functional diversity during phytoremediation (Epelde et al., 2008b; Gómez-Sagasti et al., 2021).

Phytoremediation together with organic amendments can improve soil structure and stability by increasing soil organic matter (SOM) and promoting root development, which enhances porosity, aggregation, water-holding capacity (WHC), and infiltration (Adl, 2008; Anza et al., 2019; Fagnano et al., 2020; Fiorentino et al., 2018; Marchand et al., 2016; Míguez et al., 2020). Several studies have shown that phytomanagement in combination with organic amendments improves ecosystem services such as water flow regulation through increased WHC (Burges et al., 2016, 2017, 2018; Mench et al., 2023). Phytoremediation may also improve physical properties such as bulk density, although substantial short-term improvements typically require additional soil amendments. Root systems can further enhance geotechnical properties, including shear strength and slope stability. However, increased SOM and decaying root biomass may reduce geotechnical serviceability for future construction unless additional ground improvement measures are implemented(Rehman et al., 2023).

By establishing vegetation cover and extensive root systems, phytoremediation can reduce erosion and limit off-site contaminant transport, while also decreasing contaminant leaching through enhanced water uptake and hydraulic control (Cundy et al., 2016; Drenning et al., 2022; Fiorentino et al., 2018). Poplar plantations, for example, can reduce contaminant migration by modifying site water balances through transpiration (El-Gendy et al., 2009; Fiorentino et al., 2018). In addition, phytoremediation promotes microbial activity and improves soil conditions, supporting the degradation of organic contaminants through phyto- and rhizodegradation (Gómez-Sagasti et al., 2021; Lacalle et al., 2018a; Macci et al., 2016). However, contaminant migration remains a potential concern because remediation is often a long-term process (O'Brien et al., 2017).

Phytoremediation generally improves habitat provision by supporting the recovery of microbial, faunal, and plant communities (Adl, 2008; Garaiyurrebaso et al., 2017; Touceda-González et al., 2017b). Increased microbial biomass, activity, and functional diversity are commonly reported and are often linked to root exudates and organic matter inputs (Epelde et al., 2008b; Lacalle et al., 2018a; Míguez et al., 2020). Although phytoremediation can alter microbial community composition and stimulate functional groups involved in nutrient cycling, increases in overall diversity are not always observed (Burges et al., 2020; Touceda-González et al., 2017b). By reducing contaminant bioavailability and ecotoxicity, phytoremediation can also improve habitat

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quality for plants and soil organisms, including nematodes and earthworms (Enell et al., 2016; Foucault et al., 2013; Marchand et al., 2016; Quintela-Sabarís et al., 2017).

3.2.3 Monitored natural attenuation or natural source zone depletion

Monitored natural attenuation (MNA) and natural source zone depletion (NSZD) depend on naturally occurring physical, chemical, and biological processes to reduce contaminant concentrations, toxicity, and mobility in soil and groundwater (Song et al., 2023; Wozney et al., 2022). Although considered passive remediation strategies, they require comprehensive site assessment and monitoring to verify that remediation objectives can be achieved within an acceptable timeframe (Chapelle et al., 2007; Declercq et al., 2012). ompared with active remediation technologies, MNA and NSZD cause little disturbance to soil systems and are therefore generally expected to maintain, or potentially improve, soil functionality (Meuser, 2013). However, evidence regarding their specific impacts on soil functions remains limited.

The impacts on organic matter storage and cycling on the site are expected to be positive, as the microbes on site use the contaminant as an energy source (in addition to other organic matter present) consuming it and transforming it into their own biomass. When there is oxygen present, microbes convert the organic matter and contaminant into carbon dioxide, water and biomass. Biodegradation consumes oxygen and in anaerobic conditions the organic matter that already exists in the soil degrades more slowly on its own, as microbes use also other energy sources, e.g. iron, nitrate or sulphate, to degrade organic compounds (Declercq et al., 2012). The resulting compounds from biodegradation can be recycled and used by other organisms on site (Smith et al., 2022), and the organic contaminant enters the carbon cycle (Balland-Bolou-Bi et al., 2023; Chapelle et al., 2007). Similarly, the cycling of other nutrients may be expected to continue during MNA/NSZD.

The site water balance and retention capacity is likely unaffected by MNA/NSZD, though the degradation of a hydrophobic contaminant like oils may improve the water balance since oily substances may prevent infiltration of rainwater (E. Al-Esawi et al., 2021). Similarly, MNA/NSZD is not expected to impact physical stability except for possible small positive changes when organic contaminants are removed that could improve the friction between mineral grains on site. If plants are growing on the site, they may help to reduce topsoil erosion since they are not removed or damaged during remediation.

Regarding contaminant retention and transformation, the inherent capacity of the soil is the main determining factor for whether MNA/NSZD is a viable strategy, and the suitability of the method must be ensured prior to implementation. Successful contaminant degradation or retention on site can lead to slower migration of the contaminant plume and gradual reduction of the source term. The retention and degradation rates must be monitored and ensured to be on a suitable level to not cause a risk of exposure further from the site to the direction of plume travel.

Since MNA/NSZD does not interfere with the inherent soil conditions, habitats are conserved for the macrofauna and flora. However, if the contaminant concentration is too high and exposed to biota, MNA/NSZD could be considered to deteriorate the habitat if the remediation rate is not enough to reduce the risk to an acceptable level or result in toxic intermediates.

3.2.4 Immobilization with organic amendments

Organic amendments are widely used for in-situ chemical stabilization of contaminated soils because they can simultaneously reduce contaminant mobility and improve soil quality. Materials such as compost, biochar, manure, and biosolids decrease contaminant bioavailability through sorption, complexation, precipitation, and redox processes, while enhancing soil organic matter, fertility, water retention, structure, and biological activity (Drenning et al., 2022; Kumpiene et al., 2019; Lwin et al., 2018). These properties make organic amendments particularly valuable for aided phytostabilization and ecological restoration of degraded post-mining soils and tailings (Galende et al., 2014; Garaiyurrebaso et al., 2017; Gómez-Sagasti et al., 2018; Kidd et al., 2015). Nevertheless, amendment performance varies considerably depending on material type and production characteristics, such as biochar feedstock and pyrolysis conditions, which influence their effects on soil functions (Handiso et al., 2024; Lehmann et al., 2011; Schröder et al., 2018). Given the breadth of the literature and the context-dependent nature of amendment performance, only the main trends reported for soil quality indicators and soil functions are summarized here.

The addition of organic amendments is a standard agronomic practice in organic agriculture and soil restoration that increases SOM and nutrient content thus improving plant establishment and growth (Bai et al., 2018; Gómez-Sagasti et al., 2018). Biochar, for example, can enhance nutrient acquisition and has a high cation exchange capacity (CEC) (Schröder et al., 2018); however, it has also been shown to immobilize plant-available forms of inorganic nitrogen (Brtnicky et al., 2021; Drenning et al., 2024; Rijk et al., 2024), which can become a limiting growth factor for biomass production. Some organic soil amendments such as biochar and digestate will increase soil pH, especially in acidic soils thus making the soil more suitable for certain crops, while certain types of compost have been shown to lower pH (Schröder et al., 2018). However, the change in soil pH can be transient depending on the amendment used and type of soil (Epelde et al., 2014). Overall, organic amendment addition is expected to increase soil microbial biomass and activity, since adding organic matter increases the nutrient supply for microorganisms, thus improving nutrient and carbon cycling and storage (Epelde et al., 2009; Gómez-Sagasti et al., 2018; Kidd et al., 2015; Kumpiene et al., 2009; Míguez et al., 2020; Touceda-González et al., 2017a). Several studies have demonstrated this when applying phytomanagement together with amendments like compost (Burges et al., 2016, 2017; Gómez-Sagasti et al., 2012; Kumpiene et al., 2009; Macci et al., 2013, 2016; Touceda-González et al., 2017a), but amendments applied without plants also have stimulating effects, e.g., biochar (Abou Jaoude et al., 2020; Bera et al., 2016) and lime-treated sewage sludge (Epelde et al., 2014) for chemical stabilization. Some studies have also shown that organic amendment addition can increase carbon substrate utilization (i.e., functional diversity), e.g., using Biolog community-level physiological profiles to evaluate the application of 3% biochar (Abou Jaoude et al., 2020), different doses of lime-treated sewage sludge (Epelde et al., 2014), and different organic amendments where results varied significantly (Galende et al., 2014).

Increasing SOM through the addition of organic amendments directly improves soil structure for increased stability, porosity and aggregation as well as improved water holding capacity (WHC) and infiltration (Bai et al., 2018; Fagnano et al., 2020; Fiorentino et al., 2018; Gómez-Sagasti et al., 2018; Marchand et al., 2016; Míguez et al., 2020). For example, combining organic matter amendments with calcium-carbonate rich materials was shown to stimulate soil formation at acidic mine tailing deposits by building SOM and accelerating the establishment of a functional ecosystem (Zanuzzi et al., 2009). Incorporating organic matter can also reduce soil bulk density thereby improving soil aeration and root penetration (Schröder et al., 2018). Significant additions of

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SOM likely also prevents erosion by improving or stabilizing the bulk soil structure, porosity, and water holding capacity (Gómez-Sagasti et al., 2018). Organic amendments such as biochar may improve the mechanical characteristics of the soil, e.g., cohesion and compressive strength, and decrease volume change tendency upon wetting and drying (Rehman et al., 2023). However, a complete geotechnical evaluation is required for each amendment, and the substantial addition of SOM may reduce geotechnical properties.

In general, the addition of organic soil amendments can stabilize contaminants to reduce their bioavailability and mobility through mechanisms like complex formation with humic acids or adsorption onto the organic matrix (Drenning et al., 2022; Epelde et al., 2009; Garaiyurrebaso et al., 2017; Kumpiene et al., 2019). However, some organic amendments may also increase mobility and leaching (e.g., dissolved organic carbon from non-stabilized organic amendments or 'green waste compost' may increase metal mobility) and there is a risk of adding contaminants to soil if the source material is contaminated when using, for example, sewage sludge or municipal biosolids (Gómez-Sagasti et al., 2018; Kidd et al., 2015; Lwin et al., 2018). Also, pH changes may selectively mobilize or immobilize certain contaminants such as metal(loid)s such as for biochar which decreases mobility of cations but may increase mobility of anions like arsenic (Bolan et al., 2014). It is likely that transformation and degradation of organic contaminants is also increased by providing supplemental nutrients and a carbon source for degrading microorganisms (Gómez-Sagasti et al., 2018; Lacalle et al., 2018a).

Addition of organic amendments is an integral part of soil rehabilitation that generally restores and enhances soil biodiversity and habitat, including microbial, faunal, and floral communities (Adl, 2008; Touceda-González et al., 2017b). The enhancement of soil microbial biomass, activity, and functional diversity is consistently reported and often attributed to the input of labile organic carbon from amendments and root exudates (Epelde et al., 2008b; Lacalle et al., 2018a; Míguez et al., 2020). Organic fertilization has been linked to increased activity of N-cycling bacteria (Fagnano et al., 2020). The addition of organic amendments can also improve the soil habitat by reducing contaminant bioavailability and ecotoxicity for plants, soil fauna and microorganisms (Gómez-Sagasti et al., 2018; Marchand et al., 2016; Quintela-Sabarís et al., 2017). However, recent studies have shown that certain types of biochar may have adverse, toxic effects on groups of soil organisms such as earthworms due to the presence of potentially harmful substances such as PAHs, ash, or changes in pH, with specific impacts dependent on the dose, feedstock and pyrolysis temperature (Bielská et al., 2018; Brtnicky et al., 2021; Godlewska et al., 2021). Overall, global reviews have shown that addition of organic amendments that increase SOM is generally expected to have positive impacts on soil fauna such as earthworms and increase their abundance (Bai et al., 2018; Gómez-Sagasti et al., 2018).

3.2.5 Immobilization with inorganic amendments

Inorganic amendments such as lime, gypsum, zero-valent iron (ZVI), iron slag, fly ash, bentonite, and zeolites are commonly used for in-situ chemical stabilization to reduce contaminant mobility and bioavailability through sorption, complexation, precipitation, and redox processes, thereby limiting leaching risks (Cui et al., 2023; Kumpiene et al., 2019; Lwin et al., 2018). While these amendments can also improve soil properties, their effects on soil functions differ substantially among materials. Consequently, amendment selection should balance contaminant immobilization effectiveness with potential impacts on soil functionality (Lwin et al., 2018).

The addition of inorganic amendments can have a small positive impact on SOM, but significant improvements depend on their addition together with organic amendments. For example, the addition of iron grit (Lebrun et al.,

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2019) or magnesite (Baragaño et al., 2021) had marginally positive or negative impacts compared to the unamended soils, though much greater improvements were seen in both studies when the iron amendments were applied together with compost or biochar. Some inorganic amendments such as gypsum and phosphates can provide crucial macro- and micronutrients to degraded soils that may otherwise be lacking, and liming may also indirectly induce increased plant nutrient mobilization possibly even leading to nutrient leaching (Lwin et al., 2018). Iron amendments such as nZVI have been shown to reduce the concentration of plant-available P, though this did not seem to cause any phytotoxic effects (Baragaño et al., 2020). Changes in soil pH depend on the amendment with liming materials and gypsum leading to an increase in soil pH, especially in acidic soils (Cui et al., 2023; Lwin et al., 2018), but other amendments such as iron grit and nZVI have limited or negligible impacts on soil pH (Baragaño et al., 2020; Lebrun et al., 2019; Nandillon et al., 2019), and adding ferrous sulfate as the source of Fe-oxides may even decrease soil pH (Kumpiene et al., 2019). Generally, inorganic amendment addition seems to have little to no impact on microbial activities, though studies are limited and the impacts vary depending on the amendment. For example, iron amendments like nZVI do not have any significant effect on soil microbial parameters such as basal respiration, substrate-induced respiration or carbon substrate utilization (Lacalle et al., 2018b, a). Other studies indicated that in soils amended with hydroxyapatite and composite additions, the activities of soil catalase, urease, and acid phosphatase are significantly elevated (Cui et al., 2017). The addition of 0.5% nano-montmorillonite to a Cd-contaminated agricultural soil was also shown to improve soil enzyme activity by reducing Cd toxicity to microorganisms (Liu et al., 2022).

The addition of inorganic amendments (e.g., gypsum, clay minerals and some industrial wastes) can lead an increase in water retention due to an increase in fine particulates and improved soil structure linked to reduced bulk density, increased soil aggregate stability and water holding capacity (Cui et al., 2023; Lwin et al., 2018). For instance, liming and gypsum addition, through increasing Ca saturation, can reduce dispersion of soil particles and promote flocculation to facilitate soil structure development and formation of soil aggregates and stability (Lwin et al., 2018). However, other studies have shown that addition of iron grit without other organic amendments led to a slight but significant decrease in soil water holding capacity compared to unamended technosols, which may be due to iron grit being unable to retain water (Lebrun et al., 2019; Nandillon et al., 2019). Water infiltration may also be improved with inorganic amendment addition using, for example, gypsum or clay minerals due to improvements in soil structure and porosity in a degraded soil (Cui et al., 2023; Lwin et al., 2018), but there is little direct evidence in the literature. Regarding geotechnical properties, inorganic amendments such as cement, lime, fly ash, and other types of cementing additives may improve the mechanical characteristics of the soil, e.g., cohesion and compressive strength, and decrease volume change tendency upon wetting and drying (Rehman et al., 2023). Addition of nZVI has also been shown to induce significant changes to the microstructure of soil, such as increased particle size, bubble prints, and porosity, which led to an increase in vane shear strength, stiffness, friction angle, and plasticity index of clay soil, as well as a decrease in the compression index (Liu et al., 2021).

In general, the addition of inorganic soil amendments should lead to a significant immobilization of contaminants in the soil matrix to reduce bioavailability and (Cui et al., 2023; Kumpiene et al., 2019; Lwin et al., 2018). Indeed the application of inorganic amendments such as nZVI may be necessary to immobilize anions such as As, which may be mobilized by organic amendments like compost and biochar (Baragaño et al., 2020). Addition of nZVI alone, however, may lead to a slight increase in Cu availability (Baragaño et al., 2020).

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There are limited studies assessing the impacts of inorganic amendments on soil microbial biomass and diversity, and consequent implications for habitat provision and biomass production, and the results are highly variable. For instance, studies assessing nZVI application showed that it did not have any significant effect on active soil microbial biomass (Lacalle et al., 2018b, a), while addition of 0.5% nano-montmorillonite to a Cd-contaminated agricultural soil showed a significant improvement in the microbial alpha diversity index and enhancement of the relative abundance of Bacteroidetes and *Planctomycetes* (Liu et al., 2022). Another study showed that hydroxyapatite and Thomas basic slag significantly improved the growth of the total bacterial population and promoted plant production by reducing acute ecotoxicity, with hydroxyapatite reducing plant oxidative stress and improving plant species richness and diversity (Bert et al., 2012). However, iron amendments may negatively impact revegetation as shown for magnesite treatment inhibiting the growth of *Brassica juncea* plants entirely (Baragaño et al., 2021) and amendment with iron grit alone preventing the germination of *Trifolium repens* seeds due to high soluble Fe concentrations and increased acidity in the soil pore water (Nandillon et al., 2019). Indeed, while inorganic contaminants can show great effect for reducing contaminant bioavailability, some may still be toxic for plants and soil organisms depending on the dose. For instance, 1.5% w/w iron grit amendment alone increased soil pore water (SPW) toxicity compared to the unamended control (Lebrun et al., 2019), and the toxicity of nZVI to soil microbial communities and plants was shown to vary depending on the dose (Lacalle et al., 2018b, a). Certain organic components of bone meal (e.g. various proteins), along with their degradation products (e.g. ammonia and nitrate), may also be toxic to soil organisms and have been implicated as the reason for earthworm toxicity following bone meal amendment (Kumpiene et al., 2019). Few studies, however, have assessed the impacts of amendments on soil fauna. Only one study was found that showed neither hydroxylapatite nor Thomas basic slag caused significant differences in the abundance or diversity of soil fauna (Collembola communities) compared to untreated soil (Bert et al., 2012).

4 Discussion

4.1 Evaluating and minimizing the impacts of remediation techniques on soil functions

Overall, the literature indicates that conventional remediation techniques (Table 2) often achieve effective contaminant reduction but can adversely affect soil structure, biological activity, and long-term soil functionality. In contrast, low-impact remediation approaches (Table 3) generally support or enhance SOM, microbial activity, water regulation, and biodiversity, although remediation is typically slower. These broad trends should, however, be interpreted with caution. The magnitude and direction of impacts frequently depend on site-specific conditions, contaminant characteristics, and treatment design. Moreover, many adverse effects associated with conventional techniques can be reduced through process optimization, including adjustment of operating conditions (e.g., temperature or treatment duration) and selection of less disruptive reagents or amendments.

In thermal remediation, temperature is the primary factor controlling impacts on soil properties (Biabani et al., 2023; O'Brien et al., 2017, 2018). Excessively high temperatures or prolonged heating may provide little additional contaminant removal while substantially degrading soil functionality and delaying recovery (Liao et al., 2025; O'Brien et al., 2018). By contrast, low-temperature treatment can improve post-remediation soil reuse by preserving SOM, pH, and water-holding capacity, while increasing nutrient availability (Biabani et al., 2023;

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Lee et al., 2021; Ren et al., 2020; Zhao et al., 2026), while reducing energy consumption by up to 35% compared to conventional high-temperature thermal desorption (Rehman et al., 2023). A better understanding of how thermal remediation affects soil properties can therefore help reduce costs and project duration while minimizing adverse impacts on soil functions (Biabani et al., 2023; O'Brien et al., 2017, 2018).

Similarly, soil washing is effective for contaminant removal but often degrades soil physicochemical and biological properties. Most studies have focused on the chelating agent EDTA because of its widespread use and high metal removal efficiency. However, EDTA can adversely affect several soil functions due to its persistence, residual ecotoxicity, and dose-dependent impacts, potentially delaying soil recovery (Biabani et al., 2023; Epelde et al., 2008a; Jelusic and Lestan, 2014; Lee et al., 2021; Wang et al., 2018) These effects can be reduced through improved washing processes (Gluhar et al., 2021b), using less toxic, biodegradable chelators (Wang et al., 2018), and combinations of chelators (Guo et al., 2018). For instance, biodegradable chelators such as GLDA and ISA can effectively remove metals while generally causing fewer adverse effects, although they may still alter metal mobility and bioaccessibility, creating potential risks for soil reuse (Kaurin et al., 2020; Wang et al., 2018). Modified soil-washing approaches, such as the ReSoil® process, which uses recycled EDTA at lower doses and in a less toxic Ca-EDTA form, have also shown comparatively limited impacts on soil properties, particularly when combined with rehabilitation measures such as compost addition, microbial inoculation, and phytoremediation (Gluhar et al., 2021b, a; Kaurin et al., 2020, 2021). Consequently, decisions regarding the reuse of washed soils should consider not only contaminant removal efficiency but also residual metal toxicity and impacts on soil microbial communities (Wang et al., 2018). Greater attention should likewise be given to contaminant extraction mechanisms, chemical selection, and dosage optimization to minimize adverse effects on soil functions (Biabani et al., 2023).

ISCO and ISCR frequently reduce soil organic matter (SOM), decrease aggregation, and increase leaching of dissolved organic carbon, nutrients, and mobilized metals, particularly under acidic conditions. The extent of these impacts depends on the reagents applied. For example, permanganate is often preferred because of its lower persistence and toxicity relative to hydrogen peroxide, while ISCO is generally considered more disruptive to soil properties than ISCR due to the highly reactive nature of oxidants (Wu and Lu, 2021). Several studies recommend coupling ISCO with subsequent in-situ bioremediation, using chemical treatment to reduce contaminant concentrations before biological processes are applied to support recovery and further degradation (O'Brien et al., 2017; Polli et al., 2018; Sahl and Munakata-Marr, 2006). However, chemical treatment can suppress soil microorganisms and thereby hinder biodegradation, while mobilized DOC and nutrients may compete with contaminants as preferred microbial substrates (Sutton et al., 2014).

Electrokinetic remediation can generate strong pH gradients and temporarily reduce microbial abundance. These effects are often reversible when combined with or followed by bioremediation, biostimulation, bioaugmentation, or phytoremediation. In several hybrid electrokinetic–phytoremediation systems, microbial abundance and activity recovered to levels exceeding those observed in untreated contaminated soils, suggesting that biological processes can offset short-term electrochemical stress while enhancing remediation performance (Cang et al., 2012; Couto et al., 2015; Crognale et al., 2020; Hamdi et al., 2025; Kim et al., 2010; Lima et al., 2017). Treatment duration is also important, as excessively long electrokinetic treatment periods (e.g., up to 16 days) have been reported to reduce bioremediation efficiency and inhibit electro-degradation near the cathode (Guedes et al., 2021).

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The impacts of inorganic immobilization amendments on soil functions vary considerably among amendment types. Therefore, amendment selection should consider both contaminant immobilization performance and potential effects on soil functionality (Lwin et al., 2018). For example, iron-based amendments such as iron slag or zero-valent iron (ZVI) alone may negatively affect several soil properties, whereas combining them with organic amendments such as compost or biochar can mitigate these effects while maintaining effective contaminant stabilization (Baragaño et al., 2020; Lebrun et al., 2019; Nandillon et al., 2019).

Overall, where preservation of soil functionality is an important objective, in-situ remediation methods are generally preferable. Excavation and ex-situ treatment are widely recognized as detrimental to soil functions, and although replacement soils may restore some functionality, recreating lost soil functions is often costly and difficult (Meuser, 2013; O'Brien et al., 2017). Minimizing impacts from conventional remediation therefore requires process optimization and a shift toward adaptive, hybrid treatment strategies that balance contaminant removal with soil function preservation. Integrating low-impact and nature-based approaches, such as organic amendments and phytomanagement, can reduce long-term functional degradation and improve opportunities for sustainable soil reuse. Remediation should also seek to minimize physical disturbance, extreme temperatures, aggressive chemical inputs, and other stressors, for example by adopting a functional remediation perspective that evaluates and mitigates risks to ecological receptors (Burger et al. 2016). Understanding which soil functions are most vulnerable to disruption can support the design of remediation and restoration strategies that minimize damage and promote recovery (Lee et al., 2021). Ultimately, maintaining or restoring the links between soil biodiversity and soil functioning should be a central consideration throughout the remediation process whenever feasible and cost-effective (EC, 2025; Gómez-Sagasti et al., 2012). The goal of remediation should extend beyond risk management to include the restoration of soil health and long-term ecosystem functioning (FAO et al., 2020; Gómez-Sagasti et al., 2012).

4.2 Recovery and refunctionalization of soils following remediation

The findings of this review highlight the need to integrate rehabilitation and refunctionalization into remediation planning from the outset rather than treating them as optional post-remediation measures. Although soil functions may recover naturally over time, active rehabilitation is often necessary, particularly following intensive conventional remediation. Recovery trajectories depend largely on post-treatment soil conditions and the rehabilitation measures implemented. In particular, the recovery of soil biota and associated functions is strongly influenced by soil organic matter (SOM), nutrient availability, soil moisture, and the reintroduction or recolonization of microorganisms (O'Brien et al., 2017). Indeed, without restoring SOM and favorable soil conditions, soil biodiversity recovery may take decades, even when contaminant concentrations have been reduced to acceptable levels (Adl, 2008). This challenge is especially evident after intensive methods like thermal remediation, where heating can eliminate much of the soil biota. However, recovery can be accelerated through measures such as microbial inoculation, amendment addition, natural recolonization, and mixing with uncontaminated soil (O'Brien et al., 2017, 2018; Rehman et al., 2023). Low-impact and nature-based approaches, including phytoremediation, bioremediation, and organic amendment addition, can therefore play an important role in rehabilitation when incorporated into remediation treatment chains. For example, soil washing can substantially impair soil microbial communities and their functions, but these effects are often reversible. Follow-up phytomanagement can help "polish" washed soils by reducing residual ecotoxicity and promoting

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recovery of microbial activity (Gluhar et al., 2021b; Kaurin et al., 2021). Similarly, combinations of organic amendments such as compost, vermicompost, biochar, manure, and uncontaminated soil, together with the introduction of beneficial organisms such as earthworms and microorganisms, can support post-washing soil recovery (Gluhar et al., 2021b; Jelusic and Lestan, 2014; Kaurin et al., 2020; Kaurin and Lestan, 2018). Bioremediation-based rehabilitation strategies should also be considered. One example is the use of spent mushroom substrate (SMS) in mycoremediation, which acts both as a fungal inoculum and an organic amendment rich in nutrients and organic matter. SMS can improve soil properties while enhancing the degradation of residual contaminants such as PAHs, pesticides, and chlorinated compounds (Antón-Herrero et al., 2023; Hidalgo et al., 2023).

4.3 Balancing risk reduction and soil functionality in selection of a site management strategy

Balancing risk reduction with the preservation of soil functionality requires a site-specific approach that reflects project objectives and local conditions. In practice, risk reduction often takes precedence because of regulatory requirements and liability considerations. However, the effects of remediation on soil functions should also be considered when evaluating remediation options and determining fitness-for-use, as losses in soil functionality can compromise long-term sustainability and make subsequent reclamation or restoration more difficult and costly (O'Brien et al., 2017; Volchko et al., 2013). In some cases, remediation itself may have greater impacts on ecological receptors than leaving limited contamination in place (Burger et al., 2016). Beyond effects on soil functions, remediation techniques may introduce additional environmental risks that should be considered during decision-making. Thermal treatment, for example, can increase metal mobility and residual ecotoxicity, potentially delaying ecological recovery. Techniques such as soil washing, ISCO/ISCR, and electrokinetic remediation can substantially alter soil chemistry, including pH, which may increase contaminant solubility and leaching risks. Similarly, bioremediation may generate toxic intermediate products that adversely affect microorganisms and other soil organisms (O'Brien et al., 2017).

Information on the impacts of remediation techniques on soil functions should therefore form part of site management decision-making. During option appraisal, alternatives should be evaluated not only on risk reduction, cost, and implementation time, but also on their expected effects on soil functionality and ecological recovery. This broader perspective can support the inclusion of low-impact and nature-based remediation approaches, either as standalone solutions or as components of treatment trains and soil-polishing strategies. Likewise, rehabilitation and refunctionalization should be integrated into remediation planning from the outset rather than treated as post-remediation measures. Planned interventions such as amendment addition, microbial inoculation, revegetation, or soil mixing can help restore soil properties that underpin functionality and accelerate recovery.

Ultimately, site management decisions involve trade-offs between the degree of risk reduction achieved and the level of soil functionality retained or restored. Soil health trajectories are likely to differ depending on the remediation approach and the intended future land use (Figure 1). Because remediation targets are typically defined according to acceptable risk levels for a specific land use, the required level of soil functionality will also vary. Remediation therefore sits at the intersection of human health protection and soil health management, often requiring refunctionalization measures to achieve both objectives.

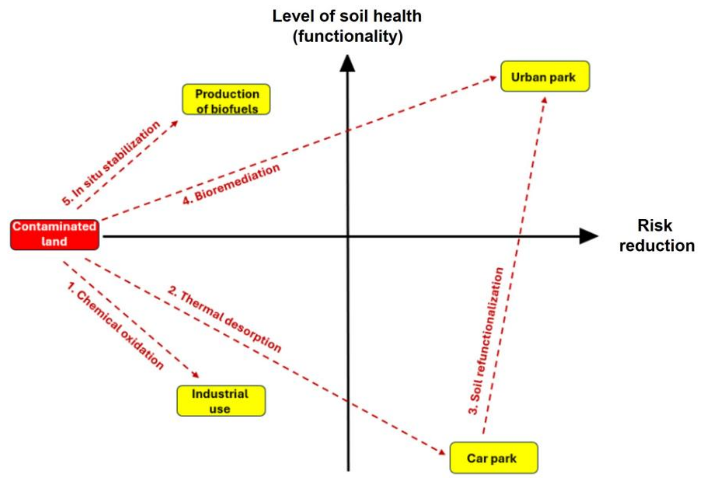

Figure 1. Conceptual illustration of different soil health trajectories towards future land uses applying different remediation techniques and refunctionalization.

4.4 Limitations and knowledge gaps

The impacts on soil quality indicators and soil functions presented in this review should be interpreted as broad qualitative trends rather than definitive outcomes. While they provide an indication of the likely effects of conventional and low-impact remediation techniques, determining quantitative thresholds beyond which soil functions cannot recover remains a major challenge. No universally accepted approach currently exists for defining soil health targets or identifying points of irreversible functional loss (Matson et al., 2024; Zwaan et al., 2026). Given these limitations, and the diversity of post-remediation land uses, a practical approach is to compare pre- and post-remediation soil measurements to evaluate relative changes in soil condition and determine whether the resulting functionality is suitable for the intended future use. However, such assessments inevitably involve value judgments, and the acceptable balance between risk reduction and soil functionality must ultimately be determined on a site-specific basis (O'Brien et al., 2017). Moreover, remediation outcomes depend on future land use objectives; techniques that are unfavorable for ecological functions may still be beneficial where geotechnical performance is the primary objective (Rehman et al., 2023). Also, the specific soil health recovery trajectories following different types of remediation have not been explored in detail but are an important area for future research.

Although the number of studies evaluating the effects of remediation on soil quality indicators has increased substantially since earlier reviews (O'Brien et al., 2017; Volchko et al., 2013), significant knowledge gaps remain. Evidence is particularly limited for techniques such as multi-phase extraction and air sparging, while emerging approaches such as mycoremediation show promise for both contaminant management and soil rehabilitation but lack sufficient field-scale validation. Research has also focused predominantly on physical and chemical soil properties, with biological indicators receiving comparatively less attention. For example, few studies have assessed the effects of remediation on soil fauna, such as earthworms, despite their importance for soil health. More broadly, assessing impacts on individual indicators and translating them into an overall evaluation of soil functionality remains challenging. Differences in site conditions, contaminants, soil characteristics, and remediation designs often result in contrasting or context-dependent outcomes, limiting the

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ability to generalize findings. Future research should therefore focus on quantitative syntheses, such as meta-analyses, to strengthen the evidence base for specific remediation techniques and soil function indicators.

5 Conclusion

This review demonstrates that remediation choices can strongly influence the long-term capacity of contaminated soils to deliver soil functions and improve soil health in accordance with the EU Soil Monitoring Law. Across the reviewed studies, conventional remediation techniques such as thermal treatment, soil washing, ISCO/ISCR and S/S generally provide rapid risk reduction but commonly do so at the expense of soil functions through often substantial impacts on physical, chemical and biological soil properties that vary depending on the specific application and operating parameters. By contrast, low-impact approaches (e.g., bioremediation, phytoremediation, monitored natural attenuation, and immobilization with organic and selected inorganic amendments) more often preserve or improve multifunctionality by supporting functions like carbon cycling, water regulation and biodiversity, albeit with slower contaminant removal and strong dependence on site conditions and management. The review further indicates that many negative impacts of conventional methods can be reduced through process optimization (e.g., reducing thermal treatment temperatures and duration, selecting less persistent or lower-toxicity reagents and chelators) and by designing hybrid, adaptive treatment chains that pair risk-reducing interventions with rehabilitation measures such as amendment addition, revegetation and (re)inoculation with microorganisms. Integrating soil health assessment and impacts on soil functions in remediation option appraisal, design, and verification monitoring provides a practical basis for balancing risk reduction and soil functionality in site management strategies. It can also support identifying areas to prioritize soil health conservation during remediation as opposed to areas with low soil functionality that can be managed with more conventional remediation techniques. Practitioners should also plan for refunctionalization of soils to promote ecological recovery, which should also be considered at the onset of a remediation project as part of the soil health trajectory. Persistent knowledge gaps, particularly on soil fauna responses, limited investigations in some remediation techniques, and quantitative thresholds for healthy versus unhealthy soils for performing various functions, highlight the need for further research and broader adoption of soil health assessment in contaminated land management practices.

6 Data availability

The data from the literature review is available in the SM.

7 Supplement link

The link to the supplement will be included by Copernicus, if applicable.

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8 Author contributions

Paul Drenning: Conceptualization, Formal analysis, Investigation, Methodology, Writing (original draft preparation), Writing (review and editing); **Kristiina Nuottimäki:** Formal analysis, Investigation, Writing (original draft preparation), Writing (review and editing); **Nazaré Couto:** Conceptualization, Funding acquisition, Project administration, Writing (review and editing)**; Virginie Derycke:** Funding acquisition, Project administration, Writing (review and editing).

9 Competing interests

The authors declare that they have no conflict of interest.

10 Disclaimer

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13 Review statement

The review statement will be added by Copernicus Publications listing the handling editor as well as all contributing referees according to their status anonymous or identified.

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