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COFA

COal to FArm

Soil Dysfunction Classification Report

D3.2

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List of abbreviations

AI4SoilHealth	Accelerating Collection and Use of Soil Health Information Using AI Technology to Support the Soil Deal for Europe and EU Soil Observatory
BENCHMARKS	Building a European Network for Soil Characterisation and Harmonisation
COFA	From COal to FArm
CZ	Czech Republic
EEA 2023	European Environment Agency
EJP Soil	European Joint Programme Soil
ENVASSO	ENVironmental ASsessment of Soil for mOnitoring
EUSO	European Soil Observatory
GR	Greece
iCOSHELLs	innovative CO-creation for Soil HEalth in Living Labs
JRC	Joint Research Centre
LUCAS	Land Use and Coverage Area frame Survey
NDVI	Normalized Difference Vegetation Index
PAH	Polycyclic Aromatic Hydrocarbons
PL	Poland
PTE	potentially toxic elements
REECOL	Ecological Rehabilitation of post-mining areas
RFCS	Research Fund for Coal and Steel
RUSLE	Revised Universal Soil Loss Equation Version
SOC	Soil organic carbon
SOM	Soil organic matter
WP	Work package

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1. Introduction

Coal and lignite mining across Europe fundamentally alters soil systems and landscapes, producing complex and persistent forms of degradation that extend far beyond the loss of agricultural productivity. Post-mining environments are characterized by the presence physical, hydrological, chemical, toxicological, and biological dysfunctions, often compounded by unfavourable landform geometry and disrupted water regimes. These dysfunctions arise from excavation, overburden dumping, compaction, erosion, exposure of geochemically reactive substrates, and long-term industrial emissions. Together, they lead to soils and substrates that frequently lack functionalities, exhibit heterogeneity, and deviate substantially from natural reference conditions.

The analysis presented in Chapter 2 of this deliverable demonstrates overview of soil dysfunctions, which rarely occur in isolation. Chapter 3 reviews key European soil health frameworks, including those underpinning the EU Soil Strategy 2030 and the Soil Monitoring Law, as well as applied, site-oriented systems developed within research projects. While these frameworks converge on a core set of physical, chemical, and biological indicators, they differ in scale, integration level, and practical applicability to post-mining environments. In particular, post-mining soils challenge conventional classifications because they may be unsuitable for food production yet still capable of supporting alternative land uses, including biomass and energy crop cultivation.

In this deliverable the From COal to FArm project (COFA) proposes a dedicated classification framework tailored to coal post-mining landscapes. This framework identifies dominant soil dysfunction types, assigns severity levels, and explicitly links soil condition to the functional role of energy crops - either as productive systems or as active reclamation tools.

1.1. Interrelations with other COFA tasks

Deliverable D3.2, developed under Task 3.2 *Defining soil dysfunctions of post-mining lands for different agricultural production*, constitutes one of the key outputs of WP3. Its primary role is to translate heterogeneous physical, chemical, hydrological, toxicological, and biological characteristics of post-mining soils into a structured and operational classification of soil dysfunctions, explicitly linked to agricultural reclamation potential.

Within Work Package (WP) 3, D3.2 provides the conceptual and diagnostic framework that connects spatial analysis, technical reclamation measures, and land-use suitability assessment.

The soil dysfunction typology and severity levels defined in D3.2 directly build upon the spatial delimitation of degraded lands generated in Task 3.1. While Task 3.1 identifies where reclamation may be feasible at the regional scale, Task 3.2 explains why specific areas are constrained, by identifying dominant limiting soil processes and their intensity. D3.2 forms the scientific basis for Task 3.3, which evaluates soil regeneration and agricultural reclamation practices. The classification of dysfunction types (physical, hydrological, chemical, toxicological, biological) allows reclamation measures to be systematically matched to soil limitations, enabling the identification of best- and worst-practice examples for specific post-mining conditions. The functional interpretation introduced in D3.2 (distinguishing between soils suitable for productive use and those requiring reclamation-oriented deployment of energy crops) directly supports Task 3.4. The assessment of carbon farming and energy crop potential relies on the soil dysfunction framework to determine whether energy crops can be applied as a production system, a reclamation tool, or both.

Beyond WP3, Deliverable D3.2 plays a key enabling role for downstream work packages. In WP5, the soil dysfunction classification developed in D3.2 is a foundational input the design of Agricultural Reclamation Scenarios. Without the diagnostic logic provided by D3.2, WP5 tools would lack a robust, site-specific link between soil condition and feasible land-use options. In WP4, D3.2 provides essential context for evaluating environmental risks, legal constraints, and social acceptability of reclamation strategies. The identification of phytotoxic versus non-phytotoxic soil conditions, contamination-driven dysfunctions, and long-term soil limitations supports legal compliance assessments (Task 4.1), stakeholder engagement discussions (Task 4.2), and environmental cost-benefit analyses (Task 4.3).

Overall, D3.2 serves as the conceptual bridge between data collection and spatial mapping (Task 3.1), technical evaluation of reclamation measures (Task 3.3), and scenario-based decision support (WP5), while simultaneously informing the social, legal, and environmental assessments in WP4. By formalising soil dysfunctions in a manner directly interpretable for agricultural reclamation and energy crop deployment, Task 3.2 ensures coherence across COFA work packages and enables the integration of scientific diagnosis with practical, policy-relevant outcomes.

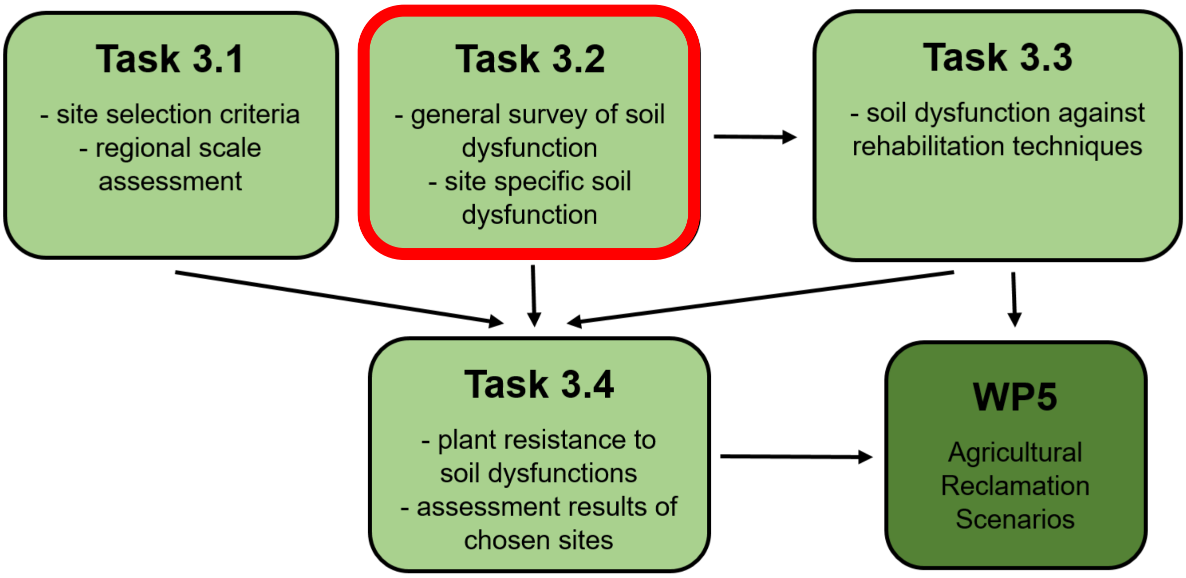

Fig. 1: Interrelations with other COFA tasks.

2. Overview of soil dysfunctions in landscapes affected by coal mining

Soil dysfunctions refer to the loss of physical, chemical, and biological properties, which limits the soil's ability to fulfil both productive and ecological functions. Different types of dysfunctions are often interconnected; one dysfunction can trigger others, creating a complex disruption of the soil system (Campbell et al., 2024; Řehoř, n.d.; Řehoř et al., 2024; Stolte et al., 2016). Dysfunctions can develop from both natural and anthropogenic influences (Feng et al., 2019).

Natural causes include erosion or contamination with hazardous elements, while anthropogenic dysfunction is often associated with coal combustion products, such as fly ash, slag, or additive fly ash granulate. (Spasić et al., 2021). The line between natural and anthropogenically induced soil dysfunctions can be very thin (Rouhani et al., 2023). For example, contamination with hazardous substances can occur naturally, but is often also caused by human activity (Řehoř et al., 2025).

2.1. Physical dysfunctions

Physical dysfunctions encompass processes that alter soil structure and mechanical properties. These include erosion, which strips away the fertile topsoil layer through the movement of soil particles, typically caused by water or wind (Krümmelbein & Raab, 2012; Kuráž et al., 2012; Řehoř et al., 2025; Rouhani et al., 2023). Anti-erosion measures may involve pits, ditches, embankments, terraces, or windbreaks, though proper crop selection and soil management remain essential (Řehoř et al., 2025).

Soil compaction presents another issue, restricting root penetration and water infiltration. Such changes diminish porosity, impair air circulation, and hinder water retention, all of which negatively affect plant growth (Campbell et al., 2024; Krümmelbein & Raab, 2012).

2.1.1. Erosion

Erosion is a natural process that disrupts the soil surface and transports soil particles via water, wind, ice, and other factors, including human activities (Campbell et al., 2024; Juliev et al., 2024). This process endangers soil productivity and other environmental components (Soil Quality Knowledge Base, 2024b). It involves three main stages:

1. Particle detachment by kinetic energy from raindrops, wind, or other agents.
2. Particle transport by water, wind, or glaciers.
3. Material deposition during energy decline.

In European continental region, water and wind erosion predominate (Vráblíková & Vráblík, 2008). Water erosion involves runoff that strips fertile topsoil from cultivated soils (Ferreira et al., 2018; Han et al., 2023). Wind erosion carries away soil, causing air pollution, root exposure and desiccation (Bullock, 2005; Tuo et al., 2023).

Other types include:

- Glacial erosion, driven by glacier movement in high mountains, transporting weathered rock.
- Soil erosion from debris flows, forming grooves and endangering valleys, settlements, or roads.
- Anthropogenic erosion, caused by direct human activity (e.g., construction, urbanization) or indirect ones (e.g., destruction of vegetation cover).

Post-mining areas are often highly susceptible to erosion, as illustrated in Fig. 2. This is mainly due to the formation of spoil heaps with steep slopes, as this allows to store a larger volume of waste. In the case of open-pit mining, the slopes of mining pits are also formed with inclines that increase the risk of erosion.

Fig. 2: Erosion furrow in ČSA area (CZ). Source: VUHU a.s., 2025.

2.1.1.1 Water erosion

Water erosion removes the most fertile layer of soil (topsoil), reducing water-holding capacity, nutrient content, and humus. It deteriorates soil physical properties, causes seed loss, silts watercourses, and threatens quality of water sources (Issaka & Ashraf, 2017; Han et al., 2023).

The intensity of water erosion increases when precipitation falls on an unprotected surface (Mishra & Singh, 2010). Raindrop impact disrupts soil structure, and during heavy rainfall, surface runoff forms rills and gullies on slopes (Soil Quality Knowledge Base, 2024b). Reducing slope gradients or dispersing runoff slows water movement and promotes sediment deposition, mitigating erosion risk.

The long-term average soil loss caused by water erosion is estimated by applying the appropriate factor values in the Universal Soil Loss Equation (USLE). If the calculated soil loss exceeds the permissible threshold, it indicates that the current land use does not provide adequate protection against erosion. In such cases, implementing anti-erosion measures becomes necessary to maintain soil sustainability and prevent further degradation.

The Wischmeier-Smith equation for expressing the average long-term soil loss (Wischmeier & Smith, 1978):

$$G = R \cdot K \cdot L \cdot S \cdot C \cdot P$$

- G average long-term soil loss
- R rainfall erosivity factor depending on frequency, duration, intensity, and kinetic energy of the rain
- K soil erodibility factor, expressed depending on topsoil texture, organic matter content, and grain size
- L slope length factor
- S slope steepness factor
- C cover management factor, expressed depending on vegetation development and applied agricultural practices
- P factor of anti-erosion measures

2.1.1.2 Erosion protection measures

Erosion cannot be fully halted but can be limited through appropriate measures. Neglecting protection results in irreversible topsoil loss, vegetation damage, silting of streams and reservoirs, and road deterioration (Soil Quality Knowledge Base, 2024b).

Measures against water erosion include organizational, such as suitable plot arrangement, field size and shape, and crop distribution; agrotechnical, like proper ploughing, leaving post-harvest residues, and mulching; and construction-technical, for example terraces, ditches, dams, and retention reservoirs. Measures against wind erosion include protective forest belts and windbreaks, which reduce wind speed and shield soil up to 20 times their height. The primary goal is to protect soil from rain and wind impacts, enhance water infiltration, lessen water's erosive force, and safely channel surface runoff (Magdoff & Van Es, 2021; Vráblíková & Vráblík, 2008).

The most effective method of limiting erosion is to reduce the slope angle of steeply sloping surfaces. However, in the case of post-mining areas, this is not always technically feasible. During the formation of mining dump, a process of compacting waste is carried out in order to reduce the risk of erosion. However, this process may have a negative impact on the physical parameters of the subsoil (Magdoff & Van Es, 2021; Vráblíková & Vráblík, 2008).

2.1.2. Soil compaction

Soil compaction is primarily caused by prolonged heavy machinery use, improper mineral fertilizer-to-organic matter ratios, and unsuitable production practices. Compaction becomes especially severe in subsoil horizons, where it persists long-term, often for decades (Lipiec & Hatano, 2003; Magdoff & Van Es, 2021; Soil Quality Knowledge Base, 2024a).

Soil compaction leads to increased soil bulk density, reduced porosity, and disrupted soil aggregates, impairing water permeability, air exchange, and root penetration resistance. This causes inadequate water infiltration, elevated surface runoff (Basset et al., 2023; Idowu & Angadi, 2013), ploughing energy costs rising by 30–80%, and crop yield losses of 10–30% (e.g., 10–20% for cereals, 20–30% for sugar beet). Subsoil compaction remains permanent and challenging to remediate (Javůrek & Vach, 2008).

2.1.2.1 Measures to prevent compaction

Agrobiological measures include applying organic fertilizers to increase organic matter content, promote soil aggregation, and increase porosity by about 8%. Liming enhances pH and stabilizes structure. Deep-rooted crops (legumes, rapeseed, corn) and cover crops improve soil aeration and structure, raising porosity by approximately 5% (Javůrek & Vach, 2008; PhycoTerra, 2022).

Mechanical compaction removal, or agromelioration, employs chiseling and loosening to alleviate soil compaction, particularly in subsoil horizons. Chiseling uses specialized chisel cultivators to fracture compacted layers without overturning the soil, thereby enhancing water infiltration, aeration, and root development (PhycoTerra, 2022). Loosening operates similarly but allows variable depths and intensities based on compaction severity (Idowu & Angadi, 2013; Magdoff & Van Es, 2021). Both methods are energy-intensive yet significantly increase porosity and restore soil structure (Javůrek & Vach, 2008).

In order to ensure suitable growing conditions for vegetation on post-mining spoil heaps, it is also possible to form the top layer of stored material with lower density.

2.1.3. Soil structure disruption

Soil structure serves as a key indicator of soil quality, influencing water and nutrient retention as well as resistance to other dysfunctions (Soil Quality Knowledge Base, 2024c). It consists of the spatial arrangement of solid particles and pores, formed by mineral particles binding with organic matter, root exudates, and microbial products (Horn et al., 1994).

Stable soil aggregates form an interconnected pore network that ensures optimal infiltration, aeration, and biological activity. Unstable aggregates readily disintegrate upon water contact, clogging macropores, increasing surface runoff and water erosion risk (Issaka & Ashraf, 2017; Juliev et al., 2024; Nemes & Rawls, 2004; Soil Quality Knowledge Base, 2024c; Tuo et al., 2023). Aggregate stability thus stands as a primary measure of soil structure quality, governed by organic matter levels, management practices, and soil cultivation intensity (Bronick & Lal, 2005; Madaras et al., 2020).

In the case of post-mining areas, we are dealing with soils whose structure has been destroyed as a result of mechanical compaction, or more often these are soilless areas (Rouhani et al., 2023). The granulometric composition of waste mining is usually not conducive to soil formation processes. In natural fertile soils, there is typically a balanced mix of particle sizes that creates a stable structure. In hard coal waste dump, the material is often dominated by large rocks and coarse gravel. Large gaps between rocks allow water to drain instantly, leaving no moisture for plants. In contrast, in the case of tailings dams the material is often extremely fine. These particles get wet and then dry, they form a hard, concrete-like crust that prevents seeds from sprouting and air from reaching the layer below. Because the particles are so small and tightly packed, there are no macropores. The substrate becomes waterlogged and lacks the oxygen required for beneficial soil microbes and root respiration. The overburden layers of open-pit mines are also frequently dominated by fine-textured fractions (silt and clay). Spoil heaps constructed from such materials exhibit physical properties that are highly detrimental to vegetation (Ahirwal & Maiti, 2016).

Farming practices critically affect soil structure stability. Incorporating organic matter, manure, crop rotations, and reduced tillage significantly enhance aggregate cohesion (Angers & Caron, 1998). Research indicates that soil structure quality can differ by up to twofold between intensive cropping systems and balanced management, particularly with organic amendments (Madaras et al., 2020).

2.1.4. Examples of physical dysfunction in study areas

An increased presence of sand was recorded on the internal spoil heap of the Bílina surface mine in the Most Basin, originating from frequent sandy layers. Higher sand content heightens the soil's susceptibility to erosion. Additional physical dysfunctions were observed at the Střimice spoil heap in the Most basin, where prominent erosion furrows were noted (Řehoř, n.d.).

2.2. Hydrological dysfunctions

Hydrological dysfunction refers to alterations in water movement and its availability within the soil system. Hydrological dysfunctions may emerge as a direct consequence of specific physical impairments in post-mining substrates. Compaction and structural degradation significantly reduce infiltration capacity, resulting in increased surface runoff and accelerated erosion (Campbell et al., 2024). Limited water retention increases drought vulnerability (Mishra & Singh, 2010), whereas inadequate drainage promotes waterlogging and oxygen depletion in the root zone.

The decline in groundwater levels represents a serious hydrological dysfunction that disrupts the natural water regime of the soil. Climate changes, particularly reduced rainfall, higher temperatures, and intensified evapotranspiration, lead to a long-term decline in groundwater reserves. This limits soil water retention capacity, heightens the risk of surface horizon desiccation, and adversely affects nutrient availability for vegetation (Jačka et al., 2021; Lu et al., 2025). Disruptions to groundwater levels can also reduce infiltration, destabilize soil structure, and inhibit biological activity, directly threatening overall ecosystem functionality and the sustainability of agricultural production (Lu et al., 2025).

Groundwater levels can undergo significant changes due to mining activities (Rouhani et al., 2023). Mine drainage often induces water shortages for plants (Ritter & Gardner, 1993), while alterations in land relief caused by extraction processes can lead to excessively wet soils or periodic/continuous flooding.

2.2.1. Infiltration disruption

Water infiltration is the process by which water enters the soil profile. A portion of this water is retained within the soil, which acts like a sponge, absorbing moisture and gradually releasing it over time. The remaining water permeates deeper through geological layers, replenishing groundwater reserves. As water moves through the soil, it undergoes natural purification. Due to its physical structure and biological activity, soil functions as an effective natural filter, trapping and decomposing pollutants (Basset et al., 2023). These properties enable the soil to contribute significantly to water retention in the landscape, mitigating flood risks, and reducing drought vulnerability. Consequently, soil is essential for ecosystem stability and sustainable water resource management. Infiltration is influenced and governed by soil structure and composition, as well as vegetation cover (de Almeida et al., 2018).

2.2.2. Decreased retention capacity

Retention refers to the soil's capacity to store water within its pores, primarily in micropores, where water remains available to plants. This reserve presents a buffer, stabilizing water availability during dry periods (Johnson, 2023).

Factors affecting water retention:

- Soil texture: Clay soils exhibit higher water retention due to their high specific surface area, whereas sandy soils have limited capacity to hold water (Mishra & Singh, 2010).
- Structure and porosity: Well-aggregated soils with a balanced distribution of macro- and micropores enable both infiltration and long-term water storage (Johnson, 2023).
- Organic matter: Enhances porosity and stabilizes aggregates, significantly improving water accumulation. Humus-rich soils, specifically, demonstrate markedly superior retention properties (Mishra & Singh, 2010).

Water retention is a key element of sustainable water and soil management. It provides an additional water source to plants during low rainfall, boosting crop yields and stabilizing production (Johnson, 2023). Adequate retention supports deeper root systems and improves plant physiology, facilitating nutrient uptake, since higher soil moisture enhances nutrient solubility. Furthermore, it promotes biological activity. Moist soils sustain microorganisms that improve fertility and structure, reinforcing the soil's ability to retain both water and nutrients (Mishra & Singh, 2010; Vráblíková et al., 2018).

2.2.3. Examples of hydrological dysfunction in study areas

The entire water balance system is critical in mining environments. Hydrological dysfunctions involve uneven temporal and spatial distribution of precipitation, leading to a water demand imbalance. Climate change is expected to reduce water potential, decreasing the dilution of pollutants and increasing their concentration (Stournaras et al., 2011). In the Kozani - Ptolemaida Basin, analysis of water samples from the irrigation canal and Aliakmon River suggested that the principal pollution sources were more likely related to adjacent agricultural, livestock activities, and urban wastewater, rather than direct emissions from coal combustion (Tsigaridas, 2014).

2.3. Chemical dysfunctions

Chemical dysfunctions manifest as changes in soil composition that impair fertility. Loss of organic matter (a chemical-biological dysfunction) restricts nutrient and water retention, while nutrient depletion (e.g., nitrogen, phosphorus, potassium) directly hinders crop growth (Campbell et al., 2024). Post-mining areas are generally characterized by insufficient levels of available nutrients

(Rouhani et al., 2023). Additional challenges include acidification and salinization, which disrupt pH and ionic balance. Carbonaceous deposits accompanying the coal seam are generally characterized by low pH and the presence of sulfur. Soil pH critically influences nutrient availability to plants (Kogelmann & Sharpe, 2006). Significant dysfunctions also include pollutant contamination. Sources range from mining and industrial activities (e.g., coal combustion residues) to natural geological factors (e.g., arsenic from Ore Mountains metamorphics via runoff) (Řehoř et al., 2025; Grygar et al., 2025).

Acidification is one of the most severe soil dysfunctions, occurring when soil buffering capacity declines due to acid formation or their external inputs. This process depletes base cations (K^+, Ca^{2+}, Mg^{2+}, Na^+). Soil pH significantly affects all element availability. Acidic conditions boost their mobility and thus phytoavailability, enabling excess uptake of both essential and potentially toxic elements (PTEs) (Goulding, 2016; Agriculture Victoria, 2025; Kogelmann & Sharpe, 2006). Phytotoxicity thresholds vary by species. While some species show no visible symptoms, other react with chlorosis, leaf brittleness, etc.(Kogelmann & Sharpe, 2006; Haurová, 2023; Munford et al., 2021).

Naturally, root exudates contribute to acidification and increased mobility of elements, potentially increasing PTEs uptake during nutrients deficiencies. Human activities further accelerate acidification through acidifying fertilizers, emissions causing acid rain (Kogelmann & Sharpe, 2006), intensive irrigation, consumption of basic elements by crops, monocultures, and limited perennial forage crops (Goulding, 2016). Consequences include lowered pH restricting nutrient availability (Kogelmann & Sharpe, 2006), heightened risk element mobility entering the food chain, structural degradation, erosion susceptibility, poorer humus quality, reduced mineral nitrogen release, and diminished phosphorus accessibility (Bradshaw, 1997; Hofman et al., 2024; Huang et al., 2023; Agriculture Victoria, 2025).

In post-coal mining areas, pyrite and other sulphur-rich minerals weathering intensifies acidity and mobility of hazardous elements, hindering soil development and revegetation (Ahirwal & Maiti, 2016; Gopinathan, Jha, et al., 2022), as illustrated in Fig. 3. This process, combined with low carbonate content, can cause the pH of the substrate to reach levels that prevent any vegetation from growing (Więckol-Ryk et al., 2023).

Fig. 3: Extremely acidic soil in Střimice dump (CZ). Source: VUHU a.s., 2025.

2.3.1. Salinization

Salinization involves the accumulation of soluble salts in the soil profile, causing degradation and reduced fertility. This process occurs naturally in high-evaporation, low-rainfall areas but accelerates due to human activities like improper irrigation and poor water management, where irrigation water's dissolved salts concentrate as it evaporates (Rengasamy, 2006).

High salt levels elevate soil solution osmotic pressure, restricting plant water uptake and inducing drought-like stress despite soil moisture. Salinization also disrupts soil structure, lowers permeability, and can lead to the formation of surface crusts (Řehoř, n.d.). These changes negatively affect plant growth, reduce yields, and, in extreme cases, can render land agriculturally unusable (Huang et al., 2023).

Globally, salinization endangers millions of hectares, particularly in arid and semi-arid zones. Prevention requires efficient irrigation, quality water, proper drainage, and adapted agricultural practices to avert permanent degradation (Rengasamy, 2006).

Post-mining waste may be characterized by increased salinity, which can have a negative impact on vegetation. In temperate climates, the salinity of mining waste in the plant root zone decreases quite rapidly as a result of salt leaching with precipitation (Więckol-Ryk et al., 2023). However, salinity is a very significant factor in degradation in post-mining areas where evaporation exceeds precipitation.

2.3.2. Examples of salinization in study areas

In the Most Basin (Czech Republic), salinity issues occur only in specific mine areas, particularly around the Nástup Tušimice Mines. In the past, significant tree die-off was observed, primarily due to extremely alkaline soil reactions. This alkaline environment impairs nutrient availability, especially magnesium, and disrupts photosynthesis. Hydrogen gypsum has been detected. Salts in upper soil horizons increase salinization, while water-bound gypsum contributes to surface drying. Thus, gypsum contamination represents the primary source of salinization (Řehoř, n.d.).

2.4. Contamination

Geogenic anomalies and other natural factors (e.g., soil or vegetation properties) account for the majority of variability in potentially toxic elements (PTEs) in soils, complicating the identification of industrial impacts (Adamec et al., 2024; Vácha et al., 2015). This variability often arises from a complex combination geogenic and anthropogenic causes (Alloway, 2012; Skala et al., 2024). Nonetheless, it is essential to acknowledge and account for the inherent complexity and heterogeneity of soils (Theocharopoulos et al., 2001). In contaminated areas, elevated PTEs concentrations hinder vegetation acclimatization and natural succession. Only a limited number of herbaceous and woody species exhibit tolerance to such adverse conditions (Nagajyoti et al., 2010; Pająk et al., 2018).

Within the context of post-mining area degradation, the persistence of these pollutants fundamentally affects the possibility of reusing the land. These landscapes frequently scarred by decades of exploitation often exhibit contaminant concentrations that far exceed the regulatory thresholds established for non-industrialized regions (Grygar et al., 2025; Wahsha et al., 2016). Numerous studies address the various forms and impacts of contamination on post-mining soils and lands (Abliz et al., 2018; Bhuiyan et al., 2010; Boahen et al., 2023; Gopinathan, Santosh, et al., 2022; Habib et al., 2019; Kou et al., 2022; Grygar et al., 2025).

2.4.1. Examples of contamination in study areas

PTEs in Most Basin soils, for example, derive from both rock geochemistry and industrial inputs (Zemanová et al., 2025; Adamec et al., 2024; Řehoř et al., 2025; Štrudl et al., 2006). Arsenic is a key risk element, with elevated levels partly geogenic and partly from coal combustion (Grygar et al., 2025; Adamec et al., 2024). In the basin, arsenic occurs mainly in coal seam soils contaminated with iron sulphides, pyrite and marcasite, though to a limited extent (Řehoř, n.d; Skala et al., 2022). The most pronounced arsenic enrichment affects Quaternary soils along the Ore Mountains foothills, sourced from runoff of metamorphites (Řehoř, n.d; Grygar et al., 2025). Anthropogenic PTE inputs stem primarily from the energy sector, including ash, slag, and desulfurization gypsum (flue gas desulfurization byproducts) deposited across the region (Řehoř, n.d; Zemanová et al., 2025). These materials can drastically alter local soil chemistry. For example, energy gypsum applied at the Prunéřov VI spoil heap near Kadaň caused sulphate buildup in the topsoil, tree mortality, and subsequent remediation (removal of 0.7 m of contaminated soil and its replacement) (Řehoř et al., 2018; Řehoř, n.d). However, analyses show that diffuse ash fallout from power plants poses minimal toxicological risk to basin agricultural soils (Adamec et al., 2024; Grygar et al., 2025). More pressing concerns are localized PTE hotspots near coal outcrops and legacy spoil heaps (Řehoř et al., 2024).

In the Kozani – Ptolemaida Basin, concentration of heavy metals and PTEs were investigated. While most concentrations were found to be low (attributed to the high effectively immobilizing the metals), areas near the coal mining facilities showed elevated levels, indicating localized chemical dysfunction hotspots (Psarraki et al., 2023). For instance, surface soil and plant samples showed increased levels of five heavy metals (Cd, Cr, Cu, Pb, and Ni), with Chromium (Cr) consistently having the highest concentration in soil, mosses, and lichens. The chemical composition suggests a common origin related to fly ash. In parallel, ash-affected soils frequently exhibit alkaline to strongly alkaline conditions, fundamentally altering soil chemistry and nutrient availability (Psarraki et al., 2023). High levels of the artificial radionuclide ^{137}Cs (attributed to the Chernobyl accident) remain present in the upper soil layers (Tsigaridas, 2014). In the Sarigkiol Basin, the alkaline nature of fly ash has caused soils to exhibit alkaline to strongly alkaline values (acidity dysfunction) (Psarraki et al., 2023). The Ptolemaida-Basin also exhibits chemical dysfunctions from organic compounds, specifically Polycyclic Aromatic Hydrocarbons (PAHs), which originate from both the lignite dust (petrogenic source) and high-temperature combustion (pyrogenic source) (Schwarzbauer & Vossen, 2024).

2.5. Biological dysfunctions

Biological dysfunction refers to a decline in soil organism activity and biodiversity loss. Reduced microbial activity impairs organic matter decomposition and humus formation, disrupting nutrient cycling (Campbell et al., 2024). Post-mining substrates are typically characterized by a profound deficiency in labile organic matter, the primary energy source for soil organisms (Rouhani et al., 2023). Furthermore, these materials are often biologically sterile, lacking the microbial consortia and macrofauna (such as earthworms and mycorrhizae) essential for driving pedogenic processes. The absence of these biological "engineers" prevents the transformation of raw mineral waste into functional, structured soil, which are necessary for the creation of productive agricultural land (Hu et al., 2020).

The decline in biological functions can also manifest as reduced vegetation cover, which is typically removed prior to mining operations. Restoring this cover can be a prolonged and challenging process availability. To fully comprehend the severity of this dysfunction, it is essential to recognize the critical role of plants In soil development, plants protect the surface against erosion and facilitate the accumulation of fine particles (Bradshaw, 1997). They also promote the accumulation of nutrients in bioavailable forms. Through their root systems, plants act as traps for otherwise inaccessible nutrients, storing them and subsequently releasing them to the soil surface via organic matter, where microbial decomposition enhances their (Frouz et al., 2008; Pająk et al., 2018).

One of the most effective measures to improve soil properties is the application of composts containing a high proportion of organic matter. Composts act as a binder for soil particles, increase the formation of pores and micropores, thereby improving water management, sorption capacity, and resistance to erosion. Beyond their physical effects, they activate the soil microflora, foster stable soil structure formation, and overall revitalizes the soil horizons. Consequently, compost serves as a key tool for restoring the biological functions of the soil and optimizing its hydro-physical properties (Zemánek & Burg, 2009).

2.5.1. Deficit of nutrients

Spoil heaps and mining dumps are generally nutrient-poor, and interactions between elements often aggravating these deficiencies. The most pronounced shortages typically involve phosphorus (both available and reserve forms) and magnesium (reserve), followed by potassium and calcium.

On light, sandy substrates, these nutrient deficiencies are particularly pronounced. Low calcium content also signals weak buffering capacity and reduced pH, which heightens the mobility of certain elements like manganese. This can lead to their excessive accumulation in leaves while simultaneously serving as an indicator of nutritional stress from macronutrient scarcity. Concurrently, signs of phytotoxicity may emerge. These relationships (low Ca → low pH → higher bioavailability of Mn/Zn; low P/Mg → excessive uptake of Mn/Zn) are being observed in spoil heaps across common tree species (birch, alder, poplar, maples, linden) and can lead to limited growth and reduced resilience of vegetation stands to climatic extremes (Alejandro et al., 2020; Bílková et al., 2023). Moreover, nutrient leaching further intensifies these deficiencies. Overall, this represents not merely a nutrient shortage, but an interconnected chemical system where deficits in basic ions, low pH, and limited P/Mg mutually reinforce each other, hindering vegetation cover stabilization during both natural succession and active reclamation activities (Matys Grygar et al., 2025).

Nutritional deficiency is often accompanied by contamination with hazardous elements, which further complicates soil recovery. In the environment of the Most Basin, nutrient-poor substrates contain trace amounts of Cd and Zn associated with coal burning (while elevated levels of As, Pb, or Cu are often geogenic), which, in combination with low pH and macro-nutrient deficiencies, increases toxicity of environment and intensifies physiological stress in plants (Grygar et al., 2025). These circumstances also slow the development of microbial communities, which play a major role in nutrient cycle.

Bradshaw, 1997 emphasized that nutrient deficiencies and toxicity represent extreme soil conditions that must first be amended, only then can natural processes accelerate restoration. Combination of natural succession and targeted adjustments of elements is recommended, specifically supplementing P and Mg, increasing Ca levels (by adjusting pH and buffering capacity) to limit excessive uptake of undesirable elements, and, at the same time, utilizing species capable of N fixation addressing nitrogen limitation (Bradshaw, 1997). Interactions between individual elements play a key role in nutrient availability and significantly affect overall plant nutrition (Kogelmann & Sharpe, 2006).

2.5.2. Loss of organic matter

Soil organic matter (SOM) serves as a critical soil component, governing fertility, structure, water retention, and biological activity (Bronick & Lal, 2005) (Merilä et al., 2010). Organic carbon (SOC), SOM's primary constituent, drives soil aggregate formation, nutrient retention, and pH regulation

(Bodlák et al., 2012). A stable organic matter content ensures a long-term soil productivity and its resilience to other soil dysfunctions (Juřicová et al., 2022).

Intensive agriculture, including prolonged tillage and monocultures, depletes SOC stocks by tens of percent over decades, degrading soil structure, decreasing water infiltration and erosion resistance. This process is particularly evident in chernozem soils, which originally contained large amounts of humus, but have become dysfunctional as a result of intensive cultivation (Abakumov et al., 2013; Juřicová et al., 2022).

Restoring soil organic matter is a long-term process that requires a change in management practices (Bartuška, 2014). Key measures include reducing intensive soil tillage, incorporating perennial forage crops, applying organic fertilizers, and using cover crops (Juřicová et al., 2022). This promotes carbon sequestration and improves soil structure, contributing to the maintenance of ecological stability and production capacity. Post-mining areas may be characterized by high organic carbon content (Ussiri et al., 2014), but this carbon is usually resistant to biological degradation (Bartuška et al., 2015).

2.5.1. Examples of nutrients deficits in study areas

Post-mining lignite mines at the Sieniawa is entire overburden mass dominated by silt formations, which can be classified as potentially productive soils suitable for agricultural reclamation, but require fertilization with lime, nitrogen, and potassium. Other examples are Quaternary and Tertiary sands and gravels having low nitrogen content and available forms of potassium and phosphorus, thus require fertilization. Clays have low nitrogen content and available forms of potassium and phosphorus. Silts and mudstones have low potassium and phosphorus content.

2.5.2. Loss of soil organisms

Soil organisms drive organic matter decomposition, nutrient mineralization, and humus formation, thereby sustaining soil fertility. Together, they create a complex network of interactions that affects soil structure, its water retention capacity, and resistance to erosion (Bartuška et al., 2015).

The decline in soil organisms is a serious problem because it disrupts the nutrient cycle and biological activity of the soil (Frouz et al., 2013). Without a sufficient population of (micro)organisms, the decomposition of organic matter decelerates, leading to soil impoverishment and reduced ability to support vegetation (Mudrák & Frouz, 2018). This process can be caused by intensive agriculture, pesticide use, soil compaction, or climate change. Substrates from post-mining coal sites exhibit variable degrees of loss of microbial activity and soil organisms. This impairment arises from the drastic disturbance during extraction processes, which strip away topsoil, expose infertile substrates, and introduce contaminants such as heavy metals and acidity, as seen on spoil heaps and mining dumps (de Quadros et al., 2016; Frouz et al., 2006).

A decline in microbial activity leads to reduced nitrogen fixation, disrupting plant symbiosis with soil bacteria and decreasing the availability of this essential nutrient (Chiurazzi et al., 2025). Nitrogen ranks as the second most critical element for plant growth and development after carbon, primarily due to its pivotal role in protein synthesis and photosynthesis. Beyond serving as nutrient sources, nitrogen-fixing bacteria function as essential ecosystem regulators by promoting plant growth, enhancing soil structure, and offering a vital tool for sustainable agriculture that reduces reliance on synthetic fertilizers. Consequently, introducing leguminous plants and pioneer species into degraded ecosystems proves strategically essential for restoring soil microbial activity and boosting biological nitrogen fixation (Bradshaw, 1997; Chiurazzi et al., 2025). On the other hand, earthworm activity has been demonstrated to boost plant biomass more markedly in immature soils than mature ones, emphasizing their importance for the early stages of restoration (Hlava et al., 2015) and reclamation practices (Háněl, 2002). The role of soil organisms in the reclamation of post-mining areas has been extensively studied, with research focusing on both microbiota (Chiurazzi et al., 2025; Harris et al., 1989) and invertebrates (Frouz et al., 2006, 2008, 2013; Háněl, 2002; Hendrychová et al., 2012; Mudrák & Frouz, 2018).

2.5.3. Examples of biological dysfunction in experimental sites

Biological dysfunctions develop in response to chemical and physical stresses. In the Kozani - Ptolemaida Basin (GR), the overall microbial load (bacterial populations) in the contaminated study area was measured to be significantly higher than that in the control area (Tsigaridas, 2014).

2.6. Other forms of post-mining land degradation beyond soil dysfunctions

Beyond soil-related constraints, post-mining landscapes are frequently affected by geomorphological, hydrological, and spatial degradations that strongly limit their suitability for agricultural use. Typical post-mining landforms, such as spoil heaps, dumping grounds, benches, and terraces, are characterized by steep slope angles, short slope lengths, abrupt breaks in slope, and irregular surface geometry. These features are inherently incompatible with conventional agricultural field layouts and mechanised farming operations.

Steep or irregular terrain directly restricts the use of standard agricultural machinery for soil cultivation, sowing, and harvesting. Where such landforms persist, they generate fragmented and patchy parcels, increase headland losses, and substantially raise unit production costs. As a result, commercial arable farming is generally economically unviable unless large-scale geomorphic reclamation and regrading are implemented (Feng et al., 2019).

Terrain configuration is a key determinant of post-mining land usability. Numerous studies indicate that slope angle controls both mechanisation feasibility and erosion risk, while microrelief and relative elevation govern runoff pathways, water redistribution, and local microclimate (Feng et al., 2019). Consequently, topography strongly influences spatial patterns of soil moisture, organic matter accumulation, and vegetation establishment during reclamation.

Post-mining areas often remain unstable for years. Such areas are prone to erosion, settlement, and episodic mass movements, particularly where constructed slopes and drainage systems do not align with natural process rates. Even sites covered with topsoil remain vulnerable if landform design concentrates runoff and accelerates surface incision, necessitating ongoing erosion control and maintenance (Spain & Hollingsworth, 2016).

Mining activities also fundamentally alter surface and subsurface hydrological regimes. Natural catchments are frequently fragmented, closed depressions and internal drainage sinks are created, and groundwater systems may be either depressed or locally perched within heterogeneous spoil bodies. This leads to high spatial variability in water availability, with waterlogged depressions coexisting alongside drought-prone convex slopes within the same reclaimed area (Qi et al., 2023).

3. Classification frameworks of soil dysfunctions

Soils across Europe face a range of biophysical and chemical dysfunctions, processes like erosion, organic matter decline, compaction, contamination, salinization, acidification, and biodiversity loss that impair key soil functions. Recognizing and classifying these soil health issues is crucial for restoration and sustainable land management.

In recent years, EU initiatives (the EU Soil Strategy 2030, the proposed Soil Monitoring Law, and the Horizon Europe “Soil Deal” Mission) have spurred the development of robust frameworks Robust indicator framework AI4SoilHealth assess soil health via measurable indicators. Multiple EU-funded projects (including Soil Mission “Living Labs & Lighthouses” (European Commission, 2025) projects and other RFCS projects) have proposed classification systems to categorize soils based on these indicators. EU soil health frameworks generally focus on biophysical and chemical indicators of degradation. These cover physical soil properties (structure, density, erosion), chemical properties (organic matter, nutrient levels, pH, salinity, pollutants), and biological factors (soil biodiversity, microbial activity). Socio-economic or land-use factors are treated separately, keeping the classification tied to measurable soil conditions (Bonfante et al., 2020).

This section provides an overview of validated classification frameworks and indicator sets related to soil health in the EU, focusing exclusively on physical, chemical, and biological soil parameters (e.g. soil organic carbon, structure, nutrients, contaminants), and compares their approaches. These classification systems will be used as a basis for designing a tailored classification system for coal post-mining soils.

3.1. Overview of existing indicator and classification frameworks for soil health in the EU

Several academic studies (Bünemann et al., 2018; Lehmann et al., 2020; Maurya et al., 2020; Stolte et al., 2016) have influenced EU projects by recommending minimum indicator sets for soil quality. Common recommendations include soil organic carbon, pH, nutrient levels (N, P), cation exchange capacity, bulk density, porosity, aggregate stability, microbial biomass, and earthworm counts, among others, as core metrics of soil health.

These insights underpin key EU initiatives, such as the Mission Board Soil Health & Food's "Caring for soil is caring for life" (European Commission, 2020), which targets 75% healthy EU soils by 2030, and the EU Soil Strategy for 2030 (European Commission, 2021), both validating such indicators under European conditions (e.g., varying optimum soil organic carbon levels between Mediterranean sandy soils and Nordic loams).

Many EU projects, supported by the European Environment Agency's soil monitoring indicators and thresholds (European Environment Agency, 2023) and the newly in force Soil Monitoring Law (European Parliament, 2025) further assign critical limits and integrate these metrics into policy frameworks.

3.1.1. EU Soil Strategy & Soil Observatory Framework

A cornerstone of EU efforts is the indicator framework developed by the Joint Research Centre (JRC) (Broothaerts et al., 2024) and the European Soil Observatory (EUSO) to support the Soil Strategy 2030 (European Commission, 2021) and the Soil Monitoring Law (European Parliament, 2025). In late 2024, JRC proposed a set of 19 key indicators representing the main soil degradation processes, each with a science-based threshold distinguishing “healthy” from “unhealthy” soil status (Broothaerts et al., 2024). These indicators align closely with the well-recognized soil threats in Europe and cover:

Soil erosion – e.g. annual loss by water, wind, tillage, or harvest exceeding 2 t/ha is deemed unsustainable.

Soil organic carbon (SOC) – Quantified as "% of optimal level," with a large (>60%) deficit from potential maximum indicating serious organic matter depletion.

Nutrient surplus or deficiency – E.g. nitrogen surplus >50 kg/ha/yr signals eutrophication risk, while plant-available phosphorus <20 mg/kg indicates nutrient depletion; excess P >50 mg/kg risks pollution.

Soil contamination – Excess heavy metals like Cu >100 mg/kg, Zn >100 mg/kg, Cd >1 mg/kg, Hg >0.5 mg/kg, or As hotspots (>5% area above 45 mg/kg) mark polluted soils.

Soil acidity – Extremely low pH (context-dependent) would be captured under nutrient/chemical indicators (the EU framework includes "soil nutrients and acidity" as a category).

Salinization – Areas at risk (e.g. >30% of land irrigated in Mediterranean climates) serve as a salinity indicator.

Soil compaction – High bulk density or "packing density" (e.g. ≥1.75 g/cm³ in topsoil) is used to flag compaction issues limiting root growth.

Soil biodiversity – A composite indicator (e.g. an index of biological functions or diversity) identifies soils with elevated risk of biodiversity loss. Because direct measurement of soil biota at scale is difficult, the JRC uses a risk modelling approach for "potential threat to biological functions" based on factors like land use and soil properties.

Soil sealing and land cover – The proportion of land that is built-up or sealed is tracked; 100% of built-up area is considered "unhealthy" by definition, since sealing completely impairs soil functions.

Each indicator has a quantitative threshold differentiating acceptable vs. problematic levels, derived from scientific literature and pan-European data. For example, soil loss >2 t/ha/yr is beyond natural regeneration rates, and Cu >100 mg/kg might exceed ecotoxicological limits. If a soil exceeds any threshold, it may be classified as "dysfunctional" or "unhealthy" for that aspect. This framework revealed that over 60% of EU soils are currently unhealthy on at least one indicator – a striking statistic underscoring widespread soil dysfunction (Broothaerts et al., 2024).

The framework intends to evolve into a composite Soil Health Index aggregating all indicators, to give an overall score per site (Broothaerts et al., 2024).

3.1.2. ENVASSO monitoring framework

The ENVASSO project (EU FP6) was an early attempt to design a European soil monitoring framework. It proposed a list of 27 soil quality indicators aligned with the eight soil threats identified in the EU Soil Thematic Strategy (erosion, organic matter decline, compaction, salinization, landslides, contamination, sealing, and biodiversity loss). Indicators included metrics like erosion rates, topsoil organic carbon content, pH (acidification), electrical conductivity (salinity), heavy metal concentrations, soil bulk density/porosity, soil biodiversity (earthworm counts, microbial respiration), etc. (Kibblewhite et al., 2008) The project evaluated feasibility and recommended methods for each indicator. ENVASSO's output highlighted that while many indicators were technically measurable, a subset of about 20 were ready and practical for immediate monitoring use. These findings laid groundwork for later programs (EEA 2023 and EJP Soil) in standardizing soil health observations.

3.1.3. EJP Soil

Pre-dating the Soil Mission, the European Joint Programme EJP Soil (a large H2020 program, 2020–2024) (*EJP Soil*, n.d.) built scientific groundwork for soil health assessment. It compiled indicator databases and tested them on long-term field experiments. Findings from EJP Soil and related EU projects feed into current frameworks. For instance, identifying soil biodiversity indicators like microbial respiration rates or invertebrate indexes, and optimal threshold values under different land uses (Faber et al., 2022).

3.1.4. Horizon Europe "Soil Mission" projects

Under the EU's "A Soil Deal for Europe" mission, several Horizon Europe projects have launched to refine and harmonize soil health indicators in real-world contexts. The EU is also directly funding many Living Labs and Lighthouses under the Soil Mission (European Commission, 2025; SOILL, 2024). These are essentially real-world test sites (farms, forests, urban sites) where soil health improvements are co-created and monitored. Soil Health Living Labs are applying the EU indicator frameworks in a practical setting, effectively validating and fine-tuning the classification of soil dysfunctions (physical, chemical, biological) at scales from field to region. To date, no living labs have been established for post-mining sites

3.1.4.1 AI4SoilHealth

AI4SoilHealth (2023–2026) focuses on developing a robust indicator framework (AI4SoilHealth, n.d.; Campbell et al., 2024) to support EU soil policy. It builds on the JRC/EUSO set and addresses gaps by identifying new or proxy indicators for soil functions that are hard to measure directly (for instance, using AI methods to correlate easily observable data with soil biodiversity or structure). The project emphasizes a comprehensive suite of indicators across all key domains – physical (e.g. bulk density, structure, infiltration capacity), chemical (organic carbon, pH, nutrient levels, salinity), biological (microbial biomass, invertebrate diversity), and hydrological (water holding capacity) – seeing these as essential to capture soil dysfunctions (Campbell et al., 2024). AI4SoilHealth is working closely with JRC and national agencies so that its framework will feed into the impending Soil Monitoring Law.

Interesting aspect of AI4SoilHealth's approach is the binary classification of soil condition for simplicity. Each monitored soil attribute is flagged as either "acceptable" or "degraded" based on agreed thresholds, similar to the JRC method. By aggregating many binary signals, the framework can map out specific threats, e.g. highlighting where soils are degraded due to erosion or pollution, and enable targeted remediation. In effect, this combines advanced data analysis with a clear-cut classification (degraded vs. not degraded for each indicator), making results accessible to policymakers and land managers (Campbell et al., 2024).

3.1.4.2 BENCHMARKS

BENCHMARKS (Building a European Network for Soil Characterisation and Harmonisation, 2023–2027) (*Soil Health Benchmarks*, n.d.) is another Soil Mission project, aimed at aligning monitoring across 24 Living Labs in Europe. It does not define a new indicator set from scratch but works on harmonizing how existing indicators are measured and interpreted across countries. This involves setting common protocols for soil sampling, analysis, and data sharing, as an essential step for consistent classification. By the end of BENCHMARKS, all participating "Soil Health Living Labs" should be using a transparent, cost-effective monitoring framework with agreed indicators and methods for soil health assessment. Indirectly, this project helps ensure that classification of soil health/dysfunction can be compared across regions (e.g. a moderate erosion soil in Spain is evaluated with the same criteria as one in Sweden).

3.1.4.3 iCOSHELLs

Project iCOSHELLs ("innovative CO-creation for Soil HEalth in Living Labs", 2024–2028) is establishing indicator-based classification schemes at the farm/landscape level. According to iCOSHELLs Deliverable 3.1, a "Catalogue of Soil Health Indicators" has been developed to guide all partner labs (Pisani & Soriano Disla, 2025). This catalogue distinguishes indicators in three categories (physical, chemical, biological) mirroring the broader frameworks.

Physical indicators – e.g. bulk density, penetration resistance, infiltration rate, aggregate stability, soil texture class, available water capacity. These diagnose structural dysfunctions like compaction or poor water regime. If bulk density is high and infiltration is low, the soil is classified as physically degraded in terms of structure.

Chemical indicators – e.g. soil organic carbon %, total nitrogen, available phosphorus and potassium, pH, electrical conductivity (salinity). These reveal fertility or contamination issues, e.g. a low SOC. Low nutrient soil may be classified as "chemically poor", whereas one with extreme pH or high salt is classified under acid or saline dysfunction.

Biological indicators – e.g. microbial biomass or activity (soil respiration), earthworm count, enzyme activities, presence of key functional groups. In practice, some living labs use simpler proxies like the abundance of soil fauna observed. These help flag biological dysfunctions (like depleted soil life).

All living labs use a common core set of such indicators to allow comparison, but each may add a few site-specific indicators relevant to local issues. For example, a Living Lab on a Mediterranean cropland might include an indicator for soil crusting or an index of drought resilience, whereas a forest soil lab might add a fungal diversity indicator. The classification of soil status in these labs is often done via scoring or rating systems rather than rigid threshold cut-offs, e.g. giving soil an index from 0 (heavily dysfunctional) to 100 (fully healthy) based on how it compares to reference values for each indicator. This is then used to track improvements over time. While individual living labs project results are still emerging, their work complements the top-down frameworks by testing how well those indicators actually detect changes in soil health on the ground. They also provide feedback on practicality, e.g. which lab tests farmers can easily perform.

3.1.5. Project REECOL

Outside of the Soil Mission, the REECOL project (RFCS-funded "Ecological Rehabilitation of post-mining areas", 2022–2025) (REECOL, n.d.) offers a targeted example of a classification framework for soil dysfunctions in coal post-mining sites. REECOL developed, on a case study (Musiałek, Szwaja, Řehoř, et al., 2024), a systematic way to classify the degree and nature of dysfunction in post-mining soils as a basis for reclamation strategies. The framework (Musiałek, Szwaja, Kania, et al., 2024) integrates a suite of indicators grouped into three domains:

Landscape indicators assess terrain and surface features that indicate degradation. REECOL uses remote-sensing indices like Standard Deviation of Elevation (micro-relief variability), Topographic Wetness Index (drainage conditions), vegetation indices (NDVI for green cover), and surface albedo or thermal anomalies (which can indicate bare or burned soils). These help identify physical state of land, e.g. very high micro-relief variability on a spoil heap suggests uneven settling, which is an obstacle to agriculture.

Geochemical indicators measure soil chemistry. Key ones include heavy metal concentrations, as mining often leaves toxic levels of As, Cd, etc., soil organic carbon content and quality, macronutrient levels, particularly nitrogen and phosphorus, which can be extremely low or imbalanced in mine spoils, soil salinity, and pH. Each is compared against critical values – many drawn from agricultural soil quality standards, to classify if the soil is chemically "normal", "degraded", or "toxic". For instance, REECOL might classify a plot as “contaminated” if heavy metals exceed safe thresholds, or as “infertile” if both organic matter and nutrients are below minimum levels for plant growth.

Geotechnical indicators evaluate physical soil properties related to structure and stability. REECOL looks at soil porosity, bulk density, and an erosion risk index (using the RUSLE factors). In practice, many mine soils are very dense, compacted by machinery or by the weight of overburden, and have low porosity – classified as a physical dysfunction (compaction). Erosion risk is also classified. If a slope has a high RUSLE score without vegetation cover, it falls into an “erosion-prone” class of dysfunction.

Using these indicators, REECOL’s framework defines degradation classes for post-mining lands. The classification has both qualitative type categories and a quantitative severity rating. Types of soil dysfunction mapped in REECOL’s case studies (Musiałek, Szwaja, Řehoř, et al., 2024) include, for example “soils contaminated with trace elements”, “soils affected by coal combustion residues” (areas where ash or slag causes toxins or pH shifts), “soils with high sand content and erosion risk”, “soils with high clay content causing poor structure”, “saline soils”, etc. Alongside, a three-tier severity scale is applied – essentially Grade 1: mildly degraded, Grade 2: moderately degraded, Grade 3: severely degraded (or “phytotoxic”). This simple scale helps decision-makers prioritize areas. A Grade 3 salinized soil, for instance, is essentially non-productive and needs intensive intervention, whereas a Grade 1 might recover with minimal assistance. By linking each class to indicator criteria, the REECOL framework is both systematic and actionable. For example, if pH < 4 and high soluble sulphate is found, the soil might be classified as “Acidic mine spoil – Severe” requiring liming and organic amendments. Thus, the classification framework directly informs the reclamation strategy (Markowska et al., 2024; Musiałek, Szwaja, Kania, et al., 2024; Musiałek, Szwaja, Řehoř, et al., 2024).

It’s worth noting that while REECOL’s focus (coal mine soils) is niche, it drew on EU-wide knowledge. The used indicators and thresholds align with those in broader frameworks, e.g. heavy metal limits were taken from EU soil screening values, and their notion of “healthy soil for revegetation” corresponds to having SOC, pH, nutrients in ranges known to support plant life. In essence, REECOL is an applied example of classifying soil dysfunctions by combining multiple indicator criteria into a map of distinct degradation types, which is slightly different from the high-level JRC approach, which flags issues indicator-by-indicator (Broothaerts et al., 2024). Both approaches complement each other. One offers an integrated classification of what is wrong and where (useful for site management), and the other provides a consistent metric for each type of problem (useful for policy and comparison).

3.2. Comparative analysis of soil health frameworks

Despite differences in context, these frameworks have strong commonalities. All are built on the premise that soil health can be assessed through a set of quantifiable biophysical indicators, and that by evaluating a soil against those indicators’ criteria, we can classify its condition. Tab. 1 compares key features of some prominent frameworks.

Tab. 1: Comparison of selected soil health classification frameworks in the EU

Framework	Focus	Key Indicators & Metrics	Classification approach
EU Soil Observatory (JRC, 2024) (Broothaerts et al., 2024)	EU-wide policy monitoring, covers all land uses across Member States. Supports Soil Strategy 2030 and Soil Monitoring Law.	19 core indicators representing main degradation processes: *Erosion:* water, wind, tillage, harvest erosion (t/ha/y) *Organic carbon:* deficit from reference (% of optimal) *Contamination:* heavy metals (Cu, Zn, Cd, Hg, As) above safe levels *Nutrients:* N surplus (kg/ha), P excess/deficit (mg/kg) *Soil acidity:* (indicator in "nutrients & acidity" group) *Compaction:* packing density (g/cm³) *Salinization:* area with high irrigation & evaporation risk *Soil biodiversity:* risk index for biological function loss *Sealing:* built-up land cover (%)	Threshold-based, binary classification. Each indicator has a defined threshold separating "healthy" vs "unhealthy" soil condition (uniform across Europe). If an indicator exceeds the threshold, the soil is classified as failing that aspect (e.g. "erosion-dysfunctional"). Overall soil health status can be judged by number and severity of failed indicators. This framework is quantitative and geospatial, enabling maps like the EUSO Soil Degradation Dashboard. The plan is to develop a composite Soil Health Index aggregating all indicators (e.g. scoring soils 0–100) for an overall class. Currently, the system effectively flags multi-dimensional classes, e.g. a soil with high erosion and low SOC is "unhealthy due to erosion & carbon loss".
AI4SoilHealth (Horizon Europe) (AI4SoilHealth, n.d.; Campbell et al., 2024)	Pan-European R&D project, refines indicators for the EU Soil Mission. Emphasis on agricultural soils but inclusively designed.	Augments core EU indicators with new data/techniques. Continues tracking standard metrics (SOC, nutrients, pH, etc.) to feed into AI models. Uses JRC's set as baseline, with added focus on: *Soil biological health:* e.g. microbial biomass, community DNA, enzyme activities, since biological metrics. *Soil structure:* e.g. infiltration rate, aggregate stability, which can serve as proxies for compaction and hydrological function. *Remote sensing proxies:* e.g. high-resolution land cover, carbon flux data, to infer soil status continuously.	Hybrid classification – follows the binary healthy/degraded flagging for each indicator (mirroring JRC thresholds), but leverages AI to integrate multiple indicators. The framework can classify soils into combined categories, e.g. identifying patterns like "likely degraded by compaction even if organic carbon is okay" via machine learning. It aims for real-time, finer-scale classification, e.g. using sensor networks and AI to update soil health class (degraded/not) frequently. Output for end-users is kept simple, e.g. a map showing degraded vs healthy areas for various threats. Ultimately, AI4SoilHealth will enhance how descriptors are interpreted across scales, but in terms of classes, it still produces clear labels like "degraded for erosion" etc., consistent with the EU Soil Law descriptors.
REECOL Framework (RFCS project) (Musiałek, Szwaja, Kania, et al., 2024)	Post-mining soils of coal mine regions being reclaimed. Focus on suitability for agriculture.	Multi-criteria indicators in 3 groups: *Landscape:* microrelief variability (std. dev. of elevation), slope angle, wetness index (drainage), vegetation cover (NDVI), surface albedo/thermal anomalies (to spot, e.g. bare dry patches). *Geochemical:* total heavy metals (As, Cd, Pb, etc.), soil organic carbon %, total N and available P, pH, electrical conductivity (salinity). *Geotechnical:* bulk density, porosity, moisture content, and an	Tiered categorical classification. REECOL combines indicator readings to assign each area a degradation category describing the dominant dysfunction, *and* a severity grade. For example, an area with low nutrients, low SOC, and high compaction might be classified as "Infertile – Structure Degraded (Moderate)". Another with extremely high metals and low pH might be "Toxic Contamination (Severe)". The framework defines several such classes (contamination-driven, structure-driven, etc.) corresponding to key limitations for reclamation. Severity is ranked 1, 2, or 3 (mild to severe).

Framework	Focus	Key Indicators & Metrics	Classification approach
		erosion risk factor (RUSLE factors: R, K, LS, C, P). Additionally, qualitative site observations, e.g. presence of acid mine drainage.	This yields a map of zones, each coloured by class (type and severity). It's a diagnostic framework – classification directly indicates what intervention is needed (lime the acidic zone, add organic matter to the infertile zone, etc.). Unlike the binary pass/fail of others, REECOL's is a multi-class system tailored to post-mining contexts, though built on standard indicators.
iCOSHELLs (Soil Health Living Labs, Horizon Europe) (Pisani & Soriano Disla, 2025)	Regional pilot sites (network of living labs and lighthouses). Agricultural and forestry living labs testing soil improvements.	Comprised of core set + site-specific indicators: Core physical: bulk density, infiltration rate, structure score, e.g. visual evaluation, water holding capacity. Core chemical: SOC, pH, N, P, K levels, CEC, salinity (if relevant). Core biological: microbial activity, e.g. respiration, earthworm count, soil biodiversity index (if available). Additionally, site-specific examples. In one lab, an erosion-prone vineyard, indicators include ground cover % and erosion pins; in another (peatland), includes peat depth and water table level.	Scorecard and rating system. Living labs (generally) typically use an index-based approach. Each indicator is scored (often 0–5 or 0–10 scale) against a benchmark – either an undisturbed reference soil or agronomic optimum. Scores are then aggregated into categories. For instance, a lab might rate soil as "Good", "Medium", or "Poor" for physical, chemical, biological health separately, via thresholds on the scores. Some use a traffic-light system (green/yellow/red) per category. The ultimate classification might be a soil health score combining all aspects, or simply a set of ratings. The emphasis is on tracking improvement, e.g. moving a soil from red (poor) to yellow (fair) in biological health after 3 years of regenerative practices. These schemes are less formalized at EU level, but generally align with the broader frameworks. For example, a "poor" rating usually correlates with failing one of the JRC thresholds.

All frameworks above share a foundation in the same core soil health parameters. Soil organic carbon, nutrient status, pH, bulk density/compaction, and evidence of erosion or pollution appear universally as critical indicators of dysfunction. This convergence is driven by decades of soil science identifying those factors as primary controls on soil functions. Accordingly, a soil with very low organic matter, or very high salt or metal content, will be flagged as degraded whether the EU's broad dashboard or a site-specific living lab assessment are used. Another similarity is the shift toward quantitative thresholds. Even if some projects present results as scores, they are usually anchored to threshold values, often drawn from research or guidelines. This introduces a degree of objectivity and allows comparison. Moreover, the purpose of these frameworks is uniformly to guide action – classification is done to inform management, e.g. policy targets, remediation priorities, or farming practices.

The differences lie in context and level of integration. The EU-level frameworks (JRC, AI4SoilHealth) treat indicators mostly independently, a soil can be simultaneously "erosion-unhealthy" and "contamination-healthy", and are intended to be aggregated over large areas for reporting statistics. In contrast, project-level frameworks (REECOL, Soil Health Living Labs) often integrate multiple indicators to define holistic classes or profiles of soil condition at a local scale. For example, REECOL's combined classes like "infertile and compacted" cover both nutrient and physical indicators. This makes them more useful for on-site decision-making. Another difference is breadth vs. depth. A policy framework must be broad but might use simplified proxies, e.g. JRC's biodiversity indicator is a modelled risk rather than direct measurement, whereas a research project might directly measure biodiversity (like counting earthworms) but within a limited area.

In general, new frameworks also seek to fill gaps in existing frameworks. For instance, the initial 19 JRC indicators did not include “vegetation cover” or “landscape heterogeneity” due to data gaps, but Living Labs inherently monitor vegetation cover on their fields as a basic indicator of soil cover. Similarly, subsoil compaction is not well covered by the EU dashboard, which uses topsoil bulk density and some modelled information, but several projects have highlighted it (deep bulk density or penetrometer readings) as important, especially for agriculture.

Nevertheless, there can be identified some remaining gaps across these frameworks. One is the difficulty of biological indicators at scale. Soil biota are inherently local and variable, so setting Europe-wide classes for “biologically healthy soil” is complicated. Though efforts like DNA metastructures or the LUCAS Soil biodiversity index are underway to address this.

Temporal aspect can also differ. Policy frameworks might classify a soil based on its current state only, whereas in Living Labs, trend over time (improving or degrading) can be part of the classification, as some initiatives talk about soils “on a regenerative trajectory” vs “degenerating”. However, these dynamic considerations are likely to be integrated into future EU monitoring as well.

Another challenge is correlating soil functions with indicators. Measuring a property is one thing but deciding how much change in that property equates to a functional loss (a dysfunction) can be complex. For example, if bulk density increases from 1.3 to 1.5 g/cm^3, at what point the soil’s root growth function is considered critically impaired. The JRC report explicitly lists plans to refine thresholds, add indicators in compliance with Mission projects.

3.3. Proposed COFA classification framework of soil dysfunctions

The COFA project proposes a dedicated classification framework for soil dysfunctions in coal post-mining landscapes, designed to support the assessment of land suitability for agricultural reuse, with particular emphasis on energy crop cultivation, including their potential use as active reclamation tools. The framework builds on experience gained in the REECOL project, while addressing the specific characteristics of post-mining substrates, which frequently lack pedogenic continuity, exhibit strong spatial heterogeneity, and present combined physical, chemical, hydrological, and biological constraints.

In contrast to conventional agricultural soils, post-mining soils may be unsuitable for food production, yet still capable of supporting selected energy crops, which often display higher tolerance to adverse conditions and can simultaneously contribute to soil improvement through biomass production, organic matter input, and contaminant uptake or stabilization. Consequently, the COFA framework explicitly distinguishes between soil suitability for biomass production and soil suitability for reclamation-oriented energy cropping.

3.3.1. Conceptual structure of the COFA classification

1st degree of dysfunction – Functional but constrained soil

Soil properties deviate from the optimal state, yet the core soil functions remain preserved. The soil can still reliably support vegetation growth, even though conditions are not ideal. Typical limitations may include mild nutrient deficiency, reduced water-holding capacity, or slight compaction.

The first degree of dysfunction is characterised by only a mild deviation from optimal site conditions. Although certain parameters may not fully meet ideal values, the upper soil horizon generally remains functional and does not require targeted interventions. Vegetation is able to establish, grow, and persist without significant instability, and while these limitations may reduce overall yields or productivity, they do not substantially hinder the area's capacity to support agricultural production.

The site conditions allow for standard agricultural use. Energy crops, in particular, can be grown successfully under these conditions, with only minor adjustments in management practices to accommodate residual limitations in soil quality or structure.

2nd degree of dysfunction – Significantly limited but non-phytotoxic soil

Soil functions are significantly restricted, leading to reduced or unstable biomass production. However, the environment is not phytotoxic, so vegetation cover is sustainable but requires support.

The second degree of dysfunction is marked by a more pronounced deterioration of key physical, chemical, or biological soil parameters. Although vegetation is generally able to establish and grow, it often does so unstably or with notably reduced yields, reflecting the limited capacity of the substrate to support sustained production. In such cases, targeted improvements to the upper soil horizon are advisable, including the addition of organic matter, pH adjustment, or measures to enhance soil structure.

Under these conditions, land use is possible only with constraints: energy crops may be cultivated, but typically as part of broader soil-improvement strategies aimed at enhancing soil properties over time. Their role is thus dual, providing limited production while contributing to surface stabilization and gradual soil recovery.

3rd degree of dysfunction – Severely dysfunction until phytotoxic soil

The soil exhibits severe impairments, which may include high toxicity, extreme physical barriers, or combined negative factors. These conditions strongly limit or prevent vegetation establishment.

The third degree of dysfunction represents a state of severe degradation in which vegetation can establish only with great difficulty or fails to develop altogether. Such sites often exhibit toxic effects or extreme physical, chemical, or biological deterioration that prevent natural soil functions from occurring. Under these conditions, substantial interventions in the upper soil horizon are necessary, such as covering or isolating unsuitable material or fully replacing the dysfunctional substrate.

Productive cultivation is generally not feasible, and even hardy or tolerant energy crops can be used only in a very limited manner. Their role is typically restricted to supporting surface stabilization as part of broader reclamation or remediation efforts rather than serving as a viable production system.

In the subsequent deliverable (D3.3), the selected indicators and their threshold values for each degree of soil dysfunction will be presented in detail. These indicators covering physical, chemical, and biological soil properties will serve as practical criteria for assigning a site to the appropriate dysfunction category and for guiding the selection of suitable reclamation practice or soil-improvement measures.

4. Concepts in post-mining land transformation

The deliverable D3.2 provides a general conceptual introduction to reclamation-related terminology and the main approaches used in post-mining land transformation. Its purpose is to establish a shared vocabulary and clarify the distinctions between key concepts that will be used across the work packages. The detailed assessment of specific measures for optimizing site conditions and improving soil properties (based on predefined environmental and technical indicators) will be addressed in deliverable D3.3, which focuses on specific restoration strategies of post-mining areas for their agricultural utilization and their applicability under various site conditions.

4.1. Definitions and terminology in post-mining land transformation

There remain significant uncertainties in the terminology used to describe the transformation of post-mining areas and the planning of their future use. Terms such as reclamation, restoration, remediation, rehabilitation and reutilization are applied inconsistently across disciplines, sectors, and regions, and are often used interchangeably. Such inconsistency complicates communication among stakeholders, hinders strategic planning, and affects the evaluation and comparability of projects. To ensure consistent terminology across all work packages, we adopt the terminology framework already applied in other RFCS projects (Galetakis et al., 2025), and the following section provides clear explanations and distinctions among these terms.

Reclamation is the broadest and most comprehensive approach, integrating physical land reshaping, chemical amendment, biological activation, vegetation establishment, and long-term management to render disturbed land fit for a defined post-mining agricultural land use. Its goal is to restore the ecological value, functionality, and overall usability of the area, either by returning it to its original state or by creating a new type of land use (Franál et al., 2024). In post-coal mining areas, reclamation typically incorporates both remediation and rehabilitation measures, with intervention intensity increasing as soil dysfunction severity increases (Sholichin et al., 2025).

According to Qi et al. (2023), **soil rehabilitation** is based on the partial restoration of key soil functions, such as water retention, aggregation, nutrient cycling, or biological activity, without aiming to reconstruct the original soil profile or the entire ecosystem and is used where full ecological restoration is not technically or economically feasible. Franál et al. (2024) further characterize rehabilitation as a technical measure aimed at ensuring the stability and safety of an area, particularly on spoil heaps and slopes vulnerable to erosion or landslides. In this context, it is not about returning to the original ecosystem, but about creating long-term stable conditions, with rehabilitation often representing a necessary first step that precedes subsequent reclamation and ecological measures.

Restoration focuses on the most faithful return of the natural ecosystem. The goal is to restore the original species composition, soil conditions, and ecological functions so that an environment similar to the original is created (Franál et al., 2024). It is usually a longer and more demanding process, applied especially in areas with high ecological or conservation value. Restoration often supports the preservation of biodiversity, natural regeneration, and the overall ecological character of the landscape but in some case a full ecological restoration cannot be carried out in some cases (Qi et al., 2023).

Soil remediation refers to targeted actions aimed at eliminating, immobilizing, or reducing the presence of contaminants such as metals, organic pollutants, or extreme acidity and salinity. Remediation primarily addresses chemical and toxicological dysfunctions and is often a prerequisite for safe agricultural use (Dileep et al., 2023).

Reutilization means a new and practical use of land after mining, this time for non-ecological purposes. This could include, for example, creating areas for renewable energy sources, recreational use, agricultural production, housing, or industry. The essence is to adapt the land to current and future societal needs (Franál et al., 2024). Reuse thus often represents an opportunity for the socio-economic revitalization of regions that were negatively affected by the decline of mining.

4.2. Overview of reclamation approaches

Post-mining landscapes can be restored through several general types of reclamation, each reflecting a different vision for the future use of the land. In practice, these approaches rarely exist in isolation. Instead, they are often combined within a single reclaimed area to achieve ecological, hydrological, social, and economic functions that no single method could deliver on its own. For example, hydric reclamation can substantially improve the local water regime, which can subsequently facilitate agricultural use on adjacent terraces. Similarly, natural succession can precede agricultural reclamation by improving soil structure, adding organic matter, and gradually reducing contamination levels, thereby creating more favourable conditions for long-term crop production.

This chapter provides a general overview of reclamation approaches illustrated on several examples. Although agricultural reclamation is primary focus of COFA project, other types of reclamation are presented to illustrate the spectrum of available strategies, highlight their complementarity or competition, and clarify how stakeholder priorities influence the selection of the final land-use concept.

4.2.1. Forestry Reclamation

Forestry reclamation represents one of the most traditional and widely applied forms of post-mining restoration. Even when forestry is not the dominant target land use, forested belts or stabilizing woodland patches are frequently combined with agricultural or recreational reclamation to improve erosion control, increase landscape permeability, and enhance ecological functions.

Forestry reclamation involves planting tree seedlings on the prepared terrain and supporting the development of a stable ecosystem. In the Most Basin (CZ), it has traditionally been applied on spoil heaps, where resilient species such as spruce, pine, birch, acacia, and poplar are used to green and stabilise slopes and reduce erosion. Over time, deciduous and native species have also been introduced to improve biodiversity and ensure the long-term ecological stability of reclaimed areas.

While forestry reclamation can be highly effective, its success depends on the quality and chemical properties of the substrate. Cases such as Prunéřov VI (CZ), where tree mortality was high due to contamination with energy gypsum, highlight the importance of thoroughly assessing soil conditions before planting.

4.2.2. Agricultural reclamation

Agricultural reclamation is often a central objective in regions with strong agricultural traditions or where maintaining food and biomass production remains a strategic priority. Its success can be significantly enhanced by complementary measures such as initial succession phases or hydrological adjustments that improve soil fertility and moisture retention. In contrast to other approaches, agricultural reclamation explicitly aims to return land to long-term productive use while gradually rebuilding soil quality.

Agricultural reclamation was common practice in the Most Basin (CZ), especially at the Radovesice, Střimice, and Jirásek spoil heaps. The terrain was leveled, grassed, and selected areas were converted into arable land. Since spoil substrates often have low fertility, lack humus, and have an acidic reaction, this type of reclamation requires pH adjustment, nutrient supplementation, application of organic or mineral fertilizers, and gradual restoration of soil structure to achieve long-term agricultural production.

In recent years, increasing attention has been paid to the use of reclaimed areas for energy crops (Usťak et al., 2019; Malinská et al., 2020; Vávrová et al., 2021), as part of the transition from mining to sustainable agricultural use. Energy crops are particularly suitable due to their relative low requirements for soil quality, ability to grow on degraded or heterogeneous substrates, and capability to stabilize the surface of the spoil heaps, as illustrated on Fig. 4. They also contribute to increasing the content of organic matter and to the long-term improvement of soil properties. Furthermore, their cultivation allows for the combination of ecological functions with the production of renewable energy, which is a significant benefit in the current energy transition context.

On reclaimed areas, common market crops are still being tested, but their yields remain limited where deeper soil dysfunctions persist. Nevertheless, agricultural reclamation represents an important way to return part of the post-mining landscape to productive use, while gradually enhancing soil quality in the long term.

Fig. 4: Harvest of energy crop forage sorrel (*Rumex* sp.) on reclaimed area

4.2.3. Hydric reclamation

Hydric reclamation creates lakes and wetlands that can serve ecological, hydrological, recreational, or aesthetic functions. Even when hydric reclamation excludes certain areas from agricultural use, it often brings broader benefits for the surrounding landscape, such as stabilizing groundwater regimes or creating microclimatic improvements that can support adjacent agricultural reclamation zones.

In Czech Republic Lake Most (flooded Ležáky guarry in 2008-2012), Lake Milada near Ústí nad Labem (flooded Chabařovice quarry in 2001-2010) have been created by flooding, and another lake is planned for the ČSA quarry near Most and Chomutov city (Fig. 5). Existing lakes currently serve as favourable locations for various recreational and sporting activities, but they have rapidly become essential as key water bird habitats and for their broader ecological functions. Free water surfaces are also currently targeted in development and strategic plans for implementing renewable energy production.

Fig. 5: The initial phase of the formation of a natural lake in the ČSA quarry (CZ). Source: VUHU a.s., 2025.

4.2.4. Other types of reclamation

Recreational and socio-economic forms of reclamation reflect stakeholder preferences for cultural, leisure, or commercial uses of post-mining landscapes. These approaches can coexist with ecological or succession-based processes, creating multifunctional mosaic landscapes. In some planning contexts, these recreational uses may compete with agricultural reclamation, depending on regional development priorities.

A significant example is the Hippodrome near the town of Most, which was created on a reclaimed spoil heap and represents one of the largest horse racing venues in the Czech Republic. Similarly, the Autodrome in Most, built on a former mining site, demonstrates the possibility of using shaped terrain for motorsports and leisure activities. Reclaimed areas can also be used for golf courses or cycling paths, which take advantage of the specific topography of the spoil heaps. These projects show that post-mining landscapes can be effectively adapted for cultural, social, and recreational purposes, contributing to a new identity and economic revitalization of the region.

4.2.5. Succession

Succession represents a natural pathway through which ecosystems re-establish themselves on disturbed substrates. Even in landscapes prioritizing agricultural reclamation, succession can play a preparatory role by improving soil conditions and reducing remediation costs. Conversely, in areas intended for conservation or recreation, succession may serve as the dominant reclamation strategy.

Succession can be either managed or spontaneous (Bradshaw, 1997). Numerous studies (Frouz et al., 2008; Prach et al., 2001; Řehoř et al., 2022; Šebelíková et al., 2016, 2019; Spasić et al., 2024) confirm the significantly positive effect of spontaneous succession. In previously inhospitable post-mining areas, natural vegetation development can be observed over the years in a number of examples.

On the Radovesice spoil heap, two experimental plots (20 ha and 32 ha) were designated in the 1980s for natural development without any reclamation interventions. These areas (Radovesice XVIIA and XVIIB) gradually greened over as pioneering grasses, shrubs, and trees established themselves. Nowadays, they represent the largest succession sites in the Czech Republic, and their soil profiles are now almost indistinguishable from those of actively reclaimed spoil heaps (Řehoř et al., 2022).

However, succession-based reclamation is not universally applicable. It is essential to know the history of the area as well as its detailed soil conditions, as the success depends on the initial substrate not being excessively toxic or excessively dry. Otherwise, the effect of succession can be significantly limited, as illustrated in the Fig. 6. In suitable locations, on the other hand, spontaneous succession has proven to be an effective and economically advantageous measure, as it enhances biodiversity and promotes the formation of mixed-age stands. The question of whether spontaneous succession or controlled reclamation of spoil heaps is more appropriate is difficult to answer (Bradshaw, 1997).

Reclamation has a long tradition in the Czech Republic, with many documented examples of both successful and unsuccessful approaches. The unsuccessful cases are especially valuable, as they highlight risks and limitations that can inform more appropriate management strategies in the future. For example, at the Prunéřov VI site in the Most Basin, a high mortality rate of planted trees occurred due to later contamination with energy gypsum, which was easily accessible in the vicinity (Řehoř, n.d). Conversely, successful reclamations, such as several sections in Radovesice (Fig. 7), serve as important reference models, and help achieve further successes (Řehoř et al., 2022). When restoring post-mining landscapes, it is essential to consider the site-specific needs, limits, and environmental history. In the case of spontaneous succession, particular attention must be given to the degree of soil dysfunction, which may prevent or significantly delay natural vegetation development. Highly acidic or even phytotoxic substrates, such as those documented at Střimice, are typically unsuitable for spontaneous succession. These areas may serve better as research or experimental plots, as establishing a stable vegetation cover is extremely challenging and sometimes nearly impossible (Lago-Vila et al., 2019). Species tolerant of acidic conditions include, for example, *Deschampsia flexuosa*, which improves soil structure and thus supports natural weathering processes that mitigate extreme acidity. On soils containing hazardous elements, natural colonization can lead to local adaptation. Species from the genera *Agrostis* and *Festuca* may develop tolerance to elevated trace elements (Bradshaw, 1997). Their biomass inputs increase organic matter content, which contributes to the complexation of available harmful elements and thus reduces toxicity over time (Vráblíková et al., 2018). In inhospitable conditions, it is more appropriate to apply controlled succession, which allows partial guidance of vegetation development and increases the likelihood of restoration success and environmental stabilization.

Fig. 6: Slow vegetation growth observed on research experimental successional area Pokrok XI established in 2010 (CZ). Source: VUHU a.s., 2025.

Fig. 7: Research experimental successional area Radovesice XVIIB established in 2000 (CZ). Source: VUHU a.s., 2025.

5. Conclusion and recommendations

Deliverable D3.2 provides a comprehensive overview of soil dysfunctions affecting post-mining landscapes in the Czech Republic, Poland, and Greece, and introduces an integrated framework for classifying their severity in relation to agricultural reclamation potential. The findings underscore that soil dysfunctions, whether physical, hydrological, chemical/toxicological, or biological - rarely occur in isolation. Their interactions shape both the feasibility of agricultural production and the selection of appropriate reclamation pathways. The proposed three-degree classification offers a clear, actionable structure for interpreting site conditions: from mildly constrained but functional soils, through significantly limited yet non-phytotoxic substrates, to severely degraded or phytotoxic areas requiring major interventions.

By formalising these dysfunctions and linking them with feasible land-use options, D3.2 establishes an essential diagnostic foundation for the COFA project. It connects the spatial identification of degraded land (Task 3.1) with the technical and agronomic measures that will be evaluated in Task 3.3, and it provides WP5 with the baseline logic needed to parameterise agricultural, carbon-farming, and scenario-based planning tools. At the same time, D3.2 supports WP4 by identifying critical environmental risks, contamination patterns, and soil limitations relevant for legal compliance, stakeholder dialogue, and socio-environmental feasibility assessments.

6. References

Abakumov, E. V., Cajthaml, T., Brus, J., & Frouz, J. (2013). Humus accumulation, humification, and humic acid composition in soils of two post-mining chronosequences after coal mining. *Journal of Soils and Sediments*, 13(3), 491–500.

Abliz, A., Shi, Q., Keyimu, M., & Sawut, R. (2018). Spatial distribution, source, and risk assessment of soil toxic metals in the coal-mining region of northwestern China. *Arabian Journal of Geosciences*, 11(24), 793.

Adamec, S., Tůmová, Š., Hošek, M., Lučić, M., & Grygar, T. M. (2024). Pitfalls of distinguishing anthropogenic and geogenic reasons for risk elements in soils around coal-fired power plants: from a case study in the Northwestern Czech Republic to general recommendations. *Journal of Soils and Sediments,* 24(3), 1274–1288.

Agriculture Victoria. (2025). Soil acidity. ONLINE: *https://agriculture.vic.gov.au/farm-management/soil/soil-acidity*

Ahirwal, J., & Maiti, S. K. (2016). Assessment of soil properties of different land uses generated due to surface coal mining activities in tropical Sal (Shorea robusta) forest, India. Catena, 140, 155–163.

AI4SoilHealth. (n.d.). Robust indicator framework. Retrieved December 12, 2026, from *https://ai4soilhealth.eu/robust-indicator-framework/*

Alejandro, S., Höller, S., Meier, B., & Peiter, E. (2020). Manganese in plants: from acquisition to subcellular allocation. *Frontiers in Plant Science*, 11, 300.

Alloway, B. J. (2012). Sources of heavy metals and metalloids in soils. In *Heavy metals in soils: trace metals and metalloids in soils and their bioavailability* (pp. 11–50). Springer.

Angers, D. A., & Caron, J. (1998). Plant-induced changes in soil structure: processes and feedbacks. *Biogeochemistry*, 42(1), 55–72.

Bartuška, M. (2014). Soil organic matter dynamics at reclaimed and unreclaimed post-mining sites.

Bartuška, M., Pawlett, M., & Frouz, J. (2015). Particulate organic carbon at reclaimed and unreclaimed post-mining soils and its microbial community composition. *Catena*, 131, 92–98.

Basset, C., Abou Najm, M., Ghezzehei, T., Hao, X., & Daccache, A. (2023). How does soil structure affect water infiltration? A meta-data systematic review. *Soil and Tillage Research*, 226, 105577.

Bhuiyan, M. A. H., Parvez, L., Islam, M. A., Dampare, S. B., & Suzuki, S. (2010). Heavy metal pollution of coal mine-affected agricultural soils in the northern part of Bangladesh. *Journal of Hazardous Materials*, 173(1–3), 384–392.

Bílková, G., Elznicová, J., Wildová, E., Hošek, M., & Grygar, T. M. (2023). Deciphering Factors Controlling Manganese Concentrations in the Leaves of Silver Birch (Betula pendula Roth) in Relation to Recent Acidification of Mountain Forest Soils. *Polish Journal of Environmental Studies*, 32(1).

Boahen, F., Száková, J., Kališová, A., Najmanová, J., & Tlustoš, P. (2023). The assessment of the soil–plant-animal transport of the risk elements at the locations affected by brown coal mining. *Environmental Science and Pollution Research*, 30(1), 337–351.

Bodlák, L., Křováková, K., Kobesová, M., Brom, J., Šťastný, J., & Pecharová, E. (2012). SOC content—An appropriate tool for evaluating the soil quality in a reclaimed post-mining landscape. *Ecological Engineering*, 43, 53–59.

Bonfante, A., Basile, A., & Bouma, J. (2020). Targeting the soil quality and soil health concepts when aiming for the United Nations Sustainable Development Goals and the EU Green Deal. *Soil*, 6(2), 453–466.

Bradshaw, A. (1997). Restoration of mined lands – using natural processes. *Ecological Engineering*, 8(4), 255–269.

Bronick, C. J., & Lal, R. (2005). Soil structure and management: a review. *Geoderma*, 124(1–2), 3–22.

Broothaerts, N., Panagos, P., & Jones, A. (2024). A proposal for soil health indicators at EU-level. https://doi.org/10.2760/8953204

Bullock, P. (2005). *Climate change impacts*.

Bünemann, E. K., Bongiorno, G., Bai, Z., Creamer, R. E., De Deyn, G., De Goede, R., Fleskens, L., Geissen, V., Kuyper, T. W., & Mäder, P. (2018). Soil quality–A critical review. *Soil Biology and Biochemistry*, 120, 105–125.

Campbell, G., Smith, P., Robinson, D. A., Shokri, N., Afshar, M., Arthur, E., & Wollesen de Jonge, L. (2024). Indicators for soil degradation and method testing for AI4SoilHealth. Deliverable D3.3.

Chiurazzi, M., Frugis, G., & Navazio, L. (2025). Symbiotic nitrogen fixation: a launchpad for investigating old and new challenges. In Journal of Experimental Botany (Vol. 76, Issue 6, pp. 1473–1477). *Oxford University Press UK.*

de Almeida, W. S., Panachuki, E., de Oliveira, P. T. S., da Silva Menezes, R., Sobrinho, T. A., & de Carvalho, D. F. (2018). Effect of soil tillage and vegetal cover on soil water infiltration. *Soil and Tillage Research*, 175, 130–138.

de Quadros, P. D., Zhalnina, K., Davis-Richardson, A. G., Drew, J. C., Menezes, F. B., Camargo, F. A. de O., & Triplett, E. W. (2016). Coal mining practices reduce the microbial biomass, richness and diversity of soil. *Applied Soil Ecology*, 98, 195–203.

Dileep, R., Kiranakumara, K., & Kumar, V. (2023). Arsenic and its management in soil. *Ecology, Environment and Conservation*, *29*, 407-410.

EJP Soil. (n.d.). https://ejpsoil.eu/

European Commission. (2020). Caring for soil is caring for life. ONLINE: *https://research-and-innovation.ec.europa.eu/knowledge-publications-tools-and-data/publications/all-publications/caring-soil-caring-life_en*

European Commission. (2021). EU Soil Strategy for 2030 (EUR-Lex: COM/2021/699). ONLINE: *https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:52021DC0699*

European Commission. (2025). Mission Soil Platform - Living Labs. ONLINE: *https://mission-soil-platform.ec.europa.eu/living-labs*

European Environment Agency. (2023). Soil monitoring in Europe – Indicators and thresholds for soil health assessments (EEA Report 08/2022).

European Parliament. (2025). Directive (EU) 2025/2360 of the European Parliament and of the Council of 12 November 2025 on soil monitoring and resilience (Soil Monitoring Law). *http://data.europa.eu/eli/C/2024/5371/oj.*

Faber, J. H., Hanegraaf, M., Gillikin, A., Hendriks, C., Kuikman, P., Cousin, I., Bispo, A., Obiang-Ndong, G., Montagne, D., Taylor, A., Bengtsson, J., Ten Damme, L., Meurer, K. H. E., Viketoft, M., Faber, J. H., Meurer, K. H. E., Hendriks, C. M. J., Bispo, A., Viketoft, M., … Bengtsson, J. (2022). Stocktaking for Agricultural Soil Quality and Ecosystem Services Indicators and their Reference Values (SIREN). *EJP SOIL Deliverable D2.* www.ejpsoil.eu

Feng, Y., Wang, J., Bai, Z., & Reading, L. (2019). Effects of surface coal mining and land reclamation on soil properties: A review. *Earth-Science Reviews*, 191, 12–25.

Ferreira, C. S. S., Keizer, J. J., Santos, L. M. B., Serpa, D., Silva, V., Cerqueira, M., Ferreira, A. J. D., & Abrantes, N. (2018). Runoff, sediment and nutrient exports from a Mediterranean vineyard under integrated production: An experiment at plot scale. Agriculture, Ecosystems & Environment, 256, 184–193.

Frantál, B., Pasqualetti, M. J., & Brisudová, L. (2024). Challenges, dilemmas and success criteria of recycling coal mining landscapes. *Moravian Geographical Reports*, 32(4), 216-232.

Frouz, J., Elhottová, D., Kuráž, V., & Šourková, M. (2006). Effects of soil macrofauna on other soil biota and soil formation in reclaimed and unreclaimed post mining sites: Results of a field microcosm experiment. *Applied Soil Ecology*, 33(3), 308–320.

Frouz, J., Livečková, M., Albrechtová, J., Chroňáková, A., Cajthaml, T., Pižl, V., Háněl, L., Starý, J., Baldrian, P., & Lhotáková, Z. (2013). Is the effect of trees on soil properties mediated by soil fauna? A case study from post-mining sites. *Forest Ecology and Management*, 309, 87–95.

Frouz, J., Prach, K., Pižl, V., Háněl, L., Starý, J., Tajovský, K., Materna, J., Balík, V., Kalčík, J., & Řehounková, K. (2008). Interactions between soil development, vegetation and soil fauna during spontaneous succession in post mining sites. *European Journal of Soil Biology*, 44(1), 109–121.

Galetakis, M., Bajcar, A., Dužeková, R., Lazăr, M., Faur F. (2025). Report on crucial factors influencing dumps and potential indicators of dump deformation and best practices and failures for re-utilization. *MiDSafe Deliverable D2.2.*

Gopinathan, P., Jha, M., Singh, A. K., Mahato, A., Subramani, T., Singh, P. K., & Singh, V. (2022). Geochemical characteristics, origin and forms of sulphur distribution in the Talcher coalfield, India. *Fuel*, 316, 123376.

Gopinathan, P., Santosh, M. S., Dileepkumar, V. G., Subramani, T., Reddy, R., Masto, R. E., & Maity, S. (2022). Geochemical, mineralogical and toxicological characteristics of coal fly ash and its environmental impacts. *Chemosphere*, 307, 135710.

Goulding, K. W. T. (2016). Soil acidification and the importance of liming agricultural soils with particular reference to the United Kingdom. *Soil Use and Management*, 32(3), 390–399.

Grygar, T. M., Adamec, S., Tůmová, Š., Burdová, H., Kříženecká, S., Bretschneiderová, D., Lučić, M., & Pavlů, I. (2025). Identification of coal combustion impacts on soil contamination by risk elements needs empirical holistic approach: case study in the Most Basin, Czech Republic. *Environmental Earth Sciences*, 84(16), 446.

Habib, M. A., Basuki, T., Miyashita, S., Bekelesi, W., Nakashima, S., Phoungthong, K., Khan, R., Rashid, M. B., Islam, A. R. M. T., & Techato, K. (2019). Distribution of naturally occurring radionuclides in soil around a coal-based power plant and their potential radiological risk assessment. *Radiochimica Acta,* 107(3), 243–259.

Han, Y., Zhao, W., Ding, J., & Ferreira, C. S. S. (2023). Soil erodibility for water and wind erosion and its relationship to vegetation and soil properties in China's drylands. *Science of the Total Environment*, 903, 166639.

Háněl, L. (2002). Development of soil nematode communities on coal-mining dumps in two different landscapes and reclamation practices. *European Journal of Soil Biology*, 38(2), 167–171.

Harris, J. A., Birch, P., & Short, K. C. (1989). Changes in the microbial community and physico-chemical characteristics of topsoils stockpiled during opencast mining. *Soil Use and Management*, 5(4), 161–168.

Haurová, J. (2023). Toxicita výsypkových materiálů a faktory, které ji ovlivňují. *Charles University*.

Hendrychová, M., Šálek, M., Tajovský, K., & Řehoř, M. (2012). Soil properties and species richness of invertebrates on afforested sites after brown coal mining. *Restoration Ecology*, 20(5), 561–567.

Hlava, J., Hlavová, A., Hakl, J., & Fér, M. (2015). Earthworm responses to different reclamation processes in post opencast mining lands during succession. *Environmental Monitoring and Assessment*, 187(1), 4108.

Hofman, V., Hruška, M., Voltr, V., Kučera, J., Podzemná, L., Jarušek, J., Havelka, J., Pavlík, F., Poláková, Š., Kubík, L., Reininger, D., Smatanová, M., Novotný, I., Vopravil, J., Čechmánková, J., Fučík, P., Kučera, J., Perglerová, M., Vojtová, I., … Gimunová, T. (2024). *Půda: situační a výhledová zpráva.*

Horn, R., Taubner, H., Wuttke, M., & Baumgartl, T. (1994). Soil physical properties related to soil structure. *Soil and Tillage Research*, 30(2–4), 187–216.

Hu, Y., Yu, Z., Fang, X., Zhang, W., Liu, J., & Zhao, F. (2020). Influence of mining and vegetation restoration on soil properties in the eastern margin of the Qinghai-Tibet Plateau. *International Journal of Environmental Research and Public Health*, 17(12), 4288.

Huang, K., Li, M., Li, R., Rasul, F., Shahzad, S., Wu, C., Shao, J., Huang, G., Li, R., & Almari, S. (2023). Soil acidification and salinity: the importance of biochar application to agricultural soils. *Frontiers in Plant Science*, 14, 1206820.

Idowu, J., & Angadi, S. (2013). Understanding and managing soil compaction in agricultural fields. NM State University, *Cooperative Extension Service*.

Issaka, S., & Ashraf, M. A. (2017). Impact of soil erosion and degradation on water quality: a review. *Geology, Ecology, and Landscapes*, 1(1), 1–11.

Jačka, L., Walmsley, A., Kovář, M., & Frouz, J. (2021). Effects of different tree species on infiltration and preferential flow in soils developing at a clayey spoil heap. *Geoderma*, 403, 115372.

Javůrek, M., & Vach, M. (2008). Negativní vlivy zhutnění půd a soustava opatření k jejich odstranění. *Výzkumný ústav rostlinné výroby*.

Johnson, R. (2023). Soil-water retention and its role in crop yield optimization. *International Journal of Advanced Chemistry Research*, 5(1), 117–120.

Juliev, M., Kholmurodova, M., Abdikairov, B., & Abuduwaili, J. (2024). A comprehensive review of soil erosion research in Central Asian countries (1993–2022) based on the Scopus database. *Soil & Water Research,* 19(4).

Juřicová, A., Chuman, T., & Žížala, D. (2022). Soil organic carbon content and stock change after half a century of intensive cultivation in a chernozem area. *Catena*, 211, 105950.

Kibblewhite, M. G., Jones, R. J. A., Montanarella, L., Baritz, R., Huber, S., Arrouays, D., Micheli, E., & Stephens, M. (2008). Environmental assessment of soil for monitoring volume VI: Soil monitoring system for Europe. EUR 23490 EN/6. (Vol. 23490).

Kogelmann, W. J., & Sharpe, W. E. (2006). Soil acidity and manganese in declining and nondeclining sugar maple stands in Pennsylvania. *Journal of Environmental Quality*, 35(2), 433–441.

Kou, J., Gan, Y., Lei, S., Meng, W., Feng, C., & Xiao, H. (2022). Soil health and ecological risk assessment in the typical coal mines on the Mongolian Plateau. *Ecological Indicators*, 142, 109189.

Krümmelbein, J., & Raab, T. (2012). Development of soil physical parameters in agricultural reclamation after brown coal mining within the first four years. *Soil and Tillage Research*, 125, 109–115.

Kuráž, V., Frouz, J., Kuráž, M., Mako, A., Shustr, V., Cejpek, J., Romanov, O. V, & Abakumov, E. V. (2012). Changes in some physical properties of soils in the chronosequence of self-overgrown dumps of the Sokolov quarry-dump complex, Czechia. *Eurasian Soil Science*, 45(3), 266–272.

Lago-Vila, M., Arenas-Lago, D., Rodríguez-Seijo, A., Andrade, M. L., & Vega, F. A. (2019). Ability of Cytisus scoparius for phytoremediation of soils from a Pb/Zn mine: Assessment of metal bioavailability and bioaccumulation. *Journal of Environmental Management*, 235, 152–160.

Lehmann, J., Bossio, D. A., Kögel-Knabner, I., & Rillig, M. C. (2020). The concept and future prospects of soil health. *Nature Reviews Earth & Environment*, 1(10), 544–553.

Lipiec, J., & Hatano, R. (2003). Quantification of compaction effects on soil physical properties and crop growth. *Geoderma*, 116(1–2), 107–136.

Lu, X., Wang, Z., Zhao, M., Peng, S., Geng, S., & Ghorbani, H. (2025). Data-driven insights into climate change effects on groundwater levels using machine learning. *Water Resources Management,* 39(7), 3521–3536.

Madaras, M., Mayerová, M., Stehlík, M., & Czakó, A. (2020). Stanovení kvality půdní struktury pomocí mobilních zařízení. *VÚRV, vvi.*

Magdoff, Fred., & Van Es, Harold. (2021). Building soils for better crops : ecological management for healthy soils. *Sustainable Agriculture Research & Education.*

Malinská, H., Pidlisnyuk, V., Nebeská, D., Erol, A., Medžová, A., & Trögl, J. (2020). Physiological response of Miscanthus x giganteus to plant growth regulators in nutritionally poor soil. *Plants*, 9(2), 194.

Markowska, M., Pierzchała, Ł., Hamerla, A., Białowąs, M., Janson, E., Łabaj, P., Szwaja, A., Musiałek, M., Řehoř, M., Schmidt, P., Roumpos, C., Louloudis, G., Mertiri, E., Servou, A., Boivin, S., Bert, V., Lamot, A., & Kamenik, M. (2024). Document on identification of main climate change impacts on post-mining rehabilitation schemes. *REECOL Deliverable D3.3.*

Matys Grygar, T., Königová, M., Adamec, S., Bílková, G., Vorm, P., & Hýlová, V. (2025). Nutrition Status of Trees on Spoil Heaps After Coal Mining Can Be Inferred From Seasonal Dynamics of Foliar Nutrient Concentrations. *Land Degradation & Development.*

Maurya, S., Abraham, J. S., Somasundaram, S., Toteja, R., Gupta, R., & Makhija, S. (2020). Indicators for assessment of soil quality: a mini-review. *Environmental Monitoring and Assessment*, 192(9), 604.

Merilä, P., Malmivaara-Lämsä, M., Spetz, P., Stark, S., Vierikko, K., Derome, J., & Fritze, H. (2010). Soil organic matter quality as a link between microbial community structure and vegetation composition along a successional gradient in a boreal forest. *Applied Soil Ecology*, 46(2), 259–267.

Mishra, A. K., & Singh, V. P. (2010). A review of drought concepts. *Journal of Hydrology*, 391(1–2), 202–216.

Mudrák, O., & Frouz, J. (2018). Earthworms increase plant biomass more in soil with no earthworm legacy than in earthworm-mediated soil, and favour late successional species in competition. *Functional Ecology*, 32(3), 626–635.

Munford, K. E., Casamatta, M., Basiliko, N., Glasauer, S., Mykytczuk, N. C. S., & Watmough, S. A. (2021). Paper birch (Betula papyrifera) nutrient resorption rates on nutrient-poor metal-contaminated soils and mine tailings. *Water, Air, & Soil Pollution,* 232(1), 33.

Musiałek, M., Szwaja, A., Kania, M., Pierzchała, Ł., Janson, E., Łabaj, P., Řehoř, M., Schmidt, P., Christos Roumpos, Louloudis, G., Mertiri, E., Servou, A., Derycke, V., & Stéphane Boivin. (2024). Document on classification of ecosystem degradation including of degraded land maps. *REECOL Deliverable D3.1.*

Musiałek, M., Szwaja, A., Řehoř, M., Roumpos, C., Louloudis, G., Servou, A., Mertiri, E., Paraskevis, N., Pierzchała, Ł., Kruczek, M., Łabaj, P., Bezak, B., Balaga, D., Lamot, A., & Kamenik, M. (2024). Report on characteristics of post-mining reclamation case study areas. *REECOL Deliverable D3.4.*

Nagajyoti, P. C., Lee, K. D., & Sreekanth, T. V. M. (2010). Heavy metals, occurrence and toxicity for plants: a review. *Environmental Chemistry Letters*, 8(3), 199–216.

Nemes, A., & Rawls, W. J. (2004). Soil texture and particle-size distribution as input to estimate soil hydraulic properties. *Developments in Soil Science*, 30, 47–70.

Pająk, M., Błońska, E., Szostak, M., Gąsiorek, M., Pietrzykowski, M., Urban, O., & Derbis, P. (2018). Restoration of vegetation in relation to soil properties of spoil heap heavily contaminated with heavy metals. *Water, Air, & Soil Pollution*, 229(12), 392.

Papadopoulou, C.-I., Loizou, E., Chatzitheodoridis, F., Michailidis, A., Karelakis, C., Fallas, Y., & Paltaki, A. (2023). What makes farmers aware in adopting circular bioeconomy practices? Evidence from a Greek rural region. *Land*, 12(4), 809.

PhycoTerra. (2022, November 9). Improve Your Soil Structure and Prevent Soil Compaction with Proper Management and Soil Amendments. ONLINE: *https://phycoterra.com/blog/improve-soil-structure-prevent-soil-compaction/*

Pisani, C., & Soriano Disla, J. M. (2025). Catalogue of Soil Health Indicators. *iCOSHELLs Deliverable D3.1.*

Prach, K., Bartha, S., Joyce, C. B., Pyšek, P., Van Diggelen, R., & Wiegleb, G. (2001). The role of spontaneous vegetation succession in ecosystem restoration: a perspective. *Applied Vegetation Science,* 4(1), 111–114.

Psarraki, D., Papazotos, P., Vasileiou, E., & Perraki, M. (2023). Mapping the Soil Chemical Properties of the Sarigkiol Basin, Western Macedonia, Greece, in View of the Transition to the Post-Lignite Era. *Materials Proceedings*, 15(1), 33.

Puettmann, W., & Schaefer, R. G. (1990). Assessment of carbonization properties of coals by analysis of trapped hydrocarbons. *Energy & Fuels*, 4(4), 339–346.

Qi, L., Sun, S., Gao, K., Ren, W., Liu, Y., Chen, Z., & Yuan, X. (2023). Effect of reclamation years on soil physical, chemical, bacterial, and fungal community compositions in an open-pit coal mine dump in grassland area of Inner Mongolia, China. *Land Degradation & Development*, *34*(12), 3568-3580.

REECOL. (n.d.). Ecological rehabilitation and long tern monitoring of post mining areas. *https://reecol.komag.eu/*

Řehoř, M. (n.d.). Description of soil degradation.

Řehoř, M., Jarolimová, M., Žižka, L., Schmidt, P., Vráblík, P., & Vráblíková, J. (2018). The Results of the Long-Term Research on the Development of Anthropomorphic Soil in the Reclaimed and Successive Areas of the Most Basin in Czech Republic. *Journal of Mining Science*, 53(4), 778–788.

Řehoř, M., Kostková, A., Fraštia, M., & Schmidt, P. (2022). Výsypka Radovesice–příklad úspěšné rekultivace v podmínkách 21. století. *Zpravodaj Hnědé Uhlí*, 2.

Řehoř, M., Pierzchała, Ł., Deska, M., Warzecha, A., Musiałek, M., Maksymowicz, M., & Goutant, J. (2024). Report on site study cases: initial assessment. *REECOL Deliverable D5.2.*

Řehoř, M., Tichá, L., & Königová, M. (2025). Půdní dysfunkce na lokalitách v mostecké pánvi. *VUHU a.s.*

Rengasamy, P. (2006). World salinization with emphasis on Australia. *Journal of Experimental Botany*, 57(5), 1017–1023.

Ritter, J. B., & Gardner, T. W. (1993). Hydrologic evolution of drainage basins disturbed by surface mining, central Pennsylvania. *Geological Society of America Bulletin*, 105(1), 101–115.

Rouhani, A., Gusiatin, M. Z., & Hejcman, M. (2023). An overview of the impacts of coal mining and processing on soil: Assessment, monitoring, and challenges in the Czech Republic. *Environmental Geochemistry and Health*, 45(11), 7459–7490.

Schwarzbauer, J., & Vossen, J. (2024). The potential of polycyclic aromatic compounds as indicators for soil contamination by open-pit mining and utilization of lignite: a case study at Ptolemaida basin, Greece. *International Journal of Earth Sciences*, 113(8), 2373–2383.

Šebelíková, L., Csicsek, G., Kirmer, A., Vítovcová, K., Ortmann-Ajkai, A., Prach, K., & Řehounková, K. (2019). Spontaneous revegetation versus forestry reclamation—Vegetation development in coal mining spoil heaps across Central Europe. *Land Degradation & Development*, 30(3), 348–356.

Šebelíková, L., Řehounková, K., & Prach, K. (2016). Spontaneous revegetation vs. forestry reclamation in post-mining sand pits. *Environmental Science and Pollution Research*, 23(14), 13598–13605.

Sholichin, M., Putri, S. E., Barlian, E., Umar, I., Syah, N., & Gusman, M. (2025). Land reclamation strategy for former coal mining in Kutai Kartanegara Regency, East Kalimantan Province. *Multidisciplinary Science Journal*, (Accepted Articles).

Skala, J., Boahen, F., Száková, J., Vacha, R., & Tlustoš, P. (2022). Arsenic and lead in soil: impacts on element mobility and bioaccessibility. *Environmental Geochemistry and Health*, 44(3), 943–959.

Skala, J., Grygar, T. M., & Achasova, A. (2024). Novel definition of local baseline values for potentially toxic elements in Czech farmland using adaptive spatial weighting. *Applied Geochemistry*, 170, 106082.

Smith, P., Robinson, D. A., Shokri, N., Afshar, M., Arthur, E., Moldrup, P., & Matson, A. (2024). Indicators for soil degradation and method testing for AI4SoilHealth. *AI4SoilHealth D3. 3.*

Soil Health Benchmarks. (n.d.). ONLINE: *https://soilhealthbenchmarks.eu/*

Soil Quality Knowledge Base. (2024a). Soil compaction. Soil Compaction. ONLINE: *https://soilqualityknowledgebase.org.au/soil-compaction/*

Soil Quality Knowledge Base. (2024b). Soil erosion. Soil Erosion. ONLINE: *https://soilqualityknowledgebase.org.au/soil-erosion/*

Soil Quality Knowledge Base. (2024c). Soil stability. ONLINE: *https://soilqualityknowledgebase.org.au/soil-stability/*

SOILL. (2024). SOILL. Support Structure for Soil Health Living Labs and Lighthouses. ONLINE: *https://soill2030.eu/*

Spain, A. V., & Hollingsworth, I. (2016, March). Selected properties of the incipient soils developing on coal mining wastes, Bowen Basin, Australia. In *Mine Closure 2016: Proceedings of the 11th International Conference on Mine Closure* (pp. 173-186). Australian Centre for Geomechanics.

Spasić, M., Borůvka, L., Vacek, O., Drábek, O., & Tejnecký, V. (2021). Pedogenesis problems on reclaimed coal mining sites. *Soil and Water Research*, 16(3), 137–150.

Spasić, M., Vacek, O., Vejvodová, K., Tejnecký, V., Vokurková, P., Križová, P., Polák, F., Vašát, R., Borůvka, L., & Drábek, O. (2024). Which trees form the best soil? Reclaimed mine soil properties under 22 tree species: 50 years later—Assessment of physical and chemical properties. *European Journal of Forest Research*, 143(2), 561–579.

Stolte, J., Tesfai, M., Keizer, J., Øygarden, L., Kværnø, S., Verheijen, F., & Hessel, R. (2016). Soil threats in Europe. EUR 27607, JRC Scientific and Technical Reports. *European Commission. Joint Research Centre.*

Stournaras, G., Skourtos, M., Kontogianni, A., Yoxas, G., Machleras, A., & Nastos, P. (2011). The risks and impacts of climate change by sector. In The environmental, economic and social impacts on climate change in Greece. *Bank of Greece.*

Štrudl, M., Borůvka, L., Dimitrovský, K., & Kozák, J. (2006). Contents of potentially risk elements in natural and reclaimed soils of the Sokolov region.

Theocharopoulos, S. P., Wagner, G., Sprengart, J., Mohr, M.-E., Desaules, A., Muntau, H., Christou, M., & Quevauviller, P. (2001). European soil sampling guidelines for soil pollution studies. *Science of the Total Environment*, 264(1–2), 51–62.

Tsigaridas, K. (2014). Study of heavy metal and radionuclide contamination in Kozani-Ptolemais basin with the use of bioindicators, water and microorganisms. *Aristotle University of Thessaloniki.*

Tuo, D., Lu, Q., Wu, B., Li, Q., Yao, B., Cheng, L., & Zhu, J. (2023). Effects of wind–water erosion and topographic factor on soil properties in the Loess Hilly Region of China. *Plants*, 12(13), 2568.

Ussiri, D. A. N., Jacinthe, P.-A., & Lal, R. (2014). Methods for determination of coal carbon in reclaimed minesoils: A review. *Geoderma*, 214, 155–167.

Usťak, S., Šinko, J., & Muňoz, J. (2019). Chrastice rákosovitá (Phalaris arundinacea L.) jako slibná energetická plodina. *Journal of Central European Agriculture*, 20(4), 1143-1168.

Vácha, R., Skála, J., Čechmánková, J., Horváthová, V., & Hladík, J. (2015). Toxic elements and persistent organic pollutants derived from industrial emissions in agricultural soils of the Northern Czech Republic. *Journal of Soils and Sediments*, 15(8), 1813–1824.

Vávrová, K., Weger, J., Outrata, D., Rumportl, D., Knápek, J., Králík, T. (2021). Metodika alokace porostů energetických plodin na zemědělské půdě zohledňující mimoprodukční funkce v krajině. 3/2021-054. *VÚKOZ, v. v. i.*

Vráblíková, J., & Vráblík, P. (2008). Aplikovaná pedologie (1st ed.). Univerzita J.E. Purkyně v Ústí nad Labem, Fakulta životního prostředí.

Vráblíková, J., Vráblík, P., Šoch, M., Blažková, M., Řehoř, M., Schmidt, P., & Wildová, E. (2018). Metodika ochrany půdy tvorbou a udržováním trvalých travních porostů a optimalizace rekultivačních postupů v antropogenně zatížené krajině regionu Podkrušnohoří. Univerzita JE Purkyně v Ústí nad Labem, Fakulta životního prostředí.

Wahsha, M., Nadimi-Goki, M., & Bini, C. (2016). Land contamination by toxic elements in abandoned mine areas in Italy. *Journal of Soils and Sediments*, 16(4), 1300–1305.

Więckol-Ryk, A., Pierzchała, Ł., Bauerek, A., & Krzemień, A. (2023). Minimising Coal Mining’s Impact on Biodiversity: Artificial Soils for Post-Mining Land Reclamation. *Sustainability*, 15(12), 9707.

Wischmeier, W. H., & Smith, D. D. (1978). Predicting rainfall erosion losses: a guide to conservation planning (Issue 537). *Department of Agriculture, Science and Education Administration*.

Zemánek, P., & Burg, P. (2009). Možnosti využití kompostů při optimalizaci hydrofyzikálních vlastností zemědělských půd. *Agritech Science*, 9, 1–4.

Zemanová, V., Menšík, L., Hangen, E., Schilling, B., Hlisnikovský, L., & Kunzová, E. (2025). Assessment of Potentially Toxic Element Pollution in Surface Soils of the Upper Ohře River Basin. *Toxics*, 13(8), 644.