AGU ADVANCING EARTH AND SPACE SCIENCES

Reviews of Geophysics®

REVIEW ARTICLE
10.1029/2025RG000883

# Rethinking Global Soil Degradation: Drivers, Impacts, and Solutions

**Nima Shokri[1,2], David A. Robinson[3], Mehdi Afshar[1,2], Christine Alewell[4], Milad Aminzadeh[1,2], Emmanuel Arthur[5], Nils Broothaerts[6], Grant A. Campbell[7], Lina Eklund[8,9,10], Surya Gupta[4], Richard Harper[11], Amirhossein Hassani[12], Cathy Hohenegger[13], Thomas Keller[14,15], Maximilian Kiener[16], Inma Lebron[3], Kaveh Madani[10], Tshilidzi Marwala[17], Francis Matthews[18,19], Per Moldrup[20], Attila Nemes[21,22], Panos Panagos[6], Remus Prăvălie[23,24,25], Matthias C. Rillig[26,27], Philipp Saggau[19], Salome M. S. Shokri-Kuehni[1,10], Pete Smith[7], Amy Thomas[3], Lis Wollesen de Jonge[5], and Dani Or[28]**

[1]Institute of Geo-Hydroinformatics, Hamburg University of Technology, Hamburg, Germany, [2]United Nations University Hub on Engineering to Face Climate Change at the Hamburg University of Technology, United Nations University Institute for Water, Environment and Health (UNU-INWEH), Hamburg, Germany, [3]UK Centre for Ecology and Hydrology, Bangor, UK, [4]Department of Environmental Sciences, University of Basel, Basel, Switzerland, [5]Department of Agroecology, Aarhus University, Tjele, Denmark, [6]European Commission, Joint Research Centre (JRC), Ispra, Italy, [7]Institute of Biological and Environmental Sciences, University of Aberdeen, Aberdeen, UK, [8]Centre for Advanced Middle Eastern Studies, Lund University, Lund, Sweden, [9]Department of Physical Geography and Ecosystem Science, Lund University, Lund, Sweden, [10]United Nations University Institute for Water, Environment and Health (UNU-INWEH), Richmond Hill, ON, Canada, [11]Centre for Crop and Food Innovation, Murdoch University, Perth, WA, Australia, [12]The Climate and Environmental Research Institute NILU, Kjeller, Norway, [13]Max Planck Institute for Meteorology, Hamburg, Germany, [14]Department of Soil and Environment, Swedish University of Agricultural Sciences, Uppsala, Sweden, [15]Agroscope, Department of Agroecology and Environment, Zürich, Switzerland, [16]Institute for Ethics in Technology, Hamburg University of Technology, Hamburg, Germany, [17]United Nations University, Tokyo, Japan, [18]Department of Earth and Environmental Sciences, KU Leuven, Belgium, [19]Department of Science, Roma Tre University, Rome, Italy, [20]Department of the Built Environment, Aalborg University, Aalborg, Denmark, [21]Norwegian Institute of Bioeconomy Research, Division of Environment and Natural Resources, Ås, Norway, [22]Norwegian University of Life Sciences, Faculty of Environmental Sciences and Natural Resource Management, Ås, Norway, [23]University of Bucharest, Faculty of Geography, Bucharest, Romania, [24]University of Bucharest, Research, Institute of the University of Bucharest (ICUB), Bucharest, Romania, [25]Academy of Romanian Scientists, Bucharest, Romania, [26]Institute of Biology, Freie Universität Berlin, Berlin, Germany, [27]Berlin-Brandenburg Institute of Advanced Biodiversity Research, Berlin, Germany, [28]Department of Civil and Environmental Engineering, University of Nevada, Reno, NV, USA

**Key Points:**

- Key soil degradation processes, for example, compaction, erosion, salinization, acidification, alkalization, or nutrient loss, are examined
- Major drivers such as deforestation and unsustainable farming, along with impacts on soil health, dust storms, and public health, are discussed
- Existing assessment methods and their limitations are reviewed, with a focus on emerging approaches for quantifying soil degradation

**Correspondence to:**

N. Shokri,
nima.shokri@tuhh.de

**Citation:**

Shokri, N., Robinson, D. A., Afshar, M., Alewell, C., Aminzadeh, M., Arthur, E., et al. (2025). Rethinking global soil degradation: Drivers, impacts, and solutions. *Reviews of Geophysics*, *63*, e2025RG000883. https://doi.org/10.1029/2025RG000883

Received 9 APR 2025
Accepted 28 OCT 2025



## Abstract

The increasing threat of soil degradation presents significant challenges to soil health, especially within agroecosystems that are vital for food security, climate regulation, and economic stability. This growing concern arises from intricate interactions between land use practices and climatic conditions, which, if not addressed, could jeopardize sustainable development and environmental resilience. This review offers a comprehensive examination of soil degradation, including its definitions, global prevalence, underlying mechanisms, and methods of measurement. It underscores the connections between soil degradation and land use, with a focus on socio-economic consequences. Current assessment methods frequently depend on insufficient data, concentrate on singular factors, and utilize arbitrary thresholds, potentially resulting in misclassification and misguided decisions. We analyze these shortcomings and investigate emerging methodologies that provide scalable and objective evaluations, offering a more accurate representation of soil vulnerability. Additionally, the review assesses both physical and biological indicators, as well as the potential of technologies such as remote sensing, artificial intelligence, and big data analytics for enhanced monitoring and forecasting. Key factors driving soil degradation, including unsustainable agricultural practices, deforestation, industrial activities, and extreme climate events, are thoroughly examined. The review emphasizes the importance of healthy soils in achieving the United Nations Sustainable Development Goals, particularly concerning food and water security, ecosystem health, poverty alleviation, and climate action. It suggests future research directions that prioritize standardized metrics, interdisciplinary collaboration, and predictive modeling to facilitate more integrated and effective management of soil degradation in the context of global environmental changes.

**Plain Language Summary** Soil degradation is a growing problem that threatens soil and food security, ecosystem functioning, and socio-economic activities. This review looks at how soil degradation is defined and measured, the causes behind it, and how it affects people and the environment. It also explores new tools like satellite data, artificial intelligence, and big data analytics that could help us better detect and predict soil degradation. The study highlights how protecting soil health is essential to achieving UN Sustainable Development Goals (UN SDGs). It calls for better ways to measure soil health and smarter strategies to manage land in a sustainable way.

## 1. Introduction

### 1.1. Rationale and Objectives

This review focuses on soil degradation distinguished from the broader term land degradation as explained in the next section. Soil degradation is a component and a driver of land degradation that has emerged as a global threat to the health and productivity of ecosystems, while undermining soil security, ecosystem services, environmental stability, and socio-economic activities (Amundson et al., 2015; FAO and ITPS, 2015; Prăvălie et al., 2024). The consequences of degraded lands include a progressive decline in productivity and a reduced capacity to sustain crops and livestock (Blaikie & Brookfield, 1987; FAO, 2019). In some cases, biodiversity may decline through habitat loss, thereby reducing ecosystem resilience. Furthermore, degraded lands are often more susceptible to natural disasters such as landslides, floods, droughts, and climate extremes.

Human-induced drivers and pressures present various threats to soil health, including loss of fertility, increased erosion, pollution, compaction, salinization, acidification, and depletion of organic matter, among others (FAO and ITPS, 2015; Hassani et al., 2024; Prăvălie et al., 2024; Figure 1). These threats heighten the susceptibility of soils to degradation processes. We focus on anthropogenic actions that contribute to soil degradation, especially doing "*the wrong action, at the wrong time, in the wrong location.*" Consequently, part of the solution involves educating stakeholders to undertake initiatives in appropriate locations and at suitable times. Achieving this necessitates a comprehensive understanding of both the processes of degradation and the inherent vulnerabilities of soils to various drivers and pressures, which can be set within a global context of the intersection between soils, biomes, and the centers of increasing human activity and population.

A pernicious aspect of soil degradation is that it often is transparent as it builds up, ultimately reaching a threshold or even tipping point beyond which severe and often devastating irreversible consequences arise (Bestelmeyer et al., 2003, 2015). Historical events, such as the salinization of Mesopotamian soil around 2000 BCE, played a role in the decline of civilizations and cities like Sumer and Akkad (Shahid et al., 2018). The Dust Bowl event in the USA in the 1930s stripped millions of acres of fertile topsoil from the land, leading to extensive human migration and widespread poverty (Baveye et al., 2011). Similarly, drought and desertification in the Sahel region in the 1960's–1980's, exacerbated by overgrazing, deforestation, and poor land management, transferred once productive land into unusable areas, leading to widespread famine (Lal, 1993). Additionally, centuries of deforestation and overgrazing on vulnerable silty soils on the Loess Plateau in China led to severe erosion of once fertile landscapes (Yu et al., 2020). Intensive agriculture on steep slopes, deforestation, and overpopulation exacerbated by political conflict also led to severe soil erosion in Rwanda in the 1990s (Karamage et al., 2016).

Once the topsoil has gone, and the land shifts into an altered, degraded state, recovery can be long-term. However, there is hope to restore, or better prevent, such occurrences with concerted action. The Chinese Government, for instance, launched a massive restoration project on the Loess plateau, successfully rehabilitating millions of hectares of degraded land (Yu et al., 2020). Across Europe, acid rain that led to soil acidification and forest dieback was addressed by the convention on long-range transboundary air pollution, which drastically reduced emissions (Grennfelt et al., 2020). In the UK, the recovery of soils from acidification has been reported, for example, from long-term monitoring (Reynolds et al., 2013) but this rebound may be halting (Seaton et al., 2023). In Asia, and especially in China, acid deposition continued to be high until the early 2000s, starting to decline in 2005 but with critical loads still being exceeded in the 2020s mainly due to high nitrogen loads (Xie, Duan, et al., 2024; Xie, Ge, et al., 2024). These examples highlight the need for consistent and long-term monitoring to observe progress and identify effective interventions and success.

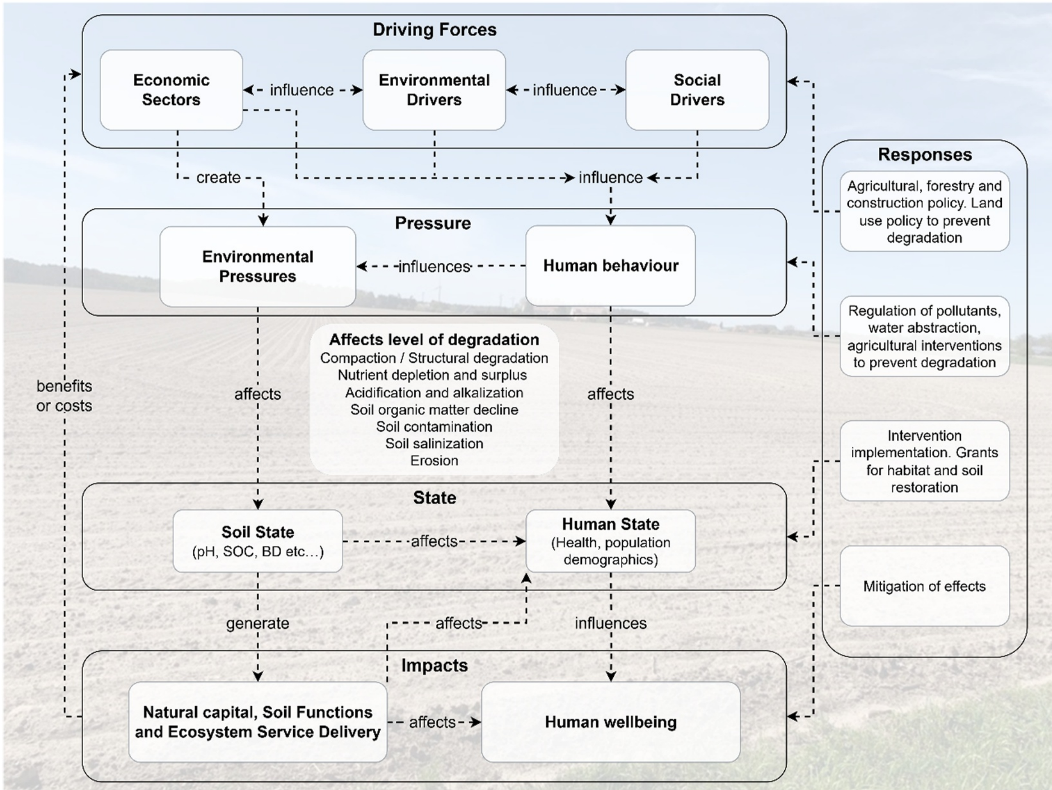


**Figure 1.** Drivers and pressures contributing to soil degradation, indicating the complex socio-economic and environmental interactions that contribute. Based on the Drivers, Pressures, State, Impact, Responses (DPSIR) framework.

Another major scientific challenge is understanding and predicting the consequences of soil and land degradation. Degradation can occur relatively rapidly, yet it may leave the land in a state that requires a long time to recover, representing a transient state, or even never recovers representing an alternative permanently degraded stable state (Bestelmeyer et al., 2015; Briske et al., 2005). The desirable option is to prevent degradation in the first place, which requires empowering policymakers to understand the risks and implement preventive measures to pre-empt problems. This requires identifying vulnerable soils and predicting how land use will respond to human activity and changing climate. Currently, our ability to model and predict the consequences of soil degradation, especially what happens due to multiple drivers and at thresholds and whether recovery will occur once pressures are removed (represented by a transient state), is limited. Tipping points by their abrupt changes in soil state are particularly difficult to anticipate, such as the consequences of overgrazing and rapid loss of fertile soil leading to desertification of the southern Sahel in the 1960s (Sinclair & Fryxell, 1985). The time it takes for soil to degrade to a certain state, and recovery asymmetry in time, or conversely if a tipping point is breached and an alternative stable degraded state are particularly challenging to predict. New opportunities presented by Industry 4.0 technologies (the so-called *fourth industrial revolution*) such as big data analytics, unprecedented computational capacities and the rapid development of Artificial Intelligence (AI) and machine learning models capable of dealing with a multitude of drivers and nonlinear processes offer new ways of addressing these gaps (Shokri et al., 2025; Tahmasebi et al., 2020). Moreover, these modern tools may guide and expand our understanding of soils' response to multiple soil degradation drivers and pressures. Current models and experiments often address individual drivers, while new experimental approaches reveal unexpected emergent responses of soils to conditions where multiple drivers act jointly (Rillig et al., 2019).

Previous reviews on soil degradation have addressed specific processes (e.g., erosion (Borrelli et al., 2021), salinization (Shokri et al., 2024), contamination (Tetteh, 2015), and integration of drivers (Kopittke et al., 2025)), and aspects (e.g., conservation tillage (Hussain et al., 2021), remote sensing (Wang, Du, et al., 2023; Wang, Min, et al., 2023; J. Wang, et al., 2023), food security (Bindraban et al., 2012), mitigation (Qi et al., 2024)), or focused on particular regions and soil types (e.g., Western Europe (Virto et al., 2014), European Mediterranean (Ferreira et al., 2022), North America (Baumhardt et al., 2015), India (Bhattacharyya et al., 2015), Africa (Diop et al., 2022), black soils (Rui et al., 2025)). However, they often suffer from fragmentation, limited spatial coverage, or narrow thematic scope. Comprehensive, globally consistent assessments are rare. Reviews also tend to treat degradation drivers in isolation, neglecting the interactions between multiple anthropogenic pressures and ecological feedbacks. Furthermore, there is limited integration of recent advances in data science, AI, and remote sensing, which can now support large-scale, real-time soil monitoring and modeling. This review addresses these gaps and provides a multi-scale assessment of soil degradation processes, synthesizing recent scientific and technological advances for improved understanding and prediction. It also draws attention to the critical need for harmonized global indicators and decision-support tools that bridge science, policy, and practice for sustainable soil and land management.

In this review, we pursue a multi-scale assessment of soil degradation processes in relation to global change drivers and pressures driven largely by anthropogenic activities. With the specific objectives of:

- Defining, describing, and delineating soil degradation processes as a concept in the context of being a driver of land degradation.
- Predicting soil degradation in response to anthropogenic drivers and pressures; assessing stability and states and how to optimally capture this in models.
- Going beyond single drivers of degradation to understand emergence in response to multiple environmental change drivers and pressures.
- Highlighting the opportunities presented by the Industry 4.0 technologies (e.g., advancements in big data analytics, remote sensing, and AI) to improve soil degradation monitoring and prediction.
- Assessing the latest research to address the social, economic and environmental drivers, pressures and impacts of soil degradation.
- Identifying solutions for restoration and prevention at multiple scales and in trans-disciplinary contexts.

By understanding degradation processes, their extent and severity, and bridging the gap between science and policy to communicate soil vulnerability, scientific research can help co-develop strategies that proactively address future social, economic, and environmental challenges.

### 1.2. Definitions of Soil Degradation

In this review, we distinguish the term "*soil degradation*" from "*land degradation.*" Land degradation includes changes to functioning of ecosystems supported by all land components such as forests, streams, grasslands, and interconnectedness for wildlife and other organisms. Moreover, land degradation is often cast primarily as a social problem (Blaikie & Brookfield, 1987). They define land degradation as "a reduction in the capability of land to satisfy a particular use." Capability implies context for a specific purpose, for example, urban, agricultural, or forestry land.

Soil degradation is a component, and often driver, of land degradation, reducing its capability through the diminished capacity of soil to provide climate and ecosystem services, often but not exclusively, due to lower productivity, loss of topsoil, compaction, salinization, and loss of biodiversity and biomass (Prăvălie et al., 2024). However, in the case of peat soils, degradation may occur through increased productivity, especially nitrogen deposition that may kill sphagnum moss, for example. Therefore, soil degradation refers to internal or external influences that result in processes which diminish the soil systems' structure, functionality, performance, efficiency, or resilience over time. Soil degradation may be physical, chemical, or biological. Degradation typically involves energy dissipation, loss of structural integrity or an increase in entropy. It is associated with increased vulnerability and a reduction in functional capacity for the system, often irreversibly making degradation processes hard to reverse. A key aspect of degradation is that a system requires either maintenance, often linked with sustainability, to counteract degradation or regeneration and renewal that usually requires external input, feedback, or intervention to restore the system's capacity to function.

According to UNEP and ITPS (2015), defined as: "Soil degradation is the decline in soil quality caused by its improper use by humans, usually for agricultural, pastoral, industrial, or urban purposes." Soil degradation includes physical, biological, and chemical deterioration by recognized soil threats, such as a decline in soil fertility, perturbed structural condition, erosion, adverse chemical changes in salinity, acidity or alkalinity, loss of biological function, and the effects of toxic chemicals, pollutants or excessive inundation. It is a process implying time and rate dependencies that may result in a degraded state.

Importantly, the baseline is set by the soil condition prior to human use or impact, or by similar soils that are unused. Soils that show deterioration features that are not a result of human activity are simply considered subject to natural environmental change. Synthesizing this, one can form an equation, similar to the one in (Blaikie & Brookfield, 1987), such that:

$$\text{Net degradation} = (\text{environmental change processes} + \text{human interference}) - (\text{natural soil formation} + \text{restorative management})$$

It is the human interference aspect that leads to an acceleration of processes well beyond their native state and which forms the focus of this review. However, the restorative management indicates the importance of people in solving the issue.

We propose that for this work soil degradation is defined as "a dynamic deficit in soil condition undermining its capacity to function under given management and climate conditions that may result in a degraded state." This definition offers a more holistic systems-based lens through which to view degradation, appropriate to the emphasis of this review.

### 1.3. Global Distribution of Degraded Soils and Hotspots

Only a few systematic evaluations of soil degradation at national and regional scales are presently available (FAO, 2015a, 2015b; Ferreira et al., 2022). Existing global assessments, such as the GLASOD map produced during the GLASOD project (1987–1990) for the United Nations Environment Program (UNEP), are now outdated (Bridges & Oldeman, 1999; Oldeman, 1992). Consequently, significant gaps remain in identifying locations of severe soil degradation and their societal and economic impacts on governments and land users (FAO and ITPS, 2015; Ferreira et al., 2022). Furthermore, data on trends in soil degradation and related risks are also limited. To address this deficiency in direct information, researchers have used proxies such as vegetation health and changes in land use/land cover to monitor degradation over time (Prăvălie, 2021).

The definition of soil degradation itself presents challenges, leading to considerable variations in estimates. The term is frequently used in a broad sense to encompass various land degradation processes, including salinization, erosion, and compaction, as well as issues like soil acidification, contamination from pesticides and heavy metals, nutrient imbalances, and waterlogging (Prăvălie et al., 2024). In contrast, some studies narrow their focus to specific degradation processes, such as soil water erosion (Panagos et al., 2015), depletion of soil organic carbon (SOC; Padarian et al., 2022; Stockmann et al., 2015), or soil salinity (Hassani, Azapagic & Shokri, et al., 2020). This inconsistency complicates comparisons across studies, especially when certain research is limited to specific soil types, such as drylands (Dregne, 2002; Hassani et al., 2021; Nachtergaele & Licona-Manzur, 2008) or soil depths (e.g., topsoil versus deeper layers), while others consider all environments.

Lambin and Meyfroidt (2011) estimated that land degradation renders between 1 and 3 million hectares non-arable each year. The UN Convention to Combat Desertification (UNCCD) found that over 100 Mha of land had been lost to land degradation each year between 2015 and 2019 (UNCCD, 2025), suggesting an increase in the rates of degradation. Olsson et al. (2023) point out that the lack of a clear definition of land degradation means that global estimates may not be reliable. Global estimates of total degraded land exhibit considerable variation, with figures ranging from under 1 Gha to more than 6 Gha, as reported in a review of global land degradation mapping by Gibbs and Salmon (2015). To evaluate degraded lands on a global scale, four main methodologies have been employed: expert assessments, satellite imagery analysis, biophysical modeling, and surveys of abandoned agricultural areas (Gibbs & Salmon, 2015). Similar to the situation with soil degradation, Gibbs and Salmon (2015) emphasize the absence of agreement on the extent of degraded land, both at the global level and within specific nations. There are few, if any, systematic national evaluations that monitor baseline conditions or track changes over time, and a universally accepted framework for conducting these assessments is lacking. Additionally, Gibbs and Salmon (2015) note that current estimates of degraded lands are frequently compromised by incomplete or unreliable data. Research is essential to improve these estimates and mitigate the risk of overestimating the areas of degraded land.

10.1029/2025RG000883

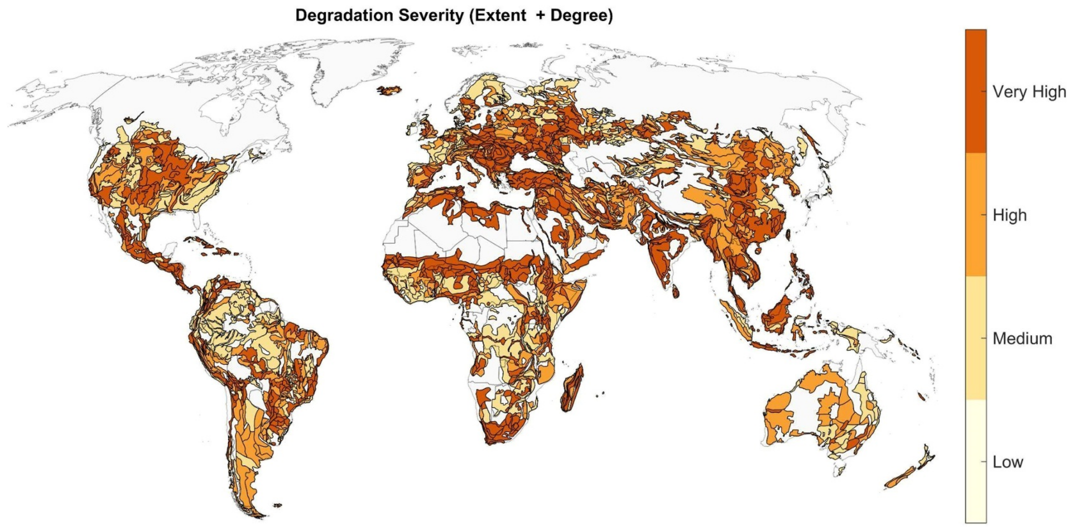


**Figure 2.** The Global Assessment of Human-induced Soil Degradation (GLASOD) map, as presented by Oldeman et al. (1990), illustrating the worldwide severity of soil degradation caused by human activities. Due to the polygonal format of the map, identifying specific locations of soil degradation is not feasible. However, it is possible to estimate the relative prevalence of various types of soil degradation within the polygons. The "severity of soil degradation" is evaluated by considering both the intensity and the extent of the affected regions.

Bateman and Muñoz-Rojas (2019) highlight that based on limited global and national data, approximately 20% of the world's soils are currently experiencing degradation, with an annual loss of 5–10 million hectares. The FAO (2015b, 2015a) indicated that soil degradation results in the loss of 12 million hectares of agricultural land each year, and currently, about 33% of soils worldwide are classified as moderately to highly degraded. 40% of these degraded areas are situated in Africa, particularly in regions grappling with poverty and food insecurity.

The GLASOD map estimates that human activities have led to the degradation of approximately 15% (1,964 million hectares) of the Earth's land surface (Bridges & Oldeman, 1999; Oldeman, 1992). However, the map's broad scale limits the ability to accurately assess degradation levels in individual countries. Soil degradation was mapped using expert judgment within broadly defined physiographic units (polygons), illustrating the type, extent, severity, rate, and primary causes of degradation on a global scale of 1:10 million (accessible at: https://files.isric.org/public/other/GLASOD.zip, accessed 14 February 2025). According to GLASOD, around 38% of agricultural land globally is impacted by human-induced soil degradation, although the degree of degradation varies significantly across different regions (Figure 2). The "severity of soil degradation" is evaluated by assessing both the intensity and the extent of the affected regions. The map indicates that light degradation affects approximately 38% of all degraded soils, which corresponds to around 749 million hectares. This level of degradation results in a slight decline in productivity, yet it remains manageable within local agricultural practices. In contrast, a more significant portion, approximately 46% (or 910 million hectares), suffers from moderate degradation, which leads to a considerable reduction in productivity. Restoring these moderately degraded soils typically necessitates considerable financial investment, often exceeding the resources available to local farmers, especially in developing nations. Among the moderately degraded lands, over 340 million hectares are located in Asia, while more than 190 million hectares are found in Africa. Severely degraded soils are approximately 296 million hectares globally, with 124 million hectares in Africa and 108 million hectares in Asia. These soils are generally deemed unfit for reclamation at the farm level and are considered nearly lost unless significant engineering interventions or international assistance are provided. Furthermore, extremely degraded soils, classified as irrecoverable, cover roughly 9 million hectares worldwide, with over 5 million hectares in Africa. Water erosion is the most prevalent degradation process, accounting for 56%, followed by wind erosion at 28%.

In India, around 147 million hectares of land are experiencing degradation (Bhattacharyya et al., 2015). This includes 94 million hectares impacted by water erosion, 16 million hectares by acidification, 14 million hectares by flooding, 9 million hectares by wind erosion, 6 million hectares by salinity, and 7 million hectares affected by a combination of these issues. In the European Union (EU), the Soil Health and Food Mission Board, in collaboration with the Joint Research Center (JRC; Veerman et al., 2020), indicates that 60%–70% of soils in the EU are deemed unhealthy due to unsustainable land management practices. Although not all of these soils are classified as degraded, their capacity to perform essential ecological functions has markedly diminished. The Mediterranean region in Europe is particularly vulnerable to soil degradation, displaying the highest rates of soil erosion within the EU, lower levels of organic matter, and significant challenges related to soil salinization (Ferreira et al., 2022; Lahmar & Ruellan, 2007).

The discrepancies in estimates across various studies highlight the challenges in accurately measuring soil degradation and the potential for over- or under-estimating the issue. Furthermore, the worldwide prevalence of degraded soils is frequently estimated within the context of land degradation assessments, which may encompass a range of broader degrading phenomena. Certain processes and factors that lead to soil degradation, including salinization, erosion, and compaction, are often interconnected with those associated with land degradation. Conversely, other elements, such as land subsidence, the decline of natural vegetation, or the invasion of non-native species, do not inherently lead to soil degradation as they may not have a direct effect on soil productivity or its physical and chemical characteristics. Nevertheless, 17 critical pathways of land degradation identified by Prăvălie (2021) including aridity, biological invasions, coastal erosion, water-induced land erosion, wind-induced land erosion, land pollution, land subsidence, landslides, permafrost thawing, salinization, soil acidification, loss of soil biodiversity, soil compaction, depletion of soil organic carbon, soil sealing, degradation of vegetation, and waterlogging, can also be regarded as significant factors contributing to soil degradation.

## 2. Consequences of Soil Degradation

### 2.1. Impact on Agricultural Productivity, Food and Soil Security

As mentioned earlier, soil degradation has significant repercussions for agricultural productivity, primarily through mechanisms such as erosion, salinization, compaction, and nutrient depletion (Bindraban et al., 2012). Erosion, in particular, diminishes agricultural output in both the short-term, by causing losses in crop yields, seedling availability, and water resources, and the long-term, by leading to topsoil loss, deterioration of soil structure, and a reduction in soil organic matter (SOM; Lal, 2001). SOM is often regarded as a key indicator of soil health, which plays a crucial role in facilitating crop growth. Nevertheless, the precise relationship between SOM and crop growth remains inadequately understood (Olsson et al., 2023). While SOM has been generally associated with crop growth on a global scale (Oldfield et al., 2019), a more pronounced correlation has been observed at regional levels, particularly during dry conditions (Kane et al., 2021; Pan et al., 2009). Additionally, soil salinization and waterlogging, both potential consequences of irrigation, have a direct impact on crop yields, especially when they occur simultaneously (Singh, 2015). Prior studies have identified a linear relationship between soil salinity and reductions in certain crop yields (Maas & Grattan, 1999). Furthermore, soil compaction disrupts the structural integrity of the soil, impeding root growth and adversely affecting soil biodiversity (Schjønning et al., 2015; Shah et al., 2017). This disruption has been shown to significantly hinder crop growth, impacting plant establishment (Tolon-Becerra et al., 2011), height (Abu-Hamdeh, 2003), and overall morphology (Grzesiak, 2009).

The interplay between soil degradation and agricultural productivity is reciprocal. Studies indicate that the pursuit of higher yields, driven by changes in land use and intensive farming practices, is contributing to soil degradation on a global scale, which subsequently jeopardizes future food security and the livelihoods of communities (DeLong et al., 2015; Lal, 2016). Food security is characterized as a condition in which “all people, at all times, have physical, social, and economic access to sufficient safe and nutritious food that meets their dietary needs and food preferences for an active and healthy life” (FAO, 2006). This concept is structured around four fundamental pillars: availability, access, utilization, and stability. The aspect of food availability, which refers to the physical presence of adequate quantities of high-quality food, is directly compromised by soil degradation, as it leads to diminished agricultural output (Bagnall et al., 2021; Lal, 2016). Furthermore, soil degradation adversely affects the quality of food produced, potentially resulting in malnutrition (Pozza & Field, 2020). Contamination of soils, particularly by heavy metals and pesticides, poses direct health risks to humans via the food chain (Alengebawy et al., 2021). Degraded soils frequently lose their ability to buffer against pollutants, rendering them more vulnerable to contamination from industrial and agricultural sources. Crops cultivated in such compromised soils may absorb harmful substances, including cadmium, mercury, and lead, which can lead to bioaccumulation and biomagnification, thereby threatening food security. For example, a study analyzing 484 rice samples from five contaminated regions in China revealed that over 18% of these samples contained cadmium levels surpassing the maximum allowable limit, with the highest incidence of 41.1% observed in samples from Hezhang, Guizhou. Prolonged consumption of such contaminated food can result in serious health issues, including kidney damage, neurological disorders, and developmental delays in children (Lamas et al., 2016; F. Wang et al., 2020; Z. Wang et al., 2020). Therefore, maintaining productive and healthy soils is essential for safeguarding food security (Bindraban et al., 2012; Jones et al., 2013).

The notion of soil security underscores the critical need to preserve and enhance global soil resources, paralleling the significance attributed to water and energy security (Koch et al., 2013; McBratney et al., 2014). This framework encompasses five distinct dimensions: Capability, which refers to the ideal state of the soil; Condition, denoting the present state of the soil; Connectivity, which examines the relationship between humans and soil; Capital, representing the financial, social, or natural worth of the soil; and codification, involving the education, policies, and regulations pertaining to soil (McBratney et al., 2014; Pozza & Field, 2020). Consequently, this concept expands the discourse on soil degradation beyond a purely biophysical viewpoint, inviting a more interdisciplinary exploration of related issues (Ball et al., 2018; Tripathi et al., 2022).

### 2.2. Impacts on Socio-Economics Stability, Human Migration and Rural Livelihoods

In the year 2000, approximately 1.33 billion individuals globally resided in regions characterized by deteriorating agricultural land, a figure that rose by 12.4% by 2010 (Barbier & Hochard, 2016). Additionally, a greater proportion of the population living in such areas diminishes the effectiveness of per capita income growth in alleviating poverty, thereby underscoring the relationship between land degradation and poverty. Soil degradation has a worldwide impact on livelihoods, yet communities engaged in subsistence agriculture, particularly in Sub-Saharan Africa, exhibit heightened vulnerability to this issue (Gashu & Muchie, 2018; Mganga et al., 2015; Reed et al., 2015). The consequences of soil and land degradation can also vary by gender; for instance, in Ghana, land degradation resulting from agricultural mechanization compelled women to adopt less sustainable livelihood strategies, such as charcoal production and fuelwood collection (Kansanga et al., 2020). However, the influence of wealth on land management practices is context-dependent. In Central Asia, lower-income households were observed to implement more sustainable land management techniques compared to their higher-income counterparts (Mirzabaev et al., 2023).

The relationships between environmental change, such as land and soil degradation, and migration are intricate and vary depending on the context (McLeman & Gemenne, 2018). Migration decisions typically arise from a blend of social, demographic, economic, environmental, and political influences, rather than being driven solely by environmental factors (Black et al., 2011; McLeman, 2014). The specific outcomes of migration are contingent upon the characteristics of the environmental change, distinguishing between gradual events (e.g., droughts or soil degradation) and sudden occurrences (e.g., floods; Kaczan & Orgill-Meyer, 2020). Additionally, the perception of environmental changes and the socioeconomic vulnerability of the impacted communities (Afifi et al., 2016; Koubi et al., 2016) significantly influence migration patterns. Consequently, linking migration directly to particular factors, such as soil degradation, presents considerable challenges. Migration encompasses a broad spectrum of human mobility (McLeman & Gemenne, 2018). Research indicates that the majority of migration occurs within national borders, or internally, irrespective of the underlying drivers (Obokata et al., 2014). Environmental factors often lead to migration toward urban centers, typically over shorter distances (Thiede et al., 2016). This trend can be attributed to the fact that migration, particularly international migration, necessitates both economic and social resources (Findlay, 2011). As a result, vulnerable groups who lack the means to migrate as a strategy for adapting to environmental changes may find themselves in a state of entrapment (Zickgraf, 2018).

In studies concerning migration driven by environmental factors, soil degradation is frequently included within the wider framework of land degradation (see McLeman, 2017). This relationship connects land degradation to rural livelihoods, which subsequently influences migration patterns (Hermans & McLeman, 2021). Although there are global estimates regarding the population affected by migration or displacement due to land degradation, McLeman (2017) emphasizes that these figures are rough approximations and should not be regarded as scientifically valid statistics.

Migration, which can involve the relocation of entire households or just individual members, serves as one of several strategies for adaptation in response to soil degradation (Khan et al., 2024; Sanfo et al., 2017).

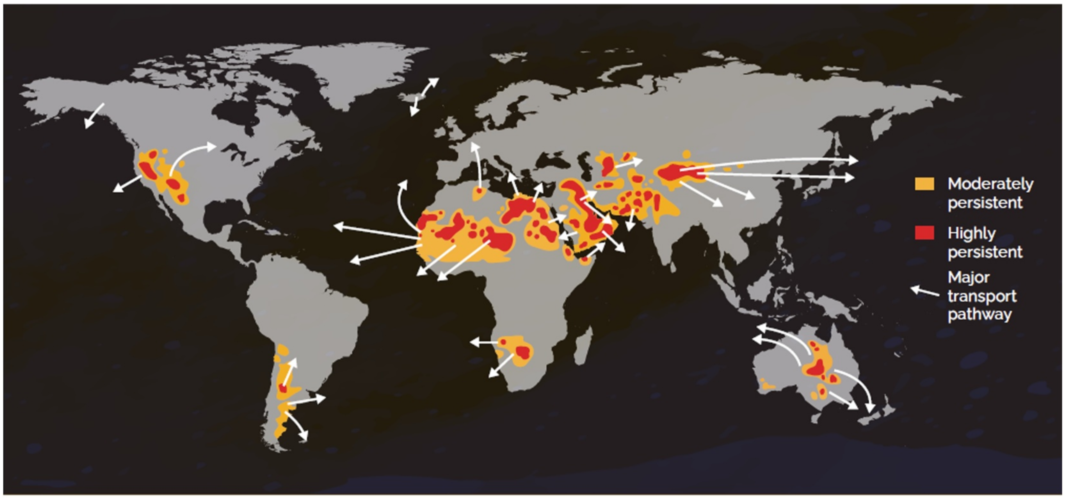


**Figure 3.** Worldwide origins of desert dust and primary routes of long-range, often transboundary transport (based on FAO (2023), with data sourced from Muhs et al. (2014)).

Environmental factors frequently interact with elements that encourage migration, such as the availability of alternative income sources in different regions (Neumann et al., 2015). The relationship between land degradation and migration is primarily mediated by the effects of altered agricultural productivity on livelihoods (Hermans et al., 2023). For instance, in coastal Bangladesh, rising soil salinity adversely affected agricultural yields, prompting households to seek income diversification and engage in the internal migration of one or more members (Chen & Mueller, 2018). While increased salinity was associated with a rise in internal migration, it correspondingly led to a decline in international migration. In Pakistan, soil erosion exacerbated labor migration as food production diminished; however, the most vulnerable households were unable to leverage migration as a means of adaptation, resulting in a state of involuntary immobility (Khan et al., 2024). In contrast, in China, soil erosion did not significantly influence migration patterns, with socio-economic factors being more decisive (Zhang & Zhuang, 2019). Likewise, in Uganda, land degradation did not appear to significantly affect migration choices; instead, climate anomalies emerged as the primary migration drivers (Call & Gray, 2020). Nonetheless, favorable soil fertility conditions facilitated non-labor-related migration.

Soil health and land dynamics are intricately connected to pastoral mobility in arid regions, where such mobility serves to alleviate the strain on soil and vegetation resources. A decline in this mobility, as noted by Liao et al. (2020), can lead to detrimental effects on both land and soil quality. The interplay between soil or land degradation and migration is multifaceted, context-specific, and operates in various directions, in line with the broader body of research on environmentally induced migration (R. Hoffmann et al., 2020; C. Hoffmann et al., 2020).

### 2.3. Impacts on Desertification, Dust Storms and Public Health

Soil degradation and desertification are intricately linked, with the former often acting as a precursor or driver of the latter. Desertification is defined by the United Nations Convention to Combat Desertification (UNCCD) as the degradation of land in arid, semi-arid, and dry sub-humid regions, primarily resulting from climatic variations and human activities (Becerril-Piña & Mastachi-Loza, 2021). Soil degradation is at the core of this process, as the loss of soil fertility, structure, and ecosystem services fundamentally undermines land productivity. Desertification exacerbates the loss of arable land, reduces vegetation cover, and increases soil vulnerability to wind erosion, which collectively increases the frequency and intensity of dust storms (Zucca et al., 2022).

Dust storms can travel thousands of kilometers (Cao et al., 2015), impacting vast areas (Figure 3). Desert dust storm definitions vary (Wang, 2015); however, here we use the WMO definition: a collection of small particles lifted by strong, turbulent winds that reduce visibility to less than 1000 m, typically measured at 1.8 m above the ground (UNEP, WMO, UNCCD, 2016). Dust storms are characterized by high concentrations of particulate matter (PM10 and PM2.5), which can far exceed recent WHO-recommended limits of 45 μg $m^{-3}$ for PM10 and 15 μg $m^{-3}$ for PM2.5 over a 24-hr period (World Health Organization, 2021).

Salt-born dust storms, originating from saline soils, are significant environmental challenges in regions prone to soil degradation (Hassani, Azapagic, D'Odorico, et al., 2020). These storms, characterized by the transport of particulate matter high in salt content, can severely affect the surrounding areas, both in terms of soil health and human and ecological well-being. Although the frequency and scale of such events may be localized, their impact is considerable, leading to a range of negative consequences on nearby regions (Abuduwaili et al., 2010). In addition to impacts on human health and biogeochemical cycles, dust emissions also influence climate and the hydrological cycle. Dust particles can affect cloud microphysics and precipitation processes, in some cases leading to rainfall suppression, and thereby contribute to feedbacks that exacerbate aridity and desertification (Rosenfeld et al., 2001).

The negative consequences of dust storms are well-documented, but there are also some positive aspects worth noting. One of the examples is the transcontinental transport of Saharan dust, which fertilizes ecosystems thousands of kilometers away (Bhattachan et al., 2012; Duce et al., 1991; Mahowald et al., 2005; Okin et al., 2004, 2011; Swap et al., 1992). The Amazon basin receives an estimated 28 (8–48) Tg of Saharan dust annually, supplying critical nutrients such as phosphorus that sustain its productivity (Yu et al., 2015). The deposited dust may supply approximately 0.022 (ranging from 0.006 to 0.037) Tg of phosphorus annually. This phenomenon shows the complex correlations between soil degradation and global biogeochemical cycles. However, the balance between such benefits and the overwhelming negative impacts on local communities must be critically examined.

Chronic exposure to the elevated PM levels as a result of dust storms has been linked to respiratory diseases, including asthma (Zheng et al., 2024), chronic obstructive pulmonary disease (COPD; Wang, Du, et al., 2023; Wang, Min, et al., 2023; J. Wang, et al., 2023), and lung cancer (Lee et al., 2022). The review of 52 experimental studies on mechanisms underlying the health effects of desert sand dust (Fussell & Kelly, 2021) shows that both virgin sand dust and remote dust storm particles induce inflammatory lung injury and worsen allergen-induced eosinophilia via cytokines, chemokines, and immunoglobulin pathways, potentially through toll-like receptor signaling. In vitro studies indicate that suspended desert dust during storms can interact with surface chemicals, increasing PM2.5 bioreactivity and generating toxic organic compounds, which enhance aerosol toxicity in urban areas (Fussell & Kelly, 2021). Dust storms also transport toxic metals (C. Zhang et al., 2022; D. Zhang et al., 2022; X. Zhang et al., 2022), microplastics (Abbasi et al., 2022), organic pollutants, and microbial pathogens (Jasim et al., 2024) over long distances. These contaminants can settle on crops (Middleton, 2024), enter water systems, or be directly inhaled, resulting in chronic exposure.

Additionally, public health could be affected by soil degradation through its impact on water resources too, as leaching from degraded soils can introduce nitrates, phosphates, and heavy metals into groundwater and surface water systems (Hossain et al., 2022; Pérez-Lucas et al., 2019). In agricultural regions of India and sub-Saharan Africa, nitrate concentrations in drinking water often exceed the WHO guideline of 50 mg $L^{-1}$, contributing to methemoglobinemia (Fewtrell, 2004), also known as "blue baby syndrome" in infants. Additionally, pesticide residues in water sources are linked to endocrine disruption and increased cancer risks (Pérez-Lucas et al., 2019). The global burden of disease attributable to waterborne pollutants from soil degradation is difficult to quantify but is estimated to contribute to millions of cases of diarrhea, cancer, and other health conditions annually (World Health Organization, 2023).

### 2.4. Impacts on Hydrological Processes

Soil degradation has repercussions for both the immediate and surrounding areas of land. The earlier sections examined the alterations in physical and chemical processes associated with different degradation mechanisms. This section will concentrate on the influence of land degradation on hydrological indicators. Generally, soil degradation leads to a reduction in infiltration rates, primarily due to compaction and the deterioration of soil structure. This decline results in heightened runoff and accelerated soil erosion. Additionally, it diminishes the soil's capacity to retain water, which in turn affects groundwater recharge, contributes to waterlogging, and increases sediment yield in streams, ultimately compromising water quality.

Changes in land use contribute to various soil degradation processes, including compaction, erosion, and a decline in soil organic carbon levels. These factors adversely affect soil structure, leading to diminished rates of soil infiltration and saturated hydraulic conductivity (Togbévi et al., 2022). Consequently, this situation exacerbates surface runoff, hampers groundwater recharge, and increases the likelihood of waterlogging. Research indicates that infiltration rates are generally higher in forested areas compared to grasslands and croplands, attributed to the preservation of soil structure and greater root density (Price et al., 2010). For instance, Robinson et al. (2022) demonstrated that forests exhibited an infiltration rate that was double that of croplands, based on over 800 global measurements. The ongoing transition of land use from forest to grassland or agricultural practices markedly diminishes the soil's infiltration capacity (Sun et al., 2018). Additionally, soil erosion has a profound effect on both infiltration rates and saturated hydraulic conductivity. Mai et al. (2023) investigated the effects of erosion on the hydraulic properties of sloped farmland, revealing that erosion degradation led to a reduction in saturated hydraulic conductivity, water holding capacity, and water supply capacity in black soils, with the 0–10 cm layer exhibiting higher values than the 10–20 cm layer. Furthermore, Yimer et al. (2008) found that in cultivated and grazed areas, infiltration capacity and soil moisture content were reduced by 70% and 45%, respectively, in comparison to forested regions, while dry bulk density increased by 13%–20%. These reductions are primarily attributed to soil compaction resulting from tillage and animal trampling, along with a decrease in soil organic carbon content.

Diminished soil infiltration and reduced water retention capacity lead to heightened surface runoff, which results in the displacement of topsoil that is, abundant in organic matter, nutrients, and microbial life. This topsoil may either be relocated within the site or carried away to drainage systems or aquatic environments (Shi et al., 2012). Such processes give rise to critical environmental challenges, including flooding, sedimentation of water bodies, loss of soil biodiversity, and pollution (Al-Wadaey & Ziadat, 2014; Gao et al., 2012). A significant consequence of these off-site effects is the transport of nutrients, especially nitrogen and phosphorus, which plays a role in the phenomenon of eutrophication (Ayele & Atlabachew, 2021; Lin et al., 2021). The ongoing degradation of ecosystems further compromises water quality. Research by Abell et al. (2019) indicated that in lakes across New Zealand, total nitrogen levels had doubled, while total phosphorus concentrations had increased fourfold compared to baseline measurements, underscoring the severity of human-induced eutrophication, a pattern that is, evident worldwide (Hou et al., 2022).

### 2.5. Costs of Soil Degradation and Restoration

Soil degradation imposes significant economic strain at a global level. Currently, the decline in soil health and productivity has resulted in diminished agricultural yields, which has led to increased food prices and contributed to high economic instability, notably in areas which are heavily reliant on agriculture for food, fiber, and fuel (Nkonya et al., 2016). The worldwide financial implications of soil degradation are rapidly increasing, including direct losses in income from decreased crop outputs and the rising costs of inputs (e.g., fertilizers and soil amendments) necessary to sustain productivity (Lal et al., 2019). Soil erosion caused by water is estimated to result in an annual global cost of 8 billion ($8bn) US dollars to the Gross Domestic Product (GDP), primarily due to the reduction in agricultural productivity. This equates to around 33.7 million tonnes of agri-food production and has led to price increases from 0.4% to 3.5%, depending on the specific sector (Sartori et al., 2019). Furthermore, the repercussions of soil degradation extend beyond mere direct economic losses.

Nkonya et al. (2016) calculated that the annual global costs associated with soil degradation, resulting from changes in land use and land cover, reach approximately 230 billion ($230bn). The most significant financial impacts were observed in countries located in Sub-Saharan Africa, incurring costs of 60 billion US dollars ($60bn) annually, followed by Latin America, totaling 53 billion US dollars ($53bn). Additionally, the same research showed that the global cost of soil fertility depletion in maize, rice, and wheat cultivation accumulated 57 billion US dollars ($57bn). In the European Union (EU), the total known costs of soil degradation (e.g., crop productivity loss, sediments, soil compaction, nitrogen/phosphorus losses, $CO_2$ emissions, management of contaminated sites, and diffuse pollution) are projected to be around €40.9 billion Euros and €72.7 billion Euros annually (Panagos et al., 2024a, 2025). In Central Asia, the effects of soil degradation were particularly severe, diminishing agricultural net profits by a factor of 4.8 compared to a scenario with no degradation (Mirzabaev et al., 2023). These costs encompass both the increased use of fertilizers and other soil amendments (Baumhardt et al., 2015) as well as the broader economic repercussions (e.g., elevated healthcare expenses linked to deteriorating water and air quality; Koo, 2024; Van der Geest, 2019). Moreover, land and soil degradation results in the loss of vital ecosystem services (e.g., water purification and climate regulation), which can have substantial economic implications.

Although soil degradation has considerable financial repercussions, restoration initiatives can provide significant economic and ecological advantages. The restoration of degraded landscapes can lead to improved agricultural yields, which increases food production and income, especially for local populations (Zerbe, 2023). This increase in agricultural output can drive economic development and improve food security (Pacheco et al., 2024; Zerbe, 2023). Furthermore, restoration efforts such as the Tigray Project in Ethiopia (Hagazi, Gebrekirstos, et al., 2020; Hagazi, Gebremedhin, et al., 2020) and the Everglades Restoration Initiatives in Florida, USA can generate employment opportunities and promote sustainable livelihoods, enhancing societal well-being.

The process of restoring degraded soil involves considerable financial investments, including funding for soil restoration approaches (e.g., reforestation (Chazdon, 2008; Lamb et al., 2005), erosion control (Montgomery, 2007), and sustainable land management strategies (Pretty et al., 2006)). Irrespective, the long-term advantages of soil restoration, such as improved ecosystem services and increased land productivity, frequently surpass these initial costs (BenDor et al., 2015; Bullock et al., 2011). However, significant obstacles in soil restoration are the simultaneous presence of severe soil degradation, nutrient loss, and low-income circumstances in numerous areas. This situation creates a detrimental cycle, where soil degradation leads to depletion in nutrients. Moreover, the lack of financial resources prevents their restoration, further worsening the decline.

In preparation for the United Nations Convention to Combat Desertification Global Outlook 2 report (UNCCD, 2022), van der Esch et al. (2021) developed several restoration scenarios with cost analyses linked to each scenario. The report shows that Sub-Saharan Africa has the highest estimated costs for their restoration commitments, ranging between 112 and 631 billion USD. The median costs for different types of restoration projects range between 185 USD/ha and 2390 USD/ha, with agroforestry having the highest costs, and forest management having the lowest costs (Verhoeven et al., 2024). While the focus is on land restoration, the links to restoration of degraded soils are compelling and useful for this review (Figure 4).

## 3. Processes of Soil Degradation

### 3.1. Physical Processes

#### 3.1.1. Soil Structure Degradation

Soil structure has been defined in multiple ways, reflecting its characteristics across various spatial scales, from the nanoscale to the pedon scale. For instance, the International Union of Soil Sciences (IUSS, 2022) defines soil structure as the spatial arrangement of soil constituents and pores. A more comprehensive definition was provided by Letey (1991), who described soil structure as "the size, shape, and arrangement of the solid particles and voids which are highly variable and associated with a complex set of interactions between mineralogical, chemical, and biological factors." Both definitions emphasize the internal organization of soil components but do not explicitly account for external influences. To address this limitation, the concept of soil architecture was introduced to incorporate the effects of anthropogenic activities. De Jonge et al. (2009) defined soil architecture as "the pore and particle networks and their interfaces which are created by interactions between biotic and abiotic solids, water and solutes, and influenced by man during soil use and management."

Rabot et al. (2018) conducted a comprehensive review of commonly used physical soil structure properties and indicators relevant to soil ecosystem functions and services, including biomass production, water storage and filtration, nutrient cycling, carbon sequestration, habitat provision for biological activity, and physical stability and support. The structural properties examined in their review included bulk density, degree of compactness (i.e., bulk density relative to a reference bulk density, representing the maximum bulk density under natural field conditions), aggregate size distribution and stability, and water retention characteristics (pore-size distribution). Additional parameters included macropore volume, air capacity, available water capacity, the Dexter S index, soil-specific surface area, and various structural indices derived from soil visualization techniques such as computed tomography (CT) imaging (Shokri, 2014; Shokri & Sahimi, 2012). Rabot et al. (2018) further identified porosity, macro-porosity, pore distances, and pore connectivity derived from imaging techniques as promising soil structure indicators. However, they highlighted the limited availability of such methods and emphasized the need for an open-access soil structure library. In response to this challenge, simpler soil visualization methods, such as VESS (Visual Evaluation of Soil Structure), SubVESS, and CoreVESS, have gained widespread global adoption as alternatives to complex CT imaging techniques. Studies by Johannes et al. (2017), Franco

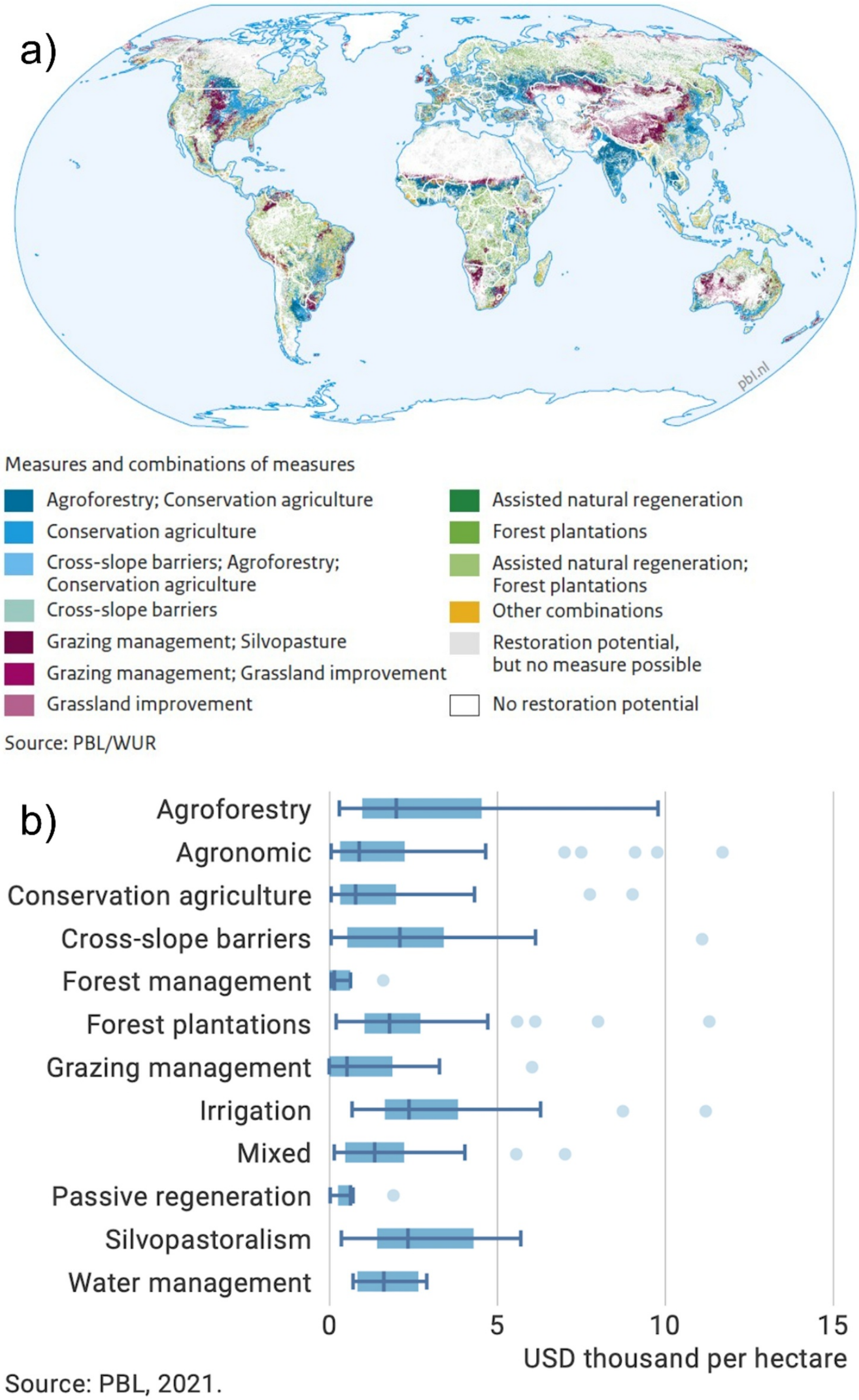


**Figure 4.** Global land restoration measures and associated costs. (a) Global distribution of improved land management and restoration measures as applied in restoration scenarios (van der Esch et al., 2021). (b) Cost ranges of major land restoration measures (in USD thousand per hectare; UNCCD, 2022).

10.1029/2025RG000883

**Table 1**
*Soil Degradation in Terms of Losses of Soil Architectural Components, Causative Threats and Impacted Soil Ecosystem Services*

| | Architectural component loss | Soil threats (primary and secondary) | Soil function or ecosystem service |
|---|---|---|---|
| 1 | Fine mineral particles (clay and silt size) | Erosion, vertical leaching | Stability, carbon sequestration, productivity |
| 2 | Organic carbon (OC) | Erosion, intensive cultivation | Stability, resilience, productivity |
| 3 | Aggregates, especially water-stable micro-aggregates | Erosion, intensive cultivation, high clay/OC ratio | Productivity, infiltrability and drainability, aeration |
| 4 | Specific surfaces and area | Erosion, intensive cultivation | Stability, chemical retention, productivity |
| 5 | Total porosity | Compaction, OC loss, aggregate loss | Infiltrability and drainability, aeration |
| 6 | Macro-porosity (>30 μm pores) | Compaction | Infiltrability and drainability, aeration, productivity |
| 7 | Meso-porosity (1–30 μm pores), including RAW (readily available water) for plants) | Compaction, aggregate loss | Water storage, nutrient availability, productivity |
| 8 | Micro-porosity (<1 μm pores), including intra-aggregate porosity | OC loss, aggregate loss | Biodiversity, nutrient availability, productivity |
| 9 | Wideness of pore-size distribution (physical diversity) | OC and mineral fines loss | Biodiversity, stability, resilience |
| 10 | Pore-network connectivity | Compaction, OC, and clay loss | Biodiversity, aeration, productivity |

et al. (2019), and Phefadu and Munjonji (2022) indicate that these visualization methods often show strong agreement with traditional laboratory measurements used to quantify soil structure.

Hu et al. (2021, 2023) highlighted the importance of considering both the condition of soil structure and its vulnerability when assessing the risk of soil structural degradation due to compaction and aggregate breakdown. They argued that frequently used structural properties are insufficient as standalone parameters, advocating instead for a focus on dynamic pore-network properties and their responses to variations in soil wetness. Several soil structure vulnerability coefficients were discussed, including the Valla et al. (2000) vulnerability coefficient, which quantifies the reduction in aggregate size under a given disaggregation force. Hu et al. (2021, 2023) emphasized that compaction and aggregate breakdown do not always negatively impact soil services. In some cases, moderate compaction may enhance carbon sequestration, plant-available water, and root-soil contact, potentially benefiting both climate regulation and crop yield. Expanding on this perspective, Hu et al. (2023), similar to Rabot et al. (2018), compared the “aggregate view” and the “pore space view” of soil structural dynamics and degradation. Their findings suggest that pore-network properties, rather than aggregation alone, serve as a more direct link to soil structural state and function. This aligns with de Jonge et al. (2009), who analyzed and visualized pore-network properties of “healthy soil” and “unhealthy soil” (characterized by organic matter depletion) using a sequence of soil-air phase-derived parameters (Figures 2–7 in their study). Key drivers of soil architectural dynamics and functional pore networks were identified, including soil-specific surface area, the mineral fines-to-organic matter ratio, soil pollution, root and microbial biomass exudates, soil-water repellency, gradients in soil wetness and matric potential, mechanical energy, and temperature. These factors were also recognized as critical influences on soil structural degradation.

Taking into account the interconnected roles of soil aggregation, pore network dynamics, and structural vulnerability, monitoring soil physical structure should focus on the architectural components, including mineral and organic particles, pores, surfaces, and their interconnections. Consequently, soil physical structure degradation can be defined as the loss of key components related to particles, surfaces, and pores within the soil's functional architecture (de Jonge et al., 2009). Table 1 provides a suggested framework for identifying structural losses contributing to physical degradation. Each loss is associated with primary and secondary soil threats, often leading to declines in essential soil functions, ecosystem services, and overall soil security for human, environmental, and climatic sustainability.

From a broader perspective, the 10 losses outlined in Table 1 could be integrated into a Physical Structure Degradation Index (PSDI) through the assignment of scores to each loss. However, the implementation of scoring functions presents considerable challenges, as not all losses are equally significant or necessarily detrimental. Moreover, the impact of structural losses may vary depending on the specific ecosystem service being considered. For instance, compaction-induced reductions in macroporosity may adversely affect aeration and drainage while

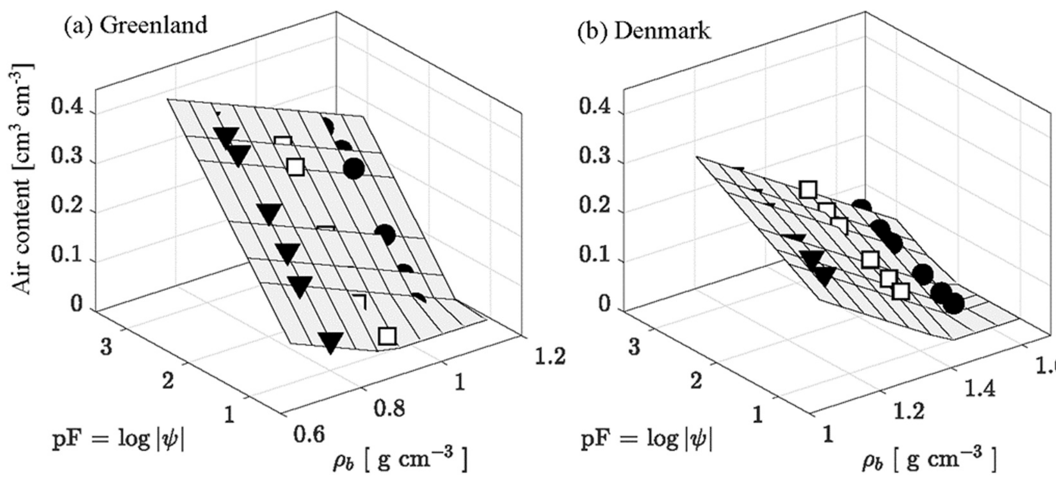


**Figure 5.** Different response surfaces of soil-air capacity or air content ($\varepsilon$) to compaction (bulk density; $\rho_b$) and soil moisture or matric potential (pF) for a high-OC (5.9%) soil (Greenland) and a low-OC (1.9%) soil (Denmark). Symbols: measured $\varepsilon$ corresponding to the lowest (▼), median (□) and highest (●) $\rho_b$ (modified from Pesch et al., 2021).

simultaneously enhancing soil resistance to further compaction. Similarly, a moderate reduction in soil aggregation may, in some cases, improve certain soil functions, such as water retention.

#### *3.1.1.1. Monitoring Soil Structure Changes Using Soil Air-Phase Parameters*

Soil-air phase parameters, including air content, gas diffusivity, and air permeability, are increasingly utilized—both individually and in combination—to assess soil structural status and monitor changes resulting from various environmental and anthropogenic threats. These threats include compaction, mineral particle loss through leaching and erosion, organic matter depletion, aggregate breakdown, and soil pollution (de Jonge et al., 2009; Müller et al., 2018; Oliveira et al., 2024; Pesch et al., 2021; Talukder et al., 2022). The high sensitivity of soil-air phase parameters to structural modifications also enables their application in quantifying the decline in soil conductivity and infiltrability due to bioactivity-induced phenomena, such as biocrust formation (L. Sun et al., 2022; F. Sun et al., 2022). Furthermore, these parameters can effectively capture both temporal and spatial variations in soil structure resulting from root development associated with different crop types (Uteau et al., 2013).

Soil-air content, also referred to as air capacity ($\varepsilon$; $m^3$ soil air/$m^3$ soil volume), can be directly measured in intact soil samples using an air pycnometer (Flint & Flint, 2002; Pulido-Moncada et al., 2019; Rüegg, 2000). Alternatively, it can be derived from the soil-water retention curve as the difference between total soil porosity and volumetric soil-water content at a given soil-water matric potential ($\psi$). Changes in air capacity within specific $\psi$ intervals at the wetter end of the retention curve provide insights into the distribution of larger pore-size classes in the soil. These pores, along with their connectivity, play a crucial role in regulating key soil functions, including water infiltration, drainage, and the mobility of solutes, air, and gases (Moldrup et al., 2001). The relationship between $\varepsilon$ and $\psi$, often expressed in pF units (where pF = log($\psi$) in hPa), is referred to as the soil-air characteristic curve (SACC; Pesch et al., 2021).

Figure 5 presents a response surface illustrating the soil-air characteristic curve (SACC) in relation to increasing compaction (bulk density) for a sandy soil from Greenland (Igaliku) with an organic carbon (OC) content of 5.9% and a silty soil from Denmark (Silstrup) with an OC content of 1.9% (Pesch et al., 2021). Across all moisture levels (pF values), air capacity ($\varepsilon$) declines rapidly as bulk density increases. Notably, increased compaction leads to a more pronounced reduction in $\varepsilon$ in the low-OC soil (Figure 5b), which reaches critical $\varepsilon$ levels below 0.1 $cm^3/cm^3$ more quickly. This threshold is frequently used as an indicator of soil aeration loss.

Soil gas transport parameters, including the gas diffusion coefficient ratio ($D_p/D_o$, the ratio of gas diffusion in soil to that in free air) and air permeability ($k_a$, $\mu m^2$), can be rapidly measured in laboratory settings (Ball & Schjønning, 2002; Lu et al., 2023; Rolston & Moldrup, 2002; Schjønning et al., 2013). Additionally, $k_a$ can be

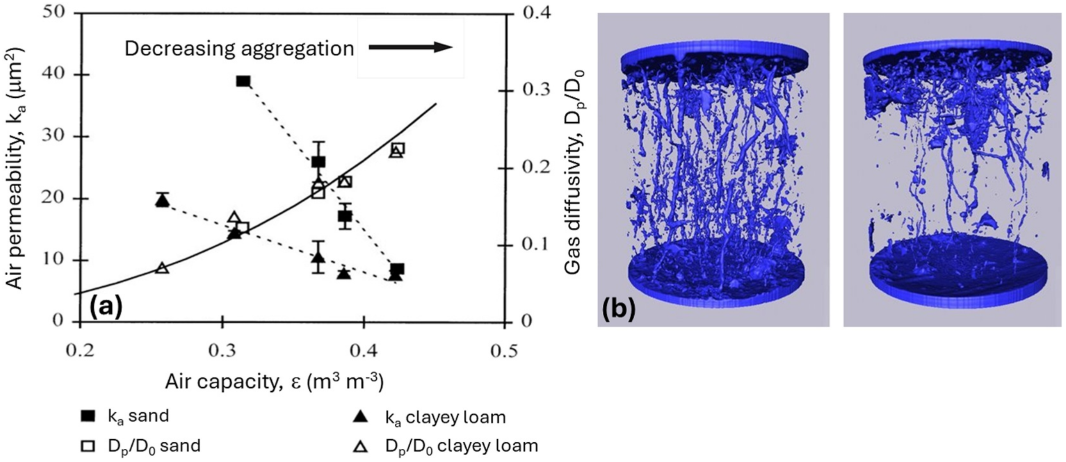


**Figure 6.** Soil air-phase parameters and X-ray CT scans can be used to follow soil structural degradation due to compaction/disaggregation. (a) Air permeability ($k_a$) and gas diffusivity ($D_p/D_o$) show opposite behavior as a function of air capacity ($\varepsilon$) when two soils become disaggregated. Air permeability decreases linearly with $\varepsilon$ while $D_p/D_o$ increases non-linearly with $\varepsilon$ during aggregate breakdown (modified from Moldrup et al., 2001). (b) X-ray CT scans show the effect of compaction on 20 by 20-cm intact core samples. On the left, a soil never compacted by machinery, showing an abundance of macropores >0.6 mm. On the right, the soil was run over by a beet harvester, eliminating most of the vertical arterial pores essential for infiltrability and drainage. Also in this case, air permeability but not gas diffusivity strongly decreased (after Lamandé et al., 2013).

efficiently and non-invasively measured in the field using a portable air permeameter (R. Chen et al., 2021; S. Chen et al., 2021; Steinbrenner, 1959). Fish and Koppi (1994) demonstrated correlations between in situ $k_a$ measurements obtained via a portable air permeameter and changes in soil morphology and structure. Since $D_p/D_o$ and $k_a$ measurements rely on the application of a gas concentration gradient or a low air pressure gradient (1–5 hPa), they do not disrupt the soil volume or sample integrity. The ability to perform rapid, non-destructive measurements makes these gas transport parameters particularly valuable for assessing soil structural status, detecting changes, and monitoring potential degradation.

Each soil-air phase parameter ($\varepsilon$, $D_p/D_o$, $k_a$) provides distinct and complementary information about soil structure and pore network architecture (de Jonge et al., 2009). Furthermore, each parameter responds differently to changes in soil structure, reflecting variations in pore size distribution, connectivity, and functionality. Figure 6a fundamentally illustrates these dynamics for a sand and a clayey loam soil. In this experiment, soil was re-packed and equilibrated at varying water contents while maintaining constant porosity ($\rho_b$), with increasing soil aggregation under wetter conditions (lower $\varepsilon$). Because oxygen ($O_2$) gas diffusion occurs through all connected air-filled pore spaces, $D_p/D_o$ decreases rapidly and non-linearly with decreasing $\varepsilon$, following a power-law model. This behavior is consistent across both soil types.

In contrast, convective air transport primarily occurs through the largest and most well-connected macropores. As soil aggregation diminishes, these macropores are reduced, leading to a rapid linear decrease in $k_a$ with decreasing $\varepsilon$. The $k_a$ reaches the same low value (around 10 $\mu m^2$) when the soils are devoid of both aggregates and water at their maximum $\varepsilon$. Counterintuitively, this pattern contrasts with the conventional expectation that both $k_a$ and $D_p/D_o$ increase with increasing $\varepsilon$, but at the same time provides a more effective means of monitoring soil structural changes by measuring both parameters. Supporting this, $k_a$ behaves differently between the two soil types, indicating a more complex relationship between soil structure, aggregation, and macropore connectivity.

Soil structural changes and degradation can be monitored more effectively by integrating conventional soil physical measurements with advanced visualization techniques such as X-ray computed tomography (X-ray CT). For instance, Naveed et al. (2013) utilized X-ray CT in conjunction with detailed soil-water retention and gas transport measurements to track structural changes along a steep field gradient in the clay-to-organic-carbon ratio. Figure 6b illustrates how X-ray CT scans can visualize the elimination of subsoil vertical macropores following compaction. In this and other studies conducted on soils from Denmark, Sweden, and Finland, the persistent and long-term effects of subsoil compaction were evident through combined measurements of water retention, gas diffusivity, and air permeability, along with X-ray CT scans of 20 × 20 cm intact soil cores (Lamandé et al., 2013).

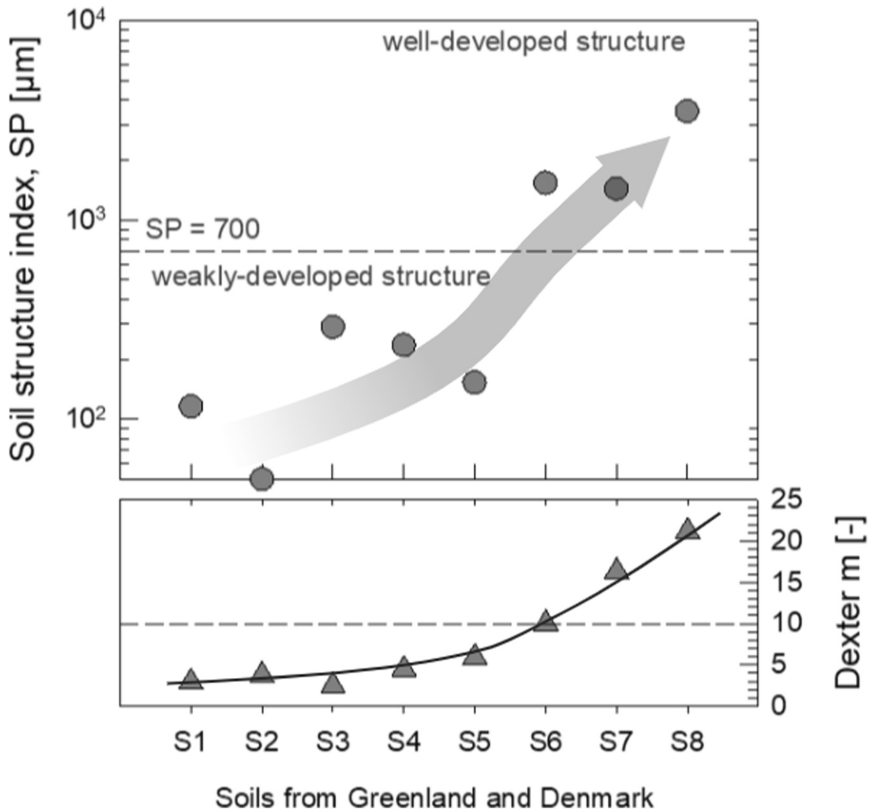


**Figure 7.** Soil structure index SP = $k_a/(D_p/D_o)$ and Dexter m = (clay + silt)/OC of eight different cultivated soils from Denmark and South Greenland. The horizontal lines represent thresholds at which soil structure (SP = 700; Kawamoto et al., 2006) and soil texture and organic matter ($m$ = 10; Schjønning et al., 2010) effects on air functions are predominant. Gray arrow depicts increasing soil structure development or complexity (Data from Pesch et al., 2021).

X-ray CT technology continues to evolve rapidly, offering new possibilities and overcoming previous limitations. Its applications and future potential in soil science and soil structure evaluation are further explored in recent reviews by Ghosh et al. (2023) and Zhang et al. (2023).

Simple and sensitive soil structure indices can also be defined by combining soil-air properties (Arthur et al., 2013; Moldrup et al., 2001; Oliveira et al., 2024). The three most commonly used and straightforward indices are: pore network tortuosity, $T$ ($\sqrt{}$ $(\varepsilon/(D_p/D_o)$, the pore organization (PO) index: PO = $k_a/\varepsilon$, and the structural parameter (SP) index: SP = $k_a/(D_p/D_o)$. Figure 7 illustrates the decrease in the SP index at a given matric potential (pF 2) across a large-scale gradient of intact soils, reflecting a decreasing ratio of fine mineral particles (clay + fine silt) to OC.

Klöffel et al. (2024) also identified clay, silt, and organic matter as primary drivers and covariates influencing soil structure evolution in Swedish and Norwegian soils. The loss of soil mineral particles, through processes such as leaching and erosion, leads to a reduction in the soil's aggregation potential, which consequently results in a lower SP index (Figure 7). This observation aligns with the findings in Figure 6, which demonstrated a dramatic reduction in air permeability ($k_a$) during aggregate breakdown.

#### *3.1.1.2. Soil Compaction - An Invisible and Insidious Soil Degradation Risk*

Soil compaction is characterized by an increase in bulk density and a decrease in porosity. This phenomenon can arise from both natural processes and human activities, such as vehicle traffic, livestock trampling, or changes in land use (Daniells, 2012; Schneider & Don, 2019). Natural compaction often occurs under periglacial conditions, leading to the formation of dense layers (Fitzpatrick, 1956) or naturally occurring pans (Needham et al., 2004). Additionally, hard-setting soils can develop impenetrable layers independent of human-induced compressive forces (Mullins et al., 1990). The forces that cause soil deformation and reduce pore spaces also modify the mechanical and hydraulic properties of the soil, negatively affecting physical, biological, and chemical processes. This can lead to decreased soil productivity, increased surface water runoff and erosion, and a decline in overall ecosystem health (Batey, 2009; Horn et al., 1995). This section will concentrate on the anthropogenic factors contributing to soil compaction.

*Anthropogenic soil compaction: degradation and recovery.* The primary causes of anthropogenic soil compaction operate on markedly different temporal scales. Firstly, land use changes, such as the transformation of grasslands into arable fields, lead to a reduction in soil organic carbon (Houghton, 1999; Poeplau et al., 2011), which subsequently results in increased bulk density (Robinson, Nemes, et al., 2022; Robinson, Thomas, et al., 2022). This process unfolds gradually over a span of years to decades (Or et al., 2021). Secondly, land management practices, particularly those involving heavy machinery or livestock trampling, cause immediate increases in bulk density (Cambi et al., 2015; Hu et al., 2021).

Soil compaction can be quantified as a state defined by the "compactness of soil" (Håkansson & Lipiec, 2000), yet this characterization merely captures a moment in an ongoing process. The overall impact of compaction arises from two opposing and simultaneous processes: mechanical degradation (compaction) and restorative actions ("de-compaction"). The rates at which these processes occur are influenced by factors such as land use, soil management practices, climate, and the inherent characteristics of the soil. The degradation time frames can vary from immediate (as seen with compaction from vehicle traffic) to several decades (as a result of decreased soil organic carbon due to changes in land management or use), while the recovery of soil structure, facilitated by both biotic and abiotic factors, may take years to decades (Or et al., 2021).

The mechanical and biological disturbances leading to soil compaction from land use changes are marked by an initial, "catastrophic" event that occurs rapidly—within days (e.g., tillage converting grassland to arable land) or weeks (e.g., deforestation)—followed by a prolonged, gradual disturbance that results in a net decrease in soil organic carbon accumulation. For instance, repeated tillage can expose previously protected soil organic carbon, increasing mineralization rates, and ultimately establishing a new mechanical and biological equilibrium (Baker et al., 2007; Reicosky, 1997).

Mechanical disturbances caused by vehicle traffic and animal trampling occur instantaneously; however, their effects accumulate over time. The frequency at which these disturbances occur is influenced by land use and soil management practices. Animal trampling may result in multiple disturbances within a single day, while there may be months between perturbations in mechanized agriculture and years between traffic events in forestry. Additionally, disturbances can arise from sporadic activities, including recreational traffic, construction projects, or military training exercises. It is important to recognize that compaction from vehicular traffic or animal trampling occurs only when the mechanical stresses applied exceed the soil's strength or critical soil stress. The interplay between the frequency of these compaction events and the time required for soil structure recovery is crucial in determining whether the soil experiences degradation over time or maintains a state of "dynamic equilibrium."

Understanding the rates at which soil structure recovers after compaction remains limited. However, various studies indicate that recovery can take months to years for near-surface soils (Blackwell et al., 1985; Radford et al., 2007), decades for subsoils in agricultural areas (Berisso et al., 2012), and even centuries in some cases (Brevik et al., 2002; Webb, 2002). The rate of soil structure recovery is influenced by several factors, including the severity of compaction—where more severe compaction leads to longer recovery times (Radford et al., 2007)—and soil depth, as recovery rates tend to diminish with increasing depth, likely due to variations in biological activity (Keller et al., 2021; Or et al., 2021). Additionally, pedo-climatic conditions such as soil type, texture, mineral composition (including the presence of expansive clays; Lehmann et al., 2021), as well as precipitation and evapotranspiration patterns, play a significant role. These factors also encompass the effects of vegetation growth on soil, which contributes carbon necessary for sustaining biological activity (e.g., earthworms) and influences abiotic processes like shrink-swell and freeze-thaw cycles. Furthermore, different soil properties exhibit varying recovery rates; properties that are sensitive to pore size and continuity (such as water infiltration and air permeability) tend to recover more quickly than those governed by soil mass per volume (like bulk density and mechanical resistance of the soil matrix; Keller et al., 2021). For instance, findings from a long-term field experiment on compaction recovery reveal that surface water infiltration returned to baseline levels within 2 years post-compaction (Keller et al., 2021). In contrast, the recovery of soil bulk density at a depth of 0.1 m takes approximately 10 years, while at 0.3 m, it is estimated to exceed two decades. Air permeability recovers more rapidly and appears to be less influenced by soil depth. Given that recovery occurs over an extended period, ranging from years to decades, it is essential to minimize the occurrence of disturbances (traffic events that lead to further soil compaction) to achieve dynamic equilibrium. Failure to do so will result in a gradual deterioration of soil structure over time.

Research indicates that tillage practices, including conventional tillage of arable topsoil, deep subsoiling, and "strategic tillage," can enhance the rates of soil recovery following compaction (Arvidsson & Håkansson, 1996; Schneider et al., 2017). However, it is important to note that tillage does not merely counteract compaction; studies have demonstrated that the restoration of compacted arable topsoil can require several years, even with regular tillage (Arvidsson & Håkansson, 1996). A meta-analysis examining the impacts of subsoiling (deep tillage) found that beneficial effects are often temporary (Schneider et al., 2017). In recent years, there has been a growing focus on utilizing plants to enhance soil structure and function after compaction, a practice sometimes referred to as "bio-tillage" (Hudek et al., 2022; Zhang & Peng, 2021). The roots of these plants can generate new macropores, which improve aeration and water infiltration, and allow subsequent crops to access deeper soil layers (e.g., Pagenkemper et al., 2013). Nonetheless, the formation of new biopores through root growth does not typically lead to a reduction in overall soil bulk density (Keller et al., 2021).

*Compaction by vehicle traffic: pressures and drivers.* The effects of soil compaction are influenced by the mechanical stresses applied and the capacity of soil to endure these stresses, which is determined by its strength. The mechanical soil stresses are dependent upon various vehicle attributes, including ground pressure, contact area, load on wheels or tracks, vehicle speed, and the degree of wheel or track slip. Soil strength is affected by soil composition—such as texture and organic matter content—as well as its structure, moisture status, and history of compaction (Horn, 1993). Furthermore, permanent deformation in soil decreases as the duration of loading decreases (Fazekas & Horn, 2005; Or & Ghezzehei, 2002).

This section does not aim to deliver an exhaustive analysis of soil stresses caused by various types of machinery; however, it is instructive to examine the historical evolution of typical soil stress levels to comprehend the increasing concern regarding soil compaction. The factors contributing to human-induced soil compaction stem from the necessity for land to provide agricultural resources for a burgeoning global population, which may necessitate changes in land use (such as converting forests or grasslands into arable land), as well as the demand for enhanced efficiency and capacity in agricultural and forestry operations (Nordfjell et al., 2019; Schjønning et al., 2015). The requirement for high-capacity machinery arises from intricate interactions among farmer preferences, private enterprises (i.e., agricultural businesses), intermediary trade, machinery manufacturers, national economies, and the global economy. Since the onset of industrialization, the power of agricultural vehicles has progressively increased, leading to a corresponding rise in the size and weight of these vehicles (see Figure 8). A similar trend in the dimensions and mass of machinery has been observed in the forestry sector (Nordfjell et al., 2019). This escalation in total mass and the associated rise in axle loads have consequently led to a gradual increase in soil stresses within the subsoil (at depths of 0.3–1.0 m) and the penetration of harmful stresses deeper into the soil (Keller et al., 2019; Keller & Or, 2022; McPhee et al., 2020; Schjønning et al., 2015), as depicted in Figure 8. The increasing magnitude of mechanical stress from increasingly heavy machinery raises the risk of exceeding soil strength thresholds, thereby exacerbating soil degradation (Keller & Or, 2022). Compaction in the subsoil is particularly concerning due to the slow recovery rates at these depths, which can lead to long-term hindrances in root development, water and gas movement, nutrient cycling, and subsequent reductions in crop yields (Batey, 2009; Håkansson & Reeder, 1994).

Soil strength, defined as the capacity of soil to endure mechanical stresses, is influenced by soil texture, increases with soil bulk density, and strongly decreases with soil moisture, while soil structure also plays a role (Horn, 1993; Horn & Lebert, 1994; Schjønning & Lamandé, 2018). The compressive strength of soil is characterized by the precompression stress, which indicates the threshold stress level beyond which the soil undergoes irreversible deformation. Typically, empirical models, such as pedo-transfer functions and random forest models, exhibit limited predictive accuracy in estimating precompression stress (Torres et al., 2024). Schjønning and Lamandé (2018) utilized a data set from Denmark to demonstrate that precompression stress remains relatively constant across different soil textures at approximately −100 hPa matric suction, with values around 117 kPa for a mean bulk density of 1,460 kg $m^{-3}$ in their study. However, it is important to note that irreversible deformation can occur even when the applied stresses are below the precompression stress threshold (Atkinson, 1993; Janbu, 1998; Keller et al., 2012; Kirby, 1994; O'Sullivan & Robertson, 1996). In their analysis of field data

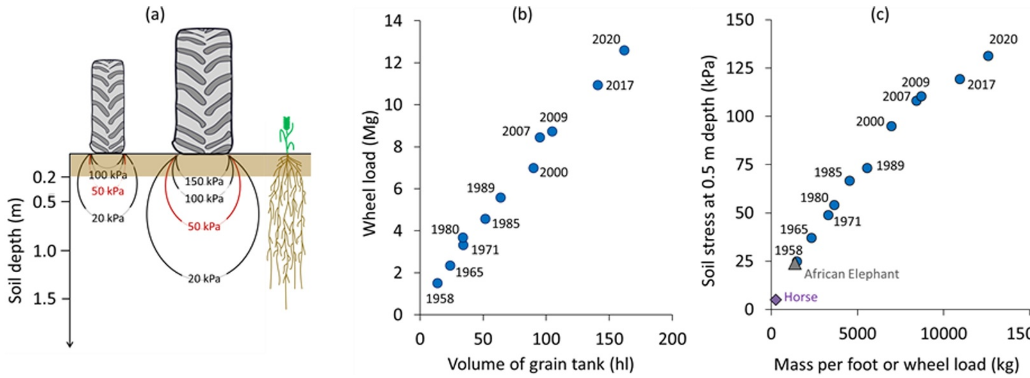


**Figure 8.** (a) The simulated vertical soil stress beneath a wheel load of 2,000 kg (left) and 8,000 kg (right), both exhibiting the same ground contact pressure, demonstrates that higher loads lead to deeper stress propagation, thereby heightening the risk of enduring subsoil compaction (with 50 kPa indicating the threshold for compaction under moist conditions). (b) The historical evolution of combine harvesters reveals that the increase in capacity, marked by the enlargement of the grain tank from 1958 to 2020, has resulted in a sixfold rise in front axle wheel loads, which has (c) correspondingly elevated subsoil stress levels by approximately five times (the years denote when specific harvesters were launched in the market; for comparative purposes, the mass and associated soil stress of a horse and an African Elephant are included; adapted from Keller & Or, 2022).

regarding soil stress and displacement caused by agricultural machinery, Keller et al. (2012) identified a critical stress threshold of about 50 kPa for arable subsoil under moist conditions, which aligns closely with the critical stress limits of 25–50 kPa proposed by Rusanov (1994).

*Effects of compaction on soil functions.* Soil compaction leads to a reduction in pore space, alters the distribution of pore sizes toward a smaller average, and diminishes both pore connectivity and continuity (Horn et al., 1995; Horn & Peth, 2011; Richard et al., 2001). This alteration limits the capacity of soil to hold air and water, while also hindering the movement of water and gases. Additionally, a reduction in porosity increases the mechanical resistance of the soil, posing challenges for plant roots and soil-dwelling organisms (Ruiz et al., 2023). As a result, soil compaction influences various soil processes and functions that rely on pore volume, structure, and mechanical properties, including the flux of gases and water, plant development, nutrient and carbon cycling, and the habitat for soil organisms. The extent of soil compaction's effects on soil characteristics, processes, and functions is context-specific and influenced by the stresses applied and the prevailing soil strength during the compaction event.

A meta-analysis conducted by Obour and Ugarte (2021) indicates that crop yields experience average reductions of approximately 10%–35% due to soil compaction, with the extent of decline varying by crop type (for instance, maize suffers greater yield losses compared to wheat) and soil texture (medium-textured soils exhibit more significant yield reductions than fine-textured soils). In a separate meta-analysis, Hernandez-Ramirez et al. (2021) reported that $N_2O$ emissions nearly doubled as a result of compaction. The long-term effects of traffic-induced compaction on soil organic carbon stocks remain under-researched. Substantial evidence indicates that compaction from agricultural traffic reduces soil water infiltration capacity, saturated hydraulic conductivity, and the ability of soil to retain water (Horn et al., 2019; Keller et al., 2019, and references therein). The reduction in infiltration rates can lead to waterlogging and increased surface runoff, increasing the risk and intensity of flooding and erosion (Alaoui et al., 2018).

Forest soils typically have a lower bulk density than soils in agricultural or grassland areas, making them more vulnerable to compaction (Panagos et al., 2024b). A meta-analysis conducted by Nazari et al. (2021) revealed that compaction from logging activities leads to a decrease in microbial biomass in subsoils and a reduction in saturated hydraulic conductivity. The adverse effects tend to escalate with the frequency of machinery passes. In hilly regions, compaction, especially in ruts, can exacerbate runoff and erosion, leading to the depletion of nutrient-rich topsoil and increased erosion (Cambi et al., 2015). This issue can significantly hinder or even prevent forest regeneration over prolonged periods (Cambi et al., 2015). Additionally, forest soils often display visible tracks from machinery, which serve as lasting signs of compaction.

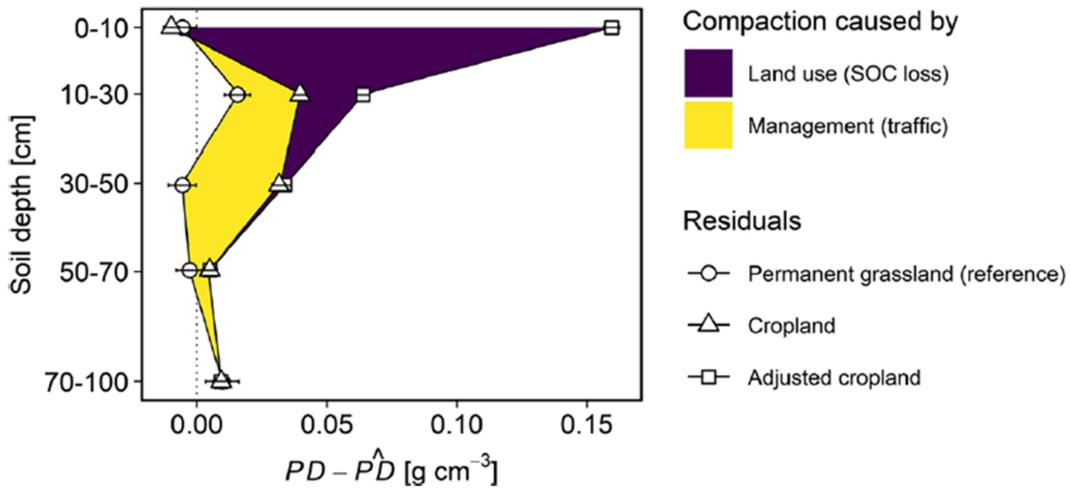


**Figure 9.** Impact of soil compaction resulting from changes in land use (indicated in violet) and from agricultural field traffic (represented in yellow) across Germany. PD and $\widehat{PD}$ refer to the measured packing densities and predicted packing densities, respectively. This analysis is based on a random forest model utilizing data from the German agricultural soil inventory, with permanent grassland serving as the uncompacted reference state. For further details, refer to Schneider and Don (2019). The packing density (PD) is calculated as bulk density plus 0.005 times the clay percentage plus 0.001 times the silt percentage (after Schneider & Don, 2019).

*Global distribution and costs of soil compaction.* Soil compaction is widely acknowledged as a significant human-induced factor contributing to the physical degradation of soil; however, the global scale and intensity of this issue remain inadequately understood. This lack of knowledge can be attributed to the absence of effective methods for quantifying the spatial distribution of compaction over extensive areas, particularly in deeper soil layers that are not easily observable. Additionally, establishing a benchmark for uncompacted soil poses challenges. While reference values for bulk density or packing density have been proposed (Lebert et al., 2007), the underlying data and statistical methodologies used to derive these values are somewhat ambiguous. Furthermore, locating "uncompacted" soil, that is, unaffected from anthropogenic influence to serve as a reference profile is difficult. Despite these challenges, some estimates regarding the prevalence of soil compaction have been made for specific regions and countries. For instance, Brus and van den Akker (2018) estimated that 43% of arable subsoils in the Netherlands are excessively compacted. In Germany, Schneider and Don (2019) reported that 51% of cropland and 32% of grassland are impacted by root-restricting layers resulting from compaction, with agricultural practices contributing to compaction in 27% of cropland cases. Their research, employing a random forest modeling technique, indicated that in the topsoil, compaction is mainly driven by land use changes that lead to a reduction in soil organic carbon, whereas in the subsoil, agricultural traffic is the primary cause of compaction (Figure 9). Similarly, a recent investigation into bulk densities in European topsoil found that density levels increase in the following order: forest soils, grasslands, and croplands (Panagos et al., 2024b). Schjønning et al. (2015), analyzing the European SPADE8 soil database (Panagos et al., 2012), discovered that approximately 25% of European agricultural subsoils (at depths of 0.25–0.7 m) exhibit critically high density levels.

The research conducted by Nazari et al. (2023) on global patterns of forest soil compaction susceptibility indicates that tropical and temperate forests, which exhibit low bulk densities and elevated soil organic carbon levels, are particularly vulnerable when moist conditions prevail during timber harvesting. A recent assessment of subsoil compaction risks in global arable crop production highlights that the highest risks are found in regions with significant mechanization, which correlates with elevated soil stress levels (see Figure 8b), alongside the presence of moist, mechanically weak soils. This includes areas in Europe, North America, Brazil, and Eastern Australia (Figure 10; Keller & Or, 2022). Furthermore, comparisons with the findings of Sonderegger and Pfister (2021) support the conclusion that mechanization is a primary factor driving soil compaction, leading to reduced crop yields in highly mechanized agricultural regions, whereas in areas with lower mechanization, erosion is the main cause of yield losses.

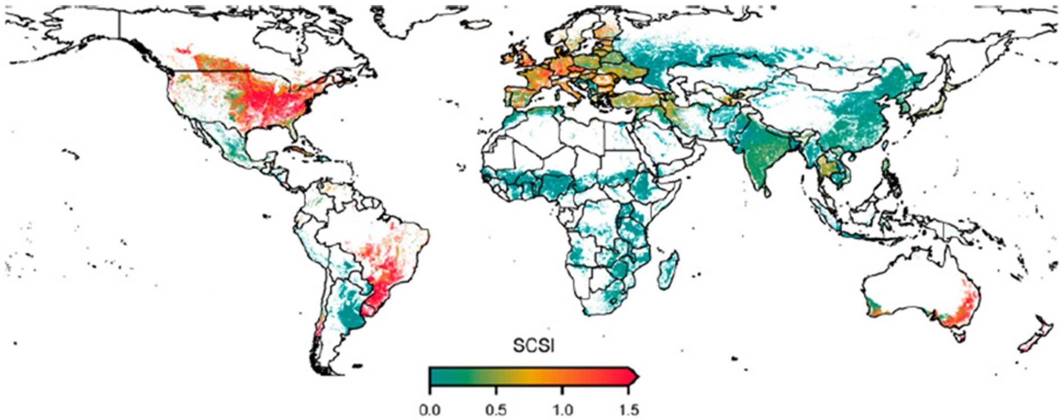


**Figure 10.** The worldwide assessment of the risk of subsoil compaction, with green indicating low risk and red signifying high risk (after Keller & Or, 2022).

Estimating the financial implications of soil compaction presents significant challenges, with limited data available. A study conducted in England and Wales approximated the annual costs of compaction to be £470 million. These costs encompass both on-site factors, such as reduced yields, inefficient fertilizer use, and increased fuel consumption during field operations, as well as off-site consequences, including flooding, greenhouse gas emissions, and nutrient leaching. Sonderegger and Pfister (2021) employed a life cycle assessment framework to estimate that high-input agriculture experiences an average annual crop yield loss of 5% across an area of 5.12 million $km^2$, which represents 60% of the total cultivated land of 8.54 million $km^2$. The global average yield for wheat is approximately 3,600 kg per hectare, translating to a total global production of 793 million metric tons from a harvested area of 220 million hectares (source: https://www.statista.com/topics/1668/wheat/). Consequently, a 5% yield loss equates to a reduction of 180 kg per hectare. If we hypothetically consider that the entire high-input crop area of 5.12 million $km^2$ is dedicated to wheat cultivation, this yield loss would result in an estimated decrease of around 93 million metric tons. With current wheat prices hovering around 200 USD per metric ton (source: https://fred.stlouisfed.org/series/PWHEAMTUSDM), the annual financial impact of yield losses due to compaction would approach 20 billion USD. It is important to note that yield losses represent only a fraction of the total costs associated with compaction. Graves et al. (2015) estimated that in England and Wales, yield losses accounted for approximately 30% of the overall costs of compaction. Assuming a similar proportion of yield loss costs relative to total costs—acknowledging that this ratio may vary across different global regions—the total annual costs of compaction could surpass 50 billion USD.

*Solutions to minimize soil compaction by vehicle traffic.* The natural recovery of soil structure occurs at a slow pace, and mechanical restoration is not only energy-intensive but also carries significant risks of yielding unsatisfactory long-term outcomes (Schneider et al., 2017). Therefore, the primary objective should be to prevent soil compaction. In theory, soil compaction can be prevented by ensuring that the mechanical stresses applied do not exceed the existing soil strength (Horn & Lebert, 1994). To assist in this endeavor, various simulation models and decision support tools have been developed to assess compaction risk by predicting soil stress and strength based on machinery characteristics and soil properties (e.g., SOCOMO, van den Akker, 2004; Terranimo®, Stettler et al., 2014; REPRO, Rücknagel et al., 2015). The uncertainties associated with compaction risk assessments mainly stem from challenges in acquiring soil data, particularly regarding soil moisture profiles, as well as the limited predictive capability of models estimating precompression stress (Torres et al., 2024). While the concept of preventing compaction is straightforward, achieving complete avoidance in practice may be challenging. This is largely due to the necessity of field traffic for agricultural production or forestry under economically feasible conditions.

The risks associated with soil compaction can be reduced by avoiding traffic on wet soil or by enhancing soil strength through effective drainage systems, plant water uptake, and root reinforcement, such as through the use of under-sown crops. Additionally, improvements in soil structure may be accelerated by fostering biological activity.

Additionally, reducing soil compaction in agriculture can be accomplished by better synchronizing cropping systems, such as crop rotation, with the specific pedo-climatic conditions of the site. This approach aims to decrease the chances of conducting field operations during wet conditions and involves adapting machinery size to the strength of the soil, often necessitating the use of lighter equipment. Nevertheless, the pursuit of economic efficiency in both agriculture and forestry typically demands high-capacity machinery, which tends to be heavier and consequently exerts greater pressure on the soil, creating a vicious cycle. The introduction of autonomous vehicles and robotic fleets presents innovative opportunities to dissociate efficiency from the reliance on heavy machinery. Lightweight autonomous vehicles show potential for advancement, although challenges persist, particularly regarding the transportation of substantial quantities of harvested crops from the field within limited timeframes (McPhee et al., 2020). The implementation of autonomous electric lightweight machinery could already prove advantageous if the costs associated with soil compaction and climate impact were taken into consideration (Lagnelöv et al., 2023).

An alternative approach to managing soil compaction involves the implementation of permanent tramlines, as seen in controlled traffic farming (McPhee et al., 2020; Tullberg et al., 2007) and in forestry practices utilizing "permanent" skid trails, which last for 50 years or more (Garland, 1983; Werder et al., 2025). In these systems, all vehicular traffic is restricted to designated tracks, with vehicles and implements tailored to specific track and working widths. In arable farming, the standard track gauge is approximately 3 m, while working widths typically vary from 9 m (for sowing and harvesting) to 27 m (for spraying; Tullberg et al., 2007). In forestry, the system comprises access lines that are 3–4 m wide and left unplanted (Werder et al., 2025), ideally occupying no more than 15% of the total area (Garland, 1983). In arable cropping, the proportion of wheeled area is about 10%–15%, in stark contrast to the over 100% seen in "random traffic systems." Research has indicated that this method can lead to increased crop yields and decreased surface water runoff and greenhouse gas emissions (Chamen et al., 2015; Tullberg et al., 2007). However, the system has drawbacks, including potentially high costs associated with modifying machinery to fit specific widths. Additionally, the tracks become significantly compacted, rendering them "permanent," which may adversely affect local hydrology and complicate future soil management and land use adjustments.

#### *3.1.1.3. Change of Soil Bulk Density and Porosity by Compaction Due To Loss of SOM*

While the major emphasis of compaction research is focused on livestock or machine traffic on soils, there is growing evidence of large-scale porosity changes that may be driven by other processes (Hirmas et al., 2018). Soil biota, vegetation, and organic matter play an important role in the development and maintenance of soil porosity (Robinson, Nemes, et al., 2022; Robinson, Thomas, et al., 2022; Ruehlmann & Körschens, 2009). Hence, this implies that activities or processes that disrupt biological activity or degrade soil organic matter, ranging from climate change, wildfires, or land use change, could all result in changes to soil porosity and potentially soil densification.

*Observations at national to continental scales.* For context at national and continental scales in this review, we refer to work on both bulk density and porosity, focusing on the factors that influence these properties and their role in densification or change in porosity of soils. In the last decade, the work of Hirmas et al. (2018) indicated continental scale change in soil macroporosity over decadal periods. They suggest a number of processes that could be driving such trends, including enhanced illuviation, mineralogy, and organic matter. After accounting for such processes, they further observed that drier and warmer climates tend to promote the development of surface-layer macroporosity, whereas more humid and cooler climates restrict the expression of macroporosity. Similarly, work by Robinson et al. (2016) provides evidence for soil structural changes in response to climate on annual to decadal timescale. These findings suggest a potential link between porosity, vegetation, and climate which remains to be fully explored.

Recent research on the bulk density and porosity of temperate soils at national (Thomas et al., 2024) and continental (Panagos et al., 2024b) scales has sought to identify environmental factors driving spatial variations in these properties. Such large-scale studies indicate the importance of SOM, but also habitat type for predicting porosity. Moreover, it shows that topsoil structural porosity is diverse and varies beyond agricultural soils. For example, Thomas et al. (2024) showed that texture, using national data from the UK, only explains about 37% of the variation in porosity for these temperate soils. However, when habitat and SOM are included, the variance increases to 70%–80%. Moreover, including both habitat and SOM improves prediction compared to models that included only one of these factors. This perhaps alludes to the role of habitat specific biota influencing the porosity as well as SOM, this could be plant roots or organisms like earthworms. Both the work by Thomas et al. (2024) and that of Panagos et al. (2024b) show clear gradients with reductions in porosity from woodland to grassland to cropland, consistent with declines in SOM commonly observed in these habitats.

*Mechanisms and processes.* Understanding observations such as Land Use/Cover Area Frame Survey data (LUCAS) across Europe, with changes in bulk density and porosity, requires a range of mechanisms and processes to be considered. Soil undergoes constant pedoturbation by biota and mechanical alteration due to shrink-swell and freeze-thaw processes. Wetting and drying cycles provide forces to bring particles together and gradually densify the soil through particle rearrangement, aggregate breakdown, pore space reduction, and biological degradation. This is especially the case as climate change and an increase in extreme events push soils beyond their normal operating range for biota in terms of floods and drought. Such processes can alter infiltration, root penetration, and soil aeration, negatively impacting plant growth and soil health. Soils requiring intervention to combat such changes are, by definition, degraded.

Different mechanisms manifest in different climates and under different management. Interfering with the co-evolving soil-vegetation ecosystem will inevitably disrupt and potentially degrade porosity, principally the structural porosity through either the loss of carbon or the reduced activity of the biota. In the organic soils of the northern latitudes, it is the fibrous geometry of the organic constituents that is, one factor leading to high porosities in peat, for example (~90%; Robinson et al., 2022), so the loss of such fibrous materials or change in orientation (for example, anisotropy) will likely trigger a change in porosity. Drainage of such systems leads to densification through shrinkage due to soil moisture loss rather than decomposition/SOM loss. Sampling and monitoring such systems is challenging, contrary results can be observed with fixed-length sampling such that an increase in carbon density in such systems is often a sign of compaction and degradation compared to the native state of a bog (Minkkinen & Laine, 1998).

In temperate latitudes, soils that are a mixture of organic and mineral material can form high porosities through aggregation (Thomas et al., 2024). The loss of organic matter from aggregates, especially the large ones, will lead to weaker structures more vulnerable to collapse through the normal process of wetting and drying. In temperate systems, especially arable soils, careful management, for example, organic matter addition, reduced tillage, or the use of cover crops can help mitigate densification and compaction risks (Blanco-Canqui et al., 2024; Blanco-Canqui & Ruis, 2018; Brady & Weil, 2008). Reduction in porosity due to soil mixing and matric forces is resisted largely by organic matter gluing aggregates, by the presence of organic matter, or by the generation of new porosity by biota, burrowing and rooting, for example. Soil organic matter may help increase soil resilience, helping to avoid soil compaction by machine traffic (Zhang et al., 2005). The study of Torres et al. (2024) showed that soil organic matter positively influences soil precompression stress—an indicator of soil strength—likely through its positive effect on soil aggregate stability. The addition of soil organic matter, however, also has a relevant secondary impact by improving soil water retention, which may increase the soil's susceptibility to compaction by decreasing precompression stress (Pereira et al., 2007).

In drier soils of the Mediterranean, organic matter is naturally less; these soils are often dominated by structures that evolve from mineral interaction and tactoid formation. The tropics' highly weathered soils may be dependent on sesquioxides maintaining structure, often initiated by old root channels. Land use change, particularly the conversion of forest, makes such soils highly vulnerable to structural degradation and porosity decline, impacting infiltration (Zimmermann et al., 2006). One of the major problems in such tropical soils is that their structure does not regenerate quickly.

Aggregation is an important concept in soil science (Berli et al., 2008; Ghezzehei & Or, 2000). Aggregates can develop naturally or be artificially induced by processes such as tillage. Aggregates can be bound by organic matter or mineral binding agents such as calcium carbonate or sesquioxides, each conferring different physical properties on the soil. Oades and Waters (1991) observed these different processes resulted in the extraction of different size fractions of aggregates in different soil types. Mollisols and Alfisols had more macroaggregates >250 μm attributed to roots and hyphae stabilizing these fractions, then root fragments and Particulate Organic Matter (POM) nucleating 20–250 μm aggregates and mineral interactions dominating for aggregates <20 μm. Hence, macroaggregates were observed in temperate Mollisols and Alfisols rich in SOM, but not in tropical Oxisols, which are SOM poor. Thomas et al. (2024) also observed that the relationship between macroaggregate

10.1029/2025RG000883

fraction and SOM followed the same curve as in Robinson et al. (2022), supporting the role of SOM in temperate aggregate structures. Hence, factors degrading SOM will likely lead to structural degradation and densification.

At the scale where clay minerals interact (∼1 μm), colloids are surrounded by an electrical double layer at the interphase between the colloid and the surrounding solution (DeCarlo & Shokri, 2014), this electrical layer changes in thickness depending on the electrical conductivity (EC), pH, and dominant cation in soil solution. When the thickness of the electrical double layer decreases within the range of attractive forces, the particles come together to form a tactoid, which is the initial stage of aggregate formation; conversely when the thickness of the double layer increases, repulsion forces are dominant and dispersion occurs. The higher the EC, the thinner the double layer, allowing the colloids to get close enough for attractive forces to be active (DeCarlo & Shokri, 2014; Shokri et al., 2015). Quirk and Schofield (1955) developed the concept of "threshold concentration" that determines the levels of EC required to maintain soil structure. The combination of high pH and the presence of sodium has detrimental effects on soil properties; this combination is associated with colloid dispersion, loss of organic carbon (Hassani et al., 2024), decrease in soil permeability, and increase in run-off and erosion (Lebron et al., 1994; U.S. Salinity lab, 1954). The process initiates with the water movement that helps to transport the dispersed colloids, clogging pores deeper in the soil profile and causing densification and a decrease in hydraulic conductivity. Addition of low EC water to sodic and alkaline soils may lead to structural collapse due to the increase in the thickness of the double layer—once the structure is lost, it can be very hard to restore. Remediation of these soils using gypsum as an amendment is the usual intervention to restore soil structure and increase porosity (Lebron et al., 2002). The impact of all these processes that lead to a decline in porosity, and often the pore size distribution, is alteration of the hydrological performance of the soils, most commonly observed through changes to infiltration rates.

Bulk density and porosity (Thomas et al., 2024) and consequently infiltration (Jarvis et al., 2013), especially in temperate soils, have a complex interplay with land use and SOC that remains to be better understood. This is most likely driven by the soils' macropore system, where macropores are present, with important contributions from the biota. Textbook knowledge (Brady & Weil, 2008) connects increased SOC with improved soil structure formation and better soil structural status, which in turn is perceived to enhance the expression of structure related to soil physical and hydrological properties. Indeed, there is ample support that suggests that SOM improves porosity and decreases soil bulk density (Robinson, Nemes, et al., 2022; Robinson, Thomas, et al., 2022). However, the positive effect on $K_{sat}$—the laboratory-measured proxy used to infer the soil's infiltration capacity—has been challenged by several studies for certain soils (e.g., Araya & Ghezzehei, 2019; Jarvis et al., 2013; Larsbo et al., 2016; Nemes et al., 2005). These studies have attempted to isolate the effect of SOC on $K_{sat}$ using a diversity of machine learning approaches and independent large data sources. The suggested mechanism is not yet clear, but is likely linked to SOC's role in developing a more tortuous pore system. Lack of clear understanding of the SOC-$K_{sat}$ relationship indicates a potentially non-trivial functional response by different soil types to loss of SOC, associated densification, and other related changes to their pore system (Nemes et al., 2005). Insight provided in Robinson et al. (2022), suggests that it's not simply the quantity of SOC but the geometry, which is likely to vary from coatings that may have the least impact to fibrous structures that have a large excluded volume. The different cited studies go to different lengths in identifying which groups of soils may be more affected, but they do not give clear delineation.

Jarvis et al. (2013) showed that infiltration rates in temperate soils were mostly dependent on porosity, SOM, and land use. While in a global meta-analysis of infiltration rates on the same soils under different land covers, Robinson et al. (2022) showed a decline in infiltration rates from woodland to cropland, supportive of Jarvis et al. (2013) findings more widely. Moreover, the work indicated that texture-based pedo-transfer functions gave on average a reasonably unbiased prediction of hydraulic conductivity for arable soils, while grasslands showed 1.30 times higher rates and woodlands showed 2.21 times higher rates. This is perhaps unsurprising, as data-driven pedotransfer functions are often developed using large data sets from largely cropland soils and drier climates (Pachepsky & Rawls, 2003); there's a need for more data from other habitats, especially woodlands. Such observations are consistent with the observations of Thomas et al. (2024), who indicate the importance of SOM and habitat on structure generation. Such studies amplify the case that degradation of SOM by climate, land use change, or management is likely to result in densification with yet to be determined impacts on soil hydrological properties.

#### *3.1.1.4. Erosion Rates and Physical Loss*

The term "erosion" refers to the geological processes involving the detachment, movement, and accumulation of soil or rock materials driven by natural forces such as gravity, ice, water, and wind. Soil erosion, a specific aspect of erosion, primarily impacts the topsoil and is significantly influenced by land use and management practices, which can lead to deviations from natural erosion rates (Lal, 2003). On-site soil erosion results in the loss of soil and rock particles, aggregates, and essential nutrients, chemicals, and organic compounds from their original locations. The consequences of this erosion extend beyond the immediate area (off-site), leading to the redistribution of these materials across the land surface and into aquatic and atmospheric systems. Soil erosion not only diminishes soil fertility, functionality, and natural productivity but also contributes to sedimentation problems, damage to infrastructure, and pollution of both aquatic and air environments due to the transport and deposition of minerals, organic matter, and soil-bound contaminants (Poesen, 2018).

Soil erosion processes can be categorized based on their driving forces, which include natural elements like wind and water erosion, as well as mechanically induced factors such as tillage erosion and soil loss due to crop harvesting (SLCH). These processes exhibit variability in their underlying mechanisms, interactions with environmental factors, and their spatial and temporal distributions (Borrelli et al., 2023), along with their implications for society. Natural erosion by wind and water tends to occur sporadically and unpredictably in both space and time, influenced by meteorological conditions. In contrast, mechanical erosion processes are also intermittent but are closely linked to specific management activities like tillage and harvesting. The intricate nature of soil erosion complicates the comprehensive understanding of its spatiotemporal dynamics, necessitating reductionist methodologies to analyze spatial patterns through long-term statistical averages (Borrelli et al., 2023). A significant research focus within the context of soil erosion and soil degradation is to investigate the synergistic or antagonistic interactions in regions where multiple erosion processes coexist (Figure 11).

*Water Erosion*. Water erosion is recognized as the most thoroughly examined process of soil loss driven by fluid dynamics (Poesen, 2018). Figure 12 illustrates examples of water erosion in agricultural landscapes. Water erosion encompasses a continuum of interrelated processes, including rainsplash erosion, interill erosion, rill erosion, and gully erosion (Aksoy & Kavvas, 2005). Piping erosion, a form of subsurface erosion occurring within hydrological conduits, involves concentrated subsurface flow that leads to the creation of linear voids, which can facilitate sediment transport and result in surface collapse (Bernatek-Jakiel & Poesen, 2018; Verachtert et al., 2011). Consequently, water erosion represents a multifaceted system where the continuum of processes illustrates the evolution of water flow dynamics, beginning with the impact of individual raindrops on the soil surface and progressing to diffuse and channelized flows capable of incising the landscape. This latter stage is linked to the movement of substantial sediment volumes over extended distances, contributing to severe land degradation (Poesen, 2018). The characteristics of soil, such as texture and soil organic carbon (SOC), determine the "erodibility" of soils (Knapen et al., 2007), which refers to their vulnerability to detachment by raindrop impacts and overland flow, as well as their hydrological properties like hydraulic conductivity that influence the generation of overland flow (Wischmeier & Smith, 1978). Splash erosion specifically entails the detachment and short-distance radial transport of soil material due to raindrop impact, resulting in a net downslope movement (Kinnell, 2005). This type of erosion alters the soil surface by breaking down soil aggregates, reducing surface roughness, and forming soil crusts, all of which increase soil erodibility and decrease infiltration rates (Le Bissonnais, 2016). Furthermore, splash erosion affects both the physical and chemical properties of soil, leading to significant consequences for soil degradation and biogeochemical cycles (García-Ruiz et al., 2015).

Surface runoff can originate from either Hortonian (infiltration-excess) overland flow (HOF) or Dunnian (saturation-excess) overland flow (SOF) processes (Buda, 2013), facilitating soil redistribution across extensive areas. When surface runoff occurs, it detaches and transports soil particles if the sediment transport capacity surpasses the sediment load, which is influenced by the flow rate and velocity (Merritt et al., 2003). Conversely, when the sediment load exceeds the transport capacity, soil particle deposition occurs (Aksoy & Kavvas, 2005).

Estimates of global gross erosion, derived from statistical analyses and model extrapolations, range from 25 to 45 Pg yr$^{-1}$ (Borrelli et al., 2017a; Quinton et al., 2010). In the context of the European Union and the United Kingdom, this figure is modeled at 0.97 Pg yr$^{-1}$, translating to an average of 2.46 t ha$^{-1}$ yr$^{-1}$ (Panagos et al., 2015). For China, the average modeled erosion rate is reported at 5.02 t ha$^{-1}$ yr$^{-1}$, with higher rates of 18.2 t ha$^{-1}$ yr$^{-1}$ observed in cultivated cropland situated on slopes (Liu et al., 2020). In the United States, similar modeling approaches yield an erosion rate of 2.32 t ha$^{-1}$ yr$^{-1}$ (Shojaeezadeh et al., 2024). Global forecasts,

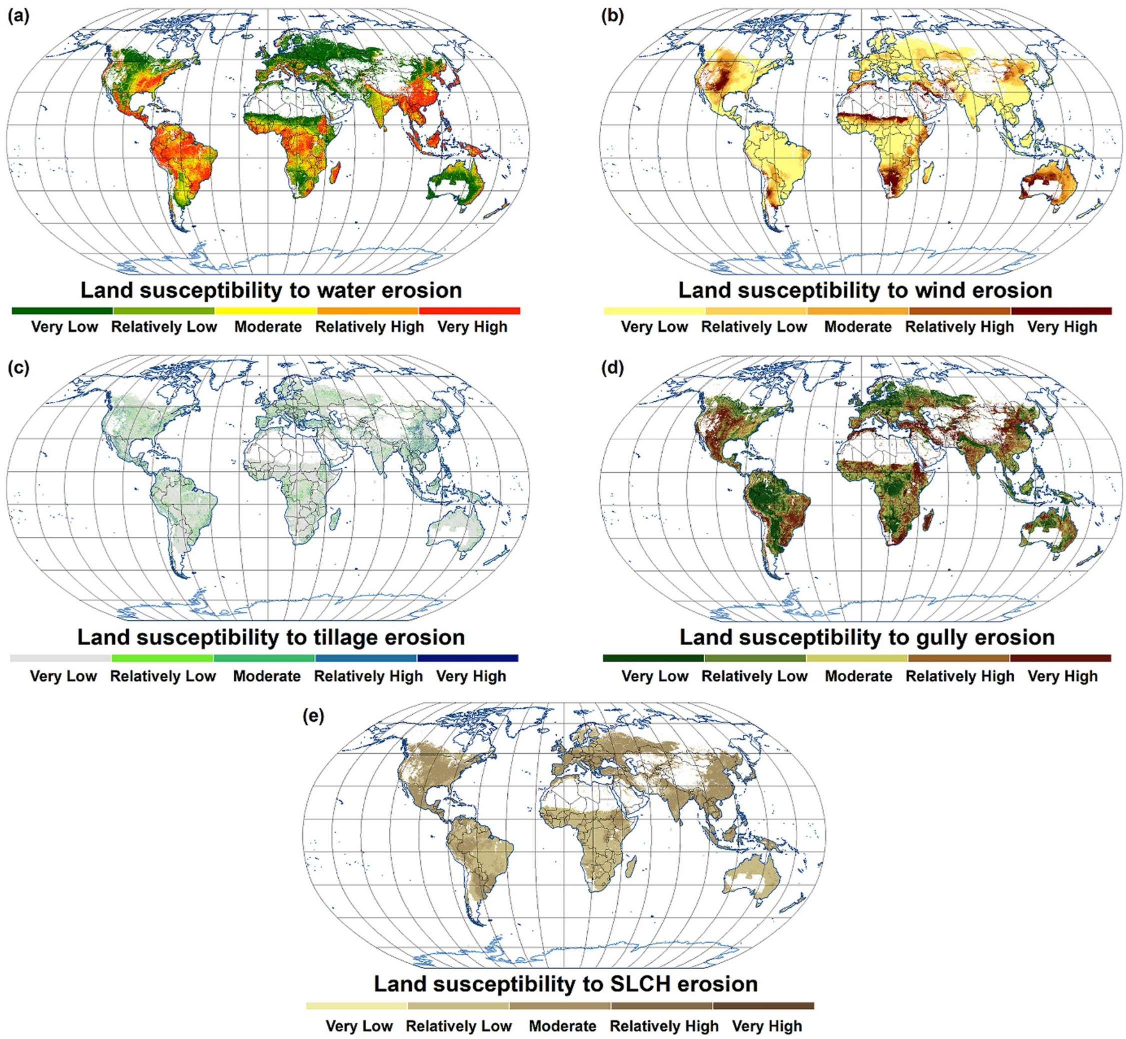


**Figure 11.** The modeled worldwide distribution of various soil erosion processes that lead to soil degradation (adapted from Borrelli et al., 2023). SLCH: Soil loss due to crop harvesting.

employing standardized methodologies, highlight considerable spatial variations in erosion rates (Borrelli et al., 2020), indicating that regions at lower latitudes are particularly vulnerable to erosion, especially in areas where agriculture is practiced on sloped terrain (Nearing et al., 2017). These long-term estimates are characterized by significant temporal variability, influenced by factors such as weather patterns, land management practices, and ecological disturbances, including wildfires and deforestation events (Kort et al., 1998; McGuire et al., 2024).

A critical factor in the quantification of large-scale water erosion is the processes that are taken into account. Linear (channelized) erosion features are the primary contributors to sediment production (Poesen et al., 2003), making their accurate prediction essential. Nevertheless, gully erosion has not yet been integrated into comprehensive quantitative evaluations of water erosion rates, even though it can lead to significant soil loss in specific areas, particularly in managed pasturelands, compared to rill and interrill erosion (Borrelli et al., 2023). Traditionally, interrill and rill erosion are characterized by small planar and linear geomorphic channels that can be altered through tillage, while gullies can reach greater depths (Merritt et al., 2003). In this context, innovative methods for predicting soil erosion that utilize machine learning classifiers and regression models to assess the presence of both ephemeral and permanent erosion features offer promising opportunities for large-scale soil erosion forecasting (Vanmaercke et al., 2021). Although quantifying erosion rates remains a complex task, the integration of rapid feature mapping with machine learning techniques marks a significant advancement in

10.1029/2025RG000883

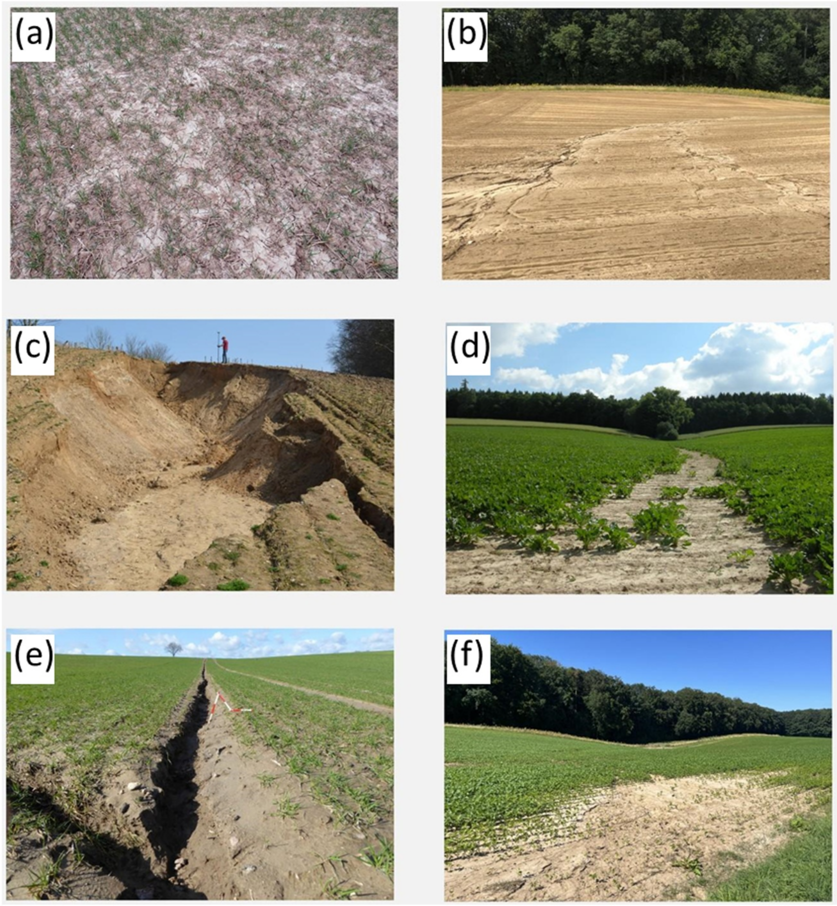


**Figure 12.** Photographic documentation of water erosion in agricultural landscapes: (a) interrill erosion leading to surface crust formation, (b) a network of rills along a drainage line prior to the emergence of crops, (c) a gully that has been deeply eroded following the harvest of maize, (d) damage to crop growth resulting from concentrated overland flow and significant soil erosion in a landscape depression (thalweg; Leibniz University of Hannover, 2021), (e) gully formation within a compacted wheel track (Leibniz University of Hannover, 2021), and (f) diminished crop growth due to the in-field accumulation of eroded sediment.

erosion research, enhancing our understanding of the conditions and trends associated with severe erosion types (De Geeter et al., 2023).

One major contributing factor of water erosion processes is soil compaction, which occurs during field traffic activity by machinery (e.g., tillage, spraying, harvest; Augustin et al., 2020; Batey, 2009). High degrees of top- and subsoil compaction by repetitive machinery passes in headlands and tramlines lead to an increased frequency and extent of soil erosion events (Evans, 2017; Prasuhn, 2020; Steinhoff-Knopp & Burkhard, 2018). The main reason for the increase in soil loss is the increasing bulk density (Saggau et al., 2022) indicating reduced pore volume and subsurface connectivity finally reducing topsoil infiltration rate and capacity (Batey, 2009; Ryken et al., 2018). This promotes accelerated generation of surface runoff during storm events and soil erosion significantly increasing phosphorus-mobilization in compacted areas (Remund et al., 2021; Withers et al., 2006). The orientation of headlands and tramlines not only influences the rates of surface runoff and soil erosion but also their connectivity within the landscape and therefore diffuse pollution (Saggau et al., 2023). Despite the importance of soil compaction for soil erosion and sediment transport, it is rarely accounted for in soil erosion models (e.g., the (R)USLE) and spatial soil erosion risk assessments (Saggau et al., 2022).

*Wind (aeolian) Erosion.* Aeolian erosion refers to the movement of soil particles through processes such as suspension (particles smaller than 0.1 mm), saltation (particles ranging from 0.1 to 0.5 mm), and creep (particles between 0.5 and 2 mm), driven by the shear force of wind acting on soil with low cohesion (Xu et al., 2024). This phenomenon is a natural component of the global dust cycle, predominantly occurring in semi-arid and arid regions (Ravi et al., 2011), where limited soil moisture and sparse vegetation leave soils vulnerable to wind action. Wind erosion significantly influences Earth systems across various time scales, with notable effects on human activities. For instance, the aeolian transport and deposition of loess in Europe, the United States, and China during past geological epochs have profoundly affected the current distribution of fertile soils essential for agriculture (X. Li et al., 2020; Y. Li et al., 2020). Unlike water erosion, wind erosion is more prevalent in flat terrains, making it a crucial factor in the overall geographic distribution of soil erosion worldwide (Borrelli et al., 2023). The consequences of wind erosion extend beyond soil degradation, impacting air quality and public health through the release of particulate matter (PM 2.5 and PM 10) into the atmosphere (Griffin et al., 2001), exacerbating desertification by diminishing the functional capacity of soils in semi-arid regions (Ravi et al., 2010), and disrupting natural ecological cycles due to the influx of dust and nutrients (Shao et al., 2011).

The process of wind erosion is influenced by several factors (Borrelli et al., 2017b), including the force of the wind necessary to initiate the movement of particles, which is contingent upon grain size, soil cohesion characteristics, vegetation cover, and root cohesion. Consequently, the susceptibility of soil to wind erosion is governed by essential geoecological factors. As wind flows, it generates shearing, uplift, and turbulent forces that can lift particles to varying heights above the ground (Jarrah et al., 2020). The transport of grains occurs through mechanisms such as creep, saltation, or suspension, depending on their elevation within the air column (Jarrah et al., 2020). While suspension is generally restricted to finer particles, it is the transport mode that allows for the longest residence times in the atmosphere and, consequently, the greatest travel distances (Ravi et al., 2010). Deposition of these particles takes place when there is a decrease in wind speed, which can result from diminished atmospheric pressure gradients or the presence of obstacles like topography, vegetation, or man-made structures.

Globally, wind erosion is estimated to be 312.5 Pg $yr^{-1}$ (Sun et al., 2024), with a significant portion originating from desert regions. The primary sources of dust emissions worldwide are found in drylands and grasslands located in (semi-)arid zones and continental interiors across Africa, Asia, North America, and South America (Ravi et al., 2011; Schepanski, 2018). While climatic factors heavily influence the global dust source areas, human activities significantly affect wind erosion rates, often surpassing natural fluctuations (Nordstrom & Hotta, 2004). For instance, the Great Plains of the United States are inherently vulnerable to wind erosion, experiencing up to 48 days of dust storms annually. However, the events of the 1920s and 1930s, which led to the infamous "Dust Bowl," exemplified the consequences of poor soil management practices (Nordstrom & Hotta, 2004). The transition from natural vegetation to heavily farmed croplands, along with the advent of agricultural mechanization before the Dust Bowl, were pivotal factors in this environmental crisis. Additional contributors to heightened wind erosion include severe or prolonged overgrazing, the removal of vegetative barriers such as hedgerows and trees, the expansion of field sizes, deforestation, and wildfires (Ravi et al., 2011).

*Tillage Erosion.* Tillage erosion refers to the movement of soil caused by repeated tillage activities, such as plowing. This phenomenon is characterized by soil displacement at the edges of fields and the downward gravitational flow of soil due to alterations in slope gradients (van Oost et al., 2000). Tillage methods significantly influence the soil's vulnerability to both wind and water erosion, while simultaneously serving as a mechanism of erosion, with the extent of erosion being contingent upon the frequency and intensity of these practices within agricultural management (van Oost et al., 2000). The immediate effects include the removal of topsoil and the mixing of nutrient- and soil organic carbon (SOC)-depleted subsoil into the upper cultivation layer, resulting in shallow, poorly structured, and nutrient-deficient topsoils. Conversely, in areas of net deposition, such as depressions, soils tend to thicken as they accumulate finer particles that are rich in nutrients and SOC (van Oost et al., 2006).

The mechanization of agriculture, which began to accelerate in the 1950s, has significantly exacerbated tillage erosion. This increase in tracking power, tillage speed, and depth has resulted in tillage erosion rates comparable to those of sheet and rill erosion, with some regions experiencing even higher rates (Poesen, 2018; van Oost et al., 2005). In the global agricultural context, tillage erosion is estimated to account for one-fifth of water erosion and is twice as prevalent as wind erosion (Quinton et al., 2010). In mechanized agricultural systems, tillage erosion rates generally fall between 3 and 70 Mg $ha^{-1}$ $yr^{-1}$, although under certain conditions, they can escalate to as much as 250 Mg $ha^{-1}$ $yr^{-1}$. While less intensive, non-mechanized tillage methods, whether powered by animals or humans, can also lead to significant tillage erosion, with recorded rates ranging from 3 to 600 Mg $ha^{-1}$ $yr^{-1}$, particularly in developing nations (Van Oost et al., 2006). The extent of tillage erosion is influenced by topographical features such as curvature and slope gradient, as well as the soil's physical properties, including water content, texture, and bulk density. Furthermore, factors such as the type of tillage implement (e.g., moldboard plow), its characteristics (e.g., shape, width, and length), management practices (e.g., tillage depth, speed, and direction), and field attributes like shape and size also play a crucial role in determining tillage erosivity (Öttl et al., 2022; van Muysen et al., 2002).

Tillage erosion predominantly occurs in convex upslope areas, such as crests and shoulder slopes, where soil is tilled in an up-and-down manner. In contrast, soil tends to accumulate in concave slopes (Govers et al., 1994; Lobb et al., 1995). This process not only smooths the landscape but also contributes material to topographic features like thalwegs, which are subject to intermittent water erosion, thereby creating a synergistic effect that enhances the lateral redistribution of soil (Quinton & Fiener, 2024). Consequently, tillage erosion plays a significant role in biogeochemical cycles, including the carbon cycle (Juřicová et al., 2025; Öttl et al., 2024; Van Oost et al., 2007). Effective strategies to mitigate tillage erosion are limited and primarily involve practices such as contour tillage and no-till methods, which minimize soil disturbance and translocation while promoting better soil structure (Van Oost et al., 2006). Contrary to popular belief, non-inversion tillage can result in significantly greater tillage erosion compared to moldboard plow tillage, with increases ranging from 130% to 210% (Öttl et al., 2022).

*Soil loss due to crop harvesting (SLCH).* Soil loss associated with crop harvesting (SLCH), also referred to as harvest erosion, pertains to the movement of topsoil from agricultural land during the collection of root and tuber crops (Kuhwald et al., 2022; Ruysschaert et al., 2004). This phenomenon occurs when crops are uprooted, bringing along loose soil, soil clumps, and rock fragments. Although a minor portion of the material clinging to the crops is redistributed back onto the field, the majority of the soil is removed with the harvested crops, resulting in a net loss of soil (Parlak & Blanco-Canqui, 2015; Ruysschaert et al., 2004). SLCH affects the entire field during harvest across various topographies and soil types, although there can be significant spatial and temporal fluctuations in SLCH rates (Saggau et al., 2024). The crops impacted by this process include groundnuts, cassava, potatoes, yams, sweet potatoes, and sugar beets. In 2019, around 1.1 million $km^2$ of crops susceptible to SLCH were grown, representing 8.4% of the global arable land, with the largest proportions found in Africa, Asia, and Europe (Kuhwald et al., 2022). In the European Union, the SLCH on average removes approximately 14.7 million tonnes per year per harvest for the 4.2 million hectares where potatoes and sugar beets are cultivated (Panagos et al., 2019). The loss of soil primarily affects the fertile topsoil, leading to soil thinning and a reduction in soil organic carbon (SOC) and nutrients (Poesen et al., 2001). This decline in soil fertility can result in increased fertilizer costs and higher production expenses (Parlak et al., 2022; Saggau et al., 2024). Current research indicates that average SLCH rates typically range from 0.008 Mg $ha^{-1}$ per harvest to 22 Mg $ha^{-1}$ per harvest (Parlak et al., 2016), with individual events potentially reaching up to 70 Mg $ha^{-1}$ per harvest (Kuhwald et al., 2022; Poesen et al., 2001). Consequently, SLCH rates are comparable to those of other soil erosion processes and are particularly significant in flat landscapes. Factors influencing SLCH rates include soil characteristics (such as water content and texture), crop attributes (including morphology and size), management practices (crop rotation and plant density), harvesting methods (type of machinery, depth, and speed), and post-harvest handling (Ruysschaert et al., 2007a, 2007b).

SLCH is influenced by the harvesting technique employed, which differs according to the type of crop and the degree of mechanization on the farm. In highly mechanized agricultural environments, self-propelled harvesters are commonly utilized and are fitted with mechanisms that clean the crops post-harvest, thus minimizing net soil losses. Conversely, in developing nations where manual harvesting is the norm, such cleaning technologies are often absent. Nonetheless, due to various uncertainties, it is not definitively established whether manual or mechanized harvesting results in greater soil loss (Li et al., 2006; Parlak et al., 2016). For numerous crops, including groundnuts, onions, asparagus, ginger, leeks, and yautia, there is a lack of data regarding SLCH rates. Additional uncertainties pertain to the impact of different soil tillage practices (such as inversion and no-till), crop rotation (including cover crops), topographical variations, and the subsequent fate of soil and its constituents (such as nutrients and pesticides) after they leave the field (Kuhwald et al., 2022). Given these circumstances, strategies to mitigate SLCH are limited, with one effective approach being the utilization of technological advancements in the cleaning systems of contemporary harvesters (Ruysschaert et al., 2007b). Since SLCH significantly escalates with increased soil moisture, it is crucial to avoid harvesting during wet soil conditions (e.g., Parlak et al., 2021; Saggau et al., 2024).

10.1029/2025RG000883

### 3.2. Chemical Processes

#### 3.2.1. pH, Acidification and Alkalization

Soil pH is fundamentally important because it controls the availability of nutrients to plants and influences soil biology. It is a measure of how acidic or alkaline the soil is; natural pH values range from 2 to 3 (very acidic) to 11–12 (very alkaline) with an optimal value of 6–7 for cultivated land (Slessarev et al., 2016). Healthy soils contain minerals that function as buffers, helping to neutralize inputs that could lead to significant pH fluctuations. However, when these inputs are excessive and deplete the soil's buffering capacity, the balance of the soil is disrupted, challenging the stability and resilience of the ecosystem (D'Odorico et al., 2013). Processes such as acidification and alkalinization contribute to soil degradation, resulting in a lasting decline in ecosystem services essential for sustaining life (Smith et al., 2024). This discussion will briefly outline some of the factors driving these processes and their impacts on soil health.

Soil acidification refers to the reduction of soil pH below 7, resulting from both natural processes and human activities. Typically, natural soil acidification occurs gradually, whereas human-induced acidification can manifest rapidly due to the significant pressures exerted on ecosystems. A notable example of anthropogenic influence is the pollution caused by acid rain, which began with the Industrial Revolution in Europe in the late 18th century and escalated into a significant environmental concern in North America and Europe during the 1950s–1970s (Grennfelt et al., 2020). The implementation of regulatory measures successfully reduced emissions of sulfur dioxide and nitrogen oxides, leading to observable improvements in soil pH levels within a decade (Kirk et al., 2010; Reynolds et al., 2013). However, recent studies suggest that this recovery has either slowed or reversed (Seaton et al., 2023). The impact of acidification from coal combustion remains a critical issue in China, India, and other industrialized countries (Bouwman et al., 2002). The transboundary nature of this pollution, along with its long-term effects on mineral weathering rates, global element cycling, and the degradation of both natural and agricultural ecosystems, positions acid rain as one of the most pressing environmental challenges linked to air pollution (Prakash et al., 2023; Singh & Agrawal, 2008). Additional human activities contributing to soil acidification include the excessive use of ammonium-based fertilizers (Arias-Navarro et al., 2024), intensive agricultural practices (Che et al., 2023), overgrazing, deforestation, and the irrigation of crops with acidic water (Goulding, 2016).

Acidification impacts all types of soils, regardless of whether they are initially acidic, neutral, or alkaline. However, soils that are already acidic are particularly vulnerable to immediate degradation from acidification. Acidic soils (pH < 5.5) possess a buffering capacity that helps them resist significant pH fluctuations, which is largely influenced by the presence of gibbsite ($Al(OH)_3$), exchangeable aluminum, and organic acids that help maintain pH levels between 4 and 6 (van Breemen et al., 1983). These acid soils are typically located in regions where precipitation surpasses evapotranspiration, predominantly in humid northern temperate zones and tropical areas with high rainfall. They account for 50% of the world's arable land, with 67% of this land supporting forests and woodlands (von Uexküll & Mutert, 1995). Nevertheless, global climate change, which alters the soil's water balance, can lead to the dissolution of carbonate minerals in alkaline soils, thereby reducing their buffering capacity and pushing the pH into the acidic range (Slessarev et al., 2016). Additional natural processes contributing to acidification include the leaching of basic cations, the decomposition of organic matter, root and microbial respiration, the nitrification of ammonium, and the oxidation of sulfides (Che et al., 2023).

Soil acidification can lead to several adverse effects, including nutrient deficiencies, the release of toxic metals, alterations in microbial communities, and degradation of soil structure. The leaching of aluminum and heavy metals into the soil solution can reach toxic concentrations for both plants and microorganisms, thereby diminishing soil fertility, affecting primary productivity, and disrupting microbial carbon cycling processes (Malik et al., 2018; Singh & Agrawal, 2008). While there are reclamation strategies available to restore and sustain soil pH levels, a comprehensive global approach is essential to tackle this issue, as it extends beyond the scope of local policy measures.

Soil alkalinization occurs when the pH level of the soil increases. This phenomenon is often due to the accumulation of basic compounds, such as calcium carbonate, which can elevate the soil's pH beyond 7. Alkaline soils (pH > 7) are predominantly located in arid and semiarid areas where evapotranspiration surpasses precipitation. Their global distribution follows a pattern influenced by climatic and geochemical factors (Jenny, 1994). In these regions, limited leaching leads to the prevalence of carbonates in the soil composition. Under atmospheric conditions, the theoretical pH of a solution containing calcite crystals is approximately 8.3 (Stumm & Morgan, 1996). It is generally accepted that soils with a pH ranging from 8.1 to 8.5 are buffered by calcite, maintaining this pH as long as calcite is present (Slessarev et al., 2016). In such environments, soils typically exhibit good structure due to the calcium released from calcite. Further alkalinization may occur when the combination of calcite and high concentration of sodic salts are present in the soil. The presence of sodium-saturated clays and the formation of sodium bicarbonates in the soil solution can raise the pH to levels as high as 11 (Smith et al., 2024). Alkalinization adversely affects soil health by reducing organic carbon, nutrient availability, and bacterial diversity, while also critically impacting soil structure, as soil colloids carry a negative charge. When repulsive forces dominate, soil particles can be easily displaced either downward in the profile or laterally through runoff, leading to compaction and erosion (Lebron et al., 1994). To mitigate these effects, soil amendments such as sulfur, organic matter, or gypsum (calcium sulfate) are effective in lowering pH, enhancing soil structure, and alleviating the impacts of alkalinity, particularly in sodic soils where sodium is the primary contributor to alkalinity (Oster & Frenkel, 1980).

The process of salinization (Shokri et al., 2024) and alkalinization serve as indicators of land degradation, which can ultimately lead to desertification, which is characterized as a "persistent loss of ecosystem services essential for sustaining life." For additional insights into the drivers and feedback mechanisms associated with global desertification, see D'Odorico et al. (2013). Given that drylands are home to over 2 billion individuals, with 25% of these areas experiencing desertification, the environmental and societal implications of these processes are significant. Moreover, projections from future climate change models anticipate an acceleration in the spread of desertification globally. Therefore, addressing and reversing land degradation should be prioritized to mitigate food insecurity and poverty (MEA, 2005).

### 3.2.2. Nutrient Depletion and Surplus

Soil fertility decline is fundamentally a gradual phenomenon characterized by the loss of vital soil nutrients through mechanisms such as harvesting, leaching, erosion, and other processes, occurring at a rate that exceeds their replenishment from both organic and inorganic sources (Andersson et al., 2011). In forest ecosystems, this decline is also influenced by natural weathering and liming. The reduction of nutrient cations, often referred to as base cations, is intricately linked to acidification processes. When plants absorb nutrient ions, they must maintain a charge balance, which involves the release of $H^+$ ions during cation uptake or $OH^-$ ions during anion uptake. As plants develop, they initially acidify the rhizosphere by selectively excreting $H^+$ ions to facilitate the uptake of nutrient cations. If these cations are not reintroduced into the soil, through methods such as fertilization or liming, following the harvest of plant biomass, the soil will experience a persistent state of acidification.

In contemporary society, the phenomenon of nutrient depletion in agricultural soils is infrequent in industrialized nations and even in some developing countries, where the use of fertilizers, and often excessive application, is prevalent. Conversely, soils in Eastern Europe, Russia, and Africa have been reported to experience ongoing depletion over recent decades, particularly concerning phosphorus levels (Chianu et al., 2012; Kalinina et al., 2009; West et al., 2014). The decline in soil fertility is a contentious topic that has garnered significant attention in discussions surrounding land degradation in Africa, posing a critical challenge for numerous farmers in sub-Saharan Africa and exacerbating issues of rural poverty and food insecurity (Andersson et al., 2011; Lal, 2009). In contrast, emerging economies, particularly China, have implemented proactive interventions and have achieved greater success in combating nutrient depletion, albeit often resulting in prolonged periods of overfertilization that lead to substantial surpluses of nitrogen and phosphorus (Andersson et al., 2011; West et al., 2014).

Depletion of soil nutrients is a threat to smallholder farmers across various regions globally. This phenomenon directly undermines agricultural productivity, food security, and rural livelihoods, while also being linked to broader environmental issues. Despite a comprehensive scientific understanding of the problem, there remains a notable disconnect between this knowledge and the implementation of effective policy measures (Chianu et al., 2012; Lal, 2009). In many developing nations, agricultural productivity is either stagnating or declining due to ongoing soil degradation, insufficient investment in management practices and institutional frameworks, and climate-related challenges (Bedeke, 2023).

In forest ecosystems, the depletion of base cations is frequently associated with soil acidification. Magnesium deficiency can result in significant needle yellowing, altered branching patterns, and an acceleration of podsolization processes. Research has indicated that nutrient deficiencies in forests correlate with the levels of exchangeable magnesium in the soil (Ende & Evers, 1997) and the concentration of magnesium in soil solutions. The ratios of calcium to aluminum and magnesium to aluminum in soil solutions or exchangeable soil fractions are commonly utilized as indicators of the harmful effects of soil acidification on trees, as well as for determining critical loads for acid deposition (Blaser et al., 1999; Løkke et al., 1996).

#### 3.2.2.1. Nutrient Depletion

Nutrient depletion is a significant outcome of soil degradation and has a negative effect on soil fertility (Prăvălie, 2021). The extent of this issue is influenced by various factors, including land use, management strategies, climatic conditions, and topographical features. Research indicates that forest soils experience less degradation thus less nutrient loss compared to grasslands and agricultural areas, attributed to their dense vegetation and porous structure (Galindo et al., 2022; Milazzo et al., 2023). Nonetheless, Sun et al. (2023) discovered that the extent of nutrient loss in forest soils is contingent upon the age of the forest. Their findings revealed that soil erosion rates, along with losses of total organic carbon (TOC) and total phosphorus (TP), diminished as forest age increased, whereas total nitrogen (TN) loss initially rose before subsequently decreasing. Additionally, forest fires represent a significant factor in nutrient loss within forest ecosystems, acting as a form of land degradation (Jhariya & Singh, 2021). Agbeshie et al. (2022) noted that low-intensity fires can enhance nutrient availability and soil pH through the deposition of ash, whereas high-intensity fires inflict considerable harm, leading to nutrient volatilization, diminished soil aggregate stability, increased bulk density, hydrophobicity, and erosion, as well as the destruction of soil biota. Changes in hydrophobicity influence the subsequent flow and transport processes in soil (Sepehrnia et al., 2024; Shokri et al., 2009, 2012). Li et al. (2020) reported that the severity of fires elevates soil pH by incinerating organic matter and leaching essential base elements, while simultaneously decreasing total organic carbon, nitrogen, and phosphorus levels. Severe burns can result in prolonged nutrient loss, with no signs of recovery even after a period of 7 years.

Grassland degradation is widespread and escalating in numerous regions across the globe (Gibbs & Salmon, 2015), with nearly 49% of the world's grassland area experiencing varying degrees of degradation (Gang et al., 2014). The primary drivers of this phenomenon are overgrazing, attributed to human activities, and soil erosion, which is influenced by climate change, both of which result in considerable nutrient depletion (Nie et al., 2013). Research by Zhang et al. (2022) indicated that intense grazing diminishes soil nitrogen, potassium, and the activities of enzymes such as urease and sucrase at depths of 0–30 cm. Furthermore, heavy grazing compromises the water stability and erosion resistance of soil aggregates, as the stability of these aggregates is contingent upon nutrient availability. The consequences of overgrazing include nutrient depletion, alterations in soil microbial communities, a slowdown in the formation of new aggregates, and a disruption of the equilibrium between the processes of aggregate formation and degradation, ultimately leading to the deterioration of soil aggregates (Ren et al., 2022). While soil erosion also plays a role in nutrient loss within grasslands, its impact is less pronounced than in agricultural lands (Milazzo et al., 2023), particularly in sloped terrains. Nie et al. (2013) noted that erosion affects soil organic carbon (SOC), total nitrogen (TN), and total phosphorus (TP) more severely in slopes characterized by low vegetation cover and heavy grazing, compared to those with denser vegetation and minimal grazing. Likewise, L. Sun et al. (2022), F. Sun et al. (2022) reported that gully erosion in alpine steppe regions leads to significant soil nutrient depletion.

Phosphorus (P) is an essential element for life, significantly contributing to growth, metabolic processes, and reproduction. However, it is also a finite resource on Earth (Figure 13). In natural ecosystems, the loss of P through soil-plant interactions is gradually compensated by the weathering of rocks (Bouwman et al., 2009). In contrast, agricultural systems that are managed by humans depend heavily on the application of fertilizers. As discussed above (Section 3.2.2.), effective management of P is crucial to alleviate the adverse effects of over-fertilization of P and safeguard freshwater and marine environments from hypoxic conditions. On the other hand, P mining, characterized by the extraction of P at rates that exceed natural replenishment, leads to a decline in soil fertility and diminishes agricultural yield potential. While natural ecosystems rely on the slow cycling of P through weathering processes to maintain growth (Izquierdo et al., 2013; Walker & Syers, 1976), sustained P mining in agricultural contexts can exhaust soil reserves. This depletion can compromise soil structure, increase erosion, and result in widespread nutrient loss, ultimately jeopardizing soil health and agricultural productivity.

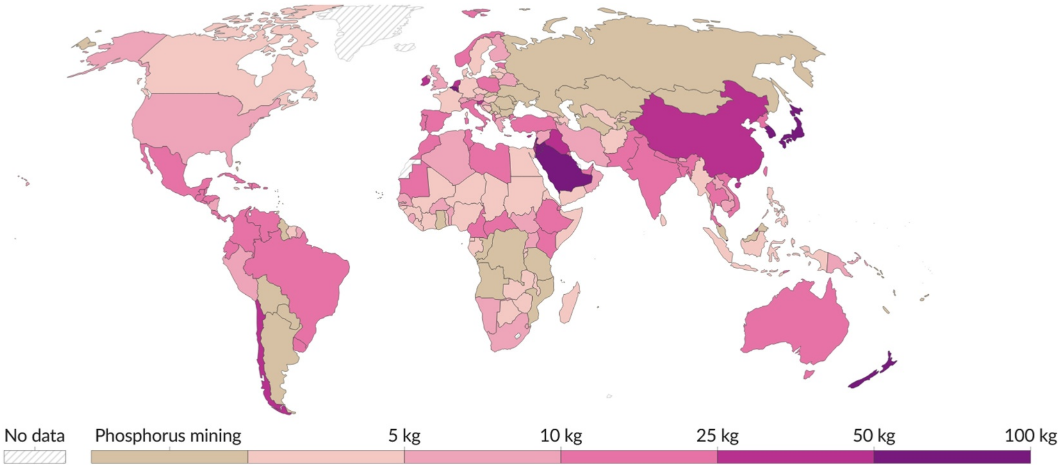


**Figure 13.** Amount of excess phosphorus per hectare of cropland. This is the difference between phosphorus inputs, and the amount harvested in crops (after West et al. (2014) with the data processed by Our World in Data).

All types of land use and land cover experience varying degrees of degradation; however, cropland is particularly susceptible when compared to forests and grasslands (Borrelli et al., 2017a). The primary factors influencing 40% and 20% of global arable land are aridity and soil erosion, respectively (Prăvălie, Patriche, et al., 2021), which result in significant nutrient depletion. A study by Moreno-Jiménez et al. (2019) examined soil micronutrients (Cu, Fe, Mn, Zn) across 143 dryland regions globally and discovered that aridity diminishes their availability by elevating soil pH and reducing organic matter content. The ratio of Fe to Zn exhibited an exponential decline with increasing aridity, suggesting a stoichiometric imbalance. Comparable findings were reported in European drylands, where aridity led to decreased Fe availability and subsequent deficiencies in Zn and Mo (after Moreno-Jiménez et al., 2022).

Another significant issue is the transformation of grassland or forest areas into cropland, which contributes to nutrient depletion. Zhang et al. (2021) employed the EPIC model to assess the impacts of converting over 2 million hectares of grassland to corn and soybeans across 12 Midwestern states in the United States from 2008 to 2016. This conversion resulted in a 7.9% increase in annual soil erosion, a 3.7% loss of nitrogen, and a 5.6% reduction in soil organic carbon. In a related study, Zhang et al. (2020) found that converting forest land to tea plantations in sloped areas led to considerable nutrient loss, with total nitrogen (TN) in runoff increasing fivefold following the removal of vegetation.

#### *3.2.2.2. Nitrogen and Phosphorus Surplus*

The linear progression of P, from mineral deposits to soils, freshwater bodies, and ultimately to oceans, has surpassed the thresholds deemed safe for sustainable development already in many parts of the world (Carpenter & Bennett, 2011). An overabundance of P can result in its accumulation in soils, which may subsequently be lost primarily through erosion (Alewell et al., 2020). This accumulation poses significant ecological risks, particularly through the process of eutrophication in nearby ecosystems.

Excessive nitrogen input has significant implications for various soil processes, including eutrophication and acidification (de Vries, 2021; Nellemann & Thomsen, 2001), nitrogen leaching (Dise & Wright, 1995; Thimonier et al., 2010), nutrient cycling, and litter decomposition (Aber et al., 1998; Knorr et al., 2005). Numerous studies indicate that chronic nitrogen input leads to alterations in the composition of soil microbial communities, often resulting in decreased activity of phenol oxidase, an enzyme produced by white-rot fungi responsible for lignin degradation (Baldrian et al., 2023; Carreiro et al., 2000). In agro-ecosystems, the nitrogen cycle is characterized by a fragile equilibrium between inputs, such as fertilization, atmospheric deposition, and symbiotic fixation, and outputs, which occur through crop harvest and leaching into deeper soil layers and aquatic systems (Shojaei et al., 2022). Effective management strategies aimed at enhancing Nitrogen Use Efficiency (NUE) are essential to prevent both over-fertilization and nutrient depletion (Oenema et al., 2016). This management is crucial not only for ensuring food security but also for safeguarding ecosystems and mitigating eutrophication in freshwater and marine environments (Shojaei et al., 2022). With the pressures of population growth and limited resources, many arable soils in Europe are subjected to intensive exploitation. The dependence on fertilizers to maintain sufficient nitrogen levels often undermines the role of soil fauna in nutrient mineralization, leading to a marked decline in soil biodiversity (Nabel et al., 2021) and the provision of ecosystem services (de Souza & Freitas, 2018). Furthermore, excessive nitrogen inputs can modify the physico-chemical properties of soil, particularly through acidification, which disrupts the balance between fungal and bacterial communities and further impacts nutrient cycling (Nabel et al., 2021). These disruptions may lead to decreased crop yields by increasing the likelihood of cereal lodging (Dahiya et al., 2018) and enhancing vulnerability to pests (Altieri & Nicholls, 2003).

Nitrogen surplus also affects forest ecosystems, altering tree nutrition (Braun et al., 2010; Waldner et al., 2015), physiology, phenology, and root systems (Braun et al., 2005). These changes weaken trees' resilience to climatic stresses and pests (Bobbink & Hettelingh, 2011). Biodiversity is impacted as nitrophilous plant species proliferate under the canopy (Bobbink et al., 2010; Pitcairn et al., 1998; Talhelm et al., 2013; Xie, Duan, et al., 2024; Xie, Ge, et al., 2024). Additionally, shifts in soil nematode (Eisenhauer et al., 2012) and microbial communities, including fungi, have been documented (Carreiro et al., 2000; Suz et al., 2021; Treseder, 2008). Over time, nitrogen saturation can lead to leaching, resulting in eutrophication and acidification in adjacent ecosystems, including aquifers, rivers, and oceans (Aber et al., 1998; Rothwell et al., 2008).

In grasslands, nitrogen surplus leads to detrimental changes such as soil acidification, eutrophication (Horswill et al., 2008), and alterations in phosphorus (Johnson et al., 1999) and carbon cycles (Stiles et al., 2017; Wedin & Tilman, 1996). While nitrogen input typically reduces species richness, it can locally increase it in nutrient-constrained ecosystems (e.g., alpine meadows). However, this often results in the displacement of adapted species by less specialized, competitive ones, leading to homogenization and community simplification (Bobbink et al., 2022). Over time, nitrogen surplus reduces seed bank diversity, affecting ecosystem regeneration and genetic diversity (Basto et al., 2015; Phoenix et al., 2012). Additionally, increased shoot-to-root ratios (Bobbink, 1991; Brouwer, 1962) and heightened lodging risk (Lillak, 2005) weaken ecosystem resilience.

Wetland soils, which sustain nutrient-deficient ecosystems characterized by distinct biodiversity, are especially susceptible to excess nitrogen (Bobbink et al., 2010; Phoenix et al., 2012). The rise in nitrogen deposition poses a threat to their structural integrity and functional capabilities, potentially converting carbon-sequestering wetlands into sources of carbon emissions (Aerts et al., 1992). Consequences of this phenomenon include diminished species diversity (Van Geel et al., 2020), increased nitrate leaching, a decline in Sphagnum cover, and a transition toward vascular plant dominance (Aerts et al., 1992; Phoenix et al., 2012). Furthermore, nitrogen surplus contributes to heightened greenhouse gas emissions (Wiedermann et al., 2007), soil acidification (Bobbink et al., 2010), and the accumulation of toxic metals, which in turn heightens susceptibility to pathogens (Limpens et al., 2004; Sheppard et al., 2009).

### 3.2.3. Soil Organic Matter

The decline in soil organic matter (SOM), frequently evaluated through soil organic carbon (SOC) metrics, arises mainly from either diminished biomass contributions to the soil or an accelerated decomposition rate of SOM. According to Smith et al. (2024), SOC loss can manifest in both mineral and organic soils, including peatlands. The primary factors contributing to SOC depletion include changes in land use, disturbances to vegetation, frequent tillage practices, the removal of crop residues, and the drainage of peatlands. While human activities are the predominant catalysts for SOC loss, environmental factors also significantly influence this process (Smith et al., 2024). Smith et al. (2024) indicate that the average SOC in India has declined from 1% to 0.3% over the past 70 years, with the Mediterranean region of Europe being particularly susceptible to SOM depletion and soil degradation. Since the advent of agriculture, mineral soils have experienced a loss of approximately 133 Pg of SOC, with a notable acceleration in this rate over the last two centuries. The most significant soil carbon losses have occurred in key agricultural zones and degraded grazing lands, each accounting for about half of the historical reductions (Sandermann et al., 2017). Furthermore, as reported by Smith et al. (2024), around 12% of global peatlands have been drained and degraded, leading to the release of substantial amounts of carbon. Consequently, these degraded peatlands are responsible for approximately 4% (equivalent to 2,000 million tonnes of carbon dioxide) of annual global anthropogenic emissions (UNEP, 2022). Each year, human activities result in the destruction of half a million hectares of peatlands. The primary threats to these ecosystems include agricultural drainage, livestock overgrazing, peat extraction for energy and horticultural purposes, infrastructure development (such as oil and gas exploitation), intentional burning, mining activities in peatland regions, and urban expansion (UNEP, 2022).

Soil salinization is among other environmental stressors of SOC. While salinity's impact on SOC can be complex, evidence suggests a predominantly negative correlation, with higher salinity levels accelerating SOC loss (Hassani et al., 2024). This relationship is particularly significant in the context of land degradation, where salinity-induced SOC depletion can further compromise soil fertility, carbon sequestration potential, and overall ecosystem stability. Based on an analysis of 43,459 mineral soil samples (SOC < 150 g $kg^{-1}$) collected across various land cover types since 1992 (Figure 14a), Hassani et al. (2024) showed that increasing soil salinity from 1 to 5 dS $m^{-1}$ in croplands reduces SOC in mineral soils (SOC < 150 g $kg^{-1}$) from 0.14 g $kg^{-1}$ above the mean predicted SOC (18.47 g $kg^{-1}$) to 0.46 g $kg^{-1}$ below it. While soil salinity explains less than 6% of SOC variability (Figure 14b), a one-standard deviation increase in topsoil salinity (0–7 cm) is associated with SOC reductions of approximately 4.4% and 9.26% in croplands and non-croplands, respectively. The most significant SOC depletion in croplands occurs in vegetation/cropland mosaics, whereas non-croplands dominated by evergreen needle-leaved trees exhibit the highest SOC losses (Hassani et al., 2024). Among all factors analyzed, soil nitrogen, precipitation Seasonality Index, and land cover were found to be the most significant factors in explaining variations in SOC content (Figure 14b).

There are important process interactions between erosion and SOC, which cause disaggregation, dislocation, transport, deposition, burial, and export of particulate matter to riverine systems. Erosion perturbs the SOC pool over different temporal (geological to decadal) and spatial (slope segment to hillslope) scales, which are important to understand for the carbon cycle (Wang et al., 2017; Zheng et al., 2025). A portion of the globally eroded SOC (0.65–3.70 Pg C $yr^{-1}$) is delivered to riverine and aquatic systems, encountering numerous decomposition and long-term burial processes outside of soil systems (Zheng et al., 2025). Debate on whether soil erosion contributes to a net source or sink of carbon to the atmosphere typifies the complexity of erosion-SOC interactions (Sanderman & Berhe, 2017; Van Oost et al., 2007). Scale-dependent emergent properties suggest that sampling hillslope segments experiencing net erosion creates sampling and quantification bias in favor of net source environments (i.e., where transport and enhanced decomposition exceed dynamic replacement), omitting the role of downslope net deposition and long-term burial of SOC, which is considered to tip the global balance of erosion to a net SOC sink of 0.47–0.61 Pg C $year^{-1}$ (Van Oost et al., 2007). Nevertheless, numerous uncertainties propagate across the process domains and impact quantifications, such as: (a) spatial biases in the erosion rates due to land use change, unconsidered processes and their global prevalence (e.g., Figure 11) in modeled estimates (e.g., the omission of gully and permafrost erosion), (b) the stability of buried SOC following lateral transport by erosion, and (c) the effect of erosion and transport processes on SOC stability in terrestrial and aquatic systems.

In a more general perspective, SOC is a key environmental indicator for soil fertility and agronomic productivity (Zomer et al., 2017), climate stability (Jungkunst et al., 2022) and food security (Ma et al., 2023), thus having multidimensional global implications in the diagnosis of several major environmental issues, such as land degradation, climate change, or the food crisis (Prăvălie, Nita, et al., 2021). Consequently, global or continental data on soil carbon dynamics are crucial for the overall understanding of SOC-related environmental disturbances.

However, there are very few large-scale (global) studies on SOC changes and their environmental stressors that act as driving forces for soil carbon losses. One such global study, focused on recent spatio-temporal trends in SOC (in the 30-cm deep profile), revealed remarkable changes across the planet during the 2001–2015 period (Prăvălie, Nita, et al., 2021). The results highlighted various rates of global SOC changes that were, however, mostly negative across continents, with annual values that even exceeded −500 t C $km^2$ $yr^{-1}$ in vast areas of Northern Eurasia, Northern North America, and Southeast Asia (Figure 15). The study revealed that most SOC negative annual rates ranged between −0.1 and −100 t C $km^2$ $yr^{-1}$ (Figure 15), a class that encompasses ~75% of all global SOC decreases or >55% of the total (negative and positive) SOC changes detected worldwide. At the opposite pole, some positive trends in SOC stock were identified across the planet, but to a much lesser extent compared to global SOC decreases (Figure 15). The geospatial results of negative and positive trends (examined in balance) revealed a total net area of SOC decline of ~1.9 million $km^2$ (~2.7 million $km^2$ with global negative SOC trends vs. ~0.8 million $km^2$ of positive SOC rates) throughout the Earth's terrestrial surface (Figure 15).

The same research (Prăvălie, Nita, et al., 2021) emphasized some remarkable findings regarding the total global amount of SOC changes, estimated as losses (~5 Pg C) and gains (over 1.8 Pg C) since the beginning of the 21st

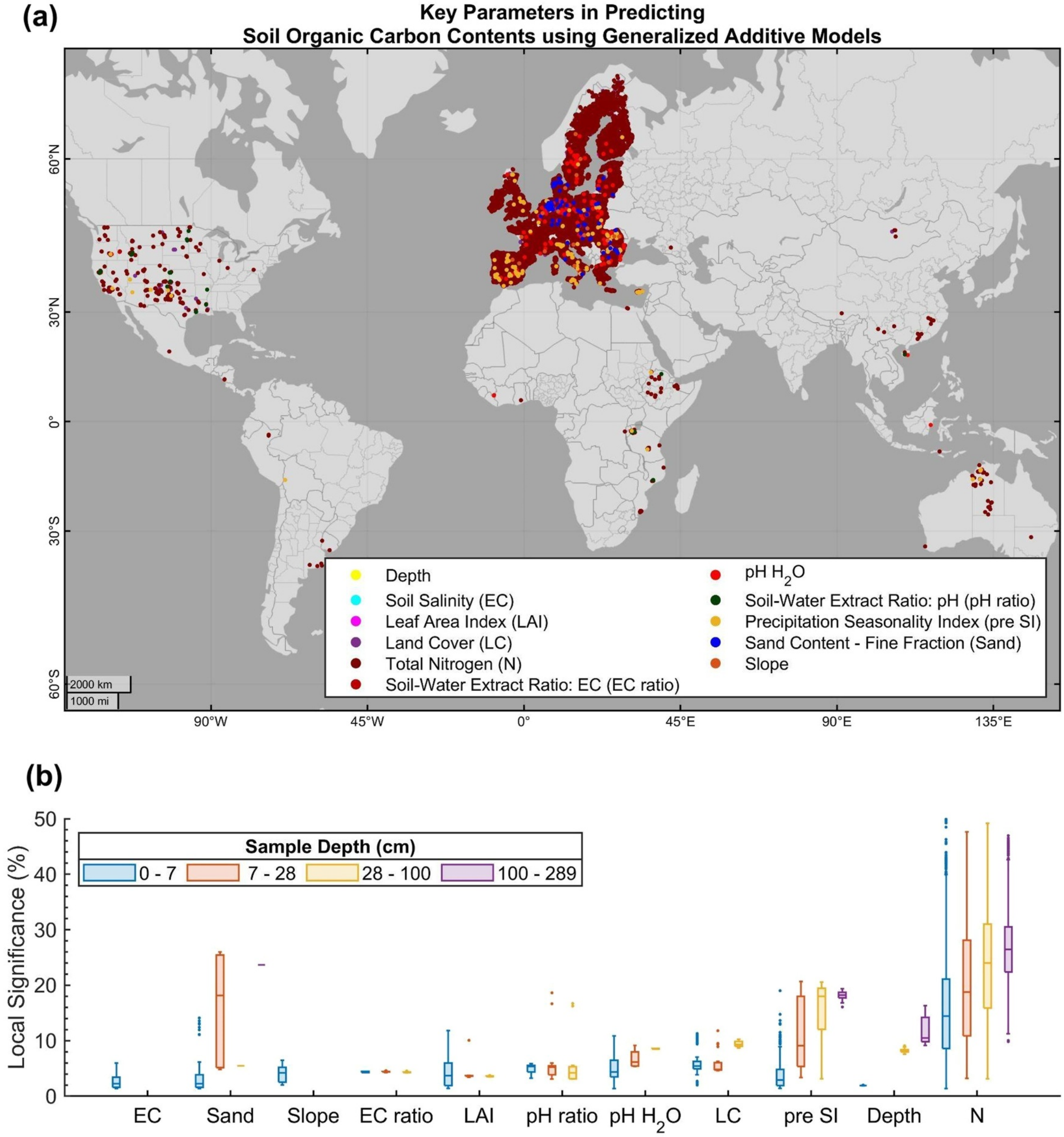


**Figure 14.** Covariates with the highest local significance in predicting SOC content based on the fitted General Additive Models fitted to the 43,459 mineral soil samples. (a) The most significant covariates explaining the topsoil SOC content (0–7 cm) variation at each observation site. (b) Box plots illustrating the distribution of local covariate significance across different soil depths. The box plots display the median, lower and upper quartiles, as well as the non-outlier minimum and maximum values. Outliers are defined as values exceeding 1.5 times the interquartile range (IQR) beyond the upper or lower quartiles. For each observation, the local significance of a covariate in predicting SOC was determined by calculating its absolute effect relative to the sum of absolute effects of all other covariates, excluding the intercept (after Hassani et al., 2024).

century. It seems that worldwide net losses of SOC over the 15-year period (>3.1 Pg C) represent ~8% of all global SOC losses recorded during the Holocene (the last 12,000 years, with total historical carbon losses estimated at 37 Pg C for the same soil profile of 0.3 m depth (Sanderman et al., 2017)), although it occurred in only ~0.1% of this time period.

The recent accelerated loss of large amounts of SOC indicates an alarming signal for climate change. This threat is greater considering that Canada and Russia were most affected by carbon leaks from soils (Figure 15), cumulatively recording 58% of all global net carbon losses after 2001 (Prăvălie, Nita, et al., 2021). These countries hold the largest permafrost areas of the planet, with frozen soils containing large amounts of carbon that can leak into

10.1029/2025RG000883


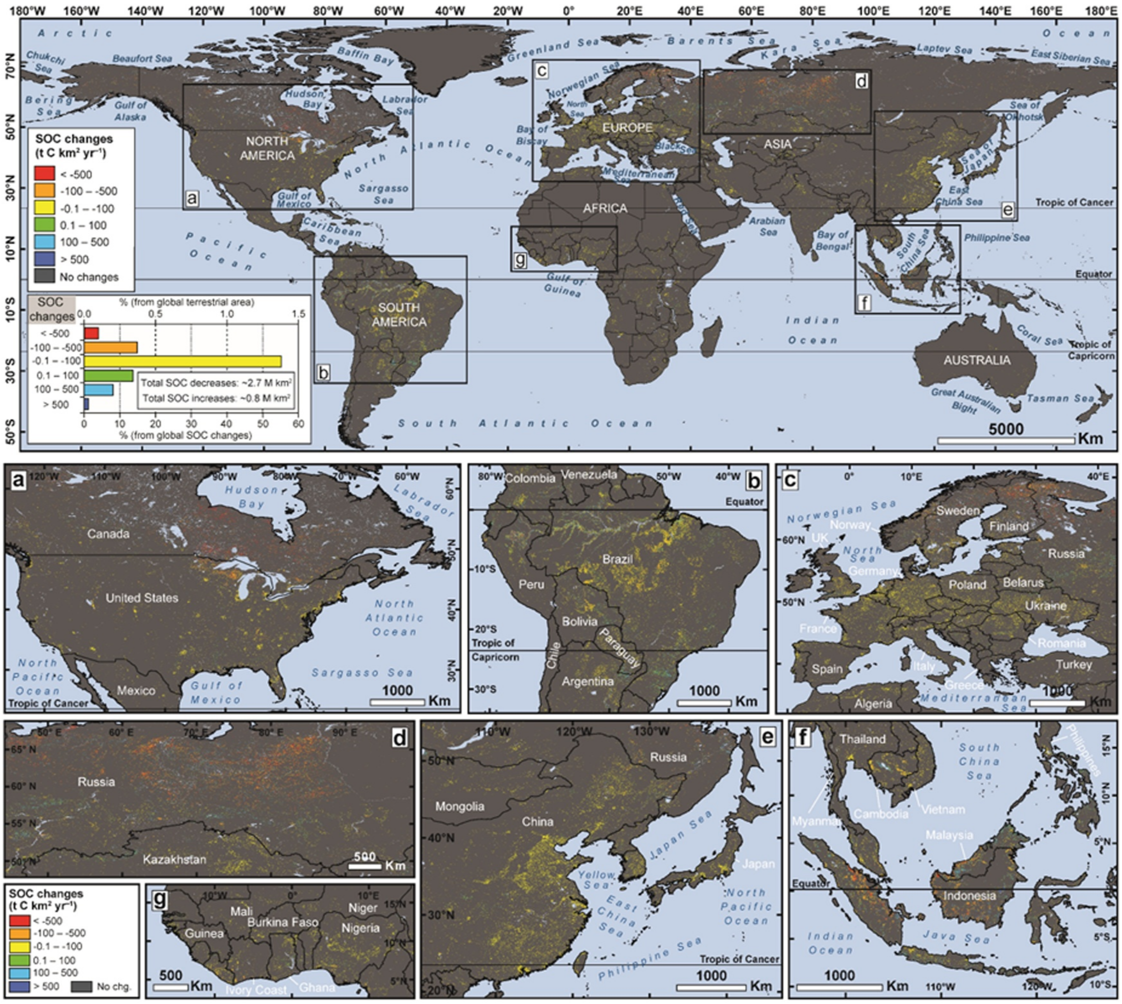


**Figure 15.** Spatio-temporal changes of SOC (pixel-level trends detected at 95% confidence level), mapped and quantified as impacted areas (in millions (M) of km$^2$ and %) across the global terrestrial surface, during 2001–2015 (after Prăvălie, Nita, et al., 2021). Panels from (a) to (g) illustrate the geographic regions where a zoom was applied to enable a better view of SOC changes across the globe.

the atmosphere, as $CO_2$ and (especially) $CH_4$ (Schuur et al., 2015). Consequently, it was reported that significant SOC losses at high latitudes could contribute to shifts in climate patterns, affecting global climate stability.

### 3.2.4. Salinization

Soil salinization represents a significant and escalating issue within the broader context of global soil degradation. This chemical phenomenon, influenced by both natural processes and human activities, adversely affects soil productivity, water quality, and the overall health of ecosystems (Shokri et al., 2024). The terms "soil salinity" and "soil salinization" are closely related yet distinct concepts that are essential for comprehending soil degradation as a chemical process. "Soil salinity" specifically refers to the concentration of soluble salts in the soil solution, which primarily includes sodium ($Na^+$), chloride ($Cl^-$), sulfate ($SO_4^{2-}$), calcium ($Ca^{2+}$), magnesium ($Mg^{2+}$), and bicarbonates ($HCO_3^-$; Abrol et al., 1988). This concentration is quantitatively assessed through the electrical conductivity (EC) of the soil solution, typically reported in deciSiemens per meter (dS m$^{-1}$). The accumulation of salts can interfere with osmotic potential (W. Hu et al., 2023; S. Hu et al., 2023; C. Zhang et al., 2022; D. Zhang et al., 2022; Z. Zhang et al., 2022), reducing water availability for plants, and can lead to ion toxicity, nutrient imbalances, and structural deterioration (Metternicht & Zinck, 2003; Shokri et al., 2024). Additionally, saline soils exhibit reduced microbial diversity and activity, which negatively affects nutrient cycling and soil fertility (Singh, 2016).

**Table 2**
*The Severity of Salt-Related Issues in Soil Is Typically Classified Based on the Electrical Conductivity of the Saturated Soil Extract ($EC_e$) and Exchangeable Sodium Ion Levels, and Soil Salt Content by Weight (Adopted From Omuto et al., 2020)*

| Intensity (FAO, 2024) | Salinity ($EC_e$ dS/m) | | Sodicity (ESP) | | |
|---|---|---|---|---|---|
| Intensity | FAO/IIASA/ISRIC/ISSCAS/JRC (2012) | Richards (1954) | Intensity | Abrol et al. (1988) | Amrhein (1996) |
| None | <0.75 | 0–2 | None | <15 | <6 |
| Slight | 0.75–2 | 2–4 | Slight | 15–30 | 6–10 |
| Moderate | 2–4 | 4–8 | Moderate | 30–50 | 10–15 |
| Strong | 4–8 | 8–16 | High/Strong | 50–70 | 15–25 |
| Very Strong | 8–15 | >16 | Extreme/Very Strong | >70 | >25 |
| Extreme | >15 | | | | |

Chinese classification scheme (weight of salt per unit kg of soil) (Chinese Academy of Sciences, 2001)

| Region | None | Light | Moderate | Severe | Solonchak |
|---|---|---|---|---|---|
| Coastal, semi-humid, semiarid, and arid | <1.0 g/kg | 1–2 g/kg | 2–4 g/kg | 4–6 g/kg | >6 g/kg |
| Semi-desert and desert regions | <2.0 g/kg | 2–3 g/kg | 3–5 g/kg | 5–10 g/kg | >10 g/kg |

Soil salinization, a significant factor in land degradation, often leads to alterations in plant diversity, which subsequently affects microbial diversity and composition (G. Zhang et al., 2024; B. Zhang et al., 2024). Unlike general soil salinity, salinization specifically denotes the process of salt accumulation, which can be progressive and potentially reversible if appropriate remediation measures are implemented. The term “soil salinization” describes the gradual increase in soil salinity over time (Hassani, Azapagic & Shokri, 2020). The presence of excessive salts in the soil solution reduces soil productivity, resulting in a series of negative impacts on ecosystem services, such as food production and the preservation of biodiversity (Shokri et al., 2024). As a result, lands affected by salinization frequently become abandoned, contributing to desertification and the destruction of habitats for both plants and animals (Sentis, 1996). Additionally, salinization impairs the ability of soil to sequester carbon, thereby intensifying greenhouse gas emissions influencing climate (Hassani et al., 2024).

Sodic soils, on the other hand, are marked by a high concentration of sodium ($Na^+$) ions in comparison to other cations, particularly calcium ($Ca^{2+}$) and magnesium ($Mg^{2+}$; Bleam, 2016). The degree of sodicity is quantitatively assessed using the “Exchangeable Sodium Percentage” (ESP), which indicates the fraction of sodium ions present in the soil's Cation Exchange Capacity (CEC). A soil is classified as sodic when its ESP exceeds 15% or when the “Sodium Adsorption Ratio (SAR)” exceeds 13 mmol $L^{-1}$, alongside conditions of pH greater than 8.5 and electrical conductivity (electrical conductivity of saturated paste extract: $EC_e$) below 4 dS $m^{-1}$ (Abrol et al., 1988; Richards, 1954; refer to Table 2 adopted from Omuto et al., 2020). In contrast to general soil salinity, which includes all soluble salts, sodicity specifically addresses the predominance of sodium and its adverse effects on soil structure and quality. Elevated sodium levels lead to the displacement of divalent cations such as calcium and magnesium from clay particles, resulting in clay dispersion and the disintegration of soil aggregates. This deterioration manifests as compromised soil structure, diminished infiltration rates, and reduced permeability, which further aggravate waterlogging and hinder root growth (De la Paix et al., 2013). The decline in physical soil properties due to sodicity adversely impacts agricultural productivity by limiting the availability of water and nutrients to plants. Additionally, sodic soils are often susceptible to surface crust formation, erosion, and decreased microbial activity, all of which contribute to a decline in soil fertility and the ability to provide ecosystem services (Singh, 2016; Wong et al., 2010). The effects of sodicity are particularly severe in irrigated and semi-arid regions, where poor water quality and insufficient drainage intensify sodium accumulation.

#### 3.2.4.1. *Drivers of Soil Salinization*

The drivers of soil salinization can be broadly categorized into "natural" (primary) and "anthropogenic" (secondary) processes, which often act synergistically to exacerbate salinity (Eswar et al., 2021). Natural salinity arises from the weathering of parent rocks, which release soluble salts into the soil. In regions with poor drainage, such as inland basins and river deltas, salts accumulate over time due to limited leaching. The rise of saline groundwater is another critical factor, particularly in low-lying areas or regions with high water tables (Rengasamy, 2006). Saline groundwater can contribute to soil salinization through capillary rise, particularly under high evapotranspiration conditions (Kamai & Assouline, 2018; Sadeghi et al., 2012; Shokri, 2019; Shokri-Kuehni et al., 2020; Vogelbacher et al., 2024). Climate plays a pivotal role in natural salinization. Arid and semi-arid climates, which account for approximately 40% of the global land surface, are particularly vulnerable due to high evaporation rates and low precipitation (Abrol et al., 1988; Hassani et al., 2021). The lack of sufficient rainfall limits salt leaching, allowing salts to concentrate in the soil, particularly close to the soil surface (Jannesarahmadi et al., 2024, 2025; Shokri-Kuehni et al., 2017a, 2017b, 2022). Geomorphological factors, such as topography and proximity to saline water bodies, also influence salinization. Coastal areas, for example, are prone to salinity due to sea spray aerosols and seawater intrusion, particularly under conditions of sea-level surges (Mazhar et al., 2022; Sobhi Gollo et al., 2024). The rate of salt deposition in coastal regions can be an order of magnitude higher than in inland areas due to the proximity to marine sources and the dynamics of coastal environments (Hillel, 2000).

On the other hand, irrigation is among the largest contributors to secondary salinization (Ritzema, 2016). The use of poor-quality irrigation water, particularly groundwater with high salt concentrations, is a primary cause (Shokri et al., 2024). Furthermore, inefficient irrigation practices, such as over-irrigation and poor drainage systems, exacerbate salt accumulation in the root zone. Excessive and unbalanced fertilizer use can indirectly contribute to soil salinization by increasing the ionic concentration of soil solutions. Additionally, conversion of natural ecosystems to croplands often leads to changes in hydrological regimes, exacerbating salinization. Deforestation, for instance, can lead to rising water tables and salt accumulation (Rengasamy, 2006). The global threat of secondary salinization is further compounded by climate change, which intensifies salinity through rising temperatures, altered precipitation patterns, sea-level rise, over-extraction of inland/coastal groundwater resources and sea salt intrusion (Hassani et al., 2021; Sobhi Gollo et al., 2024). For example, sea salt intrusion into inland water resources has led to the salinization of tracts of coastal agricultural lands in countries like Bangladesh, Vietnam, and Egypt (Ding et al., 2020; Mahmuduzzaman et al., 2014; Nguyen & Savenije, 2006).

Saline dust storms and saline aerosols are also contributors to soil salinization, especially in arid and semi-arid regions (Hassani, Azapagic, D'Odorico, et al., 2020). These phenomena occur when winds pick up and transport salt-laden particles from salt flats, dried-up lakebeds, and other saline landscapes over distances. Saline dust storms are particularly prevalent in regions where water bodies have receded due to climate change or human activities, exposing salt-encrusted soils (Borda et al., 2022; Hassani, Azapagic, D'Odorico, et al., 2020; Wurtsbaugh et al., 2017). The known examples are Lake Urmia in Iran and Owens Lake in the United States, where lake desiccation has created expansive salt flats that serve as sources for large-scale saline dust storms. When these salt-rich particles settle on soil surfaces, they increase the soil's salinity levels through dry deposition. Subsequent rainfall or irrigation dissolves the deposited salts, leading to their infiltration into the soil profile.

#### 3.2.4.2. *Global Extent of the Salt Affected Soils*

The Global Map of Salt-Affected Soils (GSASmap) V1.0.0 of FAO (2021a), compiling data from over 118 countries and 257,419 locations, with the participation of more than 350 national experts provides the latest official spatial distribution of salt-affected soils (FAO, 2021b). GSASmap considers key indicators such as electrical conductivity (EC), exchangeable sodium percentage (ESP), and pH levels for mapping the salt-affected soils at two depth intervals (0–30 cm and 30–100 cm). The data, which covers 85% of the global land area, shows that 424 million hectares of topsoil and 833 million hectares of subsoil are affected by salinity or sodicity, with the majority found in arid and semi-arid regions. The map reveals that 85% of salt-affected topsoils are saline, while 62% of subsoils are saline.

Although the map includes data from multiple countries, some regions, like Eurasia and the Near East and North Africa, faced challenges in preparing their maps due to data gaps or limited capacities. Accordingly, the global extent of salt-affected soils, including saline, sodic, and saline-sodic soils, remains highly uncertain, with estimates ranging from approximately 4.2 million to 17 million square kilometers (Shokri et al., 2024). This

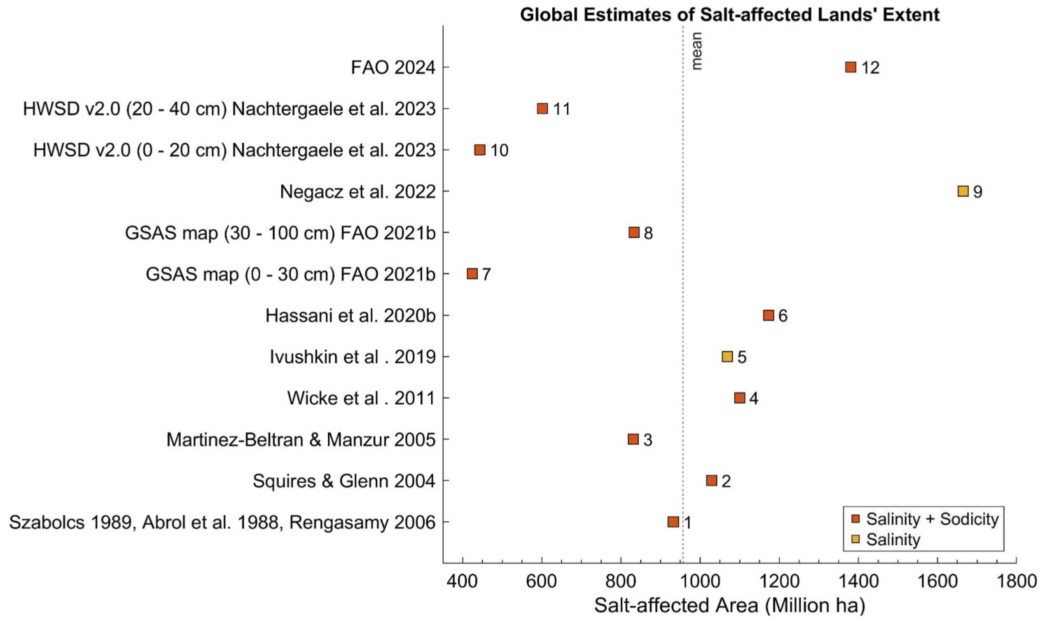


**Figure 16.** Salt-affected soil extent estimates according to different studies. Each estimate (data point) is annotated with a number (1, 2, 3, …, 12), which corresponds to specific explanations provided in the text.

variability arises from differences in measurement, modeling, mapping techniques, and the vertical depth considered in assessments. A major source of uncertainty lies in the variation of soil classification systems and the threshold values used to define salt-affected conditions. For example, soil salinity is classified using different thresholds: some systems define salinity at $EC_e$ values of 4 dS $m^{-1}$ (Abrol et al., 1988; Richards, 1954), while others, such as the Harmonized World Soil Database (HWSD; FAO/IIASA/ISRIC/ISSCAS/JRC, 2012), use 2 dS $m^{-1}$, the limit that can be intolerable for highly sensitive crops. Additionally, estimates may vary depending on whether sodic soils and soil pH are included alongside saline soils. Figure 16 presents the extent of salt-affected soils according to various studies. The numbers next to the symbols provide more information about each estimate as follows:

1. Szabolcs (1989), Abrol et al. (1988), Rengasamy (2006)—Estimates include both saline and sodic soils across more than 100 countries. Data is based on the FAO/UNESCO Soil Map of the World and other available sources at the time. Different classification and grouping systems were applied in individual countries.
2. Squires and Glenn (2011)—Covers salt-affected soils (saline, sodic, saline sodic).
3. Martinez-Beltran and Manzur (2005)—Includes salt-affected soils.
4. Wicke et al. (2011)—Extent of salt-affected soils based on a global assessment.
5. Ivushkin et al. (2019)—Multi-year salinity maps were produced; the reported value corresponds to the year 2016.
6. Hassani, Azapagic & Shokri. (2020)—Defined salt-affected soils using an $EC_e$ threshold of 4 dS $m^{-1}$ and/or an ESP of 6%. Annual maps for soil salinity and sodicity (0–30 cm depth) were generated for the period 1980–2018.
7. GSAS Map (FAO, 2021b)—Covers 75% of global land and includes data from 118 countries. Critical thresholds of $EC_e = 2$ dS $m^{-1}$, ESP = 15%, and pH = 8.2 for used to identify salt-affected soils.
8. Similar to 7, for 70–100 cm depth.
9. Negacz et al. (2022)—Estimates saline soils using $EC_e = 4$ dS $m^{-1}$ as the critical threshold.
10. HWSD v2.0 (FAO and IIASA, 2023)—Maps of salinity and sodicity were generated for different soil depths up to 2 m, but here, only the sum of saline or sodic topsoils (0–20 cm) are reported, assuming an $EC_e$ threshold of 4 dS $m^{-1}$ and an ESP of 6%.
11. Similar to 10, for 20–40 cm soil depth.

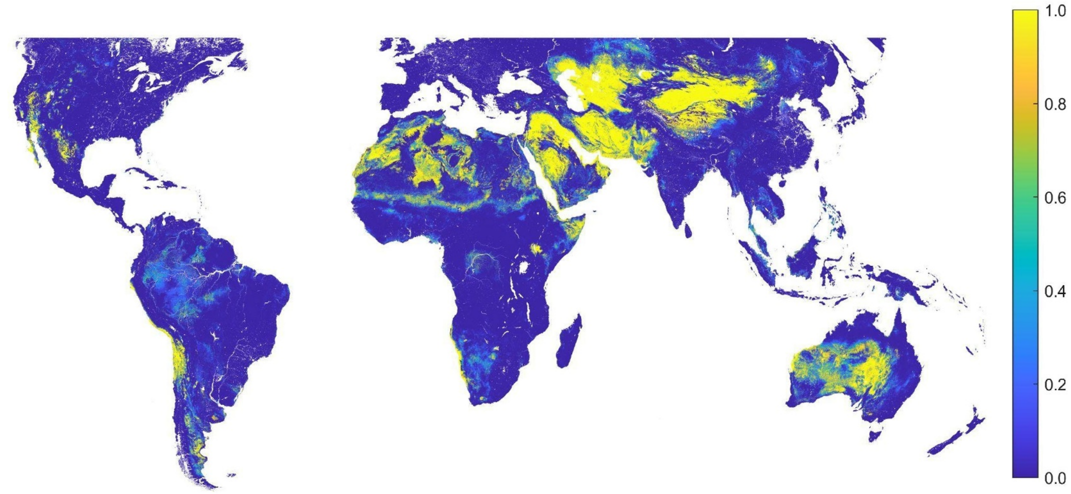


**Figure 17.** Likelihood of the surface soils (0–30 cm) with an $EC_e \geq 4$ dS $m^{-1}$ and/or ESP >6% between 1980 and 2018 (after Hassani, Azapagic & Shokri, 2020) at 1 km spatial resolution. This likelihood is dimensionless, calculated by dividing the number of years with $EC_e \geq 4$ dS $m^{-1}$ and/or ESP ≥ 6% by the total number of years studied. Hassani, Azapagic & Shokri. (2020) estimated likelihoods within the latitude range of −55° to 55°.

12. Global Status of Salt-Affected Soils Main Report (FAO, 2024)—Estimates include both saline and sodic soils. The assessment is primarily based on the GSAS map, with additional estimates for countries not covered in GSAS, derived from miscellaneous sources.

These inconsistencies between different estimates make it challenging to establish accurate local-to-global assessments of the extent, severity, and temporal changes of salt-affected soils, particularly in areas experiencing secondary salinization. Addressing these differences and uncertainties is a critical task for policymakers, land managers, and researchers (Hassani, Azapagic & Shokri, 2020; Shokri et al., 2024).

However, it is evident that salinization is particularly prevalent in arid and semi-arid regions (Figure 17), where high evapotranspiration rates exacerbate the accumulation of salts in the soil profile. Major hotspots include the Indo-Gangetic Plain in South Asia (Sharma & Mondal, 2006), the Murray-Darling Basin in Australia (Hart et al., 2020), and the San Joaquin Valley in the United States (Quinn, 2020). In such regions, salinization reduces agricultural productivity by up to 50% for crops sensitive to salinity, such as rice, wheat, and maize, leading to annual global economic losses exceeding $27 billion (Qadir et al., 2014).

### 3.2.5. Soil Contamination

Soil contamination is a major form of land degradation occurring through the presence of various (generally human-made) chemicals, which above a certain critical level causes the reduction or loss of soil ecosystem functions and services (FAO and ITPS, 2015; FAO and UNEP, 2021). It is a soil disturbance usually caused by polluting industrial and mining activities, unsustainable farming practices, or poor waste management, wearing many forms of occurrence, which may include soil contamination by heavy metals, pesticides, petroleum hydrocarbons, radionuclides, organic substances (e.g., PFAS), microplastics (Aminzadeh et al., 2025; Jannesarahmadi et al., 2023) and other groups of chemical elements or compounds (FAO and UNEP, 2021).

Globally, soil contamination is an emerging threat and a complex environmental challenge due to the vast number of potential contaminants, many of which remained unmapped. As a result, comprehensive global maps of soil contamination, both overall and for specific types, are still lacking. Global cartographic data are crucial for guiding policymakers on protecting soils, preventing further contamination (pollution) and reducing risks to ecosystems and human health (Hou & Ok, 2019). However, some contaminants have received more attention in large-scale mapping efforts across certain continents or regions.

Soil contamination with heavy metals across Europe is a key example in this regard. This aspect of land degradation has recently been explored in numerous pan-European studies, focused on spatial modeling of arsenic

(Fendrich et al., 2024), cadmium (Ballabio et al., 2024), zinc (Van Eynde et al., 2023), mercury (Ballabio et al., 2021), copper (Ballabio et al., 2018) or several heavy metals addressed simultaneously (Prăvălie et al., 2024). While significant progress has been made in understanding the spatial distribution of heavy metals in Europe, gaps remain in our knowledge of this type of soil contamination, particularly due to a lack of data in most countries outside the European Union.

Upon analysis of some spatial data used in a recent study focused on land multi-degradation in Europe (Prăvălie et al., 2024), it appears that soil contamination with arsenic is by far the greatest heavy metal pollution threat across vast pan-European (agricultural) soils (28% of which are affected by critical levels of arsenic), particularly in the developed countries located in Western and Southern Europe (e.g., France, Spain, or Italy; Figure 18b). Soil pollution with critical concentrations of nickel is also notable (especially in Greece and Italy), affecting about 3% of European agricultural landscapes (Figure 18i), which total about 2 million $km^2$ in the investigated countries (Prăvălie et al., 2024). The other toxic elements affect continental agricultural lands to a much lesser extent, corresponding to about 1% (or less) of their surface (Figure 18). Overall, it was found that about 31% of pan-European agricultural lands are affected by the heavy metal pollution (Prăvălie et al., 2024), an area resulting from joining (intersecting) the individual raster data classified as critical concentrations in topsoils (Figure 18).

Critical concentrations of heavy metals have multiple negative implications for soil health and productivity, or for human health, in Europe and worldwide. In agricultural environments, adverse ecological effects can materialize by reducing the biomass and microbial activity, disrupting nutrient availability, injuring plants via chlorosis and necrosis, inhibiting chlorophyll biosynthesis, or hindering root growth and crop yields due to phytoaccumulation and phytotoxicity (Angon et al., 2024; Nagajyoti et al., 2010). Essentially, the various negative agro-ecological effects potentially triggered by critical levels of heavy metals can significantly reduce soil fertility, impacting land productivity and food security in the affected areas of the world.

### 3.3. Soil Biological Processes

For the purposes of dividing processes for this discussion, we have provided subsections on physical and chemical processes. It should be understood, however, that many physical and chemical processes will manifest their effects in terms of biological processes, in the sense that many physico-chemical changes will in turn lead to changes in the soil microbiome, for example, in the community composition of the soil bacteria or fungi (Philippot et al., 2024).

There are also several biological processes that lead to soil degradation. There is soil biodiversity loss (Lal et al., 1989), which can be viewed variously as a result of degradation or as a driver of further degradation. Well-studied cases of soil diversity losses are from agroecosystems, where, for example, populations and diversity of arbuscular mycorrhizal fungi are often in decline in response to certain management practices, including tillage, pesticide use, and excessive fertilization.

The second biological process is the arrival and spread of antibiotic resistance genes (ARGs) within the soil bacterial community. The enhanced levels of environmental ARGs have recently been conceptualized as a factor of global change in their own right (Rillig, Lehmann, et al., 2024; Rillig, Li, et al., 2024). The reason for this proposal is that ARGs can increase in abundance and diversity in soils for reasons entirely unrelated to antibiotics, since several other drivers (e.g., heavy metals or salinity) also lead to increased ARG levels, and because ARG can spread within the microbial community, for example, when ARGs are coded on mobile genetic elements, such as plasmids.

A third biological process leading to soil degradation is the arrival of invasive species. This case is most studied for invasive plant species, as some of them can lead to the disruption of soil biota. One well-studied example is the reduction of plant root symbionts such as mycorrhizal fungi (Barto et al., 2011; Stinson et al., 2006), caused by the invasive, non-mycorrhizal plant species Alliaria petiolata (garlic mustard, an allelopathic herbaceous plant in the family Brassicaeae), with overall consequences for the functioning of the soil and the performance of native plants. On the other hand, there are also invasive microbial species (Litchman, 2010), such as invasive fungi, including ectomycorrhizal fungi (Dickie et al., 2016), that can lead to disruption of soil functioning. There is a large research gap in identifying invasive microbial species that do not form conspicuous fruiting bodies or that do not cause disease symptoms in plants or other hosts; this means that the real magnitude of this process of microbial invasion is likely underestimated at present.

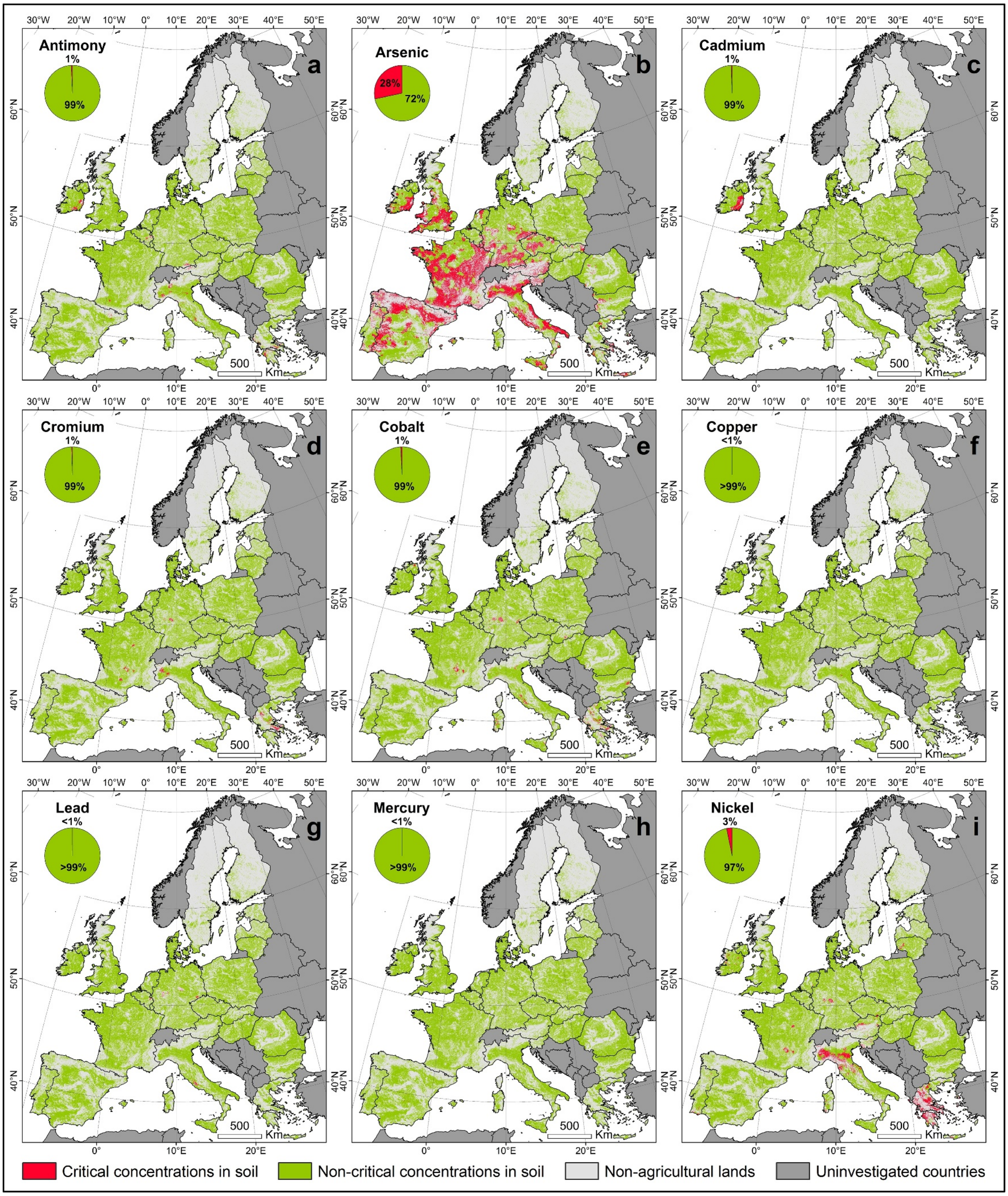


**Figure 18.** Spatial pattern and distribution of nine heavy metals in Europe, depicting critical levels of concentrations in agricultural soils for antimony (Sb) (a), arsenic (As) (b), cadmium (Cd) (c), chromium (Cr) (d), cobalt (Co) (e), copper (Cu) (f), lead (Pb) (g), mercury (Hg) (h) and nickel (Ni) (i) (data processing after Toth, Hermann, Szatmari, and Pasztor (2016), Ballabio et al. (2018, 2021), and Prăvălie et al. (2024)). Critical levels of soil pollution (red areas) were delimited based on the high concentrations (expressed in mg/kg) of heavy metals that exceed the standard guideline thresholds of toxic elements (Toth, Hermann, Szatmari, & Pasztor, 2016; Tóth, Hermann, Da Silva, & Montanarella, 2016; Prăvălie et al., 2024). Non-critical levels of soil pollution (green areas) were delimited via the lower concentrations of heavy metals, falling below the standard guideline thresholds of toxic elements. The light gray areas represent the non-agricultural lands in the investigated countries, while dark gray areas highlight uninvestigated countries across Europe, due to the unavailability or insufficiency of heavy metal data.

### 3.4. Multiple Concurrent Drivers

Drivers of global change, particularly anthropogenic factors, exert their influence on soil degradation not in isolation, but rather through their interactions. While there is a comprehensive understanding of the individual chemical, physical, and biological challenges that soils face, the collective impact of these factors on terrestrial ecosystems and their soils remains significantly less understood. The reasons for this knowledge gap are relatively straightforward: in addition to logistical challenges and the increasingly fragmented nature of research in global change biology, a fundamental issue is the combinatorial explosion problem (Katzir et al., 2019). This phenomenon refers to the rapid increase in the number of experimental treatment combinations as the number of factors considered grows. For instance, if there are 10 factors, each with two levels, the total number of treatment combinations would be $2^{10}$, which equals 1024. Conducting experiments of this scale is impractical.

A systematic mapping revealed that in the body of the published literature on soils and global change, experiments have used one or two factors only in 98% of cases, meaning we know almost nothing about the effects of more than two factors (Rillig et al., 2019). Contrast this with the high number of factors to be expected to act at the same time (Bowler et al., 2020), the true dimensionality of which is basically unknown, but given that there are an estimated 350,000 registered substances in industrial use alone (F. Wang et al., 2020; Z. Wang et al., 2020), in addition to all kinds of other factors of global change (Rillig et al., 2021), it is clear that the real number of factors affecting any soil at any time anywhere must be very high indeed. Furthermore, it is crucial to acknowledge that the influences on plants and soil can manifest at various levels within the ecological hierarchy, ranging from individual soil-plant interactions to broader landscape dynamics. These effects can propagate both upwards and downwards, resulting in complex, non-additive consequences for soil degradation (Rillig, Lehmann, et al., 2024; Rillig, Li, et al., 2024).

One way out of this conundrum is to ask a different question, and one question that has been asked is about the number of factors (Rillig et al., 2019; Yang et al., 2022). The experimental design involves creating a gradient in factor number, thereby de-emphasizing factor identity and composition. This is achieved by randomly sampling from a factor pool for each of the replicates or a given "number of factors" level. In each replicate corresponding to a specific number of factor levels, factors are selected without replacement from a designated pool, guaranteeing that each replicate is assigned the required number of factors. Conversely, within replicates of the same level, the random sampling occurs with replacement, allowing for the possibility that factors may be selected in multiple replicates. This generates a factor number gradient with de-emphasized importance of the composition of factors, since each of the replicates of each level of "factor number" will have a composition, that is, determined by chance alone. When this is done in an experiment, soils respond in a linear fashion to the increasing number of factors (with factor number alone explaining a substantial part of variability, e.g., 50%).

Examples of soil degradation-related response variables for which this has been found are several process rates, such as soil aggregation, decomposition or respiration, and also fungal biodiversity, as assessed by high-throughput sequencing (Rillig et al., 2019), and bacterial diversity (Rodriguez del Rio et al., 2025; Yang et al., 2022). Losses of biodiversity in the fungi were not random, since members of the Basidiomycota phylum were lost disproportionately compared to members of the Ascomycota or Mucoromycota phyla. Furthermore, the findings revealed unexpected results, particularly regarding soil water repellency, as assessed by water drop penetration time. Significant effects emerged when five factors were considered in the sandy soils utilized in this study. These effects could not be anticipated based on the influence of individual factors, all of which demonstrated no significant response in this parameter, except for a minor effect attributed to drought. The reason for the frequently linear response to the factor number gradient remains ambiguous, as it could potentially exhibit a curvilinear pattern or demonstrate a threshold response. It is possible that the gradient generated by 10 factors (for instance, in increments of 0, 1, 3, 5, 8, and 10 factors) was inadequate to reveal such behaviors in a statistically significant manner. Future experiments may need to explore higher dimensionality, potentially increasing the number of factors to 30 or 50, a figure, that is, not implausible considering the degree of environmental contamination in certain human-impacted settings.

Recent research has demonstrated that, beyond the mere count of factors, the degree of dissimilarity among these factors significantly influences the observed responses in soil properties and process rates (Bi et al., 2024). This conclusion was reached through the experimental creation of 150 distinct "communities" characterized by varying compositions of factors, which were assessed at three different levels of factor count and evaluated for dissimilarity based on their calculated effects derived from individual factor impacts. The study revealed that a greater likelihood of synergistic interactions, which increased with dissimilarity, accounted for these effects while controlling for the so-called sampling effect, an artifact inherent to this experimental design. The sampling effect refers to the tendency for the probability of including a disproportionately influential factor in random selections to rise as the number of factors increases.

Alongside experimental data, observational data can also be utilized to demonstrate that a specific proportion of the total variability in multifunctionality across approximately 200 ecosystems globally can be linked to the number of factors involved (Rillig et al., 2023). To facilitate this analysis, continuous data underwent transformation through thresholding (to generate counts) and grouping (to reduce multicollinearity). This indicates that greater dissimilarity among the factors influencing soil simultaneously leads to more detrimental effects on soil properties and processes, resulting in increased soil degradation.

The processes that drive the influence of various factors on soil degradation remain poorly understood, as the random sampling approach does not allow for the identification of specific combinations of factors responsible for the observed effects. Nevertheless, metagenomic analyses conducted by Rodriguez del Rio et al. (2025) indicate that certain bacterial taxa are favored in treatments involving multiple factors (eight combined factors) when compared to those involving individual factors. These selected taxa exhibit characteristics associated with stress tolerance and antibiotic resistance. Furthermore, it was observed that members of the rare biosphere present in the control samples became predominant bacterial phylotypes in the treatments with high factor combinations.

Clearly, understanding the effects of numerous, concurrently acting drivers on soil and its processes and biodiversity will remain a challenge and current results are to be seen as a starting point. A more in-depth mechanistic investigation into the combined effects of various factors acting together will be essential for advancing our knowledge of soil degradation.

## 4. Drivers of Soil Degradation

### 4.1. The Difference Between Soil Degradation Drivers and Perturbations

In this review, we frame soil degradation processes through our own conceptual distinction between drivers and perturbations. Drivers are sustained natural or human-induced forces, such as climate change, tectonic activity, or intensive agriculture, that maintain or shift soil system states over extended periods, from months to geological timescales. Perturbations are short-term disturbances such as floods, droughts, extreme weather, or sudden land-use changes that temporarily disrupt the system, testing its resilience. Over time, frequent or prolonged perturbations may become drivers. Depending on the timescale, the same factor may act as either; for example, human activity is a perturbation on geological scales but a driver in the context of modern land management.

Distinguishing between anthropogenic and natural drivers, as well as between drivers and perturbations, is essential both for accurately attributing the causes of soil degradation, and for identifying effective management or policy interventions in order to slow or reverse these degradation trends. Critically, understanding root causes of soil degradation and the magnitude of new drivers or perturbations may help society prioritize management response with limited resources (based on urgency of intervention or importance of the soil in a particular ecosystem). In practice, a goal for soil degradation research is to understand degradation rather than eliminate it, and learn how to minimize its impact. It should also help devise better soil management and land use to attain more sustainable states and increase system resilience. These topics will be considered in the following review.

### 4.2. Unsustainable Agricultural Practices

Unsustainable agricultural methods encompass various techniques that emphasize immediate profits at the expense of enduring ecological health (Adamsone-Fiskovica & Grivins, 2024). This also includes practices that are not adapted to the local environmental characteristics, such as soils, climate, or topography (Liebig et al., 2017). Such practices frequently result in considerable environmental harm, adversely affecting soil quality, water resources, and biodiversity. Several unsustainable approaches remain prevalent today, such as intensive tillage, monoculture cultivation, and the extensive application of chemical fertilizers and pesticides (Adamsone-Fiskovica & Grivins, 2024; Garnett, 2013).

The process of tillage entails the mechanical disruption of soil, which results in alterations to its structure and an escalation in erosion (Thapa & Dura, 2024). Additionally, repeated tillage fractures soil aggregates, diminishing crucial soil organic matter and microbial activity that are essential for sustaining overall soil health. This practice also heightens surface runoff, contributing to the erosion of topsoil, that is, critical for plant development (Liu et al., 2024). Ultimately, over time, tillage can lead to soil compaction, which adversely affects water infiltration and reduces soil fertility.

One unsustainable agricultural practice is the cultivation of a single crop species in a specific area, known as monoculture farming. This method can result in various environmental challenges (Uekoetter, 2008). Monoculture is deemed unsustainable because it can deplete essential soil nutrients necessary for the growth of that particular crop (Uekoetter, 2008). Additionally, it diminishes biodiversity, rendering crops more susceptible to pests and diseases (Altieri, 1999). Consequently, implementing crop rotation is crucial to avoid dependence on chemical inputs for pest and disease management and to preserve soil fertility.

The prevalent use of chemical fertilizers, pesticides, and herbicides in contemporary agriculture is primarily aimed at enhancing crop yields (R. Iqbal et al., 2020; S. Iqbal et al., 2020; Pahalvi et al., 2021). However, these substances can have harmful effects on the environment. The excessive application of fertilizers can lead to increased nutrient runoff into water bodies, such as rivers and streams, resulting in eutrophication and detrimental impacts on aquatic ecosystems (Chislock et al., 2013). Moreover, pesticides and herbicides can pollute soil and water, posing threats to various species, including beneficial insects and soil microorganisms (Douglas, 2015). Ultimately, the overuse of these chemicals can contribute to the emergence of resistant pest species, necessitating even greater chemical applications (Chislock et al., 2013).

In summary, the unsustainable practices outlined have been shown to intensify environmental degradation. Tillage and monoculture agriculture are known to diminish soil health and biodiversity, rendering ecosystems more susceptible and fragile to changes (Thapa & Dura, 2024). The heavy dependence on fertilizer inputs further deteriorates soil quality and adversely affects water resources (Zimnicki et al., 2020). Consequently, it is crucial to acknowledge the benefits of implementing sustainable farming methods, such as crop rotation, reduced tillage, and integrated pest management, as these practices are vital for alleviating these negative effects and fostering long-term agricultural sustainability (Liu et al., 2024).

### 4.3. Deforestation and Land-Use Change

Assessing soil degradation necessitates precise and detailed descriptions of land use and its evolving patterns (Borrelli et al., 2020; Panagos et al., 2021). The intricate relationships between global trade and agricultural production have led to an estimated 32% of the world's land undergoing changes in use from 1960 to 2019 (Winkler et al., 2021). This net change in land use indicates a transition from forested areas to agricultural cropland and pasture. Additionally, urban expansion is anticipated to exacerbate soil sealing (van Vliet, 2019), intensifying land take pressures and potential conflicts. There are significant geographical disparities in land use changes across different hemispheres and regions, which are crucial to understanding soil degradation. The Global South is witnessing a decline in forested areas, while the Global North is experiencing a net increase or trend toward reforestation. Consequently, deforestation disproportionately impacts soil degradation, with the Brazilian Amazon, Southeast Asia, and West Africa exhibiting the highest rates of ecosystem alteration due to deforestation (Winkler et al., 2021). These shifts in land use are critical to the soil system, potentially leading to the loss of natural vegetation, increased fertilizer application, the use of plant protection measures, drainage, and irrigation, all of which contribute to various degradation pressures.

Historical data underscores the critical role of land use changes in intensifying soil erosion rates (Figure 19). For instance, following the arrival of European settlers in the Americas, soil erosion rates increased dramatically, by one to two orders of magnitude (Montgomery, 2007). On a global scale, the drainage of peatlands has led to a significant reduction in soil organic carbon and diminished the land's ability to buffer against rising emissions (Smith et al., 2015). Currently, Borrelli et al. (2020) have determined that more than half (54%) of global water erosion occurs in areas designated for annual crops, permanent crops, and managed pastures. Changes in land use, particularly the expansion or reduction of global cropland, are major factors influencing variations in soil erosion rates (Borrelli et al., 2017a; Nearing et al., 2017), nutrient levels (De Rosa et al., 2024), soil fertility (Khaledian et al., 2017), compaction (Hamza & Anderson, 2005), and salinization (Hassani et al., 2021; Shokri et al., 2024). The exposure of bare soil to rainfall significantly accelerates the natural rates of erosion (Montgomery, 2007; Nearing et al., 2017).

Feedback loops represent a crucial aspect of the interactions between land use, climate, and the environment (Le Quéré et al., 2013). The strong interconnection between land use and soil degradation indicates that the latter is significantly influenced by factors such as dietary choices, global trade dynamics, land optimization, and the closure of yield gaps (Beyer et al., 2022). The proliferation of unsustainable land management practices into marginal regions with limited environmental capacity increases the likelihood of both acute and chronic

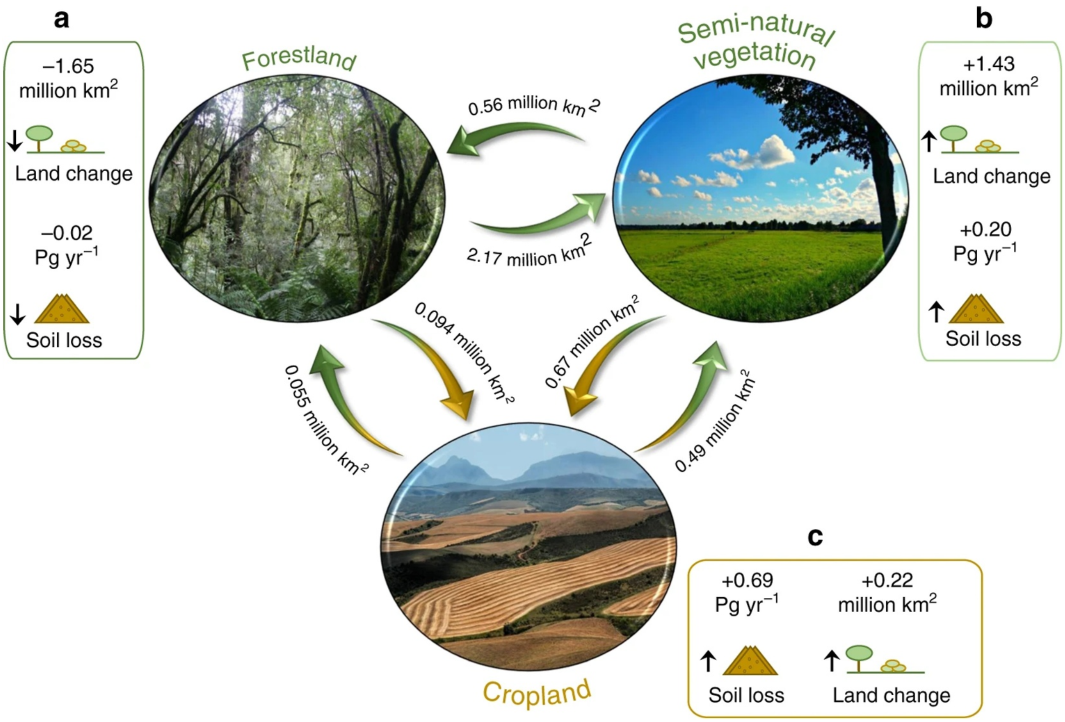


**Figure 19.** A flow diagram illustrating the alterations in land use and their impact on soil loss estimations is presented. The arrows represent the extent (in million km$^2$) of changes in land use and land cover that occurred from 2001 to 2012. Insets (a–c) highlight the net changes in land surface area (in million km$^2$) and the corresponding soil loss (in Pg yr$^{-1}$; after Borrelli et al., 2017a).

degradation, which can lead to further changes in land use or even abandonment (Vanwalleghem et al., 2017). Additionally, feedback loops are altered by climate and environmental changes, with soil degradation playing a multifaceted role in both contributing to and regulating these changes, while also affecting the planet's ability to buffer against rising carbon emissions (De Rosa et al., 2024). Therefore, large-scale trends in land use that influence soil degradation must take into account the interplay of climate change and socioeconomic factors (Stehfest et al., 2019). The critical role of land use underscores the necessity of protecting existing natural ecosystems and restoring degraded ones through methods such as rewetting, re-establishing vegetation cover, and afforestation, as essential strategies for halting and mitigating global soil degradation (Smith et al., 2016).

### 4.4. Climate Impact on Soil Degradation

#### 4.4.1. Changes in Temperature and Precipitation Patterns

There is a consensus that the average land surface temperature worldwide has risen by 1.53°C since the preindustrial era (Jia et al., 2019), and this trend is projected to persist. The rate of warming in dryland regions is approximately double that of the global average (Lickley & Solomon, 2018), and an expansion of dryland areas is anticipated. Furthermore, while it is evident that global mean precipitation is expected to rise due to alterations in the energy budget (Allen & Ingram, 2002), the specific regional variations in precipitation patterns remain uncertain.

Variations in temperature and precipitation patterns influence soil degradation processes by altering soil energy and moisture dynamics. Projected global warming and precipitation variability disrupt water availability and evapotranspiration patterns, resulting in diminished soil moisture and increased surface temperatures, which alter radiative energy partitioning over terrestrial surfaces (Aminzadeh & Or, 2014; Brutsaert, 2005). Rising soil temperature, coupled with changes in diurnal temperature variations, intensify plant stress and increase the likelihood of crop failure (Aminzadeh et al., 2023; Hultine et al., 2023; Nobel et al., 1986). Covering around 12% of global terrestrial surfaces, temperature increase and moisture deficits further influence the health and functionality of biocrusts, which serve as critical ecological and biogeochemical hotspots, particularly in arid regions (Antoninka et al., 2022; Belnap, 2013; Escolar et al., 2012; Finger-Higgens et al., 2022). Long-term experiments by Phillips et al. (2022) on the Colorado Plateau in North America showed that a 4°C rise in temperature, combined with precipitation variability, significantly hindered biocrust recovery, resulting in a 19% decline in mosses and a 4% decrease in lichens. The rise in surface temperature further impacts soil carbon dynamics with strong evidence indicating an accelerated loss of soil carbon due to warming, thereby affecting soil carbon storage and global carbon cycling (Crowther et al., 2016; Reichstein et al., 2013; Ren et al., 2024).

While declining rainfall causes soil desiccation, which negatively affects soil structure, microbial communities, soil health, and agricultural productivity, increased precipitation can accelerate soil erosion and elevate the risk of land degradation. The analysis conducted by Thackeray et al. (2022) suggests that precipitation extremes will occur approximately $32 \pm 8\%$ more frequently than at present by 2100. Such shifts in precipitation patterns could lead to a projected average increase in global rainfall erosivity of 26.2%–28.8% by 2050 and 27%–34.3% by 2070, as indicated by Panagos et al. (2022). Climate change, coupled with the anticipated increase in rainfall erosivity, is expected to be the main factor driving a projected rise in soil erosion rates of 30%–66% by 2070 (Panagos, Borrelli, et al., 2022; Panagos, Van Liedekerke et al., 2022).

#### 4.4.2. Increased Frequency of Extreme Events

The frequency and severity of extreme weather and climate events are projected to shift during the 21st century with general increases in drought, heatwave, heavy precipitation, wildfire events, and desertification on continental to global scale (AghaKouchak et al., 2020; Huning, Love, et al., 2024; Kim & Villarini, 2024; van der Wiel & Bintanja, 2021; Waidelich et al., 2024). These changes are expected to accelerate ongoing land degradation. The frequency, intensity, and duration of heatwaves have increased and will continue to increase. It is projected that between 50% and 80% of the world's land will experience significantly more severe hot extremes compared to historical data (Jia et al., 2019). Similarly, in several regions, such as the Mediterranean, North Africa and the Middle East, sub-Saharan Africa, central China, the southern Amazon, India, East and South Asia, parts of North America and eastern Australia, drought frequency and intensity have increased. Extreme precipitation events are likely to rise, as warmer atmospheric conditions enable air to retain more moisture, leading to an increase in record-breaking rainfall occurrences (Jia et al., 2019).

These extremes influence soil physiochemical environment (e.g., soil aggregate stabilization), microbial community and functionality, and soil hydrology and nutrient cycling thereby accelerate land degradation processes by intensifying floods, landslides, soil erosion, and salinization (Clarke & Rendell, 2007). Future projections indicate substantial spatio-temporal variability in these extremes, underscoring the complexity of their impacts on soil ecosystems and landscapes. During drought events, a combination of high evapotranspiration, reduced precipitation, and reliance on inappropriate irrigation methods often leads to increased soil salinity and contributes to crop failure (Hassani et al., 2021; Shokri et al., 2024). Additionally, drought conditions reduce vegetation cover, exposing soil to erosion and limiting organic matter input from plant residues, which is essential for maintaining soil fertility and structure. Research indicates that SOC is expected to decline by an average of 10.5% globally in response to climate extremes associated with a 1.5°C increase in temperature (Wang et al., 2024). On the other hand, rising atmospheric moisture content in a warming climate enhances extreme precipitation events (Pfahl et al., 2017), thus intensifying water-driven soil erosion and increasing the risk of landslides. Soil erosion is projected to increase globally 13.7% by the end of the century, with Africa anticipated to experience the most significant increase (Lal, 2012). Borrelli et al. (2020) estimated that the combined effects of land use and climate change could lead to 66% increase in global soil erosion by 2070. Climate change was identified as the primary driver behind these projected changes in soil erosion, highlighting its critical role in intensifying soil degradation processes. Changes in precipitation patterns, prolonged droughts, and increasing agricultural water demands have intensified pressure on groundwater resources (Aminzadeh et al., 2024; Nevermann et al., 2024). Over the past few decades, many croplands have experienced a decline in groundwater levels due to over-extraction and insufficient natural recharge (Davydzenka et al., 2024; Haghshenas Haghighi & Motagh, 2024; Noori et al., 2023). The cumulative stress of these factors has not only threatened crop yields and agricultural production, but has also led to land subsidence, causing irreversible damage to soil structure and further undermining land productivity (Davydzenka et al., 2024; Huning et al., 2024b).

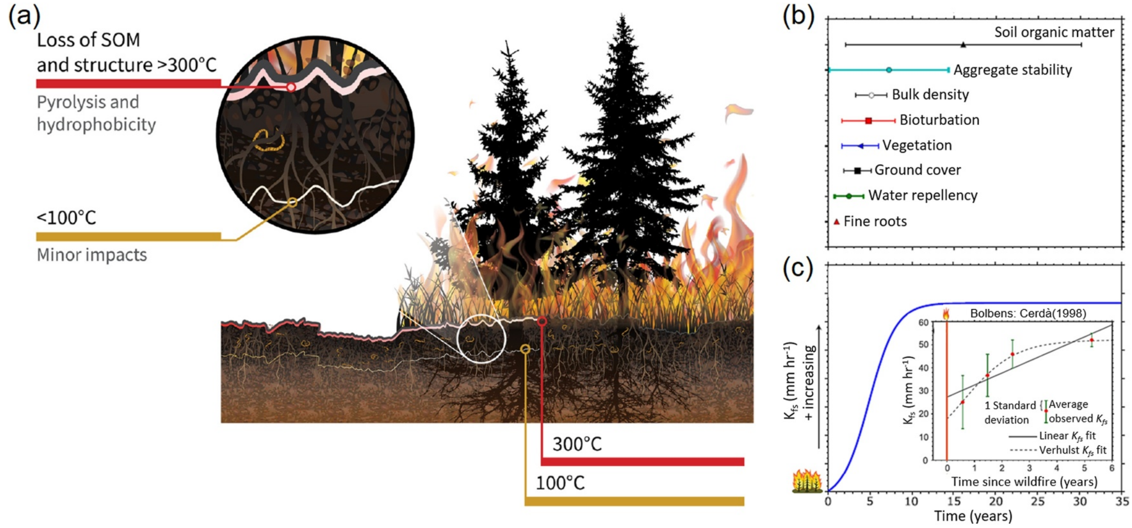


**Figure 20.** (a) Conceptual representation of changes in soil properties due to wildfire induced thermal alterations. Variations in critical temperatures and their corresponding penetration depths influence soil structure, induce hydrophobicity, modify SOM/SOC through pyrolysis, alter soil minerals, and affect soil fauna and ecological functions to varying extents. (b) Characteristic time scales for post-fire recovery of parameters that influence saturated hydraulic conductivity. (c) Hypothetical curve for recovery of saturated hydraulic conductivity; the inset depicts the values of measured saturated hydraulic conductivity from a study in Spain (adapted from Or et al., 2023).

Wildfires are among the most significant natural perturbations in vegetated regions that leave significant imprints on soil and vegetation status long after the event. Evidence suggests persistent increase in wildfire frequency and magnitude (especially in the Western US). In part, this trend is attributed to a drier climate, but also to legacy of fire suppression and fuel management (Shuman et al., 2022). The impacts of wildfires on landscapes and soil have been studied extensively, indicating long-term alterations in landscape and ecosystem functioning (DeBano et al., 1998; Keeley et al., 2011; Roces-Díaz et al., 2022). The nature and extent of wildfire impacts on soil are related to wildfire characteristics (heat release and duration of active fire over the land). In addition, vegetation loss, soil organic carbon alterations, and soil structure degradation may significantly alter soil hydrology, leading to increased runoff, accelerated erosion, and greater soil loss, while reducing infiltration and evapotranspiration (Or et al., 2023). The increase in runoff and sediment yield following wildfire events has been reported by Lamb et al. (2011), Mayor et al. (2007), and Moody and Martin (2009). The destabilization of soil combined with ash accumulation can lead to high-viscosity slurries (Burns & Gabet, 2015) that may trigger landslides (Thomas et al., 2021) and debris flows (McGuire et al., 2017). Various studies (e.g., DeBano et al., 1998; Neary et al., 1999, 2005) have quantified aspects of wildfire impacts on soil functions. In addition to wildfire characteristics that influence the impact and define the boundary conditions for recovery, climate significantly affects recovery rates. Interestingly, different soil functions may be restored at different rates, as indicated by the range of data in Figure 20.

### 4.5. Industrial Activities and Mining

Intense industrial activity during the last two centuries has had a serious impact on soil contamination. The main associated contaminants are mineral oils, trace elements (e.g., arsenic, cadmium, lead, nickel, and zinc), and organic contaminants such as halogenated and non-halogenated solvents, polychlorinated biphenyls, and polycyclic aromatic hydrocarbons (Vieira et al., 2024).

A significant aspect of global land use is the extraction of mineral and fossil fuel resources, which leads to various forms of soil degradation. This sector is distinct from other land uses in two primary respects: the intensity of soil degradation and the financial resources available for remediation. Although mining activities can considerably disrupt site hydrology and agro-ecological functions, these factors are not addressed in this review. In this analysis, we focus on three main categories of resource extraction: two forms of mining and the extraction of oil and gas. Each of these extraction methods has multiple effects on soil and presents different soil management challenges.

1. Surface mining, which constitutes the predominant method of mineral extraction, accounts for approximately 80% of mining activities in terms of ore and waste generation (Ramani, 2012; Tibbett, 2024). This process entails the complete removal of soil and underlying materials to access minerals. Examples include the extraction of coal, iron, bauxite, oil sands, as well as sand and gravel. It may also encompass placer mining, which involves extracting minerals from sediments through dredging with water.
2. Underground mining, which involves the extraction of ore along with associated waste materials. Notable examples include coal and gold mining.
3. Oil and gas extraction through well drilling.

Tang and Werner (2023) estimated mining's total global footprint as 65,585 $km^2$, this comprising pits, ore stockpiles, waste rock dumps, water ponds, tailings dams, heap leach pads, and processing/milling infrastructure. This does not include oil and gas production, with this encompassing not only oil and gas production facilities but also pipelines and processing infrastructure. For these activities there doesn't appear to be an estimate of global footprint.

Mining and oil and gas production result in soil degradation in several ways:

1. Removal of various soil horizons and exposure of subsoils and underlying substrates. Consequences include the loss of soil nutrients, organic matter, and biologic functions (Worlanyo & Jiangfeng, 2021). Where rehabilitation occurs, some of the removed soil material may be returned, or the underlying subsoil treated to enable re-establishment of vegetation. Reduce compaction or manage high bulk density, non-structured materials (Koch & Hobbs, 2007; Tibbett, 2024).
2. Management of overburden and processing wastes
   (a) Overburden may be stockpiled in waste rock dumps (Tibbett, 2024) during mining and not returned to the void following mine closure. Considerable work has been undertaken on stabilization of waste rock dumps, with an aim of developing functional soils (Kumar et al., 2023; Williams, 2022). One aim is to promote the development of plant cover to reduce the percolation of water through the waste dump and thus acidic leakage (Williams, 2022). Issues relate to the properties of the capping soil such as pH, sodicity, and compaction and soil erosivity. Several recent papers (e.g., Ruiz et al., 2020a, 2020b) have considered the properties of constructed soils (e.g., Technosols (Schad, 2018)) on mining wastes.
   (b) Several types of mining involve beneficiation of ores by physical or chemical processes to increase the concentration of or extract the target element. This results in a range of residue materials, or tailings (Tibbett, 2024) ranging from those that are inert through to toxic or radioactive contaminants. These residues may be retained in waste rock dumps or dams (Tibbett, 2024), buried under other materials (Ramani, 2012), or disposed of by incorporating into existing soils (Santini & Fey, 2018). In some cases, attempts are made to promote plant growth via establishment of a soil cover on tailings to prevent drainage of toxic elements (Karaca et al., 2018).
3. Mine infrastructure, including exploration tracks, haul roads, mining infrastructure, access roads, and power supplies.
   (a) Compaction of soils through construction of exploration tracks, roads, and easements for conveyor belts, pipelines, and power supplies.
   (b) Soil contamination with hydrocarbons is particularly an issue with oil and gas production, with recent reviews on impacts and remediation approaches (Lim et al., 2016; Ossai et al., 2020). Heavy metal contamination of soils is also common across mine sites and downstream areas; methods of remediation include removal and burial of contaminated soil, soil washing, and attempts to decrease the bioavailability of metals (Karaca et al., 2018).
4. Disruption of a site's water balance and changes in recharge, discharge, and surface runoff. For example, the discharge of mine-waters when de-watering mine voids can contaminate soils with a range of heavy metals (e.g., lead, arsenic, cadmium) and, in some cases, carbonates. The release of hydraulic fracturing fluids and also contaminated groundwater can occur in oil and gas projects. Pichtel (2016) reviews the effect of this water on soils; contaminants include salts, radioactive materials, and hydrocarbons.

Certain elements of rehabilitating soils affected by mining exhibit parallels with the restoration of soil functions in other land-use contexts. These commonalities encompass the challenges associated with the loss of nutrients and organic matter, addressing chemical concerns such as sodicity and extreme pH levels, as well as soil contamination. Additionally, various physical soil issues, including compaction, dispersion, hardsetting, and loss of soil structure, are relevant. However, the degradation of soils due to mining diverges from other land uses primarily in the extent of the damage incurred. This includes the total removal and inversion of soil profiles, the exposure of non-soil materials and substrates, and the scale and intensity of degradation, particularly regarding soil contamination linked to mining operations and oil and gas extraction.

Furthermore, the regulatory frameworks and funding mechanisms differ significantly; the financial returns from mining often allow for substantial restoration efforts. In some areas, mining activities are governed by stringent mine-closure regulations that mandate subsequent restoration (Lima et al., 2016), while in other locations, soil restoration may not take place at all (Worlanyo & Jiangfeng, 2021).

### 4.6. Biological Drivers of Soil Degradation

A comprehensive review by Sims (1990) on "Biological soil degradation" provides an authoritative overview of key soil biological processes and agents operating in soil, emphasizing how land management, toxic substances, and mining operations affect soil biology. While the title of Sims (1990) review resembles this section's heading, the emphasis is different—here we focus on how biological agents drive soil degradation. This aspect of soil degradation is relatively understudied (from the soil perspective), and it is often motivated by ecological impacts on ecosystems (e.g., species invasion). The primary examples reviewed here involve the role of invasive fauna and plant species on soil properties and functions; identifying conditions that promote soil pathogen outbreaks due to land management (e.g., crop rotation avoidance with repeated use of monoculture); and biologically driven modification of soil microbial communities (suppression of nitrogen fixers and rhizobiome keystone members).

Simberloff et al. (2013) provide a summary of the pervasiveness and impacts of biological invasions and their ecologic, economic, and social damages. The monitoring and assessment of impacts are often hampered by poor monitoring and lack of clear metrics. The primary modifiers of soil functions and services are invasive plants (nonnative species that become locally dominant). Plant invasions are often a byproduct of globalization and may have considerable ecological and economic impacts, including effects on soils (Ravi et al., 2022). We will also review invasive soil-fauna and their impacts on soil functions.

#### 4.6.1. Invasive Plants

Meta-nalyses of plant invasion impacts on their new habitats and soils (Liao et al., 2008) demonstrate a complex web of positive and negative feedback loops in which nutrient cycles, microbiomes (Reinhart & Callaway, 2006) and soil properties (Gibbons et al., 2017) can be relatively quickly modified to benefit the invading plant species. On balance, these may not amount to soil degradation in the classical sense of diminished productivity; on the contrary, studies have shown an increase in soil carbon stocks, or in nitrogen and other nutrient concentrations (Berhe, 2019; Gibbons et al., 2017; Liao et al., 2008) and overall increase in productivity. However, the alteration of belowground soil biota (Reinhart & Callaway, 2006) to benefit the invading plants, or indirectly benefiting from lesser pressures of soil-borne enemies result in facilitation of invasion while inhibiting native species function (Callaway et al., 2004). It is with this perspective that we consider the process of exotic plant invasion as a component of soil degradation.

Torres et al. (2021) list strategies for soil microbiota manipulation by exotic plants, for example, exudence of chemical substances that reduce survival and regeneration of native plants, changes in microbial communities in the rhizosphere, alteration of decomposition processes, metabolizing labile and recalcitrant substrates, and modifying soil enzyme activities. More mechanisms for plant-soil microbiota interactions are elaborated in the review of Reinhart and Callaway (2006). A global meta-analysis of the effects of plant invasion on native soil biota and microbial activity is summarized in a recent review by Negesse et al. (2025) providing quantitative trends of the effects that were also listed by Torres et al. (2021).

Some invading plants modify the soil hydrologic regimes, such as cheatgrass (*Bromus tectorum*), an invasive grass pervasive across the water-limited Intermountain Western US (Terry et al., 2024). Cheatgrass has been shown to effectively compete with native vegetation on shallow water stored after the winter. Additionally, the grass is linked to major increases in fire frequency and promotes accumulation of soil nitrogen. Cheatgrass covers ≥15% of 210,000 km$^2$ (31%) of the Intermountain West, doubling the likelihood of burning relative to native shrublands (Balch et al., 2013). Gibbons et al. (2017) have studied three invasive plant species in the Intermountain Western US. There included the spotted knapweed (*Centaurea stoebe*; perennial forb), leafy spurge

(*Euphorbia esula*; perennial forb), and the cheatgrass (*Bromus tectorum*; annual grass) discussed above. They found that spotted knapweed and leafy spurge-invaded plots had higher soil pH and potassium concentration than native sites, while cheatgrass plots had higher phosphate concentration than native plots. Then invaders imprinted their distinct signatures on soil chemistry, and on other abiotic properties (soil pH) and certainly on the microbial composition of invaded plots relative to native vegetation. The Russian thistle (Salsola tragus or tumble weed) is considered the fastest plant invader in the US history. Introduced in the late 1800s, it quickly spread across the Western US where it competes with crops and is known to consume soil water after wheat harvest, thus compromising the yield of the following crop. In its dry state it creates a fire hazard and may indirectly promote soil degradation as it tumbles and disrupts fragile soil crusts, thus promoting dust emissions.

#### 4.6.2. Earthworm Invasion Affecting Soil Properties

Hale et al. (2005) identified the leading-edge of (European) earthworm invasion in forests of northern Minnesota and documented changes in soil characteristics before and after the earthworm invasions. They observed the rapid disappearance of the O horizon with increasing thickness, bulk density, and total soil organic matter content of the A horizon. Different earthworm species assemblages affected the magnitude and type of change in these soil parameters (associated with different feeding and burrowing habits). A study by Bohlen et al. (2004) observed similar trends as reported in Hale et al. (2005) with the most dramatic effect being the loss of the forest floor at an undisturbed forest site that also altered soil nutrient cycling. Hendrix et al. (2008) reviewed earthworm invasion in a broader global context in nearly every geographic region.

The detrimental impacts of earthworm invasion into pristine lands and forests are placed here in the context of soil degradation with an important caveat. Indeed, the massive loss of soil organic carbon cited by Bohlen et al. (2004) is detrimental to these ecosystems and may propagate to other changes in the ecosystem such as loss of aboveground arthropod communities (Jochum et al., 2022). However, earthworms are also known as soil environmental engineers that support soil structure and improve soil functioning. A recent study by Fonte et al. (2023) estimated that earthworms contribute about 6.5% of global grain production.

Interestingly, the New Zealand flatworm (Arthurdendyus triangulatus) is another invasive soil organism introduced from the South Island of New Zealand into the UK and Northern Europe (Boag & Yeates, 2001). Being an effective predator of lumbricid, A. triangulatus has been estimated to reduce native earthworm communities by 20% (Murchie & Gordon, 2013), altering many soil ecosystem functions that native earthworms provide.

#### 4.6.3. Crop Rotation and Soil-Borne Pathogens

One of the oldest and most studied phenomenon of biological soil degradation is associated with the detrimental impacts of continuous monoculture that often lead to deterioration of soil conditions for that crop due to build-up of soil-borne pathogens and other factors. Evidence suggests that crop rotation increases soil health, promotes microbial diversity, and enhances soil disease suppressive capacity (Peralta et al., 2018). The exact mechanisms by which crop rotation sequence and crops affect soil health and productivity for future crops are complicated (Peralta et al., 2018), however an area of consensus relates to central role of soil microbiome composition and "biological legacy" in the root zone as studied by Zhou et al. (2023) and Neupane et al. (2021). Important examples of the massive impacts of biological soil degradation on important agricultural crops (and the effectiveness of crop rotation over chemical sterilization of soils) include the enormous economic losses caused by Fusarium oxysporum and Macrophomina phaseolina (soil borne fungi) on strawberry production worldwide (Pastrana et al., 2023). Peanut (Arachis hypogeaea L.) is another susceptible crop to soil borne pathogens due to the close association of peanut pods with soil and difficulty applying fungicides through canopy to soil profiles (Thiessen & Woodward, 2012) in addition to environmental concerns. The long history of crop rotation (practiced since Roman times and fallow practices mentioned in the Bible) involve other aspects than affecting soil microbial communities and disrupting disease inducing pathogen life cycles; in some cases, the use of legumes to boost nitrogen stocks in soil or leaving a field fallow to capitalize on soil water storage across seasons are factored into crop rotation strategies.

#### 4.6.4. Overgrazing and Soil Alteration by Domestic and Wild Ungulates

We conclude this section on biological drivers of soil degradation by briefly mentioning the critical role of domestic and wild ungulates in overgrazing and compacting rangelands, especially in arid landscapes where vegetation response is slow and landscapes are susceptible to overgrazing. Livestock and wild ungulates are also known to modify various soil biogeochemical cycles (prominently the nitrogen cycle) and alter soil biota (Sitters & Andriuzzi, 2019).

## 5. Quantification of Soil Degradation at Different Scales

### 5.1. Traditional Quantification Approaches and Their Limitation

Various approaches have been developed to evaluate soil degradation and its impact on soil ecosystem services and functions. The most commonly used methods include expert assessments (using diagnostic indicators), land users' insights (through questionnaires and interviews), field monitoring and observations, modeling, and remote sensing techniques (Kapalanga, 2008). Analytical approaches offer a more comprehensive evaluation of soil quality. For instance, the Soil Management Assessment Framework (SMAF), developed based on analytical indicators, incorporates 81 potential indicators to assess soil functions in relation to management practices and ecosystem services (Bünemann et al., 2018). However, the methodologies for assessing soil degradation (whether qualitative, quantitative, or a combination of both) often lead to variability in identifying degradation levels due to differences in indicator selection, interpretation, and environmental conditions.

Field surveys, which rely on direct observations and measurements of soil physical, chemical, and biological quality parameters, provide valuable localized data. Nonetheless, selecting suitable diagnostic indicators, establishing their thresholds, and interpreting the results pose significant challenges, as these elements frequently differ across various land types and climatic conditions (Molchanov et al., 2015). Additionally, while large-scale soil monitoring initiatives such as LUCAS surveying campaigns (Orgiazzi et al., 2018) provide extensive data sets, most field-based assessments are constrained spatially and lack scalability. The quantitative evaluations of soil degradation derived from these measurements are challenging to scale up and necessitate ongoing monitoring to capture seasonal and annual fluctuations. On the other hand, remote sensing techniques provide a wider perspective on soil degradation; however, they face challenges related to spatio-temporal resolution, data accessibility, and precision.

While these methods provide useful insights, their limitations highlight the need for modern technological advancements (Rinot et al., 2019). Integrating local measurements with high-resolution remote sensing data and leveraging AI-based tools can significantly enhance soil degradation assessments. These advancements allow for improved spatial coverage, more precise monitoring, and a better understanding of soil degradation dynamics. They effectively overcome the limitations of traditional methods, which often struggle with scalability, standardization, and real-time monitoring. This is particularly crucial in dynamic environments where soil degradation is driven by climatic variations, land management practices, and human activities.

### 5.2. Ongoing National, Continental and Global-Scale Efforts and Assessments

Evaluations of land degradation employ a variety of quantitative and qualitative methods, utilizing both bottom-up and top-down strategies, as well as hybrid approaches (Gibbs & Salmon, 2015). These evaluations have been conducted by a range of entities, including national, multinational, and intergovernmental organizations, in addition to independent research groups (Figure 21). Global estimates suggest that the area of degraded land ranges from 1 to 6 billion hectares (Gibbs & Salmon, 2015). These assessments highlight the significance of aridity and soil erosion as contributing factors (Prăvălie, Patriche, et al., 2021). Additionally, the widespread use of inorganic fertilizers to bridge the yield gap and mitigate potential declines in yields due to soil degradation is another significant factor contributing to land degradation (Gerber et al., 2024).

The Global Assessment of Soil Degradation (GLASOD), initiated by the United Nations Environment Program (UNEP), is among the first global initiatives aimed at systematically mapping the extent of soil degradation (Oldeman, 1991). As mentioned earlier, at the time of its development, the GLASOD methodology relied heavily on expert judgment, primarily due to the limitations of remote sensing technologies and computational capabilities. GLASOD was groundbreaking in its global assessment of soil degradation; however, its limitations in spatial analysis (such as the lack of physiographic regionalization) and methodological approaches (including an insufficient evaluation of driving factors) highlighted the need for enhancements in subsequent assessments.

LADA (Land Degradation Assessment in Drylands) and GLADA (Global Assessment of Land Degradation and Improvement) emerged as successors to GLASOD, concentrating on assessments of land degradation at multiple

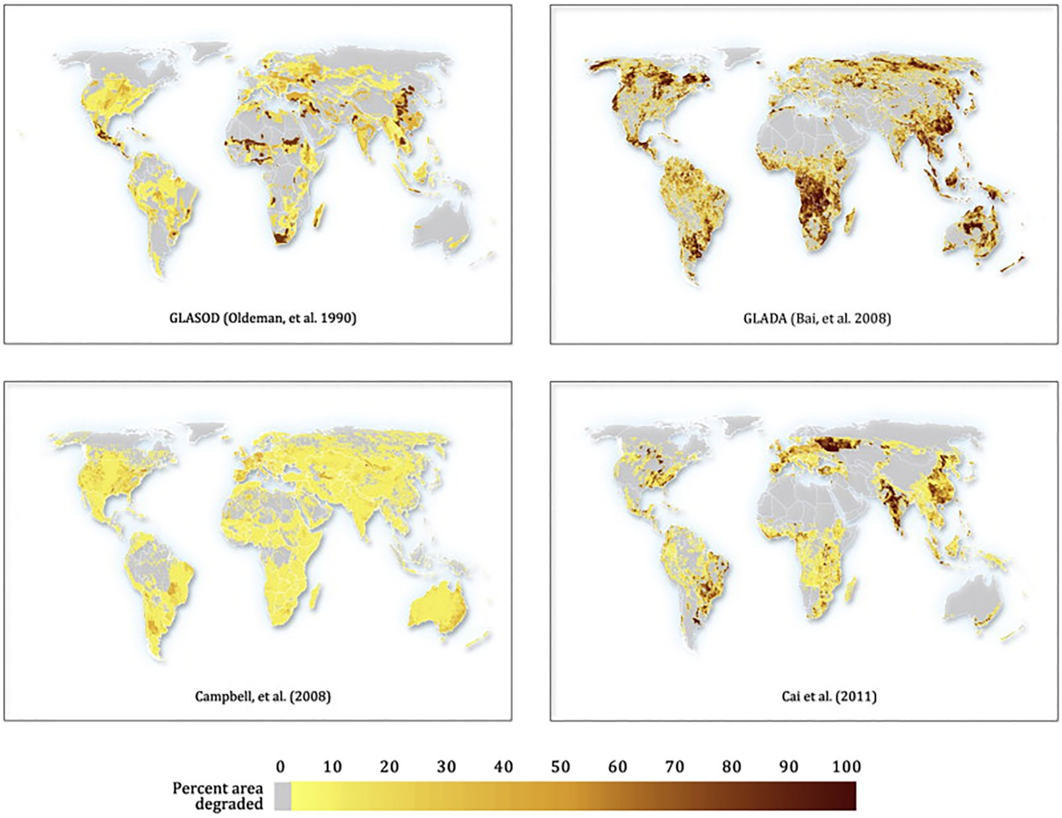


**Figure 21.** Proportion of cell area impacted by land degradation as assessed through four distinct evaluation methodologies (after Gibbs & Salmon, 2015).

scales, regional, national, and global, to evaluate its condition and trends. LADA employs the DPSIR framework, which encompasses Driving-force, Pressure, State, Impact, and Response, to characterize land degradation through a decentralized participatory approach. In contrast, GLADA utilized variations in the Normalized Difference Vegetation Index (NDVI), exemplifying data-driven methodologies for mapping degradation through remote sensing (Bai et al., 2008).

National and multinational strategies are better suited to address the specific challenges that soils encounter across various geographic and political landscapes. Prominent multinational evaluations include the National Resources Inventory (NRI) conducted by the Natural Resources Conservation Service (NRCS), the 2024 report on the state of soils in Europe, the national soil erosion census in China (Zhen, 2013), and the Desertification and Land Degradation Atlas of India, among others. Advances in data, modeling, and computational techniques are enhancing the ability of these extensive assessments to capture the detailed nuances of soil degradation. For instance, evaluations of soil degradation within the European Union suggest that 60%–70% of soils are degraded, based on soil sampling and modeling efforts (Panagos et al., 2024c; Veerman et al., 2020). This methodology employs a convergence of evidence approach (Gianoli et al., 2023), which aims to synthesize information from diverse and independent sources to ascertain the degradation status and spatially highlight the intensity and location of multiple concurrent land degradation processes (Prăvălie et al., 2024; Figure 22). The harmonization and convergence of data across various political boundaries and geographic scales will promote the comparability of national, continental, and global evaluations of soil degradation. As the volume of data increases, along with our capacity to derive significant insights from it (Bernardino et al., 2025), a persistent challenge will be to integrate and reconcile the various elements of soil degradation, particularly in capturing process interactions without incurring high complexity costs. Erosion serves as a prime example of this challenge, where numerous factors are identified, yet model coupling (i.e., the exchange of information) is lacking, hindering the understanding of process interactions and feedback mechanisms (Borrelli et al., 2023).

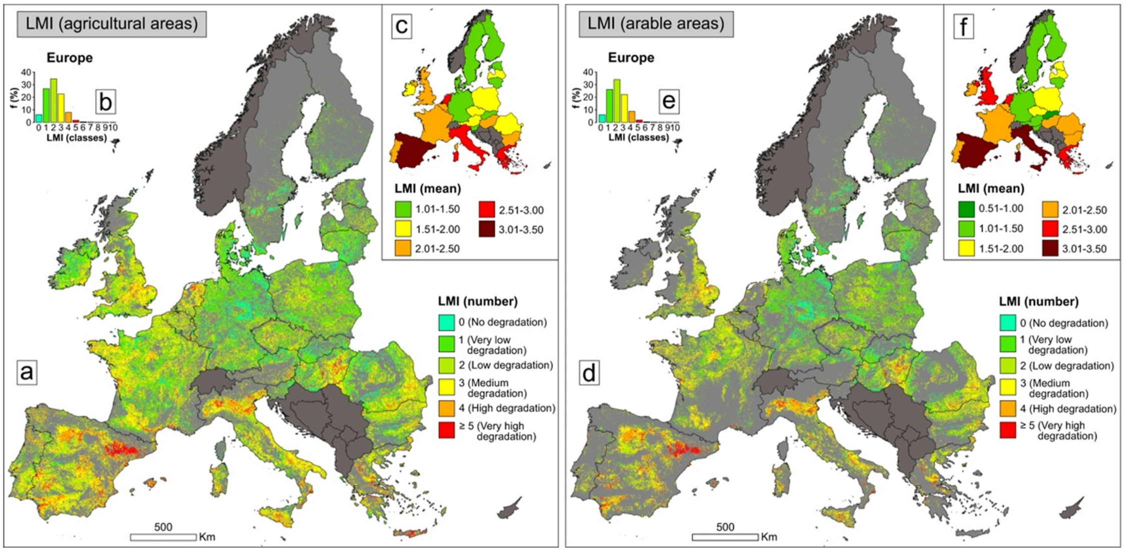


**Figure 22.** Land Multi-degradation Index (LMI) in Europe, illustrated as the number of simultaneous land degradation processes occurring in agricultural (left) and arable (right) landscapes of the continent (after Prăvălie et al., 2024). Panels from (a) to (c) illustrate the spatial distribution, histogram, and mean of agricultural LMI values across the European countries, while panels from (d) to (f) illustrate the spatial distribution, histogram, and mean of arable LMI values across Europe.

### 5.3. The Challenge of Quantification of Soil Degradation: Lack of Standardized Methods

Efforts to quantify soil degradation face limitations due to the lack of a universally accepted methodology. As a result, discrepancies in data collection, interpretation, and evaluation have arisen across various regions. The diverse processes of soil degradation, including erosion, compaction, chemical imbalances, and loss of biodiversity, necessitate tailored assessment methods. Existing approaches, such as expert assessments, remote sensing, field measurements, and feedback from land users, exhibit considerable variability in their methodologies, often lacking the necessary harmonization and comparability (Kapalanga, 2008). This inconsistency leads to fragmented data sets that hinder the development of comprehensive global assessment frameworks.

Recent efforts to improve soil degradation assessments have introduced methodologies that utilize various soil health indicators to create binary maps categorizing areas as healthy or degraded based on established thresholds, which are then aggregated to determine the overall level of soil degradation. However, the dependence of these tools on binary classifications and indicators that may be affected by local conditions could limit their ability to accurately reflect the intensity and cumulative effects of degradation processes within a given region (Panagos et al., 2024c; Prăvălie, Patriche, et al., 2021; Prăvălie et al., 2024).

Soil characteristics differ across various pedo-climatic zones. Indicators such as soil organic carbon (SOC), pH, and salinity are often subject to varying interpretations influenced by land use, environmental factors, and management objectives (Figure 23). This variability poses challenges for standardization, hindering the ability to compare soil degradation levels across diverse geographic areas (Campbell et al., 2025). Additionally, the inconsistent classification of degraded soils presents another obstacle. Employing a binary degradation model that categorizes soil as either "degraded" or "non-degraded" based on a set of thresholds might fail to account for the varying degrees of degradation intensity, which are essential for formulating effective soil restoration strategies. These challenges highlight the necessity for a cohesive, standardized methodology that incorporates multiple soil indicators to facilitate a more objective and practical assessment framework.

### 5.4. Toward Quantification of Soil Degradation

Several studies have concentrated on developing standardized, multi-scale, and objective methodologies for measuring soil degradation (Bünemann et al., 2018; Panagos et al., 2025; Poppiel et al., 2025; Prăvălie et al., 2024). These initiatives primarily focus on the incorporation of physical, chemical, and biological indicators into unified frameworks, facilitating a more thorough evaluation that can guide targeted interventions and inform policy decisions.

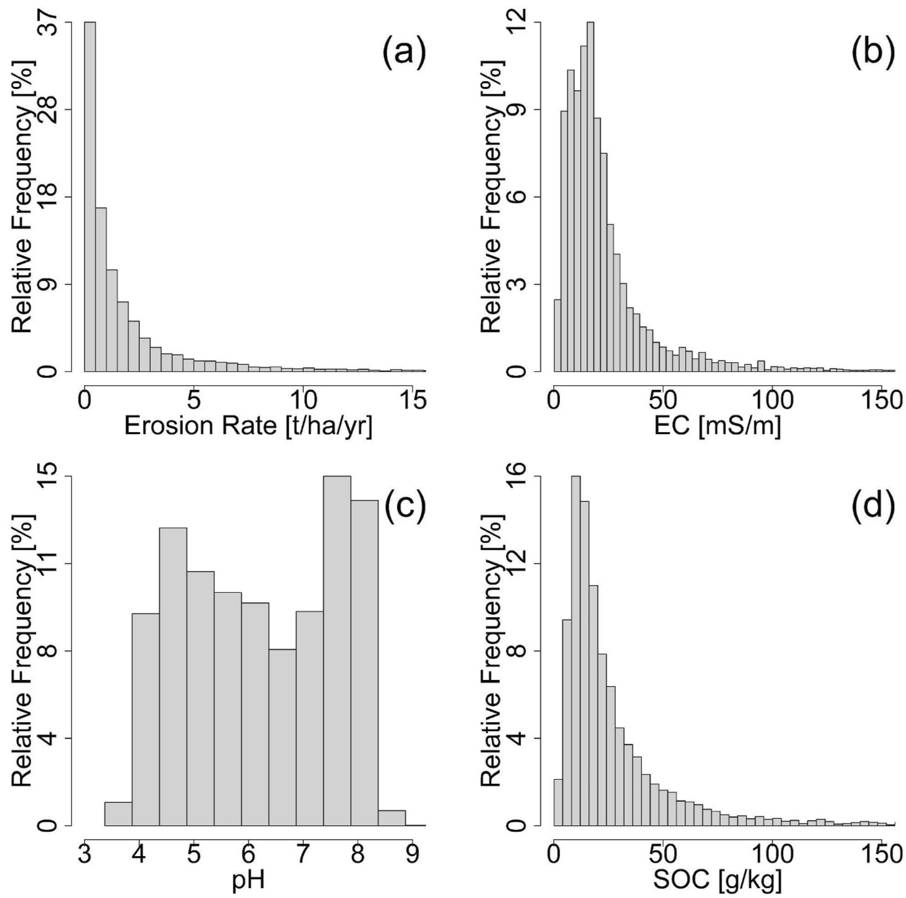


**Figure 23.** Variability of erosion rate (a), EC (b), pH (c), and SOC (d) over croplands across Land Use/Cover Area Frame Survey data (LUCAS) observations.

A prominent initiative in this domain is the Soil Degradation Dashboard, established by the EU Soil Observatory (EUSO), which aims to systematically monitor the condition of soils throughout Europe (Panagos et al., 2024c). This dashboard utilizes 19 indicators of soil health that encompass the principal degradation processes, including erosion, compaction, chemical imbalances, and loss of biodiversity (Figure 24). By establishing thresholds for these indicators, the EUSO framework employs a "one out, all out" principle, categorizing soils as degraded if any individual indicator exceeds its specified limit. This principle offers a harmonized and structured approach that could potentially be applied beyond Europe, contributing to the standardization of soil degradation assessments on a global scale.

In addition to the EUSO developments, the Land Multi-Degradation Index (LMI) serves as a framework that consolidates various degradation processes, such as soil compaction, acidification, nutrient imbalances, pollution, and different erosion mechanisms, into a singular, cohesive index (Prăvălie et al., 2024). This integration effectively captures the cumulative impacts and interactions of multiple stressors, thereby identifying degradation hotspots and prioritizing areas for remediation.

Another contribution in the quantitative evaluation of soil degradation is represented by the development of the Soil Degradation Proxy (SDP). The SDP is derived by first rescaling four indicators of soil erosion rate, electrical conductivity (EC), soil pH, and soil organic carbon (SOC) to a scale of 0–1 using their empirical cumulative distribution functions across LUCAS observations. For erosion rate and EC, higher observed values correspond to proportionately greater rescaled scores (i.e., "more is worse"). In contrast, SOC is inversely related, meaning that soils with higher organic carbon content receive lower (healthier) scores. Given soil pH values on both extremes (acidic or alkaline) negatively affect soil health, the rescaling for pH imposes penalties for deviations from the

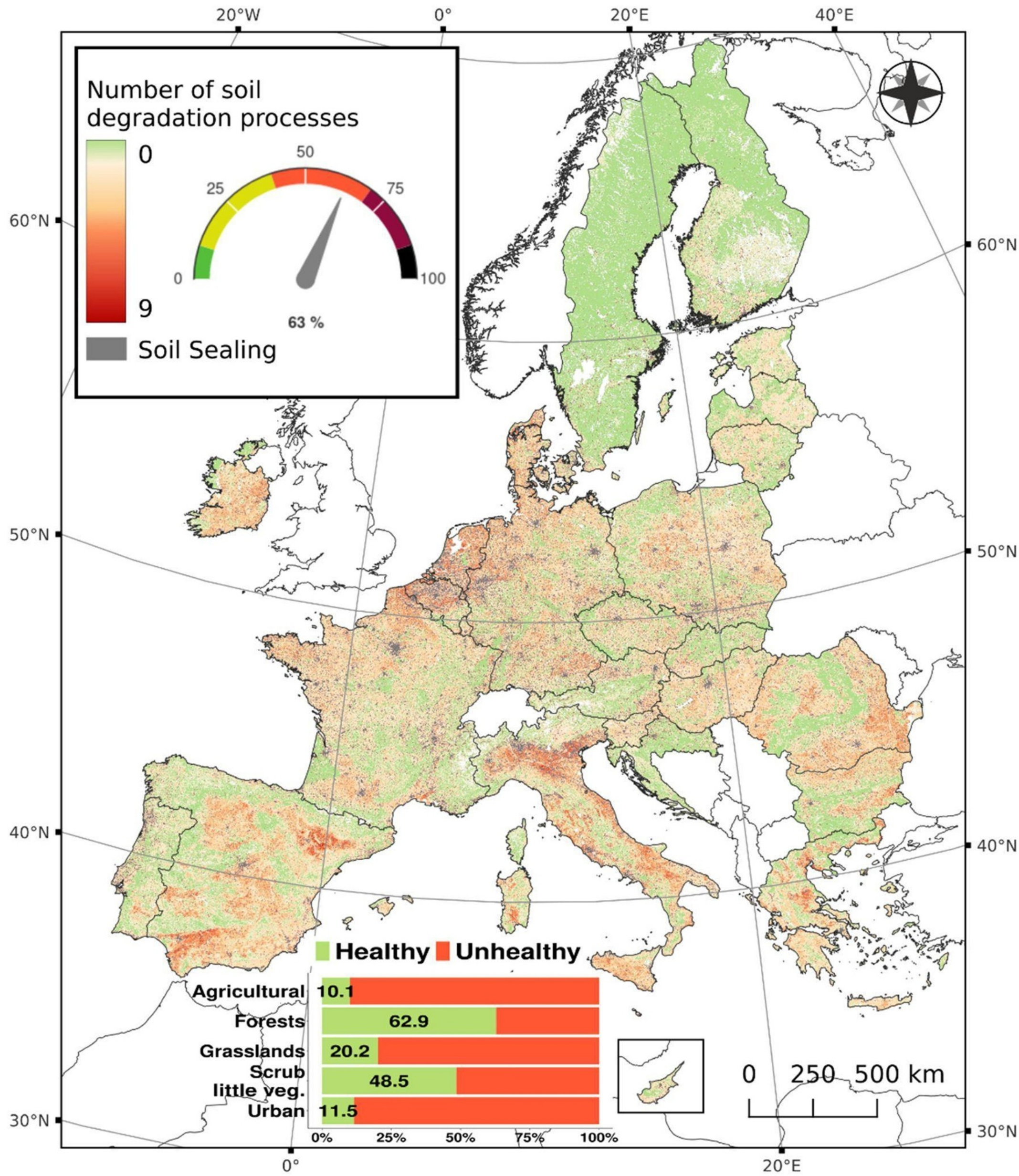


**Figure 24.** Convergence of evidence of soil degradation in the EU according to the EUSO Dashboard (after Panagos et al., 2024c).

ideal midpoint of 6.5. Ultimately, these four rescaled indicators are averaged with equal importance to yield an SDP value ranging from 0 (lowest degradation risk) to 1 (highest degradation risk).

Figure 25 compares soil degradation levels as assessed by (a) the EUSO Soil Health Dashboard, (b) the LMI, and (c) the SDP. Overall, the three maps are spatially consistent. Areas identified as having a high risk of degradation by the EUSO or LMI approaches generally also show elevated SDP values. However, whereas the EUSO and LMI typically rely on binary or threshold-based classifications to categorize soils as "degraded" or "not degraded," the SDP transitions the assessment to a continuous metric that measures the severity of soil degradation. This flexibility overcomes the shortcomings of the subjective thresholds employed in previous methodologies and provides a more objective proxy for tracking soil degradation intensity over time and across different land-use conditions.

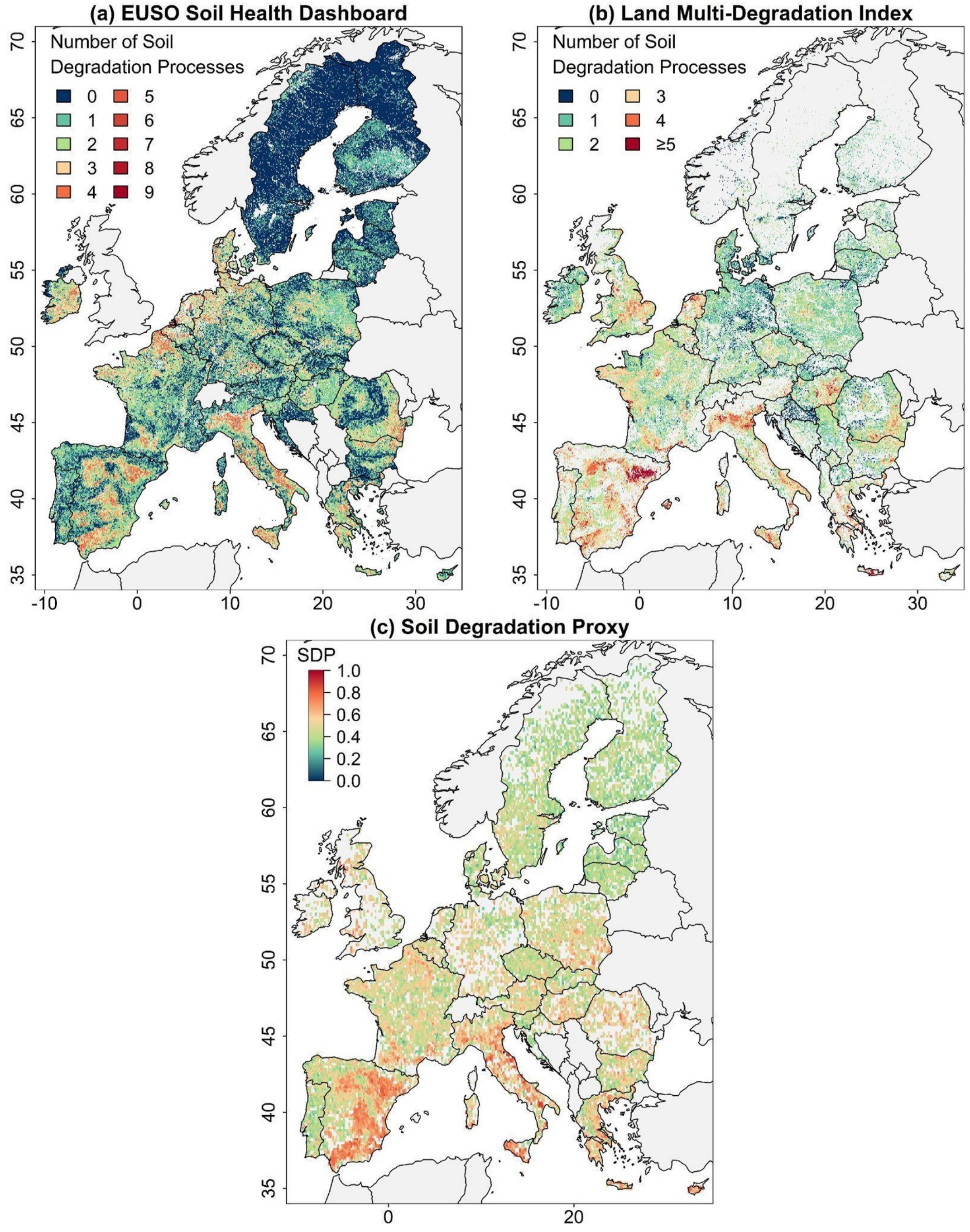


**Figure 25.** Comparison of soil degradation assessments across Europe using different methodologies. (a) The EUSO Soil Degradation Dashboard, (b) the Land Multi-Degradation Index (LMI), and (c) the Soil Degradation Proxy (SDP).

On a wider spatial scale that transcends Europe, there are also some global assessments of soil degradation that were performed in an integrated manner, in which multiple processes and environmental conditions were considered to examine soil and land degradation in general. For example, a previous study explored five key co-occurring land degradation processes (soil erosion, soil salinization, soil organic carbon decline, vegetation degradation, and aridity), which act synergistically across the world's arable environments (Prăvălie, Patriche, et al., 2021). Another integrated assessment of soil degradation can be found in the World Atlas of Desertification Report (last edition), which was focused on examining both causes and some processes that act in the complex mechanism of soil degradation through a global "convergence of evidence" approach (Cherlet et al., 2018). A final relevant example is a global study that investigated the synergistic effects of various environmental conditions potentially leading to soil degradation, using a multicriteria sensitivity model (Mediterranean Desertification and Land Use—MEDALUS) that is widely used in the multi-scale analysis of land susceptibility to degradation (Ferrara et al., 2020).

## 6. Sustainable Land Management Practices to Combat Soil Degradation

### 6.1. Soil Conservation and Agricultural Practices

Soil and agricultural practices which prioritize conservation are essential to protect natural resources and provide sustainable food production. These approaches support long-term productivity and improve resilience in the face of global challenges such as climate change and soil degradation.

Our commitment to sustainable agriculture reflects the urgent need to balance food production with environmental responsibility. It is an integrated approach aimed at producing food, fiber, and other resources in ways that are environmentally suitable, economically viable, and socially equitable (Trigo et al., 2021; Velten et al., 2015). The core objective of sustainable agriculture is to meet present needs without compromising the ability of future generations to do the same (Velten et al., 2015). A central pillar of this approach is sustainable land management, which directly addresses soil degradation. As pressures from climate change intensify, these strategies offer a pathway to secure food systems while conserving biodiversity, lowering greenhouse gas (GHG) emissions, and supporting rural livelihoods. By adopting conservation principles into farming systems, we aim to cultivate landscapes which are both productive and regenerative.

Soil health is foundational to sustainable agriculture. Practices that improve water conservation, reduce pollution, and enhance soil quality are key to maintaining environmental integrity (Shahmohamadloo et al., 2022; Tahat et al., 2020; Velasco-Muñoz et al., 2018). For sustainability to be viable, it must also be economically rewarding for farmers, ensuring profitability and stable incomes (Gurubasappa, 2022). Equally important is the welfare of farmers, workers, and communities, with safe working conditions and support for local economies.

Biodiversity conservation is another vital principle. Diverse farms contribute to ecological resilience, improving pest control, pollination, and soil function (Frison et al., 2011; Trigo et al., 2021). Efficient resource use, particularly water, energy, and nutrients, is also important. Technologies such as precision farming can enhance understanding of agricultural environments and optimize inputs.

To maintain and improve soil health, a variety of sustainable land management techniques can be used. Conservation tillage, including no-till and reduced-till methods, minimizes disturbance, preserving structure and reducing erosion (Sharma et al., 2024). Crop rotation disrupts pest cycles, enhances fertility, and reduces chemical dependency (Himmelstein et al., 2017; Raghavendra et al., 2020; Sharma et al., 2024). Agroforestry integrates trees and shrubs into farming systems, boosting biodiversity and soil stability (Raghavendra et al., 2020). Cover crops like clover and rye improve soil health, suppress weeds, and reduce herbicide use (Sharma et al., 2024).

Organic fertilizers such as compost and manure enhance microbial activity and nutrient cycling (Sharma et al., 2024). Integrated pest management, combining biological, cultural, and mechanical methods, reduces pesticide reliance and promotes ecological balance (Raghavendra et al., 2020; Sharma et al., 2024). Water management practices like drip irrigation and rainwater harvesting maintain moisture levels while preventing salinity and erosion (Sharma et al., 2024). Additional techniques such as terracing (Plekhov et al., 2024; Socci et al., 2019), contour farming (Adamsone-Fiskovica & Grivins, 2024; Garrity, 1999), mulching (El-Beltagi et al., 2022; R. Iqbal et al., 2020; S. Iqbal et al., 2020), and windbreaks (Brandle et al., 2004; Kort, 1988) offer targeted solutions for managing runoff, preventing erosion, and retaining nutrients.

Collectively, these practices encourage healthier soils that are more productive, resilient to climate change, and capable of sequestering greater amounts of soil organic carbon, contributing to reduced GHGs. They also improve biodiversity and water quality as well as mitigate risks from natural disasters such as floods and landslides. By integrating these strategies, agriculture can evolve into a more resilient and sustainable system that meets the needs of both current and future generations.

10.1029/2025RG000883

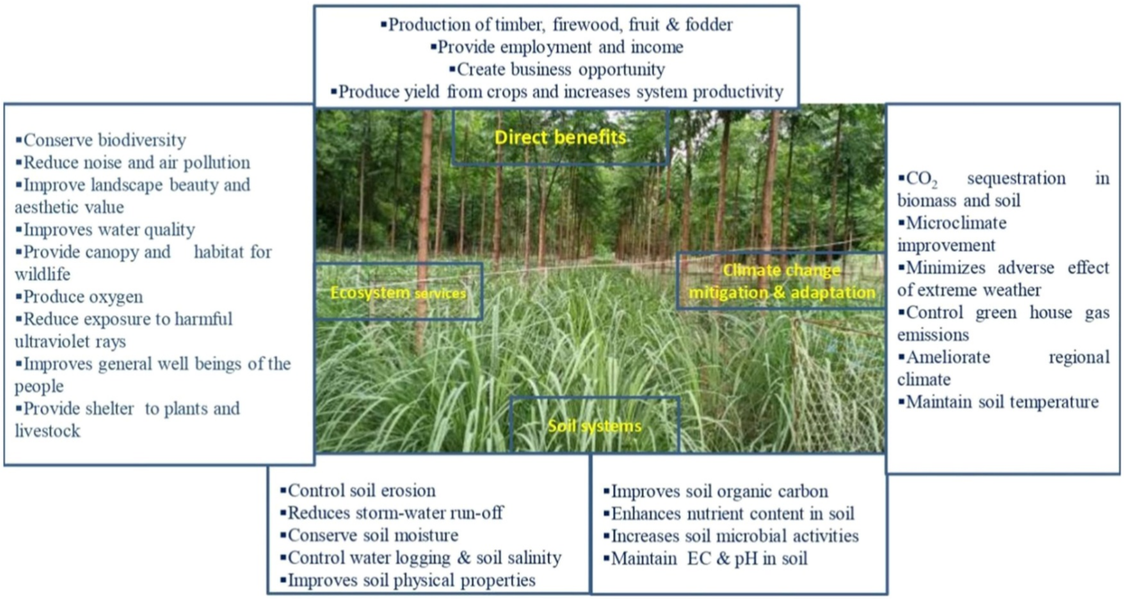


**Figure 26.** Influence of agroforestry on various direct and indirect benefits in the degraded lands (after Jinger et al., 2023).

### 6.2. Reforestation and Afforestation

Reforestation and afforestation (see Figure 26) are increasingly acknowledged as vital approaches for addressing long-term soil degradation (Cunningham et al., 2015; Khorchani et al., 2023; Luo et al., 2023). Initiatives aimed at restoring vegetation, such as China's Grain-for-Green Program (GFGP), have been effective in curtailing soil nutrient depletion. According to Zhao et al. (2020), the extent of eroded land diminished from 41.8% to 26.7%, thereby significantly alleviating erosion severity. Additionally, Chen et al. (2019) demonstrated that the transition from arable land to kiwifruit orchards led to a reduction in nitrogen and organic carbon loss, although it resulted in an increase in available phosphorus loss due to excessive fertilizer application. A global meta-analysis conducted by Shi et al. (2016) investigated variations in carbon (C), nitrogen (N), phosphorus (P), and sulfur (S) in the mineral soils of planted forests across various climatic regions, utilizing data from 139 studies. Their findings indicated that soil C and N experienced a slight decrease when grasslands were converted to forests, a moderate increase with the transformation of croplands, and a significant rise following the afforestation of barren lands. While total soil P levels remained stable, there was a marked decline in available P after the forestation of grasslands and croplands with nitrogen-fixing species.

Although afforestation offers multiple benefits, it also has certain limitations. For example, reductions in soil carbon or albedo, as well as increased fire severity (due to higher fuel loads and connectivity), can reduce the effectiveness of afforestation strategies for mitigating climate change. Additional negative impacts are also possible, such as reductions in native biodiversity and productivity, significant water yield losses, and alterations in nutrient cycles, which may exacerbate other global change drivers (Moyano et al., 2024). Therefore, while afforestation is a valuable tool for combating land degradation, it is equally important to consider and manage its potential negative consequences.

The improvement in soil quality or reduction in soil degradation also depends significantly on the vegetation restoration methods employed. For example, Qian et al. (2024) demonstrated that native tree species restoration, such as Larix principis-rupprechtii (FL), resulted in the most substantial improvements in soil quality, with higher levels of SOC, TN, and TP compared to exotic species and grassland restoration methods. In contrast, the introduction of exotic species, such as Pinus sylvestris var. mongolica (FP), showed minimal improvements. These findings highlight the superior effectiveness of native species in enhancing soil health and offer valuable insights for ecological restoration in similar transitional landscapes.

10.1029/2025RG000883

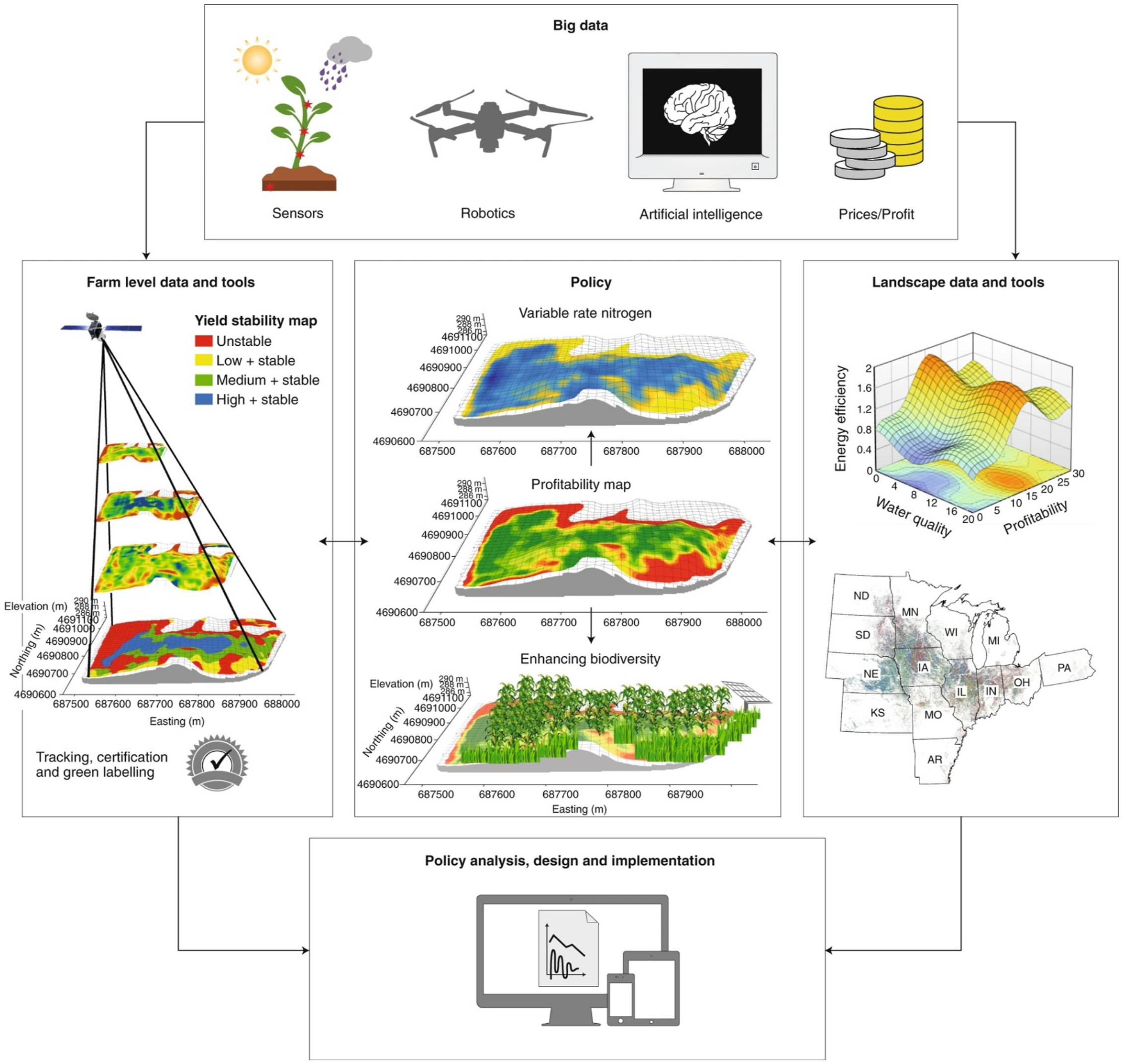


**Figure 27.** Digital agriculture in agricultural systems (after Basso & Antle, 2020).

### 6.3. Precision Agriculture

Since the beginning of the 21st century, precision agriculture has gained prominence as a significant research focus, with the goal of transforming the agricultural sector. By using cutting-edge technologies (e.g., artificial intelligence (AI), drones, robotics, and Internet of Things (IoT) sensors), precision agriculture seeks to assist farmers in optimizing water usage, reducing the requirement for chemical and fertilizer applications, and improving soil health management (Soussi et al., 2024; Xing & Wang, 2024). Consequently, these technological advancements are expected to promote more sustainable and efficient farming practices (Figure 27; Basso & Antle, 2020).

Water is an indispensable resource in the agricultural sector, and its judicious utilization is vital for promoting sustainable farming practices. Technologies such as artificial intelligence, drones, and Internet of Things (IoT) sensors are instrumental in enhancing water management through advanced irrigation systems and continuous monitoring (Duguma & Bai, 2025). While the application of chemical fertilizers and pesticides is common in agriculture, their overuse can result in environmental contamination and soil deterioration. Precision agriculture technologies facilitate the reduction of chemical inputs by allowing for targeted application and accurate monitoring (Idier et al., 2024). Moreover, healthy soil serves as the cornerstone of effective agriculture. These precision agriculture technologies help in the assessment and management of soil health by offering comprehensive insights into soil conditions and supporting sustainable farming methods. Table 3 provides a summary of how these advanced technologies can enhance precision agriculture.

### 6.4. Policies and Incentives to Address Soil Degradation

Numerous national initiatives are dedicated to safeguarding soils and enhancing soil health. In their analysis of national policies utilizing the FAO registry of soil-related legal instruments and governance legislation (SoiLEX), Smith et al. (2024) identified a total of 3,823 legal instruments and category combinations, with some instruments addressing multiple categories. Specifically, over 900 instruments focus on soil conservation, more than 600 on soil pollution, and between 100 and 400 on issues such as waterlogging, biodiversity loss, sealing, nutrient imbalance, soil organic carbon (SOC) loss, erosion, monitoring, soil quality, and restoration and remediation. Fewer than 100 instruments tackle compaction, acidification, salinization, and sodification (see Figure 28). Overall, 173 countries have enacted some form of legislation aimed at protecting and improving soil health, with Russia leading in the number of instruments, many of which are tailored to various regions within the country. Canada and Mexico also possess a diverse range of legal instruments, often specific to different regions in those nations (see Figure 29; Smith et al., 2024).

The issues surrounding soil degradation and its management are widely recognized. However, as noted by Carter (1977), even four decades after the Dust Bowl in the United States, the recurrence of erosion indicates that soil degradation remains a persistent challenge. This situation persists despite significant investments in research, practical programs, and the creation of legal frameworks. Particularly, the causes, mechanisms, and management strategies for all types of soil degradation are well understood.

Approaches to managing soil degradation generally fall into several categories (Hannam, 2020; Weersink & Wossink, 2005):

1. Education programmes; related to educating landowners on the best management practices,
2. Legislation and penalties,
3. Government incentive programs (e.g., payments to remove non-arable, or erosion prone land from production), and
4. Market-based mechanisms such as payments for ecosystem services (e.g., carbon mitigation, water quality).

Quite often land-managers are well aware of the soil degradation problems that they are confronted with, but either choose to ignore the problem or cannot deal with the problem due to a lack of capital. In some areas of the world soil degradation occurs because of extreme poverty (e.g., extensive grazing systems) or as an externality of other land-uses (e.g., deforestation). Several studies have examined the adoption gap by land-managers of soil health practices (Carlisle, 2016).

### 6.5. Case Studies of Successful Restoration

Effective land management has been instrumental in reversing land degradation across various regions globally. This section will present two case studies, highlighting the historical degradation faced in these areas, detailing the transformative measures implemented to achieve these positive changes.

*Case Study 1:* The Loess Plateau, China

Situated in north-central China, the Loess Plateau represents one of the country's most important geographical areas. This elevated region spans the provinces of Shanxi, Shaanxi, Gansu, and Henan, with an average elevation of approximately 1,200 m and covering around 635,000 km squared. The soil of the Loess Plateau consists of fine, yellow-buff silt known as loess, which was developed from dust carried by wind and deposited during the Quaternary period. These sedimentary layers can reach thicknesses of up to 100 m and are increasingly susceptible to erosion. This vulnerability is exacerbated by sparse vegetation, heavy rainfall, particularly during the summer months, and the region's rugged topography (Tang & Cai, 1991; Xia & Tang, 1992).

Historically, the Loess Plateau has served as a vital foundation for numerous Chinese societies, facilitating agricultural activities that provide and produce food, fuel, and fiber. However, centuries of extensive deforestation and unsustainable agricultural practices have resulted in increased soil erosion, land degradation, and diminished agricultural output (Encyclopaedia Britannica, 2024). By the late 1980s, the region was characterized by a huge significant ecological crisis as a result of extensive barren land areas and the formation of deep gullies.

**Table 3**
*Ways in Which Advanced Technologies Can Aid Precision Agriculture*

| Soil related issue for precision agriculture | Technologies |
|---|---|
| Optimizing Water Usage | *AI-Powered algorithms which can aid Smart Irrigation*: Analyses conducted from various sources (e.g., soil moisture sensors, weather forecasts, and historical crop data) to determine the precise water needs of specific crops. A data-driven approach ensures that water is only applied when and where appropriate, reducing water wastage and improving crop yields (Vallejo-Gómez et al., 2023; Wei et al., 2024). |
| | *Drones for Water Management*: Equipped with multispectral and thermal cameras to capture detailed images of fields, identifying areas where moisture levels are variable. This helps farmers target irrigation efforts more precisely, ensuring that water is distributed more evenly across the field. Drones can also help monitor irrigation systems for leaks or other discrepancies (Guebsi et al., 2024). |
| | *IoT Sensors for Real-Time Monitoring*: Sensors placed in the soil to continuously monitor a range of environmental factors (e.g., moisture levels, temperature). IoT sensors can provide real-time data and monitoring for farmers, enabling them to make informed decisions about irrigation schedules and water management (Sishodia et al., 2020; Tornese et al., 2024). |
| Minimize chemical and fertilizer use | *AI-Driven Algorithms to assess Pest and Disease Management*: Analyses data from a range of sources (e.g., drones, sensors, and satellite imagery) to detect early signs of pest infestations and diseases. By rapidly identifying affected areas, farmers can apply pesticides only where required, reducing overall chemical usage. AI-Driven algorithms can help this become a targeted approach which minimizes environmental impact and lowers costs (Guebsi et al., 2024; Li et al., 2024). |
| | *Drones for Precision Spraying*: Equipped with advanced sensors and AI-driven software to perform precision spraying of fertilizers and pesticides. Drones can identify specific areas that require treatment and apply chemicals with high accuracy, preventing over-application and reducing waste. This method ensures protecting the crops and the environment (Zhang, Bai, et al., 2024; Zhang, Jia, et al., 2024). |
| | *IoT Sensors for Soil and Crop Health Monitoring*: Provide continuous data and information on a range of factors (e.g., soil nutrient levels, crop health, and environmental conditions). IoT sensors allow farmers to tailor their practices to the specific needs of their crops, applying the right amount of nutrients when appropriate. Through this approach, crop growth is seen to be improved while the risk of nutrient runoff and soil contamination is reduced (Chaudhari et al., 2022; Tornese et al., 2024). |
| Improving soil health management | *AI-Powered Soil Analysis*: Processes data from a range of information (e.g., soil sensors, drones, and satellite imagery) in order to assess soil health parameters (e.g., nutrient status, pH, and soil organic matter/carbon). AI-Powered soil analysis can help farmers understand the specific needs of their soil and implement appropriate management practices. The AI can recommend certain strategies (e.g., crop rotations, cover cropping, and organic amendments) to improve soil fertility and structure (Awais et al., 2023; Huere-Peña et al., 2024). |
| | *Drones for Soil Monitoring*: Equipped with hyperspectral and thermal sensors to capture high-resolution images of fields, showing the variations in soil properties. Allows farmers to identify areas categorized as having poor soil health and take appropriate action(s) (e.g., adjusting irrigation, applying organic matter, or implementing erosion control measures). It can also facilitate effective soil sampling by offering precise locations for collection of samples (Najdenko et al., 2024). |
| | *IoT Sensors for Continuous Soil Monitoring*: Can provide real-time data (e.g., moisture levels, temperature, and nutrient content) in the soil. Allows farmers to track changes in soil health over time and make appropriate decisions to maintain or improve soil quality based on this data-driven approach. Furthermore, farmers can receive actionable recommendations for soil management, ensuring that their practices are both sustainable and effective (Guo, 2021; Tornese et al., 2024). |

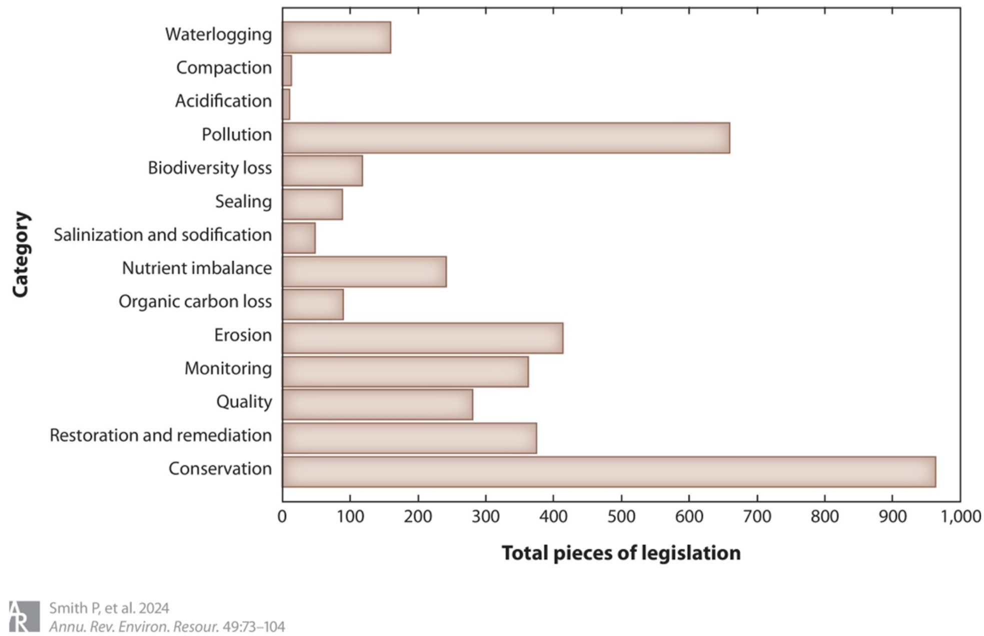


**Figure 28.** Soil-related legal instruments and soil governance legislation by category, showing totals across all countries. Data are from SoiLEX, a global database from the Food and Agriculture Organization of the United Nations that aims to facilitate access to information on existing legal instruments on soil protection and prevention of soil degradation (after Smith et al., 2024).

In response to severe environmental degradation, the Chinese government initiated a series of extensive restoration projects in the 1990s, focusing on reforestation, terracing, and sustainable land management practices (Tsunekawa et al., 2014). These initiatives aimed to stabilize soil, mitigate erosion, and improve the ecological conditions of the region while promoting sustainable development (Lu et al., 2012). In 1994, the Chinese government, in partnership with the World Bank, launched the Loess Plateau Watershed Rehabilitation Project (Chen et al., 2004; World Bank, 2006). This initiative was recognized as one of the largest erosion control programs worldwide, with the goal of transforming the degraded landscape into a sustainable agricultural zone, benefiting both the environment and the livelihoods of local populations.

The reforestation efforts concentrated on planting indigenous tree species to stabilize the soil and minimize erosion. Additionally, terracing techniques were created to reshape the landscape into stepped formations, preventing runoff and retaining water (Zhang et al., 2017). Furthermore, sustainable agricultural practices, such as crop rotation and organic fertilizer application, were encouraged to enhance soil fertility and productivity (Lu et al., 2012). The Loess Plateau Watershed Rehabilitation Project also implemented policies prohibiting deforestation on steep slopes and free-range grazing, both of which had previously exacerbated degradation (World Bank, 2006). These strategies, alongside community education and active participation, resulted in notable improvements within the region. Over time, there was an increase in vegetation cover, a reduction in soil erosion, and a rise in agricultural productivity (Liu et al., 2018), ultimately enhancing local livelihoods and restoring ecological balance and biodiversity in the Loess Plateau area.

The restoration of the Loess Plateau faced numerous and significant obstacles. The region experienced severe degradation, complicating the identification and establishment of vegetation necessary for soil stabilization (Zhao et al., 2024). Additionally, the limited availability of water and resources posed challenges for both reforestation and agricultural initiatives (Zhao et al., 2018). Ensuring an adequate water supply for newly planted trees and crops was essential. Furthermore, many local communities depended on unsustainable agricultural practices for their livelihoods, making the shift to more sustainable methods a challenge. This required various incentives, motivations, and financial assistance (Zhang, 2018). The execution of activities such as large-scale terracing and reforestation demanded considerable technical expertize and resources. The rugged landscape and loess deposits further complicated the restoration efforts. Finally, engaging the community in this restoration initiative presented its own set of challenges (Wang, Du, et al., 2023; Wang, Min, et al., 2023; J. Wang, et al., 2023). This involved

Reviews of Geophysics 10.1029/2025RG000883

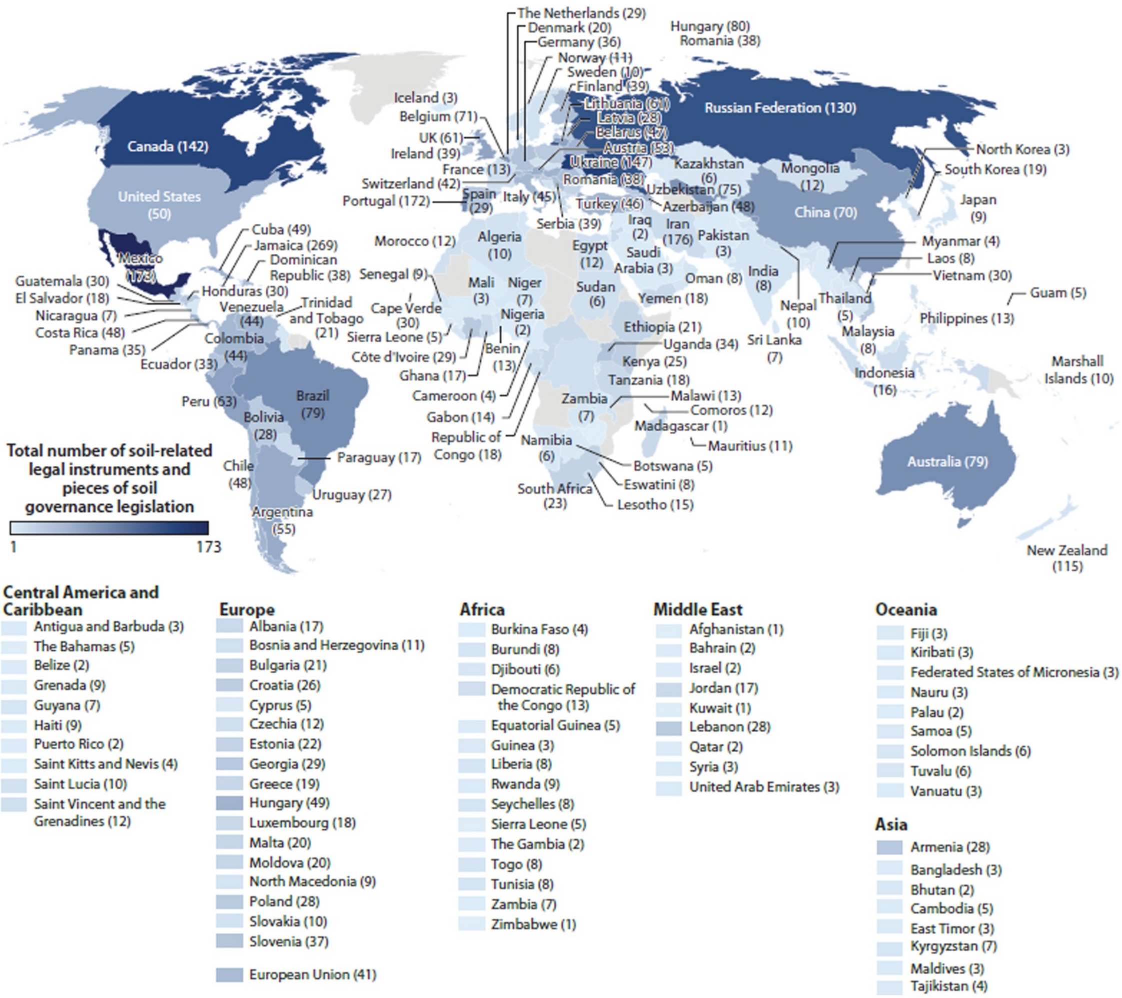


**Figure 29.** Global map showing the total number of soil-related legal instruments and pieces of soil governance legislation in each country included in the SoiLEX database. In areas of the map where labels would be overlapping and illegible, countries appear below the map in alphabetical order by region (after Smith et al., 2024).

building trust, cooperation, and support among local residents, as well as educating farmers on sustainable practices. The active involvement of these stakeholders was crucial for the project's success. Despite these hurdles, the restoration project of the Loess Plateau achieved notable success, largely due to the backing of the Chinese government, international partnerships, and community participation.

The restoration of the Loess Plateau, completed in 2002, brought about significant environmental and economic changes in the region. It is estimated that the incomes of most farmers more than doubled, increasing from around \$70 to \$200 per year per person, primarily due to enhanced agricultural productivity (World Bank,; 2007). From an ecological perspective, the project led to a substantial increase in vegetation cover, with perennial plants doubling in number. The initiatives of reforestation and the adoption of sustainable farming techniques contributed to a marked decrease in soil erosion and sedimentation in waterways, resulting in a reduction of sediment flow into the Yellow River by over 100 million tons each year (Zhao et al., 2020). This not only mitigated flooding risks but also improved water storage for agricultural use and local communities (Zhang et al., 2017).

Moreover, the project fostered an increase in employment rates as both on-farm and off-farm job opportunities became more accessible. The employment rate rose from 70% to 87%, with notable improvements in job opportunities for women (Khan et al., 2020; World Bank, 2007). Food security also saw enhancements, as terracing

10.1029/2025RG000883

and sustainable agricultural practices led to more reliable and higher crop yields, thereby reducing the reliance on emergency food aid during periods of drought (Kosmowski (2018)).

*Case Study 2:* The Tigray Region, Ethiopia

The Tigray Region, situated in northern Ethiopia, has historically played a crucial role in the development of Ethiopian civilization. Covering an area of approximately 53,000 square kilometers, it is bordered by Eritrea to the north, the Amhara region to the south, the Afar region to the east, and Sudan to the west. The region's topography is marked by elevated plateaus, steep mountains, and profound valleys, with altitudes ranging from 1,500 to 3,300 m. The Tekeze and Mareb rivers traverse the area, serving essential functions in agriculture and water provision. Despite its difficult landscape, Tigray has long been a center for agricultural practices, particularly in farming and livestock rearing (Gebru et al., 2024). In recent years, Tigray has encountered significant environmental and conflict-related issues.

The Tigray Region of Ethiopia has been the focal point of numerous restoration efforts aimed at mitigating severe land degradation and enhancing the livelihoods of local communities. A prominent initiative in this regard is the EthioTrees project (EthioTrees, 2020). This highland region has experienced various environmental challenges, including drought, erosion, and overgrazing, which have significantly exacerbated land degradation. Initiated in 2016 and completed in 2022, EthioTrees emphasizes natural regeneration through the planting of indigenous trees and the creation of "e*xclosures*" to restrict livestock grazing.

The active participation of the community has been essential, with local inhabitants providing labor and receiving compensation from carbon credit revenues. These funds have been reinvested into the community to support diverse development initiatives (Mekuria et al., 2011). This strategy has not only contributed to environmental restoration but has also bolstered the economic stability of the local populace. The achievements of the EthioTrees project highlight the effectiveness of integrated, community-oriented approaches to land restoration, offering important insights for other regions facing comparable challenges (Teshome et al., 2016).

Restoration initiatives in Tigray encountered substantial challenges, akin to those faced in the Loess Plateau Watershed Rehabilitation Project in China. A primary obstacle was the severe environmental degradation caused by extensive deforestation, overgrazing, and unsustainable agricultural methods (Nyssen et al., 2009). These factors hindered the establishment of vegetation and the stabilization of soil. The Tigray region is also susceptible to frequent and intense droughts, which significantly complicates reforestation and agricultural efforts (Kassa et al., 2022). Consequently, securing a reliable water supply for newly planted crops and trees was essential for the success of these initiatives. Moreover, many local communities depended on unsustainable farming practices for their livelihoods, necessitating financial support and incentives to facilitate a shift towards sustainable methods for effective restoration (Mekuria et al., 2011). The implementation of large-scale restoration activities, such as terracing and reforestation, required specialized technical knowledge and resources (Nyssen et al., 2015), further complicated by the region's uneven terrain. Lastly, building trust and garnering support from local communities was vital for the success of the projects (Wang, Du, et al., 2023; Wang, Min, et al., 2023; J. Wang, et al., 2023). By educating farmers on sustainable practices and actively involving them in the restoration process, the overall effectiveness of the project was significantly enhanced.

Despite these challenges, the restoration efforts in Tigray achieved notable progress, attributed to the collaborative efforts of the local Ethiopian government, communities, and international partners. The implementation of exclosures and reforestation efforts resulted in enhanced vegetation cover, which contributed to soil stabilization, diminished erosion, and bolstered overall ecological health (Tefera et al., 2024). Additionally, the construction of terraces and other conservation measures improved soil fertility and moisture retention, thereby augmenting agricultural productivity and resilience to drought conditions. These restoration activities also fostered increased biodiversity, particularly through the proliferation of various native plant species, while simultaneously enhancing habitat conditions for diverse local wildlife (Solomon et al., 2024). Furthermore, projects like EthioTrees generated economic advantages for local communities by creating numerous job opportunities and enhancing livelihoods (Hagazi, Gebrekirstos, et al., 2020; Hagazi, Gebremedhin, et al., 2020), which in turn boosted agricultural output and contributed to greater food security.

## 7. Big Data Analytics for Studying Soil Degradation

### 7.1. Remote Sensing and GIS

Technological advancements, particularly the accessibility of high-resolution remote sensing data, Geographic Information System (GIS) software, and platforms such as Google Earth Engine, have significantly enhanced the modeling and mapping of soil degradation processes (Khasanov et al., 2023). The ability to quantify and map various soil degradation phenomena, including soil erosion, landslide susceptibility, heavy metal contamination, soil compaction, salinity, and the depletion of nitrogen and phosphorus, has markedly progressed over time. A notable instance of this progress is the development of soil erosion models, such as the Revised Universal Soil Loss Equation (RUSLE). Panagos et al. (2014) emphasized the advancements in the vegetation factor and illustrated the application of satellite data from sources like NOAA-AVHRR (1.1 km), SPOT-VGT (1 km), MODIS (500 m), ENVISAT MERIS (300 m), and Proba-V (333 m) for vegetation and land cover mapping at low to moderate resolutions. Furthermore, medium- and high-resolution sensors, including Landsat-8/9 (30 m) and Sentinel-2 (10–20 m), offer superior imagery for monitoring vegetation, soil, and water, benefiting from enhanced spectral bands and more frequent revisit intervals. Similar progress is evident in digital elevation models (DEMs), which serve as critical inputs for soil hydrological models. Contemporary unmanned aerial vehicles (UAVs) equipped with high-resolution cameras can now capture imagery at the centimeter scale, facilitating the creation of ultra-high-resolution DEMs that are applicable in mapping soil degradation processes (Backes & Teferle, 2020).

Shokri et al. (2024) have delineated the primary factors contributing to soil salinization across different scales, which encompass climate, geomorphology, hydrology, land use, and anthropogenic influences. The utilization of remote sensing data pertaining to these factors is crucial for the effective mapping of soil salinization. Particularly, the volume of research in this area has surged from 14 studies in 2014 to 81 in 2022, employing various satellite technologies such as RADAR, Hyperspectral, Sentinel-2, Sentinel-1, UAVs, MODIS, and Landsat (Sahbeni et al., 2023). The application of UAVs equipped with multispectral and hyperspectral sensors for the purpose of mapping soil salinization is on the rise (Hu et al., 2019; Zhu et al., 2021), although its current application is primarily confined to field and catchment scales.

In recent times, numerous studies have concentrated on the mapping of soil compaction at the field level through remote sensing techniques. For instance, Demattê et al. (2010) assessed soil density by analyzing spectral reflectance to investigate the impacts of compaction. Their findings indicated that soils subjected to artificial compaction exhibited greater spectral intensity in comparison to those that were less compacted, revealing a linear relationship between spectral data and soil bulk density. Additionally, Mishra et al. (2020) detected transient water bodies throughout West Africa utilizing medium- and high-resolution satellite imagery. Straffelini et al. (2021) employed UAV-based Structure from Motion (SfM) technology to identify regions susceptible to surface water accumulation in an agricultural field in Italy. Furthermore, Amanor et al. (2024) utilized a hyperspectral sensor to map the spatial distribution of soil compaction, offering significant insights for informed tillage practices.

Remote sensing products serve as valuable covariates for the mapping of heavy metal concentrations, which include elements such as arsenic (As), cadmium (Cd), chromium (Cr), copper (Cu), mercury (Hg), lead (Pb), zinc (Zn), antimony (Sb), cobalt (Co), and nickel (Ni). For instance, studies by Ballabio et al. (2018, 2021, 2024) and Toth, Hermann, Szatmari & Pasztor. (2016) employed the MODIS Enhanced Vegetation Index and Digital Elevation Model (DEM) data as covariates to map heavy metals utilizing the LUCAS data set. In recent developments, hyperspectral imagery has gained traction for the detection of heavy metals in soil samples. Research conducted by Guo et al. (2022) and Q. Yang et al. (2024), N. Yang et al. (2024) utilized Gaofen-5 (GF-5) hyperspectral images to assess heavy metal concentrations in opencast coal mines in Northern China and urban soils, respectively, revealing a strong correlation between the predicted and actual concentrations.

### 7.2. Soil Databases and Models

Numerous point databases, including WOSIS (Batjes et al., 2024), LUCAS (Orgiazzi et al., 2018), SPADE (Kristensen et al., 2019), and NABODAT (Rehbein et al., 2011), are essential for analyzing soil degradation on both regional and global levels. For instance, WOSIS has been extensively utilized in various research efforts, such as mapping bulk density as an indicator of compaction in Mediterranean agro-ecosystems (Schillaci et al., 2021), modeling soil salinity (Shi et al., 2023), monitoring changes in global soil salinity (Ivushkin et al., 2019), and creating multi-hazard maps (Pouyan et al., 2021). A detailed compilation of existing point soil databases is accessible through SoilHydroDB and SoilChemDB, which can serve as valuable resources for investigating soil degradation.

LUCAS stands out as the sole database that encompasses three distinct time periods, 2009, 2015, and 2018, and its Soil Module represents the only harmonized and systematic soil survey across the European Union (Orgiazzi et al., 2018), rendering it exceptionally valuable for analyzing trends in land degradation. In 2022, approximately 38,000 soil samples were gathered, which is double the amount collected during the 2018 campaign (19,000 samples), although this data has not yet been made publicly accessible (Panagos et al., 2024c, 2024d). LUCAS is extensively employed in the formulation of soil degradation indicators and the assessment of soil health. Furthermore, this data set is frequently utilized for the mapping of soil degradation indicators through remote sensing and machine learning techniques (Ballabio et al., 2021, 2024). Some of these indicators, along with additional ones, indicate that nearly 60%–70% of soils in Europe are affected by one or more soil degradation processes and can be considered unhealthy (Panagos et al., 2024c). Recently, De Rosa et al. (2024) leveraged LUCAS data to estimate soil organic carbon (SOC) loss in both cropland and grassland across Europe from 2009 to 2018.

These data sets have facilitated the creation of soil property maps, which are routinely employed for soil degradation mapping. For instance, WOSIS data contributed to the development of SOILGRIDS, a globally recognized soil data set with a spatial resolution of 250 m (Poggio et al., 2021). Additionally, Borrelli et al. (2017a) utilized this data to produce a global soil erosion map. Similarly, spatial maps of physical properties (Ballabio et al., 2016) and chemical properties (Ballabio et al., 2019) were derived from LUCAS data and are widely applied throughout Europe. The SSURGO soil survey data from the United States was instrumental in creating the POLARIS soil property maps, which have a spatial resolution of 30 m (Chaney et al., 2016). More recently, Nauman et al. (2024) developed the Soil Landscapes of the United States (SOLUS) database at a 100-m spatial resolution, incorporating multiple U.S. soil surveys. Likewise, a high-resolution organic carbon map was generated using the Soil Data Federator, developed by CSIRO in Australia (Wadoux et al., 2023).

These data sets play a crucial role in the modeling of soil degradation phenomena, including soil erosion, runoff, carbon depletion, compaction, and salinity, at both regional and global levels. Borrelli et al. (2021) conducted a review of various soil erosion models, all of which necessitate soil data as a fundamental input. Their findings indicated that 31% of researchers employed the Revised Universal Soil Loss Equation (RUSLE) and the Universal Soil Loss Equation (USLE), both of which utilize soil property data to assess soil erodibility, a measure of soil's vulnerability to erosion.

### 7.3. Machine Learning and AI for Soil Degradation

Machine learning and artificial intelligence are increasingly recognized as influential instruments in the field of soil science and the assessment of land degradation processes (Davydzenka et al., 2022; Hassani et al., 2024; Hengl et al., 2017; Minasny et al., 2024; Tahmasebi et al., 2020; Yulianto et al., 2023). Machine learning algorithms have demonstrated high efficiency in identifying the key drivers of soil degradation, such as climatic shifts, land use changes, and alterations in soil biochemical properties (Chen et al., 2024). These models integrate diverse data sets, ranging from remote sensing imagery and climate factors to field observations, allowing for a more comprehensive and scalable assessment of soil health. Integration of multi-scale data, in particular, Earth Observation data, with AI-powered tools can fill the gap between local soil conditions and global degradation trends, offering continuous maps of soil degradation processes worldwide. Figure 30 presents a typical workflow for an AI-driven framework to predict soil properties and soil health index.

Barakat et al. (2023) successfully employed various machine learning techniques, including random forest, k-nearest neighbor, and extreme gradient boosting, to evaluate soil erosion vulnerability across 3,034 erosion sites. Khosravi et al. (2023) utilized deep learning methodologies, such as convolutional neural networks (CNN), recurrent neural networks (RNN), and long short-term memory networks (LSTM), to forecast soil water erosion. Ballabio et al. (2021, 2024) implemented deep neural networks and ensemble machine learning models to map the distribution of heavy metals, leveraging the LUCAS data set. Bakhshian et al. (2025) developed a deep learning model to predict near-term soil moisture dynamics using soil parameters and climatic variables, which were evaluated against field measurements (Hohenegger, Ament, et al., 2023; Hohenegger, Korn, et al., 2023). The results illustrated the robustness and efficiency of machine learning formalism for the spatio-temporal prediction

10.1029/2025RG000883

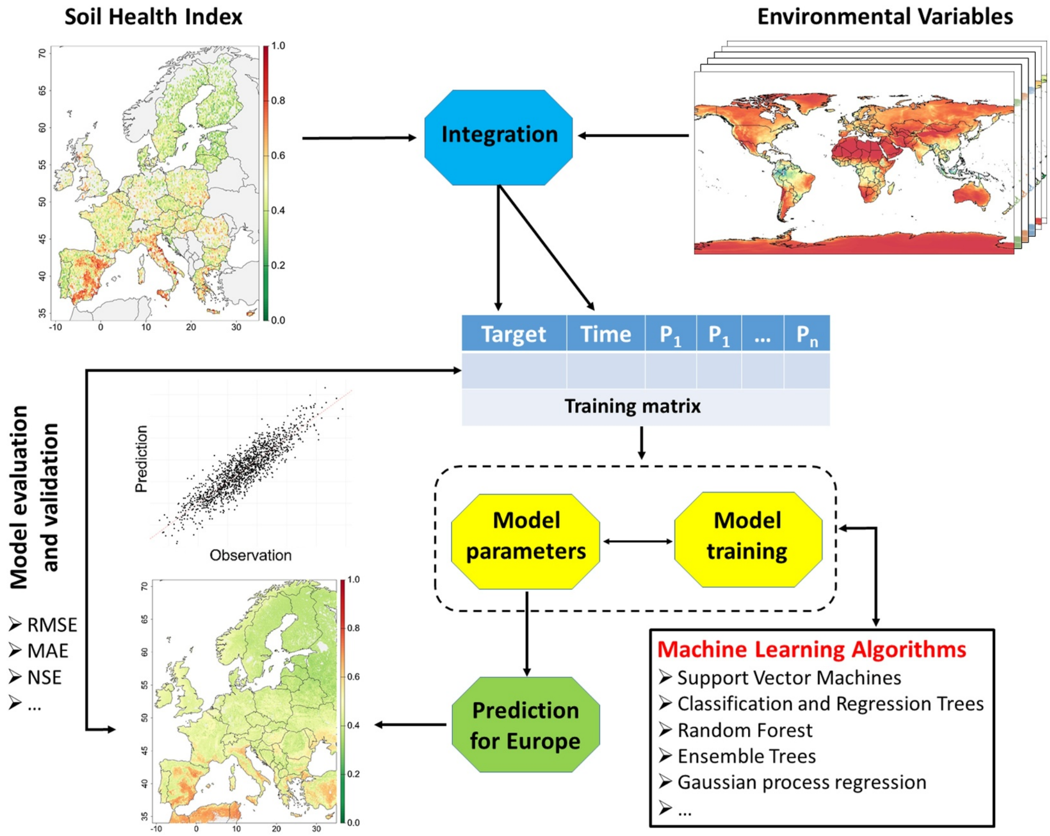


**Figure 30.** A typical workflow for an AI-driven framework to predict soil health.

of soil moisture. Achu et al. (2023) applied machine learning to model landslide susceptibility, generating accurate maps that identify areas at high risk. Additionally, Hassani, Azapagic & Shokri. (2020) created tree-based predictive models to assess global soil salinity and sodicity over a span of four decades, achieving a resolution of approximately 1 $km^2$. Collectively, these investigations underscore the considerable promise of artificial intelligence and machine learning in the mapping of land degradation processes.

Artificial Intelligence (AI) and Machine Learning (ML) play a crucial role in pinpointing areas of soil degradation, thereby enabling policymakers to respond swiftly (Hassani et al., 2021). Gholami et al. (2024) employed deep learning techniques to assess global susceptibility to wind erosion, uncovering that 26.1% of the world's land is at a very high risk. According to Hassani, Azapagic & Shokri. (2020), from 1980 to 2018, approximately 11.73 million square kilometers of non-frigid soils were affected by salinity in at least 75% of the observed years, which included 0.16 million square kilometers of agricultural land. Ballabio et al. (2024) detected cadmium contamination hotspots in Ireland, Poland, and Slovenia, while Ballabio et al. (2021) reported elevated mercury concentrations near mining operations in Spain, Italy, Slovenia, and Slovakia. Saha et al. (2020) identified that 7% of the Hinglo River basin in India is highly vulnerable to gully erosion. A multitude of studies underscores the application of AI and ML in recognizing both current and potential land degradation hotspots (Hassani et al., 2021). Furthermore, Hassani et al. (2021) forecasted future soil salinity levels for the periods 2031–2060 and 2071–2100 under various climate scenarios (RCP 4.5, RCP 8.5, SSP 2–4.5, SSP 5–8.5), pinpointing salinization hotspots in the arid regions of South America, Australia, Mexico, the southwestern United States, and South Africa, while anticipating a decrease in salinity in areas such as the northwestern United States, the Horn of Africa, Eastern Europe, Turkmenistan, and Kazakhstan. In a similar vein, Panagos et al. (2022) projected a global increase in rainfall erosivity for the periods 2041–2060 and 2061–2080, estimating a rise of 26.2%–34.3% compared to the baseline of 2010, a change attributed to climate change. This escalation could result in a 30%–66% increase in soil erosion rates by 2070. Additionally, Park and Lee (2021) utilized ML to assess landslide

10.1029/2025RG000883

susceptibility in South Korea under climate change scenarios (RCP 8.5), noting an upward trend in susceptibility over time.

The ability to obtain real-time data on rainfall, soil samples, and various remote sensing products now enables the continuous enhancement of soil degradation maps and real-time assessments through the application of evolutionary machine learning and artificial intelligence. For instance, Panagos et al. (2017) produced a global map of rainfall erosivity based on information from 3,625 monitoring stations, which was subsequently updated in 2023 (Panagos et al., 2023) with the addition of 314 new stations. Kumar et al. (2021) developed a straightforward ML framework for modeling streamflow in real time, utilizing satellite-derived rainfall and soil moisture data while controlling for other variables. Likewise, Ghadekar et al. (2024) employed the UNET deep learning architecture to identify real-time soil erosion by analyzing satellite imagery sourced from Google Earth Engine.

Although these technologies are new tools for environmental analysis, it is essential to recognize their limitations. Field validation and expert interpretation remain irreplaceable components of soil degradation assessment. The integration of AI-driven approaches should be seen as complementary, not a substitute, for ground-based measurements and domain knowledge.

## 8. Soil Degradation and the UN Sustainable Development Goals (SDGs)

Soil degradation represents a major challenge to soil health, directly impacting the achievement of the United Nations SDGs, which aim to address critical issues such as poverty, hunger, public health, and environmental sustainability (Bouma & Montanarella, 2016; Shokri et al., 2025). This situation underscores the urgent need for effective strategies to safeguard soils from degradation, as healthy soil is fundamental for food production, the preservation of water resources, the provision of ecosystem services, and climate regulation, all vital components for a sustainable future (Robinson et al., 2014; Smith et al., 2015). These initiatives are essential for the sustainability of soil and natural resources. The FAO has highlighted that healthy soil is "a prerequisite to achieving the SDGs."

As extensively examined in prior research (Hou et al., 2020; Lal et al., 2021), the health of soil is intricately connected to several United Nations SDGs. Healthy soils are vital for agricultural productivity, which in turn provides income and financial security for millions globally. Consequently, soil health is essential for achieving SDG 1 (No Poverty), as the degradation of soil diminishes agricultural output, thereby intensifying poverty and economic difficulties. Likewise, SDG 2 (Zero Hunger) is inherently linked to soil health; nutrient-dense soils promote plant growth and crop yields, whereas degraded soils lead to diminished harvests, heightening the risk of malnutrition, particularly among vulnerable populations. The advancement of innovative strategies for sustainable soil management and the monitoring of soil conditions is crucial for ensuring long-term food security and resilience. Furthermore, degraded soils and eroded landscapes contribute to the occurrence of sand and dust storms, increase exposure to harmful pollutants, and limit access to nutritious crops, all of which have direct implications for public health, thereby connecting soil degradation to SDG 3 (Good Health and Well-being). Additionally, in numerous areas, women represent a significant portion of the agricultural labor force. When soil degradation leads to reduced crop yields and agricultural efficiency, it disproportionately impacts women's income, economic autonomy, and food security, underscoring its relevance to SDG 5 (Gender Equality). Healthy soils also function as natural water filtration systems, preventing contamination and enhancing access to clean drinking water, which is vital for SDG 6 (Clean Water and Sanitation). In contrast, soil degradation results in erosion and water pollution, adversely affecting sanitation. Moreover, soils are integral to sustainable industrial practices, especially in the fields of agriculture and construction.

The quality of soil influences the frequency of natural disasters, which can compromise infrastructure and economic stability, thereby highlighting its connection to Sustainable Development Goal 9 (Industry, Innovation, and Infrastructure). Additionally, the degradation of soil adversely affects agricultural livelihoods, exacerbating social inequalities and economic instability, which are critical aspects of Sustainable Development Goal 10 (Reduced Inequalities). Furthermore, deteriorating soil conditions heighten the risk of natural disasters such as landslides and dust storms, presenting substantial challenges for urban environments, thus underscoring the importance of soil health for Sustainable Development Goal 11 (Sustainable Cities and Communities). Sustainable Development Goal 12 (Responsible Consumption and Production) relies on the maintenance of healthy soils to support sustainable food systems. As soil degradation diminishes agricultural productivity, effective soil management becomes essential for ensuring long-term agricultural viability. In addition, soils are crucial for climate regulation, influencing the water cycle, land-atmosphere interactions, and vegetation dynamics. They serve as carbon sinks; however, when degraded, they can release stored carbon, thereby impacting climate, which is a key focus of Sustainable Development Goal 13 (Climate Action). The implications of soil health for Sustainable Development Goal 15 (Life on Land) are significant, as soil degradation threatens biodiversity, disrupts ecosystems, and diminishes ecosystem services. When soil degradation results in resource scarcity and economic downturns, it can lead to socio-economic instability (Chen & Mueller, 2018), conflicts, and even forced migration, making the health of soil a fundamental component of Sustainable Development Goal 16 (Peace, Justice, and Strong Institutions).

## 9. Future Needs and Research Directions

We have performed thorough analyses of multiple dimensions of soil degradation, addressing its definition, implications, fundamental processes, and driving factors. Furthermore, we discussed techniques for measuring soil degradation, sustainable land management strategies aimed at reducing its impacts, the application of big data analytics in research related to soil degradation, and its relevance to the UN SDGs. In the subsequent sections, we will outline challenges, open issues, and prospective research directions for soil degradation.

### 9.1. Multidisciplinary Investigations

Addressing soil degradation effectively requires a cross-disciplinary approach that integrates insights from various fields, yet such collaborations remain limited. Climate science plays a crucial role in understanding how changing weather patterns, extreme events, and shifting precipitation trends contribute to soil erosion, loss of organic matter, and desertification. Economics and social sciences help assess the financial and societal impacts of soil degradation, guiding the development of policies and incentives that encourage sustainable land management. Soil science provides the foundational knowledge of soil degradation processes, while IoT and sensing technologies offer real-time monitoring solutions, enabling precise data collection on soil health parameters such as moisture levels, nutrient content, and microbial activity. Meanwhile, advancements in computer science, including artificial intelligence and machine learning, can enhance predictive modeling of degradation trends, while engineering innovations contribute to the design of sustainable land management practices and soil restoration techniques. Despite the potential benefits of integrating these disciplines, challenges such as lack of standardized methodologies and limited collaboration between researchers from different fields hinder progress. Moving forward, fostering interdisciplinary research initiatives and creating frameworks that facilitate knowledge exchange will be essential for developing more comprehensive strategies to mitigate soil degradation.

### 9.2. Interactions Between Climate, Land Use and Soil Degradation Processes

The interplay between climate change and land use practices often results in intricate feedback mechanisms that intensify soil degradation. For instance, climate change-induced loss of vegetation cover can increase the risk of erosion, which subsequently reduces the organic matter content of the soil and its overall fertility. Soil erosion not only removes essential nutrients but also compromises soil structure, thereby accelerating the degradation process. Conversely, soils that are already degraded, become increasingly susceptible to climate-related stressors, including extreme weather events, temperature variations, and precipitation patterns. These stressors can diminish the soil's capacity to retain moisture, hindering its ability to support plant growth, which creates a vicious cycle of declining soil health.

Furthermore, alterations in land use, such as deforestation or intensified agricultural practices, can result in a decrease in soil biodiversity. This loss of biodiversity undermines the soil's capacity for regeneration and adversely impacts the populations of crucial organisms, such as earthworms, microbes, and fungi, that are vital for sustaining soil fertility and structure. The decline of these essential organisms further restricts the soil's resilience to climate variability, worsening the degradation process.

Despite the recognition of these interconnections, the specific dynamics among climate-related factors, land use practices, and soil degradation are not yet fully understood. There is a pressing need for further research to clarify how various climate scenarios and land management approaches affect soil degradation across different regions and ecosystems. Future investigations should focus on these complexities through both mechanistic modeling and data-driven techniques. This approach will enhance our understanding of the diverse environmental and anthropogenic factors contributing to soil degradation and help with developing more effective mitigation strategies.

10.1029/2025RG000883

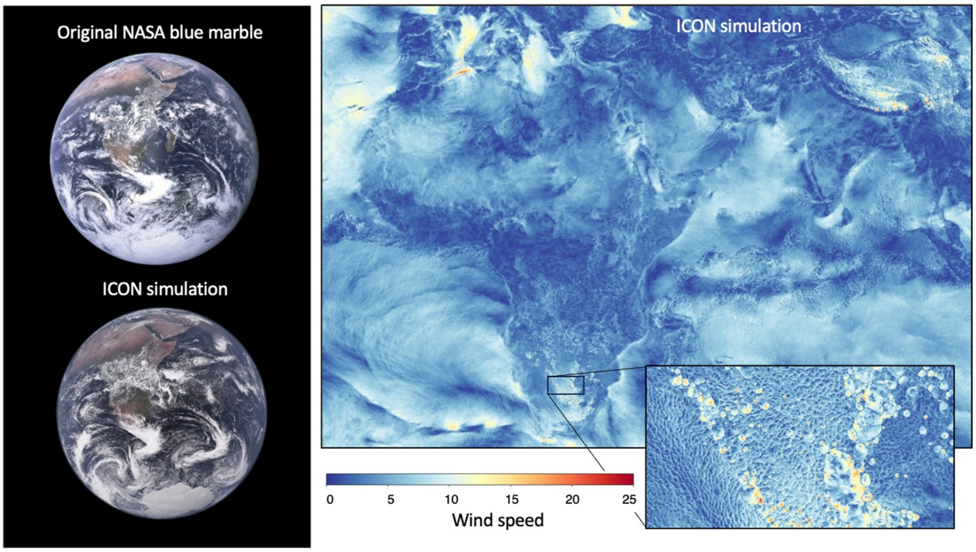


**Figure 31.** Global simulation conducted with the next-generation climate model ICON. The simulation was integrated with coupled atmosphere, land and ocean processes and using a grid spacing of 1.25 km. On the left, the original NASA Blue Marble image is presented alongside a visualization derived from the ICON output (Visualization credit: MPI-M, DKRZ and NDVIA). On the right, the simulation illustrates wind speed, featuring a detailed view of wind patterns over southern Africa. These plots represent snapshots captured on 11 December 1972, at 10:39 UTC (left) and 11:30 UTC (right).

### 9.3. Integration of New Generation of km-Scale Climate Models

Predicting climate conditions essential for studying soil degradation has traditionally been conducted on a global scale using models that simulate the interactions among various components of the Earth system, including land, atmosphere, ocean, and particles, with a grid resolution of 100 km. Recently, a new generation of climate models has been developed (Figure 31), capable of simulating these interactions at a global level with a much finer grid spacing of 10 km or less (Hohenegger et al., 2023; Rackow et al., 2025). Additionally, advancements have been made in modeling the transport of particles, extending beyond hydrometeors (Weiss et al., 2025). This progress for instance enables the simulation of dust release due to soil degradation, its subsequent transport by large-scale wind patterns, and its impacts on radiation, cloud formation, precipitation, and the feedback mechanisms influencing the climatic factors that drive soil degradation.

In the context of examining and forecasting soil degradation, employing global climate models at the kilometer scale presents two significant advantages over coarse-resolution models. Firstly, these models provide a more accurate representation of land surface heterogeneity. Secondly, they allow for the explicit modeling of convective storms by directly solving the fundamental fluid dynamics equations, rather than relying on uncertain statistical approximations known as convective parameterizations. This methodological shift has notable implications for climate simulations, as it alters the characteristics of precipitation from continuous light rain to brief, intense localized storms. Additionally, convective storms generate gusty winds, which are typically not captured in coarse-resolution models. The ability to simulate the organization and movement of mesoscale convective systems, such as squall lines in Africa, is now feasible. It is important to note that a kilometer-scale climate model is not merely a refined version of its coarse-resolution equivalent; evidence suggests that explicitly resolving convection, particularly over land, modifies the interactions between the land and atmosphere, resulting in a reduced sensitivity of the atmosphere to land surface conditions (Yoon & Hohenegger, 2025). Furthermore, unlike km-scale atmosphere-only regional climate models, the latest generation of global coupled km-scale climate models allows for the large-scale circulation to evolve organically in response to localized land changes, including their remote impacts on the ocean. Although the implications of resolving the Earth system at kilometer scales for soil degradation have yet to be explored, the aforementioned distinctions indicate that research on soil degradation could greatly benefit from the application of these advanced modeling tools.

### 9.4. Interdisciplinary Investigations for Socio-Economic Research

The examination of the socio-economic consequences of soil degradation reveals significant deficiencies in the existing literature that must be addressed to enhance the understanding of these impacts and to refine policy interventions. A primary issue is the absence of a precise definition of soil degradation, which is often conflated with land or environmental degradation. This ambiguity complicates the assessment of degradation's effects (Olsson et al., 2023) and leads to inconsistencies that hinder comparative analyses across various studies, where the terms may carry different interpretations and methodologies. Therefore, establishing standardized definitions related to land and soil degradation would facilitate more effective comparisons over time and across different regions. Much of the existing research concerning the effects of soil degradation on crop yields and agricultural productivity originates from the 1970s (e.g., Eriksson et al., 1974; Gill, 1971), indicating a pressing need for contemporary evaluations that utilize modern methodologies for assessing the impacts of soil degradation.

Although numerous localized studies have explored the various aspects of soil degradation and its socio-economic ramifications, there is a critical need for more comparative research to gain a comprehensive understanding of the broader societal impacts of soil degradation, particularly concerning rural livelihoods and the adaptation strategies employed by affected populations, such as migration and income diversification. The concept of soil security (Koch et al., 2013) may serve as a valuable framework for conducting such interdisciplinary investigations.

Furthermore, the role of soil or land degradation as a driver of migration has been less scrutinized compared to climate-related factors, despite the recognized significance of healthy soils for agricultural productivity (Hermans et al., 2023). Additional research is necessary to explore the ways in which land degradation influences migration and mobility, taking into account the complexities surrounding these phenomena, including reduced mobility and the role of soil health as a facilitator of movement. The field can benefit from insights gained from the climate-migration literature, which will aid in identifying the key research questions pertaining to the relationship between soil degradation and migration.

### 9.5. Better Data Sharing and Integration Policy

One of the key challenges in advancing soil degradation assessments is the lack of a standardized and comprehensive data sharing and integration policy. Fragmented data sources form a major obstacle toward a holistic understanding of the drivers, impacts, and solutions of soil degradation. Several factors currently prevent proper data sharing and integration. Soil data is often fragmented across different institutions, countries, and research projects. Also, many soil databases are proprietary or restricted, for instance, for privacy issues linked to soil pollution (see e.g., Gobezie & Biswas, 2024). In addition, shared soil data sets are not always accompanied with standardized, machine-readable metadata, and not all research articles are published together with an open-access data set. Reusing soil data from previous studies also faces other problems, such as the lack of data harmonization and the absence of communication between data providers and data aggregators (see e.g., Todd-Brown et al., 2022).

Addressing these issues requires (a) soil data providers to follow the FAIR data principles, (b) open-access soil data repositories, (c) metadata standards and data harmonization, and (d) collaboration between data owners and data collectors. Firstly, the FAIR data principles aim to improve the Findability, Accessibility, Interoperability, and Reuse of scientific data (Wilkinson et al., 2016). These are essential guidelines to increase the reusability and longevity of soil data. Formulated more than a decade ago, most soil scientists are aware of the FAIR principles, but their application is still not widespread. The FAIR data principles should also be applied for restricted data. For example, soil data that are subjected to privacy issues and cannot be openly shared should still meet the FAIR principles. Secondly, making soil data FAIR requires the use of trustworthy digital soil data repositories, following the TRUST principles (i.e., Transparency, Responsibility, User focus, Sustainability, and Technology (Lin et al., 2020)). Examples are Zenodo, the general-purpose open repository maintained by CERN, and the ISRIC WDC-Soils data repository, an example of a domain-specific repository for soil data. Thirdly, widely accepted metadata standards for soil data should form the base for data management and are crucial for successful data sharing (R. Hoffmann et al., 2020; C. Hoffmann et al., 2020). Consistency between the methodology used to gather and analyze soil samples is also important, and harmonization of field and laboratory measurement techniques are needed, especially for large-scale predictive modeling (see e.g., Hassani, Azapagic & Shokri, 2020). Fourthly, improving the willingness and ability to share and re-use soil data is not sufficient. To increase the reusability of soil data and the combined use of soil data from different sources, collaboration and conversation between data owners and data collectors are needed. Also, collaboration between different types of data owners, such as governments, research institutions, and private entities is needed. Working groups and stakeholder interactions, as well as working toward common goals, can help to achieve this.

Several open-access initiatives are actively working along these lines and are addressing these issues. For instance, the World Soil Information Service (WoSIS; Batjes et al., 2020; Ribeiro et al., 2018) provides freely available, standardized, and harmonized soil data. The European Soil Data Center (ESDAC), gathering and disseminating more than 130 soil related data sets and knowledge at the EU-level, such as the LUCAS Soil data set, applies an open access data policy and provides associated metadata and documentation (Panagos, Borrelli, et al., 2022; Panagos, Van Liedekerke, et al., 2022). The ESDAC will work in the upcoming years toward a new portal, the EU Soil Health Portal, reflecting the requirements specified in the proposed EU Soil Monitoring and Resilience Directive. This portal will integrate data from European and national soil monitoring schemes with other data streams coming from earth observation, citizen science, proximal sensing, as well as from research projects funded by the Mission Soil. In addition, the Soil Mission promotes the open access principle for the data sets that will be produced within the 60+ projects currently running and the 100 Living Labs.

### 9.6. Opportunities From Industry 4.0 Technologies

AI and predictive modeling frameworks are currently transforming the evaluation of soil degradation by creating novel avenues for detecting early warning signs of degradation prior to the attainment of critical thresholds worldwide. Through the analysis of large-scale environmental data sets, AI models present an effective and scalable method for monitoring soil health, thereby minimizing the necessity for comprehensive field surveys. AI models can also analyze high-resolution spatial and temporal data to detect expected changes in soil salinity, organic carbon content, erosion risks, and other factors that often trigger soil degradation processes. Moreover, AI-driven simulations allow for scenario-based projections of soil conditions under different climate and land use scenarios, providing policymakers and land managers with actionable insights for preventive interventions (Borrelli et al., 2020; Hassani et al., 2021; Shokri et al., 2024).

Limitations still need to be addressed for these technologies to be fully effective in tackling soil degradation. AI models often struggle to quantify the uncertainty in predictions, which can be critical when making decisions that impact soil management practices. Providing reliable confidence intervals would help stakeholders understand the reliability of the results and the potential risks associated with predictive models. Moreover, the effectiveness of AI in soil degradation assessments depends on the availability and quality of input data. AI and machine learning models are highly dependent on the data they are trained on. Several global soil monitoring data sets provide essential input for AI-driven predictive modeling, including the Land Use/Cover Area Frame Survey (LUCAS), the SoilGrids data set from ISRIC, the Global Soil Organic Carbon Map (GSOC), the European Soil Data Center (ESDAC), the Harmonized World Soil Database (HWSD), and the World Soil Information Service (WoSIS). These data sets can be used to facilitate large-scale and high-resolution assessments of soil degradation trends worldwide by integrating field-based observations with remote sensing products. When the data used to develop the models does not adequately represent the full range of environmental, climatic, and soil conditions, the models may generate skewed or context-specific results. As a result, the predictions may not generalize well to new or different regions, making them less effective for broader applications in soil degradation monitoring.

Errors in input data, such as satellite imagery or field observations, can propagate through AI models, leading to inaccurate predictions. These errors may amplify, especially when integrating large-scale data sets from diverse sources with varying levels of quality, which can result in unreliable assessments of soil health (Hassani, Azapagic & Shokri, 2020). Also, different AI and machine learning models might identify different sets of critical parameters or drivers of soil degradation, making it difficult for practitioners to trust or implement AI-driven recommendations across diverse regions and contexts. Ensuring reproducibility across different models and data sets is a key challenge.

### 9.7. Citizen Science in Soil Science for Enhanced Soil Degradation Monitoring and Prevention

Citizen science is an emerging research approach that actively involves non-expert participants in scientific investigations, allowing for large-scale data collection and fostering public engagement. In the context of soil science, citizen science holds great potential for improving the monitoring, understanding, and mitigation of soil degradation. Traditional soil assessment methods rely on expert-led field surveys and remote sensing, but citizen-driven data collection can complement these approaches by increasing spatial and temporal coverage. This is particularly relevant for soil degradation, which requires continuous observation to detect early warning signs and implement timely interventions.

One significant initiative embracing citizen science in soil monitoring is the EU Mission *A Soil Deal for Europe* (Mission Soil), which aims to establish 100 *living labs* and *lighthouses* to drive the transition toward healthier soils by 2030 (Panagos et al., 2024c). A *living lab* is an open-innovation ecosystem where researchers, policy-makers, industries, farmers, and citizens co-develop and test sustainable soil management practices in real-world settings (Veeckman et al., 2013). These initiatives emphasize participatory approaches, bridging the gap between scientific knowledge and practical applications.

### 9.8. Lack of Time-Series Data for Predictive Modeling

One limitation of the available input data sets to assess soil degradation using AI-driven models or other approaches is the lack of dynamism in data collection. Dynamic soil parameters, such as SOC or salinity data, are often sampled at different locations, with limited efforts to revisit the same sites in subsequent data collection campaigns. This reduces the ability of AI models to make dynamic predictions or conduct longitudinal evaluations of soil degradation trends over time. The LUCAS soil sampling campaigns represent one of the attempts to establish periodic resampling, aiming to provide temporal insights into soil condition changes (Orgiazzi et al., 2018). However, even within LUCAS, certain parameters and locations have not been consistently revisited, leading to gaps in long-term monitoring. This limitation poses challenges for AI-driven modeling, as robust trend analysis and predictive capabilities require data sets with temporal continuity. Future advancements in AI-based soil degradation assessments would greatly benefit from standardized, frequent, and spatially consistent resampling efforts to enhance the reliability of long-term projections.

Additionally, the future advancements in AI-driven soil degradation assessments must prioritize the refinement of predictive frameworks by integrating high-resolution data sets and enhancing model precision. Incorporating additional soil degradation indicators, such as microbial activity and soil chemical properties (e.g., soil pH), can improve the sensitivity and reliability of AI models in capturing different aspects of soil degradation. Expanding the spatial and temporal resolution of global soil monitoring networks can further enhance predictive capabilities of the AI models to detect soil degradation hotspots under different scenarios.

### 9.9. Data Integration Platforms and GIS Tools

As mentioned earlier, the amalgamation of Geographic Information Systems (GIS) with diverse data sets, such as satellite imagery, field observations, and climate information, is on the rise. This interdisciplinary methodology is progressing swiftly, empowering researchers to refine their analyses and acquire more in-depth understanding of soil degradation. A variety of initiatives have been initiated and are continuously developing. For instance, contemporary GIS software incorporates integrated packages for different models aimed at estimating spatial soil erosion and surface runoff within catchment areas. Noteworthy examples include ArcSWAT (Yadav & Singh, 2024) and QSWAT (Tanksali & Soraganvi, 2021), which utilize the Soil Water Assessment Tool (SWAT) model within ArcGIS and QGIS, respectively. Additionally, several geospatial platforms facilitate the implementation of the Water Erosion Prediction Project (WEPP) model across various software environments and scenarios, including GeoWEPP (Renschler, 2003), GEMSE (Baigorria & Romero, 2007), QWEPP (Miller et al., 2022), and WEPPcloud (Lew et al., 2022). Recently, QGIS has launched QGeoWEPP to support high-resolution, watershed-level assessments of soil erosion (Zhang & Renschler, 2024). These software tools allow users to leverage high-resolution satellite and climate data for research purposes without necessitating programming skills.

High-resolution Geographic Information System (GIS) analysis requires substantial computational resources, which can limit access for researchers, particularly in developing countries. To mitigate this issue, there is a growing trend among researchers to utilize cloud-based platforms such as Google Earth Engine (GEE) and Microsoft Planetary Computer. These platforms provide scalable computing capabilities, extensive geospatial data sets, and AI-enhanced analytics for processing large volumes of environmental data (Lukacz, 2022; Zhao et al., 2021). For instance, Chen et al. (2021) employed GEE to assess the degradation of temperate forests through Landsat time series analysis. Nonetheless, a significant obstacle associated with these platforms is the requirement for coding skills in programming languages such as JavaScript and Python, which can hinder researchers who lack such expertise.

Additionally, high-resolution satellite imagery can be costly, and while free data sets like Sentinel and Landsat are available, they often suffer from limitations such as reduced temporal resolution or issues with cloud cover. To overcome these challenges, NASA's Harmonized Landsat-Sentinel (HLS) initiative integrates data from both satellite systems, delivering consistent, high-quality, and frequent imagery for global environmental monitoring (McCormick et al., 2025). Furthermore, there is a notable deficiency in hyperspectral data sets for monitoring land degradation. Presently, only two hyperspectral data sets are accessible at no cost: the PRISMA (PRecursore IperSpettrale della Missione Applicativa) satellite and the EnMAP (Environmental Mapping and Analysis Program) satellite. Both data sets provide a spatial resolution of 30 m, with PRISMA featuring a swath width of 20 km and a 5-day revisit cycle, while EnMAP offers a 30-km swath and a 4-day revisit cycle (Chabrillat et al., 2024; Delogu et al., 2023). Future missions, such as the European Copernicus Next Generation Hyperspectral Satellite (CHIME) and Hyperfield-1a, are anticipated to enhance this data set availability (Rast et al., 2021; Tikka et al., 2023).

### 9.10. Ethical Dimensions of Soil Degradation and Conservation

Soil degradation presents numerous ethical challenges, particularly in light of our enhanced ability to quantitatively assess soil conditions. The following discussion will highlight three of the most urgent challenges and identify future avenues for ethical inquiry.

#### 9.10.1. Data Protection

The quantitative assessment methods presented in this paper yield novel types of soil data that can be very sensitive, enabling a broader spectrum of predictions and thereby also holding significant economic implications. For example, soil degradation data can affect land valuation, agricultural subsidies, investment choices, and even aspects of food and national security, necessitating a specialized framework for both analysis and protection. The governance of such soil data, particularly through artificial intelligence technologies, emerges as a vital area of research that intersects with data ethics, environmental justice, and global sustainability. Future investigations should focus on designing regulatory frameworks that guarantee access to data, prevent the misuse of sensitive information, and leverage soil intelligence for the advantage of local farmers and overall ecological stability. A solely individualistic approach to data ownership, which treats soil data as private property of landowners, may prove inadequate, especially considering that soil degradation has far-reaching environmental repercussions, including cross-border desertification, sand and dust storms, and biodiversity loss. However, conceptualizing soil data solely as a public resource could raise fairness concerns, particularly for farmers who might be forced to share their data with corporations or governmental entities without adequate safeguards. The prevalent consent-based model of data governance appears particularly problematic in this scenario: farmers frequently lack negotiating power, may be coerced into agreements that validate exploitative data practices, or may not fully grasp the implications of for example, AI-driven data analytics on their land's value and their future resource access (Cofone, 2024). An alternative framework, such as Helen Nissenbaum's Contextual Integrity, offers a potentially more effective approach by emphasizing that both data acquisition and data flow should align with the appropriate norms of the given context. This perspective transcends the transactional focus of traditional consent models. Contextual Integrity posits that privacy encompasses more than merely controlling access to information through individual consent; it involves ensuring that data flows are appropriate and respectful of prevailing social, ethical, and political norms (Nissenbaum, 2004). This perspective prompts essential inquiries regarding the governance of soil data: What norms should guide the collection of soil data? Should various categories of soil data be managed differently according to their ecological, economic, or security significance? How can data sharing or restrictions be implemented to reconcile private land ownership with collective environmental stewardship?

#### 9.10.2. Responsibility

The enhanced accuracy of soil degradation assessments has also revealed its transboundary impacts, and it will now raise ethical and geopolitical questions about responsibility, prevention, and reparations. Degradation occurring in one country can adversely affect other countries; for example, salinization transported by wind (Hassani, Azapagic, D'Odorico, et al., 2020) or water, desertification that crosses borders, and carbon emissions from compromised land can influence climatic conditions in distant regions. This situation complicates traditional models of responsibility, which typically concentrate on a narrow range of causal factors or agents. Soil depletion arises from a multifaceted interaction of local agricultural practices, global market dynamics, historical environmental policies, and broader environmental changes. Moreover, the repercussions of soil degradation extend beyond national boundaries, particularly when they influence food security, migration patterns, and economic stability across various regions. In light of this complexity, a simplistic liability model for assigning responsibility may no longer suffice; instead, it is essential to develop a framework that recognizes diverse types of responsibilities and obligations based on causal contributions and possibly factors such as vulnerability and economic power (Kiener, 2024). Critical questions that emerge include: If a less economically developed country is responsible for degradation but lacks the means to address it, should wealthier nations offer financial assistance? If high-income countries are responsible for industrial agriculture and deforestation that lead to depletion elsewhere, do they have an obligation to support soil restoration efforts? When degraded soil exacerbates climate change, should nations face penalties for inadequate land management, or would a global incentive system prove more effective? Tackling these issues asks for a reevaluation of how concepts of sovereignty, responsibility, and economic principles intersect with ethics in a context where soil degradation is increasingly acknowledged as a collective challenge rather than solely a national concern.

### 9.10.3. Sustainability and Food Security

A significant challenge lies in reconciling sustainability with food security, as the need to ensure immediate food availability often conflicts with soil conservation, leading to potentially complex trade-offs (Van de Poel, 2015; Van den Hoven et al., 2015). In emergency scenarios, such as those marked by food shortages that jeopardize livelihoods, it may be justifiable to prioritize agricultural production over short-term soil sustainability. Nevertheless, there exists a critical threshold beyond which continued exploitation of soil resources becomes ethically indefensible, particularly when empirical evidence reveals irreversible environmental degradation or when the adverse effects disproportionately impact marginalized communities that do not gain from the exploitation of the soils in question. The complexity of these issues is further compounded when food production is outsourced globally, where some countries may deplete soil resources in other regions while safeguarding their own. This situation raises important questions regarding global environmental justice, as the immediate agricultural gains for some often result in long-term soil sustainability challenges for all. Drawing the line between short-term advantages and long-term sustainability, as well as equitably distributing the associated benefits and burdens, requires a framework that integrates scientific insights with ethical principles. Critical inquiries include: At what point should nations that rely on external soil resources play a role in supporting their sustainability and restoration? Is there a threshold at which the environmental damage inflicted by a nation becomes so severe that its sovereignty over agricultural policies must yield to a collective global obligation for sustainability? Ultimately, are sustainability and food security inherently conflicting objectives, or can they be reconceptualized as interdependent responsibilities?

## Conflict of Interest

The authors declare no conflicts of interest relevant to this study.

## Data Availability Statement

All data used to create the maps presented in this manuscript are based on previously published work. The relevant data are available in the original publications, which have been properly cited in the figure captions and throughout the manuscript. Those papers include Oldeman et al. (1990), Muhs et al. (2014), Pesch et al. (2021), Lamandé et al. (2013), Keller and Or (2022), Schneider and Don (2019), Borrelli et al. (2023), West et al. (2014), Hassani et al. (2024), Prăvălie, Nita, et al. (2021), Hassani, Azapagic & Shokri. (2020), Toth, Hermann, Szatmari & Pasztor, (2016), Ballabio et al. (2018, 2021), Prăvălie et al. (2024), Borrelli et al. (2017a), Or et al. (2023), Gibbs and Salmon (2015), Panagos et al. (2024c), Jinger et al., 2023, Basso and Antle (2020), Smith et al. (2024), and FAO (2021a).

**Acknowledgments**
Funding for the project AI4SoilHealth by the European Union's Horizon Europe Research and Innovation Programme under Grant 101086179 is gratefully acknowledged. Nima Shokri acknowledges support from the Deutsche Forschungsgemeinschaft (DFG, German Research Foundation), project number 497539130. David Robinson was supported by the Natural Environment Research Council as part of the NC for Global Challenges programme [NE/X006247/1] delivering National Capability. Also, Remus Prăvălie has received funding from the EU's NextGenerationEU instrument through the National Recovery and Resilience Plan of Romania—Pillar III-C9-2022-I8, managed by the Ministry of Research, Innovation, and Digitalization, within the project entitled "Complex modeling of multiple land degradation processes in Europe. Towards an integrative scientific framework for sustainable land management across the continent," contract no. 760051/23.05.2023, code CF 216/29.11.2022. We thank DKRZ and NDVIA for the visualization in Figure 31. During the preparation of this work, the authors used the Ahrefs sentence rewriting tool to improve language consistency across contributions from multiple international authors. After using this tool, the authors reviewed and edited the content as needed and take full responsibility for the final publication. Open Access funding enabled and organized by Projekt DEAL.

## References

Abbasi, S., Rezaei, M., Ahmadi, F., & Turner, A. (2022). Atmospheric transport of microplastics during a dust storm. *Chemosphere*, *292*, 133456. https://doi.org/10.1016/j.chemosphere.2021.133456

Abell, J. M., Ozkundakci, D., Hamilton, D. P., van Dam-Bates, P., & Mcdowell, R. W. (2019). Quantifying the extent of anthropogenic eutrophication of lakes at a national scale in New Zealand. *Environmental Science and Technology*, *53*(16), 9439–9452. https://doi.org/10.1021/acs.est.9b03120

Aber, J., McDowell, W., Nadelhoffer, K., Magill, A., Berntson, G., Kamakea, M., et al. (1998). Nitrogen saturation in temperate forest ecosystems: Hypotheses revisited. *BioScience*, *48*(11), 921–934. https://doi.org/10.2307/1313296

Abrol, I. P., Yadav, J. S. P., & Massoud, F. I. (1988). *Salt-affected soils and their management* (Vol. 39). Food and Agriculture Organization.

Abuduwaili, J., DongWei, L. I. U., & GuangYang, W. U. (2010). Saline dust storms and their ecological impacts in arid regions. *Journal of arid land*, *2*(2), 144–150. https://doi.org/10.3724/sp.j.1227.2010.00144

Abu-Hamdeh, N. H. (2003). Compaction and subsoiling effects on corn growth and soil bulk density. *Soil Science Society of America Journal*, *67*(4), 1213–1219. https://doi.org/10.2136/sssaj2003.1213

Achu, A. L., Aju, C. D., Di Napoli, M., Prakash, P., Gopinath, G., Shaji, E., & Chandra, V. (2023). Machine-learning based landslide susceptibility modelling with emphasis on uncertainty analysis. *Geoscience Frontiers*, *14*(6), 101657. https://doi.org/10.1016/j.gsf.2023.101657

Adamsone-Fiskovica, A., & Grivins, M. (2024). Understanding the potential of sustainability turn in farming: Review of sociotechnical adoption factors of agri-environmental cropping practices. *Renewable Agriculture and Food Systems*, *39*, e16. https://doi.org/10.1017/S1742170524000085

Aerts, R., Wallén, B., & Malmer, N. (1992). Growth-limiting nutrients in sphagnum-dominated bogs subject to low and high atmospheric nitrogen supply. *Journal of Ecology*, *80*(1), 131–140.

Afifi, T., Milan, A., Etzold, B., Schraven, B., Rademacher-Schulz, C., Sakdapolrak, P., et al. (2016). Human mobility in response to rainfall variability: Opportunities for migration as a successful adaptation strategy in eight case studies. *Migration and Development*, *5*(2), 254–274. https://doi.org/10.1080/21632324.2015.1022974

Agbeshie, A. A., Abugre, S., Atta-Darkwa, T., & Awuah, R. (2022). A review of the effects of forest fire on soil properties. *Journal of Forestry Research*, *33*(5), 1419–1441. https://doi.org/10.1007/s11676-022-01475-4

AghaKouchak, A., Chiang, F., Huning, L. S., Love, C. A., Mallakpour, I., Mazdiyasni, O., et al. (2020). Climate extremes and compound hazards in a warming world. *Annual Review of Earth and Planetary Sciences*, *48*(1), 519–548. https://doi.org/10.1146/annurev-earth-071719-055228

Aksoy, H., & Kavvas, M. L. (2005). A review of hillslope and watershed scale erosion and sediment transport models. *Catena*, *64*(2–3), 247–271. https://doi.org/10.1016/j.catena.2005.08.008

Alaoui, A., Rogger, M., Peth, S., & Blöschl, G. (2018). Does soil compaction increase floods? A review. *Journal of Hydrology*, *557*, 631–642. https://doi.org/10.1016/j.jhydrol.2017.12.052

Alengebawy, A., Abdelkhalek, S. T., Qureshi, S. R., & Wang, M.-Q. (2021). Heavy metals and pesticides toxicity in agricultural soil and plants: Ecological risks and human health implications. *Toxics*, *9*(3), 42. https://doi.org/10.3390/toxics9030042

Alewell, C., Ringeval, B., Ballabio, C., Robinson, D. A., Panagos, P., & Borrelli, P. (2020). Global phosphorus shortage will be aggravated by soil erosion. *Nature Communications*, *11*(1), 4546. https://doi.org/10.1038/s41467-020-18326-7

Allen, M. R., & Ingram, W. J. (2002). Constraints on future changes in climate and the hydrologic cycle. *Nature*, *419*(6903), 224–232. https://doi.org/10.1038/nature01092

Altieri, M. A. (1999). The ecological role of biodiversity in agroecosystems. *Agriculture, Ecosystems & Environment*, *74*(1–3), 19–31. https://doi.org/10.1016/s0167-8809(99)00028-6

Altieri, M. A., & Nicholls, C. I. (2003). Soil fertility management and insect pests: Harmonizing soil and plant health in agroecosystems. *Soil and Tillage Research*, *72*(2), 203–211. https://doi.org/10.1016/s0167-1987(03)00089-8

Al-Wadaey, A., & Ziadat, F. (2014). A participatory GIS approach to identify critical land degradation areas and prioritize soil conservation for mountainous olive groves (case study). *Journal of Mountain Science*, *11*(3), 782–791. https://doi.org/10.1007/s11629-013-2827-x

Amanor, I. N., Ricardo, O. A., & Noguchi, N. (2024). Assessment of remote sensing in measuring soil parameters for precision tillage. *Journal of Terramechanics*, *113*, 100973. https://doi.org/10.1016/j.jterra.2024.100973

Aminzadeh, M., Friedrich, N., Narayanaswamy, S. G., Madani, M., & Shokri, N. (2024). Evaporation loss from small agricultural reservoirs: An overlooked component of water accounting. *Earth's Future*, *12*(1), e2023EF004050. https://doi.org/10.1029/2023EF004050

Aminzadeh, M., Kokate, T., & Shokri, N. (2025). Microplastics in sandy soils: Alterations in thermal conductivity, surface albedo, and temperature. *Environmental Pollution*, *372*, 125956. https://doi.org/10.1016/j.envpol.2025.125956

Aminzadeh, M., & Or, D. (2014). Energy partitioning dynamics of drying terrestrial surfaces. *Journal of Hydrology*, *519*, 1257–1270. https://doi.org/10.1016/j.jhydrol.2014.08.037

Aminzadeh, M., Or, D., Stevens, B., AghaKouchak, A., & Shokri, N. (2023). Upper bounds of maximum land surface temperatures in a warming climate and limits to plant growth. *Earth's Future*, *11*(9), e2023EF003755. https://doi.org/10.1029/2023EF003755

Amrhein, C. (1996). Australian sodic soils: Distribution, properties, and management. *Soil Science*, *161*(6), 412. https://doi.org/10.1097/00010694-199606000-00010

Amundson, R., Berhe, A. A., Hopmans, J. W., Olson, C., Sztein, A. E., & Sparks, D. L. (2015). Soil and human security in the 21st century. *Science*, *348*(6235), 1261071. https://doi.org/10.1126/science.1261071

Andersson, E., Brogaard, S., & Olsson, L. (2011). The political ecology of land degradation. *Annual Review of Environment and Resources*, *36*(1), 295–319. https://doi.org/10.1146/annurev-environ-033110-092827

Angon, P. B., Islam, M. S., Shreejana, K. C., Das, A., Anjum, N., Poudel, A., & Suchi, S. A. (2024). Sources, effects and present perspectives of heavy metals contamination: Soil, plants and human food chain. *Heliyon*, *10*(7), e28357. https://doi.org/10.1016/j.heliyon.2024.e28357

Antoninka, A., Chuckran, P. F., Mau, R. L., Slate, M. L., Mishler, B. D., Oliver, M. J., et al. (2022). Responses of biocrust and associated soil bacteria to novel climates are not tightly coupled. *Frontiers in Microbiology*, *13*, 821860. https://doi.org/10.3389/fmicb.2022.821860

Araya, S. N., & Ghezzehei, T. A. (2019). Using machine learning for prediction of saturated hydraulic conductivity and its sensitivity to soil structural perturbations. *Water Resources Research*, *55*(7), 5715–5737. https://doi.org/10.1029/2018wr024357

Arias-Navarro, C., Baritz, R., & Jones, A. (Eds.). (2024). *The state of soils in Europe*. Publications Office of the European Union. https://doi.org/10.2760/7007291

Arthur, E., Moldrup, P., Schjønning, P., & de Jonge, L. W. (2013). Water retention, gas transport, and pore network complexity during short-term regeneration of soil structure. *Soil Science Society of America Journal*, *77*(6), 1965–1976. https://doi.org/10.2136/sssaj2013.07.0270

Arvidsson, J., & Håkansson, I. (1996). Do effects of soil compaction persist after ploughing? Results from 21 long-term field experiments in Sweden. *Soil and Tillage Research*, *39*, 175–197.

Atkinson, J. (1993). *An Introduction to the mechanics of soils and foundations through critical state soil mechanics*. McGraw-Hill International Series in Civil Engineering (p. 337).

Augustin, K., Kuhwald, M., Brunotte, J., & Duttmann, R. (2020). Wheel load and wheel pass frequency as indicators for soil compaction risk: A four-year analysis of traffic intensity at field scale. *Geosciences*, *10*(8), 292. https://doi.org/10.3390/geosciences10080292

Awais, M., Naqvi, S. M. Z. A., Zhang, H., Li, L., Zhang, W., Awwad, F. A., et al. (2023). AI and machine learning for soil analysis: An assessment of sustainable agricultural practices. *Bioresources and Bioprocessing*, *10*(1), 90. https://doi.org/10.1186/s40643-023-00710-y

Ayele, H. S., & Atlabachew, M. (2021). Review of characterization, factors, impacts, and solutions of Lake eutrophication: Lesson for lake Tana, Ethiopia. *Environmental Science and Pollution Research*, *28*(12), 14233–14252. https://doi.org/10.1007/s11356-020-12081-4

Backes, D. J., & Teferle, F. N. (2020). Multiscale integration of high-resolution spaceborne and drone-based imagery for a high-accuracy digital elevation model over Tristan da Cunha. *Frontiers of Earth Science*, *8*, 319. https://doi.org/10.3389/feart.2020.00319

Bagnall, D. K., Shanahan, J. F., Flanders, A., Morgan, C. L. S., & Honeycutt, C. W. (2021). Soil health considerations for global food security. *Agronomy Journal*, *113*(6), 4581–4589. https://doi.org/10.1002/agj2.20783

Bai, Z. G., Dent, D. L., Olsson, L., & Schaepman, M. E. (2008). Proxy global assessment of land degradation. *Soil Use & Management*, *24*(3), 223–234. https://doi.org/10.1111/j.1475-2743.2008.00169.x

Baigorria, G. A., & Romero, C. C. (2007). Assessment of erosion hotspots in a watershed: Integrating the WEPP model and GIS in a case study in the Peruvian Andes. *Environmental Modelling & Software*, *22*(8), 1175–1183. https://doi.org/10.1016/j.envsoft.2006.06.012

Baker, J. M., Ochsner, T. E., Venterea, R. T., & Griffis, T. J. (2007). Tillage and soil organic carbon sequestration – What do we really know? *Agriculture, Ecosystems & Environment*, *118*(1–4), 1–5. https://doi.org/10.1016/j.agee.2006.05.014

Bakhshian, S., Zarepakzad, N., Nevermann, H., Hohenegger, C., Or, D., & Shokri, N. (2025). Field-scale soil moisture dynamics predicted by deep learning. *Advances in Water Resources*, *201*, 104976. https://doi.org/10.1016/j.advwatres.2025.104976

Balch, J. K., Bradley, B. A., D'Antonio, C. M., & Gómez-Dans, J. (2013). Introduced annual grass increases regional fire activity across the arid western USA (1980–2009). *Global Change Biology*, *19*(1), 173–183. https://doi.org/10.1111/gcb.12046

Baldrian, P., López-Mondéjar, R., & Kohout, P. (2023). Forest microbiome and global change. *Nature Reviews Microbiology*, *21*(8), 487–501. https://doi.org/10.1038/s41579-023-00876-4

Ball, B., & Schjønning, P. (2002). Air permeability. In J. H. Dane & G. C. Topp (Eds.), *Methods of soil analysis. Part 4*. *SSSA Book Series 5* (pp. 1141–1158). SSSA.

Ball, B. C., Hargreaves, P. R., & Watson, C. A. (2018). A framework of connections between soil and people can help improve sustainability of the food system and soil functions. *Ambio*, *47*(3), 269–283. https://doi.org/10.1007/s13280-017-0965-z

Ballabio, C., Jiskra, M., Osterwalder, S., Borrelli, P., Montanarella, L., & Panagos, P. (2021). A spatial assessment of mercury content in the European Union topsoil. *Science of the Total Environment*, *769*, 144755. https://doi.org/10.1016/j.scitotenv.2020.144755

Ballabio, C., Jones, A., & Panagos, P. (2024). Cadmium in topsoils of the European Union–an analysis based on LUCAS topsoil database. *Science of the Total Environment*, *912*, 168710. https://doi.org/10.1016/j.scitotenv.2023.168710

Ballabio, C., Lugato, E., Fernández-Ugalde, O., Orgiazzi, A., Jones, A., Borrelli, P., et al. (2019). Mapping LUCAS topsoil chemical properties at European scale using Gaussian process regression. *Geoderma*, *355*, 113912. https://doi.org/10.1016/j.geoderma.2019.113912

Ballabio, C., Panagos, P., Lugato, E., Huang, J. H., Orgiazzi, A., Jones, A., et al. (2018). Copper distribution in European topsoils: An assessment based on LUCAS soil survey. *Science of the Total Environment*, *636*, 282–298. https://doi.org/10.1016/j.scitotenv.2018.04.268

Ballabio, C., Panagos, P., & Monatanarella, L. (2016). Mapping topsoil physical properties at European scale using the LUCAS database. *Geoderma*, *261*, 110–123. https://doi.org/10.1016/j.geoderma.2015.07.006

Barakat, A., Rafai, M., Mosaid, H., Islam, M. S., & Saeed, S. (2023). Mapping of water-induced soil erosion using machine learning models: A case study of Oum Er Rbia Basin (Morocco). *Earth Systems and Environment*, *7*(1), 151–170. https://doi.org/10.1007/s41748-022-00317-x

Barbier, E. B., & Hochard, J. P. (2016). Does land degradation increase poverty in developing countries? *PLoS One*, *11*(5), e0152973. https://doi.org/10.1371/journal.pone.0152973

Barto, E. K., Antunes, P. M., Stinson, K., Koch, A. M., Klironomos, J. N., & Cipollini, D. (2011). Differences in arbuscular mycorrhizal fungal communities associated with sugar maple seedlings in and outside of invaded garlic mustard forest patches. *Biological Invasions*, *13*(12), 2755–2762. https://doi.org/10.1007/s10530-011-9945-6

Basso, B., & Antle, J. (2020). Digital agriculture to design sustainable agricultural systems. *Nature Sustainability*, *3*(4), 254–256. https://doi.org/10.1038/s41893-020-0510-0

Basto, S., Thompson, K., Phoenix, G., Sloan, V., Leake, J., & Rees, M. (2015). Long-term nitrogen deposition depletes grassland seed banks. *Nature Communications*, *6*(1), 6185. https://doi.org/10.1038/ncomms7185

Bateman, A. M., & Muñoz-Rojas, M. (2019). To whom the burden of soil degradation and management concerns. In *Advances in chemical pollution, environmental management and protection* (Vol. 4, pp. 1–22). Elsevier

Batey, T. (2009). Soil compaction and soil management – A review. *Soil Use & Management*, *25*(4), 335–345. https://doi.org/10.1111/j.1475-2743.2009.00236.x

Batjes, N. H., Calisto, L., & de Sousa, L. M. (2024). Providing quality-assessed and standardised soil data to support global mapping and modelling (WoSIS snapshot 2023). *Earth System Science Data*, *16*(10), 4735–4765. https://doi.org/10.5194/essd-16-4735-2024

Batjes, N. H., Ribeiro, E., & Van Oostrum, A. (2020). Standardised soil profile data to support global mapping and modelling (WoSIS snapshot 2019). *Earth System Science Data*, *12*(1), 299–320. https://doi.org/10.5194/essd-12-299-2020

Baumhardt, R. L., Schwartz, R. C., & Howell, T. A. (2015). Soil, water, and crop management practices for sustainable dryland agriculture. *Agronomy Journal*, *107*(1), 1–12.

Baveye, P. C., Rangel, D., Jacobson, A. R., Laba, M., Darnault, C., Otten, W., et al. (2011). From dust bowl to dust bowl: Soils are still very much a Frontier of science. *Soil Science Society of America Journal*, *75*(6), 2037–2048. https://doi.org/10.2136/sssaj2011.0145

Becerril-Piña, R., & Mastachi-Loza, C. A. (2021). Desertification: Causes and countermeasures. *Life on land*, 219–231. https://doi.org/10.1007/978-3-319-95981-8_81

Bedeke, S. B. (2023). Climate change vulnerability and adaptation of crop producers in sub-Saharan Africa: A review on concepts, approaches and methods. *Environment, Development and Sustainability*, *25*(2), 1017–1051. https://doi.org/10.1007/s10668-022-02118-8

Belnap, J. (2013). Some like it hot, some not. *Science*, *340*(6140), 1533–1534. https://doi.org/10.1126/science.1240318

BenDor, T., Lester, T. W., Livengood, A., Davis, A., & Yonavjak, L. (2015). Estimating the size and impact of the ecological restoration economy. *PLoS One*, *10*(6), e0128339. https://doi.org/10.1371/journal.pone.0128339

Berhe, A. A. (2019). Drivers of soil change (Chapter 3). In M. Busse, C. P. Giardina, D. M. Morris, & D. S. Page-Dumroese (Eds.), *Developments in soil science* (Vol. 36, pp. 27–42). Elsevier. https://doi.org/10.1016/b978-0-444-63998-1.00003-3

Berisso, F. E., Schjønning, P., Keller, T., Lamandé, M., Etana, A., De Jonge, L. W., et al. (2012). Persistent effects of subsoil compaction on pore size distribution and gas transport in a loamy soil. *Soil and Tillage Research*, *122*, 42–51. https://doi.org/10.1016/j.still.2012.02.005

Berli, M., Carminati, A., Ghezzehei, T. A., & Or, D. (2008). Evolution of unsaturated hydraulic conductivity of aggregatedsoils due to compressive forces. *Water Resources Research*, *44*(5), W00C09. https://doi.org/10.1029/2007WR006501

Bernardino, P. N., de Keersmaecker, W., Horion, S., Oehmcke, S., Gieseke, F., Fensholt, R., et al. (2025). Predictability of abrupt shifts in dryland ecosystem functioning. *Nature Climate Change*, *15*(1), 86–91. https://doi.org/10.1038/s41558-024-02201-0

Bernatek-Jakiel, A., & Poesen, J. (2018). Subsurface erosion by soil piping: Significance and research needs. *Earth-Science Reviews*, *185*, 1107–1128. https://doi.org/10.1016/j.earscirev.2018.08.006

Bestelmeyer, B. T., Brown, J. R., Havstad, K. M., Alexander, R., Chavez, G., & Herrick, J. E. (2003). Development and use of state-and-transition models for rangelands. *Rangeland Ecology & Management/Journal of Range Management Archives*, *56*(2), 114–126. https://doi.org/10.2458/azu_jrm_v56i2_bestelmeyer

Bestelmeyer, B. T., Okin, G. S., Duniway, M. C., Archer, S. R., Sayre, N. F., Williamson, J. C., & Herrick, J. E. (2015). Desertification, land use, and the transformation of global drylands. *Frontiers in Ecology and the Environment*, *13*(1), 28–36. https://doi.org/10.1890/140162

Beyer, R. M., Hua, F., Martin, P. A., Manica, A., & Rademacher, T. (2022). Relocating croplands could drastically reduce the environmental impacts of global food production. *Communications Earth & Environment*, *3*(1), 49. https://doi.org/10.1038/s43247-022-00360-6

Bhattachan, A., D'Odorico, P., Baddock, M. C., Zobeck, T. M., Okin, G. S., & Cassar, N. (2012). The Southern Kalahari: A potential new dust source in the Southern Hemisphere? *Environmental Research Letters*, *7*(2), 024001. https://doi.org/10.1088/1748-9326/7/2/024001

Bhattacharyya, R., Ghosh, B. N., Mishra, P. K., Mandal, B., Rao, C. S., Sarkar, D., et al. (2015). Soil degradation in India: Challenges and potential solutions. *Sustainability*, *7*(4), 3528–3570. https://doi.org/10.3390/su7043528

Bi, M., Li, H., Meidl, P., Zhu, Y., Ryo, M., & Rillig, M. C. (2024). Number and dissimilarity of global change factors influences soil properties and functions. *Nature Communications*, *15*(1), 8188. https://doi.org/10.1038/s41467-024-52511-2

Bindraban, P. S., van der Velde, M., Ye, L., van den Berg, M., Materechera, S., Kiba, D. I., et al. (2012). Assessing the impact of soil degradation on food production. *Current Opinion in Environmental Sustainability*, *4*(5), 478–488. https://doi.org/10.1016/j.cosust.2012.09.015

Black, R., Adger, W. N., Arnell, N. W., Dercon, S., Geddes, A., & Thomas, D. (2011). The effect of environmental change on human migration. *Global Environmental Change*, *21*, S3–S11. https://doi.org/10.1016/j.gloenvcha.2011.10.001

Blackwell, P. S., Ward, M. A., Lefevre, R. N., & Cowand, D. J. (1985). Compaction of a swelling clay soil by agricultural traffic: Effects upon conditions for growth of winter cereals and evidence for some recovery of structure. *Journal of Soil Science*, *36*(4), 633–650. https://doi.org/10.1111/j.1365-2389.1985.tb00365.x

Blaikie, P., & Brookfield, H. (Eds.). (1987). *Land degradation and society* (1st ed.). Routledge. https://doi.org/10.4324/9781315685366

Blanco-Canqui, H., & Ruis, S. J. (2018). No-tillage and soil physical environment. *Geoderma*, *326*, 164–200. https://doi.org/10.1016/j.geoderma.2018.03.011

Blanco-Canqui, H., Ruis, S. J., & Francis, C. A. (2024). Do organic farming practices improve soil physical properties? *Soil Use & Management*, *40*(1), e12999. https://doi.org/10.1111/sum.12999

Blaser, P., Zysset, M., Zimmermann, S., & Luster, J. (1999). Soil acidification in southern Switzerland between 1987 and 1997: A case study based on the critical load concept. *Environmental Science and Technology*, *33*(14), 2383–2389. https://doi.org/10.1021/es9808144

Bleam, W. F. (2016). *Soil and environmental chemistry*. Academic Press.

Boag, B., & Yeates, G. W. (2001). The potential impact of the New Zealand flatworm, a predator of earthworms, in western Europe. *Ecological Applications*, *11*(5), 1276–1286. https://doi.org/10.2307/3060919

Bobbink, R. (1991). Effects of nutrient enrichment in Dutch chalk grassland. *Journal of Applied Ecology*, *28*(1), 28–41. https://doi.org/10.2307/2404111

Bobbink, R., & Hettelingh, J.-P. (2011). Review and revision of empirical critical loads and dose-response relationships. National Institute for Public Health and the Environment (RIVM). RIVM Report.

Bobbink, R., Hicks, K., Galloway, J., Spranger, T., Alkemade, R., Ashmore, M., et al. (2010). Global assessment of nitrogen deposition effects on terrestrial plant diversity: A synthesis. *Ecological Applications*, *20*(1), 30–59. https://doi.org/10.1890/08-1140.1

Bobbink, R., Loran, C., & Tomassen, H. (2022). *Review and revision of empirical critical loads of nitrogen for Europe*. German Environment Agency.

Bohlen, P. J., Groffman, P. M., Fahey, T. J., Fisk, M. C., Suárez, E., Pelletier, D. M., & Fahey, R. T. (2004). Ecosystem consequences of exotic earthworm invasion of north temperate forests. *Ecosystems*, *7*, 1–12. https://doi.org/10.1007/s10021-003-0126-z

Borda, L. G., Cosentino, N. J., Iturri, L. A., Garcia, M. G., & Gaiero, D. M. (2022). Is dust derived from shrinking saline lakes a risk to soil sodification in southern South America? *Journal of Geophysical Research: Earth Surface*, *127*(4), e2021JF006585. https://doi.org/10.1029/2021jf006585

Borrelli, P., Alewell, C., Alvarez, P., Anache, J. A. A., Baartman, J., Ballabio, C., et al. (2021). Soil erosion modelling: A global review and statistical analysis. *Science of the Total Environment*, *780*, 146494. https://doi.org/10.1016/j.scitotenv.2021.146494

Borrelli, P., Alewell, C., Yang, J. E., Bezak, N., Chen, Y., Fenta, A. A., et al. (2023). Towards a better understanding of pathways of multiple co-occurring erosion processes on global cropland. *International Soil and Water Conservation Research*, *11*(4), 713–725. https://doi.org/10.1016/j.iswcr.2023.07.008

Borrelli, P., Lugato, E., Montanarella, L., & Panagos, P. (2017). A new assessment of soil loss due to wind erosion in European agricultural soils using a quantitative spatially distributed modelling approach. *Land Degradation & Development*, *28*(1), 335–344. https://doi.org/10.1002/ldr.2588

Borrelli, P., Robinson, D. A., Fleischer, L. R., Lugato, E., Ballabio, C., Alewell, C., et al. (2017). An assessment of the global impact of 21st century land use change on soil erosion. *Nature Communications*, *8*(1), 1–13. https://doi.org/10.1038/s41467-017-02142-7

Borrelli, P., Robinson, D. A., Panagos, P., Lugato, E., Yang, J. E., Alewell, C., et al. (2020). Land use and climate change impacts on global soil erosion by water (2015-2070). *Proceedings of the National Academy of Sciences of the United States of America*, *117*(36), 21994–22001. https://doi.org/10.1073/pnas.2001403117

Bouma, J., & Montanarella, L. (2016). Facing policy challenges with inter- and transdisciplinary soil research focused on the UN Sustainable Development Goals. *Soils*, *2*, 135–145. https://doi.org/10.5194/soil-2-135-2016

Bouwman, A. F., Beusen, A. H. W., & Billen, G. (2009). Human alteration of the global nitrogen and phosphorus soil balances for the period 1970–2050. *Global Biogeochemical Cycles*, *23*(4). https://doi.org/10.1029/2009gb003576

Bouwman, A. F., Van Vuuren, D. P., Derwent, R. G., & Posch, M. (2002). A global analysis of acidification and eutrophication of terrestrial ecosystem. *Water, Air, & Soil Pollution*, *141*(1–4), 349–382. https://doi.org/10.1023/a:1021398008726

Bowler, D. E., Bjorkman, A. D., Dornelas, M., Myers-Smith, I. H., Navarro, L. M., Niamir, A., et al. (2020). Mapping human pressures on biodiversity across the planet uncovers anthropogenic threat complexes. *People and Nature*, *2*(2), 380–394. https://doi.org/10.1002/pan3.10071

Brady, N. C., & Weil, R. R. (2008). *The nature and properties of soils* (Vol. 13, pp. 662–710). Prentice Hall.

Brandle, J. R., Hodges, L., & Zhou, X. H. (2004). Windbreaks in sustainable agricultural systems. *Agroforestry Systems*, *61*(1), 65–78. https://doi.org/10.1023/B:AGFO.0000028990.31801.62

Braun, S., Cantaluppi, L., & Flückiger, W. (2005). Fine roots in stands of Fagus sylvatica and Picea abies along a gradient of soil acidification. *Environment and Pollution*, *137*(3), 574-9.

Braun, S., Thomas, V. F. D., Quiring, R., & Flückiger, W. (2010). Does nitrogen deposition increase forest production? The role of phosphorus. *Environmental Pollution*, *158*(6), 2043–2052. https://doi.org/10.1016/j.envpol.2009.11.030

Brevik, E., Fenton, T., & Moran, L. (2002). Effect of soil compaction on organic carbon amounts and distribution, South-Central Iowa. *Environmental Pollution*, *116*, S137–S141. https://doi.org/10.1016/s0269-7491(01)00266-4

Bridges, E. M., & Oldeman, L. R. (1999). Global assessment of human-induced soil degradation. *Arid Soil Research and Rehabilitation*, *13*(4), 319–325. https://doi.org/10.1080/089030699263212

Briske, D. D., Fuhlendorf, S. D., & Smeins, F. E. (2005). State-and-transition models, thresholds, and rangeland health: A synthesis of ecological concepts and perspectives. *Rangeland Ecology & Management*, *58*(1), 1–10. https://doi.org/10.2458/azu_rangelands_v58i1_smeins

Brouwer, R. (1962). Nutritive influences on the distribution of dry matter in plant. *Netherlands Journal of Agricultural Science*, *10*(5), 399–408. https://doi.org/10.18174/njas.v10i5.17581

Brus, D. J., & van den Akker, J. J. H. (2018). How serious a problem is subsoil compaction in the Netherlands? A survey based on probability sampling. *Soil*, *4*(1), 37–45. https://doi.org/10.5194/soil-4-37-2018

Brutsaert, W. (2005). *Hydrology: An Introduction*. Cambridge University Press. Retrieved from https://www.cambridge.org/highereducation/books/hydrology/59A90AFF36F02ECDE7DB2EE21C740612

Buda, A. R. (2013). Surface-runoff generation and forms of overland flow. In *Mountain and hillslope geomorphology* (pp. 73–84). Elsevier Inc.

Bullock, J. M., Aronson, J., Newton, A. C., Pywell, R. F., & Rey-Benayas, J. M. (2011). Restoration of ecosystem services and biodiversity: Conflicts and opportunities. *Trends in Ecology & Evolution*, *26*(10), 541–549. https://doi.org/10.1016/j.tree.2011.06.011

Bünemann, E. K., Bongiorno, G., Bai, Z., Creamer, R. E., De Deyn, G., de Goede, R., et al. (2018). Soil quality – A critical review. *Soil Biology and Biochemistry*, *120*, 105–125. https://doi.org/10.1016/j.soilbio.2018.01.030

Burns, K., & Gabet, E. J. (2015). The effective viscosity of slurries laden with vegetative ash. *Catena*, *135*, 350–357. https://doi.org/10.1016/j.catena.2014.06.008

Call, M., & Gray, C. (2020). Climate anomalies, land degradation, and rural out-migration in Uganda. *Population and Environment*, *41*(4), 507–528. https://doi.org/10.1007/s11111-020-00349-3

Callaway, R., Thelen, G., Rodriguez, & Holben, W. E. (2004). Soil biota and exotic plant invasion. *Nature*, *427*(6976), 731–733. https://doi.org/10.1038/nature02322

Cambi, M., Certini, G., Neri, F., & Marchi, E. (2015). The impact of heavy traffic on forest soils: A review. *Forest Ecology and Management*, *338*, 124–138. https://doi.org/10.1016/j.foreco.2014.11.022

Campbell, G. A., Smith, P., Broothaerts, N., Panagos, P., Jones, A., Ballabio, C., et al. (2025). Continental scale soil monitoring: A proposed multi-scale framing of soil quality. *European Journal of Soil Science*, *76*(4), e70174. https://doi.org/10.1111/ejss.70174

Cao, H., Amiraslani, F., Liu, J., & Zhou, N. (2015). Identification of dust storm source areas in West Asia using multiple environmental datasets. *Science of the Total Environment*, *502*, 224–235. https://doi.org/10.1016/j.scitotenv.2014.09.025

Carlisle, L. (2016). Factors influencing farmer adoption of soil health practices in the United States: A narrative review. *Agroecology and Sustainable Food Systems*, *40*(6), 583–613. https://doi.org/10.1080/21683565.2016.1156596

Carpenter, S. R., & Bennett, E. M. (2011). Reconsideration of the planetary boundary for phosphorus. *Environmental Research Letters*, *6*(1), 014009. https://doi.org/10.1088/1748-9326/6/1/014009

Carreiro, M., Sinsabaugh, R., Repert, D., & Parkhurst, D. (2000). Microbial enzyme shifts explain litter decay responses to simulated nitrogen deposition. *Ecology*, *81*(9), 2359–2365. https://doi.org/10.2307/177459

Carter, L. J. (1977). Soil erosion: The problem persists despite the billions spent on it. *Science*, *196*(4288), 409–411. https://doi.org/10.1126/science.196.4288.409

Chabrillat, S., Foerster, S., Segl, K., Beamish, A., Brell, M., Asadzadeh, S., et al. (2024). The EnMAP spaceborne imaging spectroscopy mission: Initial scientific results two years after launch. *Remote Sensing of Environment*, *315*, 114379. https://doi.org/10.1016/j.rse.2024.114379

Chamen, T. W. C., Moxey, A. P., Towers, W., Balana, B., & Hallett, P. D. (2015). Mitigating arable soil compaction: A review and analysis of available cost and benefit data. *Soil and Tillage Research*, *146*, 10–25.

Chaney, N. W., Wood, E. F., McBratney, A. B., Hempel, J. W., Nauman, T. W., Brungard, C. W., & Odgers, N. P. (2016). POLARIS: A 30-meter probabilistic soil series map of the contiguous United States. *Geoderma*, *274*, 54–67. https://doi.org/10.1016/j.geoderma.2016.03.025

Chaudhari, S. K., Patra, A., Dey, P., Bal, S. K., Gorantiwar, S., & Parsad, R. (2022). Sensor-based monitoring for Improving agricultural productivity and sustainability: A review. *Journal of the Indian Society of Soil Science*, *70*(2), 121–141. https://doi.org/10.5958/0974-0228.2022.00013.5

Chazdon, R. L. (2008). Beyond deforestation: Restoring forests and ecosystem services on degraded lands. *Science*, *320*(5882), 1458–1460. https://doi.org/10.1126/science.1155365

Che, J., Zhao, X. Q., & Shen, R. F. (2023). Molecular mechanisms of plant adaptation to acid soils: A review. *Pedosphere*, *33*(1), 14–22. https://doi.org/10.1016/j.pedsph.2022.10.001

Chen, H., Rahmati, M., Montzka, C., Gao, H., & Vereecken, H. (2024). Soil physicochemical properties explain land use/cover histories in the last sixty years in China. *Geoderma*, *446*, 116908. https://doi.org/10.1016/j.geoderma.2024.116908

Chen, H., Shaojun, C., & Wang, Y. (2004). The Loess Plateau watershed rehabilitation project. In *Scaling up poverty reduction: A Global learning process and conference*. The World Bank.

Chen, J., & Mueller, V. (2018). Coastal climate change, soil salinity and human migration in Bangladesh. *Nature Climate Change*, *8*(11), 981–985. https://doi.org/10.1038/s41558-018-0313-8

Chen, R., Huang, J.-W., Zhou, C., Ping, Y., & Chen, Z.-K. (2021). A new simple and low-cost air permeameter for unsaturated soils. *Soil and Tillage Research*, *213*, 105083. https://doi.org/10.1016/j.still.2021.105083

Chen, S., Woodcock, C. E., Bullock, E. L., Arévalo, P., Torchinava, P., Peng, S., & Olofsson, P. (2021). Monitoring temperate forest degradation on Google Earth Engine using Landsat time series analysis. *Remote Sensing of Environment*, *265*, 112648. https://doi.org/10.1016/j.rse.2021.112648

Chen, Z., Wang, L., Wei, A., Gao, J., Lu, Y., & Zhou, J. (2019). Land-use change from arable lands to orchards reduced soil erosion and increased nutrient loss in a small catchment. *Science of the Total Environment*, *648*, 1097–1104. https://doi.org/10.1016/j.scitotenv.2018.08.141

Cherlet, M., Hutchinson, C., Reynolds, J., Hill, J., Sommer, S., & vonMaltitz, G. (Eds.). (2018). *World atlas of desertification* (3rd ed.). Publication Office of the European Union.

Chianu, J. N., Chianu, J. N., & Mairura, F. (2012). Mineral fertilizers in the farming systems of sub-Saharan Africa. A review. *Agronomy for Sustainable Development*, *32*(2), 545–566. https://doi.org/10.1007/s13593-011-0050-0
Chinese Academy of Sciences. (2001). *Chinese soil taxonomy. Coordinated by Institute of Soil Science* (p. 203). Chinese Academy of Sciences.
Chislock, M. F., Doster, E., Zitomer, R. A., & Wilson, A. E. (2013). Eutrophication: Causes, consequences, and controls in aquatic ecosystems. *Nature Education Knowledge*, *4*(4), 10.
Clarke, M. L., & Rendell, H. M. (2007). Climate, extreme events and land degradation. In M. V. K. Sivakumar & N. Ndiang'ui (Eds.), *Climate and Land Degradation* (pp. 137–152). Springer. https://doi.org/10.1007/978-3-540-72438-4_7
Cofone, I. N. (2024). *The privacy fallacy: harm and power in the information economy*. Cambridge University Press.
Crowther, T. W., Todd-Brown, K. E. O., Rowe, C. W., Wieder, W. R., Carey, J. C., Machmuller, M. B., et al. (2016). Quantifying global soil carbon losses in response to warming. *Nature*, *540*(7631), 104–108. https://doi.org/10.1038/nature20150
Cunningham, S. C., Mac Nally, R., Baker, P. J., Cavagnaro, T. R., Beringer, J., Thomson, J. R., & Thompson, R. M. (2015). Balancing the environmental benefits of reforestation in agricultural regions. *Perspectives in Plant Ecology, Evolution and Systematics*, *17*(4), 301–317. https://doi.org/10.1016/j.ppees.2015.06.001
Dahiya, S., Kumar, S., Chaudhary, C., & Chaudhary, C. (2018). Lodging: Significance and preventive measures for increasing crop production. *International Journal of Chemical Studies*, *6*(2), 700–705.
Daniells, I. G. (2012). Hardsetting soils: A review. *Soil Research*, *50*(5), 349–359. https://doi.org/10.1071/sr11102
Davydzenka, T., Tahmasebi, P., & Carroll, M. (2022). Improving remote sensing classification: A deep-learning-assisted model. *Computers & Geosciences*, *164*, 105123. https://doi.org/10.1016/j.cageo.2022.105123
Davydzenka, T., Tahmasebi, P., & Shokri, N. (2024). Unveiling the global extent of land subsidence: The sinking crisis. *Geophysical Research Letters*, *51*(4), e2023GL104497. https://doi.org/10.1029/2023GL104497
DeBano, L. F., Neary, D. G., & Ffolliott, P. F. (1998). *Fire effects on ecosystems*. John Wiley and Sons.
DeCarlo, K. F., & Shokri, N. (2014). Salinity effects on cracking morphology and dynamics in 3-D desiccating clays. *Water Resources Research*, *50*(4), 3052–3072. https://doi.org/10.1002/2013WR014424
De Geeter, S., Verstraeten, G., Poesen, J., Campforts, B., & Vanmaercke, M. (2023). A data driven gully head susceptibility map of Africa at 30 m resolution. *Environmental Research*, *224*, 115573. https://doi.org/10.1016/j.envres.2023.115573
de Jonge, L. W., Moldrup, P., & Schjønning, P. (2009). Soil infrastructure, interfaces and translocation processes in inner space ("Soil-it-is"): Towards a road map for the constraints and crossroads of soil architecture and biophysical processes. *Hydrology and Earth System Sciences*, *13*(8), 1485–1502. Retrieved from www.hydrol-earth-syst-sci.net/13/1485/2009/
De la Paix, M. J., Lanhai, L., Xi, C., Varenyam, A., Nyongesah, M. J., & Habiyaremye, G. (2013). Physicochemical properties of saline soils and aeolian dust. *Land Degradation & Development*, *24*(6), 539–547. https://doi.org/10.1002/ldr.1148
Delogu, G., Caputi, E., Perretta, M., Ripa, M. N., & Boccia, L. (2023). Using PRISMA hyperspectral data for land cover classification with artificial intelligence support. *Sustainability*, *15*(18), 13786. https://doi.org/10.3390/su151813786
DeLong, C., Cruse, R., & Wiener, J. (2015). The soil degradation paradox: Compromising our resources when we need them the most. *Sustainability*, *7*(1), 866–879. https://doi.org/10.3390/su7010866
Demattê, J. A. M., Nanni, M. R., Da Silva, A. P., de Melo Filho, J. F., Dos Santos, W. C., & Campos, R. C. (2010). Soil density evaluated by spectral reflectance as an evidence of compaction effects. *International Journal of Remote Sensing*, *31*(2), 403–422. https://doi.org/10.1080/01431160902893469
De Rosa, D., Ballabio, C., Lugato, E., Fasiolo, M., Jones, A., & Panagos, P. (2024). Soil organic carbon stocks in European croplands and grasslands: How much have we lost in the past decade? *Global Change Biology*, *30*(1), e16992. https://doi.org/10.1111/gcb.16992
de Souza, T. A. F., & Freitas, H. (2018). Long-term effects of fertilization on soil organism diversity. In S. Gaba, B. Smith, & E. Lichtfouse (Eds.), *Sustainable agriculture reviews 28:* Ecology for agriculture (pp. 211–247). Springer International Publishing.
de Vries, W. (2021). Impacts of nitrogen emissions on ecosystems and human health: A mini review. *Current Opinion in Environmental Science and Health*, *21*, 100249. https://doi.org/10.1016/j.coesh.2021.100249
Dickie, I. A., Nuñez, M. A., Pringle, A., Lebel, T., Tourtellot, S. G., & Johnston, P. R. (2016). Towards management of invasive ectomycorrhizal fungi. *Biological Invasions*, *18*(12), 3383–3395. https://doi.org/10.1007/s10530-016-1243-x
Ding, Z., Koriem, M. A., Ibrahim, S. M., Antar, A. S., Ewis, M. A., He, Z., & Kheir, A. M. S. (2020). Seawater intrusion impacts on groundwater and soil quality in the northern part of the Nile Delta, Egypt. *Environmental Earth Sciences*, *79*(13), 1–11. https://doi.org/10.1007/s12665-020-09069-1
Diop, M., Chirinda, N., Beniaich, A., El Gharous, M., & El Mejahed, K. (2022). Soil and water conservation in Africa: State of play and potential role in tackling soil degradation and building soil health in agricultural lands. *Sustainability*, *14*(20), 13425. https://doi.org/10.3390/su142013425
Dise, N. B., & Wright, R. F. (1995). Nitrogen leaching from European forests in relation to nitrogen deposition. *Forest Ecology and Management*, *71*(1), 153–161. https://doi.org/10.1016/0378-1127(94)06092-w
D'Odorico, P., Bhattachan, A., Davis, K. F., Ravi, S., & Runyan, C. W. (2013). Global desertification: Drivers and feedbacks. *Advances in Water Resources*, *51*, 326–344. https://doi.org/10.1016/j.advwatres.2012.01.013
Douglas, A. E. (2015). Multiorganismal insects: Diversity and function of resident microorganisms. *Annual Review of Entomology*, *60*(1), 17–34. https://doi.org/10.1146/annurev-ento-010814-020822
Dregne, H. E. (2002). Land degradation in the drylands. *Arid Land Research and Management*, *16*(2), 99–132. https://doi.org/10.1080/153249802317304422
Duce, R., Liss, P., Merrill, J., Atlas, E., Buat-Menard, P., Hicks, B., et al. (1991). The atmospheric input of trace species to the world ocean. *Global Biogeochemical Cycles*, *5*(3), 193–259. https://doi.org/10.1029/91gb01778
Duguma, A. L., & Bai, X. (2025). How the internet of things technology improves agricultural efficiency. *Artificial Intelligence Review*, *58*(2), 63. https://doi.org/10.1007/s10462-024-11046-0
Eisenhauer, N., Cesarz, S., Koller, R., Worm, K., & Reich, P. B. (2012). Global change belowground: Impacts of elevated, nitrogen, and summer drought on soil food webs and biodiversity. *Global Change Biology*, *18*(2), 435–447. https://doi.org/10.1111/j.1365-2486.2011.02555.x
El-Beltagi, H. S., Basit, A., Mohamed, H. I., Ali, I., Ullah, S., Kamel, E. A. R., et al. (2022). Mulching as a sustainable water and soil saving practice in agriculture: A review. *Agronomy*, *12*(8), 1881. https://doi.org/10.3390/agronomy12081881
Encyclopaedia Britannica. (2024). *Loess Plateau*. Encyclopaedia Britannica. Retrieved from https://www.britannica.com/place/Loess-Plateau
Ende, H. P., & Evers, F. H. (1997). Visual magnesium deficiency symptoms (coniferous, deciduous trees) and threshold values (foliar, soil). In R. F. Hüttl & W. Schaaf (Eds.), *Magnesium deficiency in forest ecosystems*. *Nutrients in ecosystems* (Vol. 1, pp. 3–21). Springer. https://doi.org/10.1007/978-94-011-5402-4_1

Eriksson, J., Hakansson, I., & Danfors, B. (1974). The effect of soil compaction on soil structure and crop yields. *Swedish Institute of Agricultural Engineering Bulletin*.

Escolar, C., Martínez, I., Bowker, M. A., & Maestre, F. T. (2012). Warming reduces the growth and diversity of biological soil crusts in a semi-arid environment: Implications for ecosystem structure and functioning. Philosophical Transactions of the Royal Society B. *Biological Science*, *367*(1606), 3087–3099. https://doi.org/10.1098/rstb.2011.0344

Eswar, D., Karuppusamy, R., & Chellamuthu, S. (2021). Drivers of soil salinity and their correlation with climate change. *Current Opinion in Environmental Sustainability*, *50*, 310–318. https://doi.org/10.1016/j.cosust.2020.10.015

EthioTrees. (2020). *EthioTrees Project*. *EthioTrees.* Web Archive. Retrieved from https://web.archive.org/web/20200215154255/https://ethiotrees.com/

Evans, R. (2017). Factors controlling soil erosion and runoff and their impacts in the upper Wissey catchment, Norfolk, England: A ten year monitoring programme. *Earth Surface Processes and Landforms*, *42*(14), 2266–2279. https://doi.org/10.1002/esp.4182

FAO. (2006). *Food Security* (Policy Brief). Retrieved from https://www.fao.org/fileadmin/templates/faoitaly/documents/pdf/pdf_Food_Security_Cocept_Note.pdf

FAO. (2015a). *Agroecology to reverse soil degradation and achieve food security*. Food and Agricultural Organization of the United Nations. Retrieved from https://openknowledge.fao.org/server/api/core/bitstreams/bb2a86db-7f53-4e70-91ca-35ceb9d777db/content

FAO. (2015b). World Fertilizer Trends and Outlook to 2018 Food Agriculture Organization United Nations, Rome, Italy.

FAO. (2019). Soil erosion: the greatest challenge to sustainable soil management (p. 100).

FAO. (2021a). Food and Agriculture Organization of the United Nations, CB3670EN/1/03.21. Retrieved from https://www.fao.org/3/cb3670en/cb3670en.pdf

FAO. (2021b). Global map of salt-affected soils GSASmap v1.0. Retrieved from https://www.fao.org/documents/card/en/c/cb7247en

FAO. (2023). Sand and dust storms - A guide to mitigation, adaptation, policy and risk management measures in agriculture. https://doi.org/10.4060/cc8071en

FAO. (2024). Global status of salt-affected soils – Main report. https://doi.org/10.4060/cd3044en

FAO and IIASA. (2023). Harmonized World Soil Database version 2.0. Rome and Laxenburg. https://doi.org/10.4060/cc3823en

FAO/IIASA/ISRIC/ISSCAS/JRC. (2012). *Harmonized world soil database (version 1.2)*.

FAO and ITPS. (2015). *Status of the world's soil resources (SWSR)*. Main report FAO ITPS. Food and Agriculture Organization of the United Nations, - FAO.

FAO and UNEP. (2021). *Global assessment of soil pollution: Report*. Food and Agriculture Organization of the United Nations and United Nations Environment Programme.

Fazekas, O., & Horn, R. (2005). Zusammenhang zwischen hydraulischer und mechanischer Bodenstabilität in Abhängigkeit von der Belastungsdauer. *Journal of Plant Nutrition and Soil Science*, *168*(1), 60–67. https://doi.org/10.1002/jpln.200421381

Fendrich, A. N., Van Eynde, E., Stasinopoulos, D. M., Rigby, R. A., Mezquita, F. Y., & Panagos, P. (2024). Modeling arsenic in European topsoils with a coupled semiparametric (GAMLSS-RF) model for censored data. *Environment International*, *185*, 108544. https://doi.org/10.1016/j.envint.2024.108544

Ferrara, A., Kosmas, C., Salvati, L., Padula, A., Mancino, G., & Nolè, A. (2020). Updating the MEDALUS-ESA framework for worldwide land degradation and desertification assessment. *Land Degradation & Development*, *31*(12), 1593–1607. https://doi.org/10.1002/ldr.3559

Ferreira, C. S. S., Seifollahi-Aghmiuni, S., Destouni, G., Ghajarnia, N., & Kalantari, Z. (2022). Soil degradation in the European Mediterranean region: Processes, status and consequences. *Science of the Total Environment*, *805*, 150106. https://doi.org/10.1016/j.scitotenv.2021.150106

Fewtrell, L. (2004). Drinking-water nitrate, methemoglobinemia, and global burden of disease: A discussion. *Environmental Health Perspectives*, *112*(14), 1371–1374. https://doi.org/10.1289/ehp.7216

Findlay, A. M. (2011). Migrant destinations in an era of environmental change. *Global Environmental Change*, *21*, S50–S58. https://doi.org/10.1016/j.gloenvcha.2011.09.004

Finger-Higgens, R., Duniway, M. C., Fick, S., Geiger, E. L., Hoover, D. L., Pfennigwerth, A. A., et al. (2022). Decline in biological soil crust N-fixing lichens linked to increasing summertime temperatures. *Proceedings of the National Academy of Sciences*, *119*(16), e2120975119. https://doi.org/10.1073/pnas.2120975119

Fish, A. N., & Koppi, A. J. (1994). The use of a simple field air permeameter as a rapid indicator of functional soil pore space. *Geoderma*, *63*(3–4), 255–264. https://doi.org/10.1016/0016-7061(94)90067-1

Fitzpatrick, E. A. (1956). AN indurated soil horizon formed by permafrost. *Journal of Soil Science*, *7*(2), 248–254. https://doi.org/10.1111/j.1365-2389.1956.tb00882.x

Flint, L. E., & Flint, A. L. (2002). Porosity (2.3.2.3. Volumetric method with gas pycnometry). In J. H. Dane & G. C. Topp (Eds.), *Methods of soil analysis. Part 4. Physical methods* (pp. 241–254). SSSA.

Fonte, S. J., Hsieh, M., & Mueller, N. D. (2023). Earthworms contribute significantly to global food production. *Nature Communications*, *14*(1), 5713. https://doi.org/10.1038/s41467-023-41286-7

Franco, H. H. S., Guimaraes, R. M. L., Tormena, C. A., Cherubin, M. R., & Favilla, H. S. (2019). Global applications of the Visual Evaluation of Soil Structure method: A systematic review and meta-analysis. *Soil and Tillage Research*, *190*, 61–69. https://doi.org/10.1016/j.still.2019.01.002

Frison, E. A., Cherfas, J., & Hodgkin, T. (2011). Agricultural Biodiversity Is Essential for a Sustainable Improvement in Food and Nutrition Security. *Sustainability*, *3*(1), 238–253. https://doi.org/10.3390/su3010238

Fussell, J. C., & Kelly, F. J. (2021). Mechanisms underlying the health effects of desert sand dust. *Environment International*, *157*, 106790. https://doi.org/10.1016/j.envint.2021.106790

Galindo, V., Giraldo, C., Lavelle, P., Armbrecht, I., & Fonte, S. J. (2022). Land use conversion to agriculture impacts biodiversity, erosion control, and key soil properties in an Andean watershed. *Ecosphere*, *13*(3), e3979. https://doi.org/10.1002/ecs2.3979

Gang, C., Zhou, W., Chen, Y., Wang, Z., Sun, Z., Li, J., et al. (2014). Quantitative assessment of the contributions of climate change and human activities on global grassland degradation. *Environmental Earth Sciences*, *72*(11), 4273–4282. https://doi.org/10.1007/s12665-014-3322-6

Gao, H., Li, Z., Li, P., Jia, L., & Zhang, X. (2012). Quantitative study on influences of terraced field construction and check-dam siltation on soil erosion. *Journal of Geographical Sciences*, *22*(5), 946–960. https://doi.org/10.1007/s11442-012-0975-5

García-Ruiz, J. M., Beguería, S., Nadal-Romero, E., González-Hidalgo, J. C., Lana-Renault, N., & Sanjuán, Y. (2015). A meta-analysis of soil erosion rates across the world. *Geomorphology*, *239*, 160–173. https://doi.org/10.1016/j.geomorph.2015.03.008

Garland, J. J. (1983). *Designated skid trails minimize soil compaction*. Oregon State University Extension service.

Garnett, T. (2013). Food sustainability: Problems, perspectives and solutions. *Proceedings of the Nutrition Society*, *72*(1), 29–39. https://doi.org/10.1017/S0029665112002947

Garrity, D. P. (1999). Contour farming based on natural vegetative strips: Expanding the scope for increased food crop production on sloping lands in Asia. *Environment, Development and Sustainability*, *1*(3–4), 323–336. https://doi.org/10.1023/A:1010091904395

Gashu, K., & Muchie, Y. (2018). Rethink the interlink between land degradation and livelihood of rural communities in Chilga district, Northwest Ethiopia. *Journal of Ecology and Environment*, *42*(1), 17. https://doi.org/10.1186/s41610-018-0077-0

Gebru, A. L., Abay, T. K., & Asmerom, A. T. (2024). Post-conflict challenges and aspirations of Tigray Youth, Ethiopia: A needs assessment and recovery roadmap. *African Journal of Political Science and International Relations*, *18*(3), 83–95. https://doi.org/10.5897/ajpsir2024.1472

Gerber, J. S., Ray, D. K., Makowski, D., Butler, E. E., Mueller, N. D., West, P. C., et al. (2024). Global spatially explicit yield gap time trends reveal regions at risk of future crop yield stagnation. *Nature Food*, *5*(2), 125–135. https://doi.org/10.1038/s43016-023-00913-8

Ghadekar, P., Kamble, A., Arole, R., Chandra, A., Singh Bhatti, S., & Jiby, B. (2024). Real-time soil erosion detection using satellite imagery and analysis. *Multimedia Tools and Applications*, *84*(24), 1–14. https://doi.org/10.1007/s11042-024-20215-w

Ghezzehei, T. A., & Or, D. (2000). Dynamics of soil aggregate coalescence governed by capillary and rheological processes. *Water Resources Research*, *36*(2), 367–379. https://doi.org/10.1029/1999WR900316

Gholami, H., Mohammadifar, A., Song, Y., Li, Y., Rahmani, P., Kaskaoutis, D. G., et al. (2024). An assessment of global land susceptibility to wind erosion based on deep-active learning modelling and interpretation techniques. *Scientific Reports*, *14*(1), 18951. https://doi.org/10.1038/s41598-024-70125-y

Ghosh, T., Maity, P., Rabbi, S. M. F., Das, T. K., & Bhattacharyya, R. (2023). Application of X-ray computed tomography in soil and plant – A review. *Frontiers in Environmental Science*, *11*, 1216630. https://doi.org/10.3389/fenvs.2023.1216630

Gianoli, F., Weynants, M., & Cherlet, M. (2023). Land degradation in the European Union—Where does the evidence converge? *Land Degradation & Development*, *34*(8), 2256–2275. https://doi.org/10.1002/ldr.4606

Gibbons, S. M., Lekberg, Y., Mummey, D. L., Sangwan, N., Ramsey, P. W., & Gilbert, J. A. (2017). Invasive plants rapidly reshape soil properties in a grassland ecosystem. *mSystems*, *2*, e00178-16. https://doi.org/10.1128/mSystems.00178-16

Gibbs, H. K., & Salmon, J. M. (2015). Mapping the world's degraded lands. *Applied Geography*, *57*, 12–21. https://doi.org/10.1016/j.apgeog.2014.11.024

Gill, W. R. (1971). Economic assessment of soil compaction. *Compaction of Agricultural Soils*.

Gobezie, T. B., & Biswas, A. (2024). Preserving soil data privacy with SoilPrint: A unique soil identification system for soil data sharing. *Geoderma*, *442*, 116795. https://doi.org/10.1016/j.geoderma.2024.116795

Goulding, K. W. T. (2016). Soil acidification and the importance of liming agricultural soils with particular reference to the United Kingdom. *Soil Use & Management*, *32*(3), 390–399. https://doi.org/10.1111/sum.12270

Govers, G., Vandaele, K., Desmet, P., Poesen, J., & Bunte, K. (1994). The role of tillage in soil redistribution on hillslopes. *European Journal of Soil Science*, *45*(4), 469–478. https://doi.org/10.1111/j.1365-2389.1994.tb00532.x

Graves, A. R., Morris, J., Deeks, L. K., Rickson, R. J., Kibblewhite, M. G., Harris, J. A., et al. (2015). The total costs of soil degradation in England and Wales. *Ecological Economics*, *119*, 399–413. https://doi.org/10.1016/j.ecolecon.2015.07.026

Grennfelt, P., Engleryd, A., Forsius, M., Hov, Ø., Rodhe, H., & Cowling, E. (2020). Acid rain and air pollution: 50 years of progress in environmental science and policy. *Ambio*, *49*(4), 849–864. https://doi.org/10.1007/s13280-019-01244-4

Griffin, D. W., Kellogg, C. A., & Shinn, E. A. (2001). Dust in the wind: Long range transport of dust in the atmosphere and its implications for global public and ecosystem health. *Global Change & Human Health*, *2*(1), 20–33. https://doi.org/10.1023/A:1011910224374

Grzesiak, M. T. (2009). Impact of soil compaction on root architecture, leaf water status, gas exchange and growth of maize and triticale seedlings. *Plant Root*, *3*, 10–16. https://doi.org/10.3117/plantroot.3.10

Guebsi, R., Mami, S., & Chokmani, K. (2024). Drones in precision agriculture: A comprehensive review of applications, technologies, and challenges. *Drones*, *8*(11), 686. https://doi.org/10.3390/drones8110686

Guo, B., Guo, X., Zhang, B., Suo, L., Bai, H., & Luo, P. (2022). Using a two-stage scheme to map toxic metal distributions based on GF-5 Satellite hyperspectral images at a northern Chinese opencast coal mine. *Remote Sensing*, *14*(22), 5804. https://doi.org/10.3390/rs14225804

Guo, M. (2021). Soil health assessment and management: Recent development in science and practices. *Soil Systems*, *5*(4), 61. https://doi.org/10.3390/soilsystems5040061

Gurubasappa, S. (2022). Assessing the economic viability of sustainable farming practices: A study. *International Journal of Research and Analytical Reviews*, *9*(2), 2855. Retrieved from https://ijrar.org/papers/IJRAR22A2855.pdf

Hagazi, N., Gebrekirstos, A., Birhane, E., Bongers, F., Kelly, R., & Bräuning, A. (2020). Land restoration requires a shift from quantity to quality: Lessons from Tigray, Ethiopia. ETFRN News 60 - Restoring African Drylands. CIFOR-ICRAF.

Hagazi, N., Gebremedhin, B., & Tesfay, G. (2020). Community-based soil and water conservation in Tigray, Ethiopia: Impacts and implications. *Journal of Environmental Management*, *260*, 110–118. https://doi.org/10.1016/j.jenvman.2020.110118

Haghshenas Haghighi, M., & Motagh, M. (2024). Uncovering the impacts of depleting aquifers: A remote sensing analysis of land subsidence in Iran. *Science Advances*, *10*(19), eadk3039. https://doi.org/10.1126/sciadv.adk3039

Håkansson, I., & Lipiec, J. (2000). A review of the usefulness of relative bulk density values in studies of soil structure and compaction. *Soil and Tillage Research*, *53*(2), 71–85. https://doi.org/10.1016/s0167-1987(99)00095-1

Håkansson, I., & Reeder, R. C. (1994). Subsoil compaction by vehicles with high axle load—Extent, persistence and crop response. *Soil and Tillage Research*, *29*(2), 277–304. https://doi.org/10.1016/0167-1987(94)90065-5

Hale, C. M., Frelich, L. E., Reich, P. B., & Pastor, J. (2005). Effects of European earthworm invasion on soil characteristics in northern hardwood forests of Minnesota, USA. *Ecosystems*, *8*, 911–927. https://doi.org/10.1007/s10021-005-0066-x

Hamza, M. A., & Anderson, W. K. (2005). Soil compaction in cropping systems: A review of the nature, causes and possible solutions. *Soil and Tillage Research*, *82*(2), 121–145. https://doi.org/10.1016/j.still.2004.08.009

Hannam, I. (2020). Soil legislation in Australia. In H. Yahyah, H. Ginzky, E. Kasimbazi, R. Kibugi, & O. C. Ruppel (Eds.), *Legal Instruments for sustainable soil management in Africa* (pp. 181–212). Springer International Publishing. https://doi.org/10.1007/978-3-030-36004-7_10

Hart, B., Walker, G., Katupitiya, A., & Doolan, J. (2020). Salinity management in the murray–darling Basin, Australia. *Water*, *12*(6), 1829. https://doi.org/10.3390/w12061829

Hassani, A., Azapagic, A., D'Odorico, P., Keshmiri, A., & Shokri, N. (2020). Desiccation crisis of saline lakes: A new decision-support framework for building resilience to climate change. *Science of the Total Environment*, *703*, 134718. https://doi.org/10.1016/j.scitotenv.2019.134718

Hassani, A., Azapagic, A., & Shokri, N. (2020). Predicting long-term dynamics of soil salinity and sodicity on a global scale. *Proceedings of the National Academy of Sciences*, *117*(52), 33017–33027. https://doi.org/10.1073/pnas.2013771117

Hassani, A., Azapagic, A., & Shokri, N. (2021). Global predictions of primary soil salinization under changing climate in the 21st century. *Nature Communications*, *12*(1), 6663. https://doi.org/10.1038/s41467-021-26907-3

Hassani, A., Smith, P., & Shokri, N. (2024). Negative correlation between soil salinity and soil organic carbon variability. *Proceedings of the National Academy of Sciences*, *121*(18), e2317332121. https://doi.org/10.1073/pnas.2317332121

Hendrix, P. F., Callaham, M. A., Jr., Drake, J. M., Huang, C. Y., James, S. W., Snyder, B. A., & Zhang, W. (2008). Pandora's Box contained bait: The global problem of introduced earthworms. *Annual Review of Ecology and Systematics*, *39*(1), 593–613. https://doi.org/10.1146/annurev.ecolsys.39.110707.173426

Hengl, T., Mendes de Jesus, J., Heuvelink, G. B. M., Ruiperez Gonzalez, M., Kilibarda, M., Blagotić, A., et al. (2017). SoilGrids250m: Global gridded soil information based on machine learning. *PLoS One*, *12*(2), e0169748. https://doi.org/10.1371/journal.pone.0169748

Hermans, K., & McLeman, R. (2021). Climate change, drought, land degradation and migration: Exploring the linkages. *Current Opinion in Environmental Sustainability*, *50*, 236–244. https://doi.org/10.1016/j.cosust.2021.04.013

Hermans, K., Müller, D., O'Byrne, D., Olsson, L., & Stringer, L. C. (2023). Land degradation and migration. *Nature Sustainability*, *6*(12), 1503–1505. https://doi.org/10.1038/s41893-023-01231-4

Hernandez-Ramirez, G., Ruser, R., & Kim, D.-G. (2021). How does soil compaction alter nitrous oxide fluxes? A meta-analysis. *Soil and Tillage Research*, *211*, 105036. https://doi.org/10.1016/j.still.2021.105036

Hillel, D. (2000). *Salinity management for sustainable irrigation: integrating science, environment, and economics*. World Bank Publications.

Himmelstein, J., Ares, A., Gallagher, D., & Myers, J. (2017). A meta-analysis of intercropping in Africa: Impacts on crop yield, farmer income, and integrated pest management effects. *International Journal of Agricultural Sustainability*, *15*(1), 1–10. https://doi.org/10.1080/14735903.2016.1242332

Hirmas, D. R., Giménez, D., Nemes, A., Kerry, R., Brunsell, N. A., & Wilson, C. J. (2018). Climate-induced changes in continental-scale soil macroporosity may intensify water cycle. *Nature*, *561*(7721), 100–103. https://doi.org/10.1038/s41586-018-0463-x

Hoffmann, C., Schulz, S., Eberhardt, E., Grosse, M., Stein, S., Specka, X., et al. (2020). Data standards for soil- and agricultural research. *BonaRes Series*. https://doi.org/10.20387/BonaRes-ARM4-66M2

Hoffmann, R., Dimitrova, A., Muttarak, R., Crespo Cuaresma, J., & Peisker, J. (2020). A meta-analysis of country-level studies on environmental change and migration. *Nature Climate Change*, *10*(10), 904–912. https://doi.org/10.1038/s41558-020-0898-6

Hohenegger, C., Ament, F., Beyrich, F., Löhnert, U., Rust, H., Bange, J., et al. (2023). FESSTVaL: The field experiment on submesosale saptio-temporal variability in Lindenberg. *Bulletin America Meteorology Social*, *104*(10), E1875–E1892. https://doi.org/10.1175/BAMS-D-21-0330.1

Hohenegger, C., Korn, P., Linardakis, L., Redler, R., Schnur, R., Adamidis, P., et al. (2023). ICON-Sapphire: Simulating the components of the Earth system and their interactions at kilometer and subkilometer scales. *Geoscientific Model Development*, *16*(2), 779–811. https://doi.org/10.5194/gmd-16-779-2023

Horn, R. (1993). Mechanical properties of structured unsaturated soils. *Soil Technology*, *6*, 47–75.

Horn, R., Domżżał, H., Słowińska-Jurkiewiecz, A., & van Owerkerk, C. (1995). Soil compaction processes and their effects on the structure of arable soils and the environment. *Soil and Tillage Research*, *35*, 23–36.

Horn, R., & Lebert, M. (1994). Soil compactibility and compressibility. In B. D. Soane & C. vanOuwerkerk (Eds.), *Soil Compaction in Crop Production* (pp. 45–69). Elsevier.

Horn, R., Mordhorst, A., Fleige, H., Zimmermann, I., Burbaum, B., Filipinski, M., & Cordsen, E. (2019). Soil type and land use effects on tensorial properties of saturated hydraulic conductivity in Northern Germany. *European Journal of Soil Science*, *71*(2), 179–189. https://doi.org/10.1111/ejss.12864

Horn, R., & Peth, S. (2011). Mechanics of unsaturated soils for agricultural applications. In *Handbook of Soil Science* (2nd ed.). CRC Press.

Horswill, P., O'Sullivan, O., Phoenix, G. K., Lee, J. A., & Leake, J. R. (2008). Base cation depletion, eutrophication and acidification of species-rich grasslands in response to long-term simulated nitrogen deposition. *Environment and Pollution*, *155*(2), 336–349. https://doi.org/10.1016/j.envpol.2007.11.006

Hossain, M. E., Shahrukh, S., & Hossain, S. A. (2022). Chemical fertilizers and pesticides: Impacts on soil degradation, groundwater, and human health in Bangladesh. In *Environmental degradation: Challenges and strategies for mitigation* (pp. 63–92). Springer.

Hou, D., Bolan, N. S., Tsang, D. C. W., Kirkham, M. B., & O'Connor, D. (2020). Sustainable soil use and management: An interdisciplinary and systematic approach. *Science of the Total Environment*, *729*, 138961. https://doi.org/10.1016/j.scitotenv.2020.138961

Hou, D., & Ok, Y. S. (2019). Soil pollution—speed up global mapping. *Nature*, *566*(7745), 455. https://doi.org/10.1038/d41586-019-00669-x

Hou, X., Feng, L., Dai, Y., Hu, C., Gibson, L., Tang, J., et al. (2022). Global mapping reveals increase in lacustrine algal blooms over the past decade. *Nature Geoscience*, *15*(2), 130–134. https://doi.org/10.1038/s41561-021-00887-x

Houghton, R. A. (1999). The annual net flux of carbon to the atmosphere from changes in land use 1850–1990. *Tellus*, *51B*(2), 298–313. https://doi.org/10.1034/j.1600-0889.1999.00013.x

Hu, J., Peng, J., Zhou, Y., Xu, D., Zhao, R., Jiang, Q., et al. (2019). Quantitative estimation of soil salinity using UAV-borne hyperspectral and satellite multispectral images. *Remote Sensing*, *11*(7), 736. https://doi.org/10.3390/rs11070736

Hu, S., Zhang, C., & Lu, N. (2023). Quantifying coupling effects between soil matric potential and osmotic potential. *Water Resources Research*, *59*(2), e2022WR033779. https://doi.org/10.1029/2022WR033779

Hu, W., Cichota, R., Beare, M., Müller, K., Drewry, J., & Eger, A. (2023). Soil structural vulnerability: Critical review and conceptual development. *Geoderma*, *430*, 116346. https://doi.org/10.1016/j.geoderma.2023.116346

Hu, W., Drewry, J., Beare, M., Eger, A., & Müller, K. (2021). Compaction induced soil structural degradation affects productivity and environmental outcomes: A review and New Zealand case study. *Geoderma*, *395*, 115035. https://doi.org/10.1016/j.geoderma.2021.115035

Hudek, C., Putinica, C., Otten, W., & De Baets, S. (2022). Functional root trait-based classification of cover crops to improve soil physical properties. *European Journal of Soil Science*, *73*(1), e13147. https://doi.org/10.1111/ejss.13147

Huere-Peña, J. L., Castrejon-Valdez, M., Castañeda-Campos, C., Leon-Gomez, R., Mateu-Mateo, W. A., Bautista-Gómez, R., et al. (2024). Innovations in soil health monitoring: Role of advanced sensor technologies and remote sensing. *Journal of Experimental Biology and Agricultural Sciences*, *12*(5), 653–667. https://doi.org/10.18006/2024.12(5).653.667

Hultine, K. R., Hernández-Hernández, T., Williams, D. G., Albeke, S. E., Tran, N., Puente, R., & Larios, E. (2023). Global change impacts on cacti (Cactaceae): Current threats, challenges and conservation solutions. *Annals of Botany*, mcad040. https://doi.org/10.1093/aob/mcad040

Huning, L. S., Bateni, S. M., Hayes, M., Ho, S. Q.-G., Jayasinghe, S., Kumar, R., et al. (2024). Sustainability nexus analytics, informatics, and data (AID): Drought. *SNF*, *32*(1), 18. https://doi.org/10.1007/s00550-024-00546-w

Huning, L. S., Love, C. A., Anjileli, H., Vahedifard, F., Zhao, Y., Chaffe, P. L. B., et al. (2024). Global land subsidence: Impact of climate extremes and human activities. *Reviews of Geophysics*, *62*(4), e2023RG000817. https://doi.org/10.1029/2023RG000817

Hussain, S., Hussain, S., Guo, R., Sarwar, M., Ren, X., Krstic, D., et al. (2021). Carbon sequestration to avoid soil degradation: A review on the role of conservation tillage. *Plants*, *10*(10), 2001. https://doi.org/10.3390/plants10102001

Idier, H., Dehhaoui, M., Maatala, N., & El Kadi, K. A. (2024). Assessing the impact of precision farming technologies: A literature review. *World Journal of Agricultural Science and Technology*, *2*(4), 161–179. https://doi.org/10.11648/j.wjast.20240204.17

Iqbal, R., Raza, M. A. S., Valipour, M., Saleem, M. F., Zaheer, M. S., Ahmad, S., et al. (2020). Potential agricultural and environmental benefits of mulches—A review. *Bulletin of the National Research Centre*, *44*(1), 75. https://doi.org/10.1186/s42269-020-00290-3

Iqbal, S., Riaz, U., Murtaza, G., Jamil, M., Ahmed, M., Hussain, A., & Abbas, Z. (2020). Chemical fertilizers, formulation, and their influence on soil health. In *Microbiota and biofertilizers* (pp. 1–15). Springer. https://doi.org/10.1007/978-3-030-48771-3_1

IUSS Working Group WRB. (2022). *World Reference Base for Soil Resources. International soil classification system for naming soils and creating legends for soil maps* (4th ed.). International Union of Soil Sciences (IUSS).

Ivushkin, K., Bartholomeus, H., Bregt, A. K., Pulatov, A., Kempen, B., & De Sousa, L. (2019). Global mapping of soil salinity change. *Remote Sensing of Environment*, *231*, 111260. https://doi.org/10.1016/j.rse.2019.111260

Izquierdo, J. E., Houlton, B. Z., & van Huysen, T. L. (2013). Evidence for progressive phosphorus limitation over long-term ecosystem development: Examination of a biogeochemical paradigm. *Plant and Soil*, *367*(1), 135–147. https://doi.org/10.1007/s11104-013-1683-3

Janbu, N. (1998). Sediment deformations – A classical approach to stress-strain-time behaviour of granular media as developed at NTH over 50 year period. *NTNU, Department of Geotechnical Engineering, Trondheim, Bulletin*, *35*, 86.

Jannesarahmadi, S., Aminzadeh, M., Helmig, R., Or, D., Oesterle, B., & Shokri, N. (2025). The role of wind velocity in saline water evaporation from porous media and surface salt crystallization Dynamics. *ACS Earth and Space Chemistry*, *9*(7), 1938–1945. https://doi.org/10.1021/acsearthspacechem.5c00130

Jannesarahmadi, S., Aminzadeh, M., Helmig, R., Or, D., & Shokri, N. (2024). Quantifying salt crystallization impact on evaporation dynamics from porous surfaces. *Geophysical Research Letters*, *51*(22), e2024GL111080. https://doi.org/10.1029/2024GL111080

Jannesarahmadi, S., Aminzadeh, M., Raga, R., & Shokri, N. (2023). Effects of microplastics on evaporation dynamics in porous media. *Chemosphere*, *311*, 137023. https://doi.org/10.1016/j.chemosphere.2022.137023

Jarrah, M., Mayel, S., Tatarko, J., Funk, R., & Kuka, K. (2020). A review of wind erosion models: Data requirements, processes, and validity. *Catena*, *187*, 104388. https://doi.org/10.1016/j.catena.2019.104388

Jarvis, N., Koestel, J., Messing, I., Moeys, J., & Lindahl, A. (2013). Influence of soil, land use and climatic factors on the hydraulic conductivity of soil. *Hydrology and Earth System Sciences*, *17*(12), 5185–5195. https://doi.org/10.5194/hess-17-5185-2013

Jasim, S. A., Mohammadi, M. J., Patra, I., Jalil, A. T., Taherian, M., Abdullaeva, U. Y., et al. (2024). The effect of microorganisms (bacteria and fungi) in dust storm on human health. *Reviews on Environmental Health*, *39*(1), 65–75. https://doi.org/10.1515/reveh-2022-0162

Jenny, H. (1994). Climate as a soil forming factor. In *Factors of soil formation: A system of quantitative pedology* (pp. 104–196). Dover.

Jhariya, M. K., & Singh, L. (2021). Effect of fire severity on soil properties in a seasonally dry forest ecosystem of Central India. *International journal of Environmental Science and Technology*, *18*(12), 3967–3978. https://doi.org/10.1007/s13762-020-03062-8

Jia, G., Shevliakova, E., Artaxo, P., De Noblet-Ducoudré, N., Houghton, R., House, J., et al. (2019). Land-climate interactions. In P. R. Shukla, J. Skea, E. Calvo Buendi, V. Masson-Delmotte, H.-O. Pörtner, D. C. Roberts, et al. (Eds.), *Climate change and land, (IPCC Special Report)* (pp. 131–247).

Jinger, D., Kaushal, R., Kumar, R., Paramesh, V., Verma, A., Shukla, M., et al. (2023). Degraded land rehabilitation through agroforestry in India: Achievements, current understanding, and future prospectives. *Frontiers in Ecology and Evolution*, *11*, 1088796. https://doi.org/10.3389/fevo.2023.1088796

Jochum, M., Thouvenot, L., Ferlian, O., Zeiss, R., Klarner, B., Pruschitzki, U., et al. (2022). Aboveground impacts of a belowground invader: How invasive earthworms alter aboveground arthropod communities in a northern North American forest. *Biological Letters*, *18*(3), 20210636. https://doi.org/10.1098/rsbl.2021.0636

Johannes, A., Weisskopf, P., Schulin, R., & Boivin, P. (2017). To what extent do physical measurements match with visual evaluation of soil structure? *Soil and Tillage Research*, *173*, 24–32. https://doi.org/10.1016/j.still.2016.06.001

Johnson, D., Leake, J., & Lee, J. (1999). The effects of quantity and duration of simulated pollutant nitrogen deposition on root-surface phosphatase activities in calcareous and acid grasslands: A bioassay approach. *New Phytologist*, *141*(3), 433–442. https://doi.org/10.1046/j.1469-8137.1999.00360.x

Jones, D. L., Cross, P., Withers, P. J. A., DeLuca, T. H., Robinson, D. A., Quilliam, R. S., et al. (2013). REVIEW: Nutrient stripping: The global disparity between food security and soil nutrient stocks. *Journal of Applied Ecology*, *50*(4), 851–862. https://doi.org/10.1111/1365-2664.12089

Jungkunst, H. F., Göpel, J., Horvath, T., Ott, S., & Brunn, M. (2022). Global soil organic carbon–climate interactions: Why scales matter. *WIREs Climate Change*, *13*(4), e780. https://doi.org/10.1002/wcc.780

Juřicová, A., Öttl, L. K., Wilken, F., Chuman, T., Žížala, D., Minařík, R., & Fiener, P. (2025). Tillage erosion as an underestimated driver of carbon dynamics. *Soil and Tillage Research*, *245*, 1–11. https://doi.org/10.1016/j.still.2024.106287

Kaczan, D. J., & Orgill-Meyer, J. (2020). The impact of climate change on migration: A synthesis of recent empirical insights. *Climatic Change*, *158*(3–4), 281–300. https://doi.org/10.1007/s10584-019-02560-0

Kalinina, O., Goryachkin, S. V., Karavaeva, N. A., Lyuri, D. I., Najdenko, L., & Giani, L. (2009). Self-restoration of post-agrogenic sandy soils in the southern Taiga of Russia: Soil development, nutrient status, and carbon dynamics. *Geoderma*, *152*(1–2), 35–42. https://doi.org/10.1016/j.geoderma.2009.05.014

Kamai, T., & Assouline, S. (2018). Evaporation from deep aquifers in arid regions: Analytical model for combined liquid and vapor water fluxes. *Water Resources Research*, *54*(7), 4805–4822. https://doi.org/10.1029/2018WR023030

Kane, D. A., Bradford, M. A., Fuller, E., Oldfield, E. E., & Wood, S. A. (2021). Soil organic matter protects US maize yields and lowers crop insurance payouts under drought. *Environmental Research Letters*, *16*(4), 044018. https://doi.org/10.1088/1748-9326/abe492

Kansanga, M. M., Mkandawire, P., Kuuire, V., & Luginaah, I. (2020). Agricultural mechanization, environmental degradation, and gendered livelihood implications in northern Ghana. *Land Degradation & Development*, *31*(11), 1422–1440. https://doi.org/10.1002/ldr.3490

Kapalanga, T. S. (2008). A review of land degradation assessment methods. Land Restoration Training Programme, 2011. Retrieved from https://www.grocentre.is/static/gro/publication/374/document/taimi.pdf

Karaca, O., Cameselle, C., & Reddy, K. R. (2018). Mine tailing disposal sites: Contamination problems, remedial options and phytocaps for sustainable remediation. *Reviews in Environmental Science and Biotechnology*, *17*(1), 205–228. https://doi.org/10.1007/s11157-017-9453-y

Karamage, F., Zhang, C., Ndayisaba, F., Shao, H., Kayiranga, A., Fang, X., et al. (2016). Extent of cropland and related soil erosion risk in Rwanda. *Sustainability*, *8*(7), 609. https://doi.org/10.3390/su8070609

Kassa, H., Abiyu, A., Hagazi, N., Mokria, M., Kassawmar, T., & Gitz, V. (2022). Forest landscape restoration in Ethiopia: Progress and challenges. *Frontiers in Forests and Global Change*, *5*, 96106. https://doi.org/10.3389/ffgc.2022.796106

Katzir, I., Cokol, M., Aldridge, B. B., & Alon, U. (2019). Prediction of ultra-high-order antibiotic combinations based on pairwise interactions. *PLoS Computational Biology*, *15*(1), e1006774. https://doi.org/10.1371/journal.pcbi.1006774

Kawamoto, K., Moldrup, P., Schjønning, P., Iversen, B. V., Komatsu, T., & Rolston, D. E. (2006). Gas transport parameters in the vadose zone: Development and tests of power-law models for air permeability. *Vadose Zone Journal*, *5*(4), 1205–1215. https://doi.org/10.2136/vzj2006.0030

Keeley, J. E., Bond, W. J., Bradstock, R. A., Pausas, J. G., & Rundel, P. W. (2011). *Fire in mediterranean ecosystems: Ecology, evolution and management*. Cambridge University Press.

Keller, T., Arvidsson, J., Schjønning, P., Lamandé, M., Stettler, M., & Weisskopf, P. (2012). In situ subsoil stress-strain behavior in relation to soil precompression stress. *Soil Science*, *177*(8), 490–497. https://doi.org/10.1097/ss.0b013e318262554e

Keller, T., Colombi, T., Ruiz, S., Schymanski, S., Weisskopf, P., Koestel, J., et al. (2021). Soil structure recovery following compaction – Short-term evolution of soil physical properties in a loamy soil. *Soil Science Society of America Journal*, *85*(4), 1002–1020. https://doi.org/10.1002/saj2.20240

Keller, T., & Or, D. (2022). Farm vehicles approaching weights of sauropods exceed safe mechanical limits for soil functioning. *Proceedings of the National Academy of Sciences of the United States of America*, *119*(21), e2117699119. https://doi.org/10.1073/pnas.2117699119

Keller, T., Sandin, M., Colombi, T., Horn, R., & Or, D. (2019). Historical increase in agricultural machinery weights enhanced soil stress levels and adversely affected soil functioning. *Soil and Tillage Research*, *194*, 104293. https://doi.org/10.1016/j.still.2019.104293

Khaledian, Y., Kiani, F., Ebrahimi, S., Brevik, E. C., & Aitkenhead-Peterson, J. (2017). Assessment and monitoring of soil degradation during land use change using multivariate analysis. *Land Degradation & Development*, *28*(1), 128–141. https://doi.org/10.1002/ldr.2541

Khan, S. A., Doevenspeck, M., & Sass, O. (2024). Climate (im)mobilities in the Eastern Hindu Kush: The case of Lotkuh Valley, Pakistan. *Population and Environment*, *46*(1), 2. https://doi.org/10.1007/s11111-023-00443-2

Khan, S. U., Xia, X., Zhang, H., & Guo, C. (2020). Screening of agricultural land productivity and returning farmland to forest area for sensitivity to rural labor outward migration in the ecologically fragile Loess Plateau region. *Environmental Science and Pollution Research*, *27*, 9022–9036. https://doi.org/10.1007/s11356-020-09022-6

Khasanov, S., Kulmatov, R., Li, F., van Amstel, A., Bartholomeus, H., Aslanov, I., et al. (2023). Impact assessment of soil salinity on crop production in Uzbekistan and its global significance. *Agriculture, Ecosystems & Environment*, *342*, 108262. https://doi.org/10.1016/j.agee.2022.108262

Khorchani, M., Gaspar, L., Nadal-Romero, E., Arnaez, J., Lasanta, T., & Navas, A. (2023). Effects of cropland abandonment and afforestation on soil redistribution in a small Mediterranean mountain catchment. *International Soil and Water Conservation Research*, *11*(2), 339–352. https://doi.org/10.1016/j.iswcr.2022.10.001

Khosravi, K., Rezaie, F., Cooper, J. R., Kalantari, Z., Abolfathi, S., & Hatamiafkoueieh, J. (2023). Soil water erosion susceptibility assessment using deep learning algorithms. *Journal of Hydrology*, *618*, 129229. https://doi.org/10.1016/j.jhydrol.2023.129229

Kiener, M. (Ed.). (2024). *The Routledge handbook of philosophy of responsibility*. *Routledge Handbooks in Philosophy Series* (1st ed.). Routledge.

Kim, H., & Villarini, G. (2024). Higher emissions scenarios lead to more extreme flooding in the United States. *Nature Communications*, *15*(1), 237. https://doi.org/10.1038/s41467-023-44415-4

Kinnell, P. I. A. (2005). Raindrop-impact-induced erosion processes and prediction: A review. *Hydrological Processes*, *19*(14), 2815–2844. https://doi.org/10.1002/hyp.5788

Kirby, J. M. (1994). Simulating soil deformation using a critical state model: I. Laboratory tests. *European Journal of Soil Science*, *45*(3), 239–248. https://doi.org/10.1111/j.1365-2389.1994.tb00506.x

Kirk, G. J. D., Bellamy, P. H., & Lark, M. (2010). Changes in soil pH across England and Wales in response to decrease acid deposition. *Global Change Biology*, *16*(11), 3111–3119. https://doi.org/10.1111/j.1365-2486.2009.02135.x

Klöffel, T., Barron, J., Nemes, A., Giminez, D., & Jarvis, N. (2024). Soil, climate, time and site factors as drivers of soil structure evolution in agricultural soils from a temperate-boreal region. *Geoderma*, *442*, 116772. https://doi.org/10.1016/j.geoderma.2024.116772

Knapen, A., Poesen, J., Govers, G., Gyssels, G., & Nachtergaele, J. (2007). Resistance of soils to concentrated flow erosion: A review. *Earth-Science Reviews*, *80*(1–2), 75–109. https://doi.org/10.1016/j.earscirev.2006.08.001

Knorr, M., Frey, S. D., & Curtis, P. S. (2005). Nitrogen additions and litter decomposition: A meta-analysis. *Ecology*, *86*(12), 3252–3257. https://doi.org/10.1890/05-0150

Koch, A., McBratney, A., Adams, M., Field, D., Hill, R., Crawford, J., et al. (2013). Soil security: Solving the global soil crisis. *Global Policy*, *4*(4), 434–441. https://doi.org/10.1111/1758-5899.12096

Koch, J. M., & Hobbs, R. J. (2007). Synthesis: Is Alcoa successfully restoring a jarrah forest ecosystem after bauxite mining in Western Australia? *Restoration Ecology*, *15*(s4), S137–S144. https://doi.org/10.1111/j.1526-100x.2007.00301.x

Koo, J. (2024). The economic implications of environmental degradation. *Journal of Environmental Economics*, *45*(2), 123–145.

Kopittke, P. M., Harper, S. M., Asio, L. G., Asio, V. B., Batalon, J. T., Batuigas, A. M. T., et al. (2025). Soil degradation: An integrated model of the causes and drivers. *International Soil and Water Conservation Research*, *13*(4), 744–755. https://doi.org/10.1016/j.iswcr.2025.07.010

Kort, J. (1988). Benefits of windbreaks to field and forage crops. *Agriculture, Ecosystems & Environment*, *22–23*, 165–190. https://doi.org/10.1016/0167-8809(88)90017-5

Kort, J., Collins, M., & Ditsch, D. (1998). A review of soil erosion potential associated with biomass crops. *Biomass and Bioenergy*, *14*(4), 351–359. https://doi.org/10.1016/S0961-9534(97)10071-X

Kosmowski, F. (2018). Soil water management practices (terraces) helped to mitigate the 2015 drought in Ethiopia. *Agricultural Water Management*, *204*, 11–16. https://doi.org/10.1016/j.agwat.2018.02.025

Koubi, V., Stoll, S., & Spilker, G. (2016). Perceptions of environmental change and migration decisions. *Climatic Change*, *138*(3), 439–451. https://doi.org/10.1007/s10584-016-1767-1

Kristensen, J. A., Balstrøm, T., Jones, R. J., Jones, A., Montanarella, L., Panagos, P., & Breuning-Madsen, H. (2019). Development of a harmonised soil profile analytical database for Europe: A resource for supporting regional soil management. *Soils*, *5*(2), 289–301. https://doi.org/10.5194/soil-5-289-2019

Kuhwald, M., Busche, F., Saggau, P., & Duttmann, R. (2022). Is soil loss due to crop harvesting the most disregarded soil erosion process? A review of harvest erosion. *Soil and Tillage Research*, *215*, 105213. https://doi.org/10.1016/j.still.2021.105213

Kumar, A., Das, S. K., Nainegali, L., & Reddy, K. R. (2023). Phytostabilization of coalmine overburden waste rock dump slopes: Current status, challenges, and perspectives. *Bulletin of Engineering Geology and the Environment*, *82*(4), 130. https://doi.org/10.1007/s10064-023-03159-7

Kumar, A., Ramsankaran, R. A. A. J., Brocca, L., & Muñoz-Arriola, F. (2021). A simple machine learning approach to model real-time streamflow using satellite inputs: Demonstration in a data scarce catchment. *Journal of Hydrology*, *595*, 126046. https://doi.org/10.1016/j.jhydrol.2021.126046

Lagnelöv, O., Larsson, G., Larsolle, A., & Hansson, P.-A. (2023). Impact of lowered vehicle weight of electric autonomous tractors in a systems perspective. *Smart Agricultural Technology*, *4*, 100156. https://doi.org/10.1016/j.atech.2022.100156

Lahmar, R., & Ruellan, A. (2007). Soil degradation in the Mediterranean region and cooperative strategies. https://doi.org/10.5555/20073206722
Lal, B., Gautam, P., Nayak, A. K., Panda, B. B., Bihari, P., Tripathi, R., et al. (2019). Energy and carbon budgeting of tillage for environmentally clean and resilient soil health of rice-maize cropping system. *Journal of Cleaner Production*, *226*, 815–830. https://doi.org/10.1016/j.jclepro.2019.04.041
Lal, R. (1993). Soil erosion and conservation in West Africa. *World soil erosion and conservation*, 7–25.
Lal, R. (2001). Soil degradation by erosion. *Land Degradation & Development*, *12*(6), 519–539. https://doi.org/10.1002/ldr.472
Lal, R. (2003). Soil erosion and the global carbon budget. *Environment International*, *29*(4), 437–450. https://doi.org/10.1016/S0160-4120(02)00192-7
Lal, R. (2009). Soils and food sufficiency. A review. *Agronomy for Sustainable Development*, *29*(1), 113–133. https://doi.org/10.1051/agro:2008044
Lal, R. (2012). Climate change and soil degradation mitigation by sustainable management of soils and other natural resources. *Agricultural Research*, *1*(3), 199–212. https://doi.org/10.1007/s40003-012-0031-9
Lal, R. (2016). Feeding 11 billion on 0.5 billion hectare of area under cereal crops. *Food and Energy Security*, *5*(4), 239–251. https://doi.org/10.1002/fes3.99
Lal, R., Bouma, J., Brevik, E., Dawson, L., Field, D. J., Glaser, B., et al. (2021). Soils and sustainable development goals of the United Nations: An International Union of Soil Sciences perspective. *Geoderma Regional*, *25*, e00398. https://doi.org/10.1016/j.geodrs.2021.e00398
Lal, R., Hall, G. F., & Miller, F. P. (1989). Soil degradation: I. Basic processes. *Land Degradation & Development*, *1*(1), 51–69. https://doi.org/10.1002/ldr.3400010106
Lamandé, M., Wildenschild, D., Berisso, F. E., Garbout, A., Marsh, M., Møldrup, P., et al. (2013). X-ray CT and laboratory measurements on glacial till subsoil cores – Assessment of inherent and compaction-affected soil structure characteristics. *Soil Science*, *178*(7), 359–368. https://doi.org/10.1097/SS.0b013e3182a79e1a
Lamas, G. A., Navas-Acien, A., Mark, D. B., & Lee, K. L. (2016). Heavy metals, cardiovascular disease, and the unexpected benefits of chelation therapy. *Journal of the American College of Cardiology*, *67*(20), 2411–2418. https://doi.org/10.1016/j.jacc.2016.02.066
Lamb, D., Erskine, P. D., & Parrotta, J. A. (2005). Restoration of degraded tropical forest landscapes. *Science*, *310*(5754), 1628–1632. https://doi.org/10.1126/science.1111773
Lamb, M. P., Scheingross, J. S., Amidon, W. H., Swanson, E., & Limaye, A. (2011). A model for fire-induced sediment yield by dry ravel in steep landscapes. *Journal of Geophysical Research*, *116*(3), F03006. https://doi.org/10.1029/2010JF001878
Lambin, E. F., & Meyfroidt, P. (2011). Global land use change, economic globalization, and the looming land scarcity. *Proceedings of the National Academy of Sciences*, *108*(9), 3465–3472. https://doi.org/10.1073/pnas.1100480108
Larsbo, M., Koestel, J., Kätterer, T., & Jarvis, N. (2016). Preferential transport in macropores is reduced by soil organic carbon. *Vadose Zone Journal*, *15*(9), 1–7. https://doi.org/10.2136/vzj2016.03.0021
Lebert, M., Böken, H., & Glante, F. (2007). Soil compaction—Indicators for the assessment of harmful changes to the soil in the context of the German Federal Soil Protection Act. *Journal of Environmental Management*, *82*(3), 388–397. https://doi.org/10.1016/j.jenvman.2005.11.022
Le Bissonnais, Y. (2016). Aggregate stability and assessment of soil crustability and erodibility: I. Theory and methodology. *European Journal of Soil Science*, *67*(1), 11–21. https://doi.org/10.1111/ejss.4_12311
Lebron, I., Suarez, D. L., & Alberto, F. (1994). Stability of a calcareous saline-sodic soil during reclamation. *Soil Science Society of America Journal*, *58*(6), 1753–1762. https://doi.org/10.2136/sssaj1994.03615995005800060025x
Lebron, I., Suarez, D. L., & Yoshida, T. (2002). Gypsum effect on the aggregate size and geometry of three sodic soils under reclamation. *Soil Science Society of America Journal*, *66*(1), 92–98. https://doi.org/10.2136/sssaj2002.9200
Lee, H. W., Kang, S.-C., Kim, S.-Y., Cho, Y.-J., & Hwang, S. (2022). Long-term exposure to PM10 increases lung cancer risks: A cohort analysis. Cancer Research and Treatment. *Official Journal of Korean Cancer Association*, *54*(4), 1030–1037. https://doi.org/10.4143/crt.2021.1030
Lehmann, P., Leshchinsky, B., Gupta, S., Mirus, B. B., Bickel, S., Lu, N., & Or, D. (2021). Clays are not created equal: How clay mineral type affects soil parameterization. *Geophysical Research Letters*, *48*(20), e2021Glo95311. https://doi.org/10.1029/2021gl095311
Leibniz University of Hanover. (2021). © Long-Term Soil Erosion Monitoring in Lower Saxony, Germany 2000-2023: Digital Foto.
Le Quéré, C., Andres, R. J., Boden, T., Conway, T., Houghton, R. A., House, J. I., et al. (2013). The global carbon budget 1959–2011. *Earth System Science Data*, *5*(1), 165–185. https://doi.org/10.5194/essd-5-165-2013
Letey, J. (1991). The study of soil structure - Science or art. *Australian Journal of Soil Research*, *29*(6), 699–707. https://doi.org/10.1071/sr9910699
Lew, R., Dobre, M., Srivastava, A., Brooks, E. S., Elliot, W. J., Robichaud, P. R., & Flanagan, D. C. (2022). WEPPcloud: An online watershed-scale hydrologic modeling tool. Part I. Model description. *Journal of Hydrology*, *608*, 127603. https://doi.org/10.1016/j.jhydrol.2022.127603
Li, X., Jin, H., Wang, H., Wu, X., Huang, Y., He, R., et al. (2020). Distributive features of soil carbon and nutrients in permafrost regions affected by forest fires in northern Da Xing'anling (Hinggan) Mountains, NE China. *Catena*, *185*, 104304. https://doi.org/10.1016/j.catena.2019.104304
Li, Y., Chen, J., Zhang, X., Wang, D., Qin, S., Li, A., et al. (2024). Intelligent agriculture: Deep learning in UAV-based remote sensing for pest detection. *Frontiers in Plant Science*, *15*, 1435016. https://doi.org/10.3389/fpls.2024.1435016
Li, Y., Ruysschaert, G., Poesen, J., Zhang, Q. W., Bai, L. Y., Li, L., & Sun, L. F. (2006). Soil losses due to potato and sugar beet harvesting in NE China. *Earth Surface Processes and Landforms*, *31*(8), 1003–1016. https://doi.org/10.1002/esp.1304
Li, Y., Shi, W., Aydin, A., Beroya-Eitner, M. A., & Gao, G. (2020). Loess genesis and worldwide distribution. *Earth-Science Reviews*, *201*, 102947. https://doi.org/10.1016/j.earscirev.2019.102947
Liao, C., Agrawal, A., Clark, P. E., Levin, S. A., & Rubenstein, D. I. (2020). Landscape sustainability science in the drylands: Mobility, rangelands and livelihoods. *Landscape Ecology*, *35*(11), 2433–2447. https://doi.org/10.1007/s10980-020-01068-8
Liao, C., Peng, R., Luo, Y., Zhou, X., Wu, X., Fang, C., et al. (2008). Altered ecosystem carbon and nitrogen cycles by plant invasion: A meta-analysis. *New Phytologist*, *177*(3), 706–714. https://doi.org/10.1111/j.1469-8137.2007.02290.x
Lickley, M., & Solomon, S. (2018). Drivers, timing and some impacts of global aridity change. *Environmental Research Letters*, *13*(10), 104010. https://doi.org/10.1088/1748-9326/aae013
Liebig, M. A., Herrick, J. E., Archer, D. W., Dobrowolski, J., Duiker, S. W., Franzluebbers, A. J., et al. (2017). Aligning land use with land potential: The role of integrated agriculture. *Agricultural & Environmental Letters*, *2*(1), 170007. https://doi.org/10.2134/ael2017.03.0007
Lillak, R. (2005). Integrating efficient grassland farming and biodiversity: Proceedings of the 13th International Occasional Symposium of the European Grassland Federation. Tartu, Estonia, 29 - 31 August 2005. Tartu: European Grassland Federation (Grassland science in Europe, 10).
Lim, M. W., Lau, E. V., & Poh, P. E. (2016). A comprehensive guide of remediation technologies for oil contaminated soil — Present works and future directions. *Marine Pollution Bulletin*, *109*(1), 14–45. https://doi.org/10.1016/j.marpolbul.2016.04.023

Lima, A. T., Mitchell, K., O'Connell, D. W., Verhoeven, J., & Van Cappellen, P. (2016). The legacy of surface mining: Remediation, restoration, reclamation and rehabilitation. *Environmental Science & Policy*, *66*, 227–233.

Limpens, J., Berendse, F., & Klees, H. (2004). How phosphorus availability affects the impact of nitrogen deposition on sphagnum and vascular plants in bogs. *Ecosystems*, *7*(8), 793–804. https://doi.org/10.1007/s10021-004-0274-9

Lin, D., Crabtree, J., Dillo, I., Downs, R. R., Edmunds, R., Giaretta, D., et al. (2020). The TRUST principles for digital repositories. *Scientific Data*, *7*(1), 144. https://doi.org/10.1038/s41597-020-0486-7

Lin, S. S., Shen, S. L., Zhou, A., & Lyu, H. M. (2021). Assessment and management of lake eutrophication: A case study in Lake Erhai, China. *Science of the Total Environment*, *751*, 141618. https://doi.org/10.1016/j.scitotenv.2020.141618

Litchman, E. (2010). Invisible invaders: Non-pathogenic invasive microbes in aquatic and terrestrial ecosystems. *Ecology Letters*, *13*(12), 1560–1572. https://doi.org/10.1111/j.1461-0248.2010.01544.x

Liu, B., Xie, Y., Li, Z., Liang, Y., Zhang, W., Fu, S., et al. (2020). The assessment of soil loss by water erosion in China. *International Soil and Water Conservation Research*, *8*(4), 430–439. https://doi.org/10.1016/j.iswcr.2020.07.002

Liu, J., Gao, G., Wang, S., Jiao, L., Wu, X., & Fu, B. (2018). The effects of vegetation on runoff and soil loss: Multidimensional structure analysis and scale characteristics. *Journal of Geographical Sciences*, *28*(1), 59–78. https://doi.org/10.1007/s11442-018-1459-z

Liu, Z., Deng, K., Zheng, H., Zhu, Y., & Shi, Z. (2024). Effects of tillage practices on runoff and soil losses in response to different crop growth stages in the red soil region of southern China. *Journal of Soils and Sediments*, *24*(5), 2199–2212. https://doi.org/10.1007/s11368-024-03790-1

Lobb, D. A., Kachanoski, R. G., & Miller, M. H. (1995). Tillage translocation and tillage erosion on shoulder slope landscape positions measured using 137 Cs as a tracer. *Canadian Journal of Soil Science*, *75*(2), 211–218. https://doi.org/10.4141/cjss95-029

Løkke, H., Jesper, B., Falkengren-Grerup, U., Finlay, R. D., Ilvesniemi, H., Nygaard, P. H., & Starr, M. (1996). Critical loads of acidic deposition for forest soils: Is the current approach adequate? *Ambio*, *25*(8), 510–516.

Lu, S.-F., Han, Z.-J., Xu, L., Lan, T.-G., Wei, X., & Zhao, T.-Y. (2023). On measuring methods and influencing factors of air permeability of soils: An overview and a preliminary database. *Geoderma*, *435*, 116509. https://doi.org/10.1016/j.geoderma.2023.116509

Lu, Y., Fu, B., Feng, X., Zeng, Y., Liu, Y., Chang, R., et al. (2012). A policy-driven large scale ecological restoration: Quantifying ecosystem services changes in the Loess Plateau of China. *PLoS One*, *7*(2), e31782. https://doi.org/10.1371/journal.pone.0031782

Lukacz, P. M. (2022). Data capitalism, Microsoft's planetary computer, and the biodiversity informatics community. In *International Conference on Information* (pp. 355–369). Springer International Publishing.

Luo, X., Hou, E., Zhang, L., Kuang, Y., & Wen, D. (2023). Altered soil microbial properties and functions after afforestation increase soil carbon and nitrogen but not phosphorus accumulation. *Biology and Fertility of Soils*, *59*(6), 645–658. https://doi.org/10.1007/s00374-023-01726-4

Ma, Y., Woolf, D., Fan, M., Qiao, L., Li, R., & Lehmann, J. (2023). Global crop production increase by soil organic carbon. *Nature Geoscience*, *16*(12), 1159–1165. https://doi.org/10.1038/s41561-023-01302-3

Maas, E. V., & Grattan, S. (1999). Crop yields as affected by salinity. *Agricultural Drainage*, *38*, 55–108. https://doi.org/10.2134/agronmonogr38.c3

Mahmuduzzaman, M., Ahmed, Z. U., Nuruzzaman, A. K. M., & Ahmed, F. R. S. (2014). Causes of salinity intrusion in coastal belt of Bangladesh. *International Journal of Plant Research*, *4*(4A), 8–13.

Mahowald, N. M., Baker, A. R., Bergametti, G., Brooks, N., Duce, R. A., Jickells, T. D., et al. (2005). Atmospheric global dust cycle and iron inputs to the ocean. *Global Biogeochemical Cycles*, *19*(GB4025). https://doi.org/10.1029/2004gb002402

Mai, J., Wang, Z., Hu, F., Huang, J., & Zhao, S. W. (2023). Study on soil hydraulic properties of slope farmlands with different degrees of erosion degradation in a typical black soil region. *PeerJ*, *11*, e15930. https://doi.org/10.7717/peerj.15930

Malik, A. A.,Puissant, J.,Buckeridge, K. M., Goodall, T.,Jehmlich, N.,Chowdhury, S., et al. (2018). Land use driven change in soil pH affects microbial carbon cycling processes. *Nature Communications*, *9*(1), 3591. https://doi.org/10.1038/s41467-018-05980-1

Martinez-Beltran, J., & Manzur, C. L. (2005). Overview of salinity problems in the world and FAO strategies to address the problem. In *Proceedings of the international salinity forum* (pp. 311–313).

Mayor, A. G., Bautista, S., Llovet, J., & Bellot, J. (2007). Post-fire hydrological and erosional responses of a Mediterranean landscpe: Seven years of catchment-scale dynamics. *Catena*, *71*(1), 68–75. https://doi.org/10.1016/j.catena.2006.10.006

Mazhar, S., Pellegrini, E., Contin, M., Bravo, C., & De Nobili, M. (2022). Impacts of salinization caused by sea level rise on the biological processes of coastal soils-a review. *Frontiers in Environmental Science*, *10*, 909415. https://doi.org/10.3389/fenvs.2022.909415

McBratney, A., Field, D. J., & Koch, A. (2014). The dimensions of soil security. *Geoderma*, *213*, 203–213. https://doi.org/10.1016/j.geoderma.2013.08.013

McCormick, R., Thenkabail, P. S., Aneece, I., Teluguntla, P., Oliphant, A. J., & Foley, D. (2025). Artificial neural network multi-layer perceptron models to classify California's crops using Harmonized Landsat Sentinel (HLS) Data. *Photogrammetric Engineering & Remote Sensing*, *91*(2), 91–100. https://doi.org/10.14358/pers.24-00072r3

McGuire, L. A., Ebel, B. A., Rengers, F. K., Vieira, D. C. S., & Nyman, P. (2024). Fire effects on geomorphic processes. *Nature Reviews Earth & Environment*, *5*(7), 486–503. https://doi.org/10.1038/s43017-024-00557-7

McGuire, L. A., Rengers, F. K., Kean, J. W., & Staley, D. M. (2017). Debris flow initiation by runoff in a recently burned basin: Is grain-by-grain sediment bulking or en masse failure to blame? *Geophysical Research Letters*, *44*(14), 7310–7319. https://doi.org/10.1002/2017GL074243

McLeman, R. (2017). Migration and land degradation: Recent experience and future trends. (Global Land Outlook Working Paper).

McLeman, R., & Gemenne, F. (2018). Environmental migration research: Evolution and current state of the science. In *Routledge handbook of environmental displacement and migration*. Routledge.

McLeman, R. A. (2014). *Climate and human migration: Past experiences, future challenges*. Cambridge University Press.

McPhee, J. E., Antille, D. L., Tullberg, J. N., Doyle, R. B., & Boersma, M. (2020). Managing soil compaction – A choice of low-mass autonomous vehicles or controlled traffic? *Biosystems Engineering*, *195*, 227–241. https://doi.org/10.1016/j.biosystemseng.2020.05.006

MEA. Millennium Ecosystem Assessment. (2005). *Ecosystems and human wellbeing: desertification synthesis* (Vol. 2005). World Resource Institute.

Mekuria, W., Veldkamp, E., Tilahun, M., Olschewski, R., & Muys, B. (2011). Economic valuation of land restoration: The case of exclosures established on communal grazing lands in Tigray, Ethiopia. *Land Degradation & Development*, *22*(3), 334–344. https://doi.org/10.1002/ldr.1001

Merritt, W. S., Letcher, R. A., & Jakeman, A. J. (2003). A review of erosion and sediment transport models. *Environmental Modelling & Software*, *18*(8–9), 761–799. https://doi.org/10.1016/S1364-8152(03)00078-1

Metternicht, G. I., & Zinck, J. A. (2003). Remote sensing of soil salinity: Potentials and constraints. *Remote Sensing of Environment*, *85*(1), 1–20. https://doi.org/10.1016/s0034-4257(02)00188-8

Mganga, K. Z., Musimba, N. K. R., & Nyariki, D. M. (2015). Combining sustainable land management technologies to combat land degradation and improve rural livelihoods in semi-arid lands in Kenya. *Environmental Management*, *56*(6), 1538–1548. https://doi.org/10.1007/s00267-015-0579-9

Middleton, N. (2024). Impacts of sand and dust storms on food production. Environmental Research. *Food Systems*, *1*(2), 022003. https://doi.org/10.1088/2976-601x/ad63ac

Milazzo, F., Francksen, R. M., Zavattaro, L., Abdalla, M., Hejduk, S., Enri, S. R., et al. (2023). The role of grassland for erosion and flood mitigation in Europe: A meta-analysis. *Agriculture, Ecosystems & Environment*, *348*, 108443. https://doi.org/10.1016/j.agee.2023.108443

Miller, M. E., Breffle, W. S., Battaglia, M., Banach, D., Robichaud, P. R., Elliot, W. J., et al. (2022). Socio-economic impact of the Rapid Response Erosion Database (RRED). *Journal of Geoscience and Environment Protection*, *10*, 103–125. https://doi.org/10.4236/gep.2022.1010009

Minasny, B., Bandai, T., Ghezzehei, T. A., Huang, Y. C., Ma, Y., McBratney, A. B., et al. (2024). Soil science-informed machine learning. *Geoderma*, *452*, 117094. https://doi.org/10.1016/j.geoderma.2024.117094

Minkkinen, K., & Laine, J. (1998). Effect of forest drainage on the peat bulk density of pine mires in Finland. *Canadian Journal of Forest Research*, *28*(2), 178–186. https://doi.org/10.1139/cjfr-28-2-178

Mirzabaev, A., Strokov, A., & Krasilnikov, P. (2023). The impact of land degradation on agricultural profits and implications for poverty reduction in Central Asia. *Land Use Policy*, *126*, 106530. https://doi.org/10.1016/j.landusepol.2022.106530

Mishra, V., Limaye, A. S., Muench, R. E., Cherrington, E. A., & Markert, K. N. (2020). Evaluating the performance of high-resolution satellite imagery in detecting ephemeral water bodies over West Africa. *International Journal of Applied Earth Observation and Geoinformation*, *93*, 102218. https://doi.org/10.1016/j.jag.2020.102218

Molchanov, E. N., Savin, I. Y., Yakovlev, A. S., Bulgakov, D. S., & Makarov, O. A. (2015). National approaches to evaluation of the degree of soil degradation. *Eurasian Soil Science*, *48*(11), 1268–1277. https://doi.org/10.1134/S1064229315110113

Moldrup, P., Olesen, T., Komatsu, T., Schjønning, P., & Rolston, D. E. (2001). Tortuosity, diffusivity, and permeability in the soil liquid and gaseous phases. *Soil Science Society of America Journal*, *65*(3), 613–623. https://doi.org/10.2136/sssaj2001.653613x

Montgomery, D. R. (2007). Soil erosion and agricultural sustainability. *Proceedings of the National Academy of Sciences*, *104*(33), 13268–13272. https://doi.org/10.1073/pnas.0611508104

Moody, J. A., & Martin, D. A. (2009). Synthesis of sediment yields after wildland fire in different rainfall regimes in the western United States. *International Journal of Wildland Fire*, *18*(1), 96–115. https://doi.org/10.1071/WF07162

Moreno-Jiménez, E., Orgiazzi, A., Jones, A., Saiz, H., Aceña-Heras, S., & Plaza, C. (2022). Aridity and geochemical drivers of soil micronutrient and contaminant availability in European drylands. *European Journal of Soil Science*, *73*(1), e13163. https://doi.org/10.1111/ejss.13163

Moreno-Jiménez, E., Plaza, C., Saiz, H., Manzano, R., Flagmeier, M., & Maestre, F. T. (2019). Aridity and reduced soil micronutrient availability in global drylands. *Nature Sustainability*, *2*(5), 371–377. https://doi.org/10.1038/s41893-019-0262-x

Moyano, J., Dimarco, R. D., Paritsis, J., Peterson, T., Peltzer, D. A., Crawford, K. M., et al. (2024). Unintended consequences of planting native and non-native trees in treeless ecosystems to mitigate climate change. *Journal of Ecology*, *112*(11), 2480–2491. https://doi.org/10.1111/1365-2745.14300

Muhs, D. R., Prospero, J. M., Baddock, M. C., & Gill, T. E. (2014). Identifying sources of aeolian mineral dust: Present and past. In *Mineral dust* (pp. 51–74). Springer.

Müller, K., Dal Ferro, N., Katuwal, S., Tregurtha, C., Zanini, F., Carmignato, S., et al. (2018). Effect of long-term irrigation and tillage practices on X-ray CT and gas transport derived pore-network characteristics. *Soil Research*, *57*(6), 657–669. https://doi.org/10.1071/SR18210

Mullins, C. E., MacLeod, D. A., Northcote, K. H., Tisdall, J. M., & Young, I. M. (1990). Hardsetting Soils: Behavior, Occurrence, and Management. In R. Lal & B. A. Stewart (Eds.), Advances in Soil Science: *Soil Degradation Volume 11* (pp. 37–108). Springer.

Murchie, A. K., & Gordon, A. W. (2013). The impact of the 'New Zealand flatworm', Arthurdendyus triangulatus, on earthworm populations in the field. *Biological Invasions*, *15*(3), 569–586. https://doi.org/10.1007/s10530-012-0309-7

Nabel, M., Selig, C., Gundlach, J., von Der Decken, H., & Klein, M. (2021). Biodiversity in agricultural used soils: Threats and options for its conservation in Germany and Europe. *Soil Organisms*, *93*(1), 1–11.

Nachtergaele, F. O., & Licona-Manzur, C. (2008). The Land Degradation Assessment in Drylands (LADA) Project: Reflections on indicators for land degradation assessment. In *The Future of Drylands:* International Scientific Conference on Desertification and Drylands Research Tunis, Tunisia, 19-21 June 2006 (pp. 327–348). Springer.

Nagajyoti, P. C., Lee, K. D., & Sreekanth, T. V. (2010). Heavy metals, occurrence and toxicity for plants: A review. *Environmental Chemistry Letters*, *8*(3), 199–216. https://doi.org/10.1007/s10311-010-0297-8

Najdenko, E., Lorenz, F., Dittert, K., & Olfs, H. W. (2024). Rapid in-field soil analysis of plant-available nutrients and pH for precision agriculture—A review. *Precision Agriculture*, *25*(6), 3189–3218. https://doi.org/10.1007/s11119-024-10181-6

Nauman, T. W., Kienast-Brown, S., Roecker, S. M., Brungard, C., White, D., Philippe, J., & Thompson, J. A. (2024). Soil landscapes of the United States (SOLUS): Developing predictive soil property maps of the conterminous United States using hybrid training sets. *Soil Science Society of America Journal*, *88*(6), 2046–2065. https://doi.org/10.1002/saj2.20769

Naveed, M., Moldrup, P., Arthur, E., Wildenschild, D., Eden, M., Lamandé, M., et al. (2013). Revealing soil structure and functional macroporosity along a clay gradient using X-ray computed tomography. *Soil Science Society of America Journal*, *77*(2), 403–411. https://doi.org/10.2136/sssaj2012.0134

Nazari, M., Arthur, E., Lamandé, M., Keller, T., Bilyera, N., & Bickel, S. (2023). A meta-analysis of soil susceptibility to machinery-induced compaction in forest ecosystems across global climatic zones. *Current Forestry Reports*, *9*(5), 370–381. https://doi.org/10.1007/s40725-023-00197-y

Nazari, M., Eteghadipour, M., Zarebanadkouki, M., Ghorbani, M., Dippold, M. A., Bilyera, N., & Zamanian, K. (2021). Impacts of logging-associated compaction on Forest soils: A meta-analysis. *Frontiers in Forests and Global Change*, *4*, 780074. https://doi.org/10.3389/ffgc.2021.780074

Nearing, M. A., Xie, Y., Liu, B., & Ye, Y. (2017). Natural and anthropogenic rates of soil erosion. *International Soil and Water Conservation Research*, *5*(2), 77–84. https://doi.org/10.1016/j.iswcr.2017.04.001

Neary, D. G., Klopatek, C. C., DeBano, L. F., & Ffolliott, P. F. (1999). Fire effects on belowground sustainability: A review and synthesis. *Forest Ecology and Management*, *122*(1), 51–71. https://doi.org/10.1016/S0378-1127(99)00032-8

Neary, D. G., Ryan, K. C., & DeBano, L. F. (2005). Wildland fire in ecosystems: Effects of fire on soils and water. Gen. Tech. Rep. RMRS-GTR-42-Vol.4. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 250 (p. 42). https://doi.org/10.2737/RMRS-GTR-42-V4

Needham, P., Scholz, G., & Moore, G. (2004). Physical restrictions to root growth. In G. Moore (Ed.), *Soil guide: a handbook for understanding and managing agricultural soils*, *Dept of Agriculture, Western Australia. Bulletin No. 4343* (pp. 109–124).

Negacz, K., Malek, Ž., de Vos, A., & Vellinga, P. (2022). Saline soils worldwide: Identifying the most promising areas for saline agriculture. *Journal of Arid Environments*, *203*, 104775. https://doi.org/10.1016/j.jaridenv.2022.104775

Negesse, Z., Pan, K., Guadie, A., Justine, M. F., Azene, B., Pandey, B., et al. (2025). Plant invasions alter soil biota and microbial activities: A global meta-analysis. *Plant and Soil*, *513*(1), 1031–1050. https://doi.org/10.1007/s11104-025-07227-7

Nellemann, C., & Thomsen, M. G. (2001). Long-term changes in forest growth: Potential effects of nitrogen deposition and acidification. *Water, Air, and Soil Pollution*, *128*(3), 197–205. https://doi.org/10.1023/a:1010318800180

Nemes, A., Rawls, W. J., & Pachepsky, Y. A. (2005). Influence of organic matter on the estimation of saturated hydraulic conductivity. *Soil Science Society of America Journal*, *69*(4), 1330–1337. https://doi.org/10.2136/sssaj2004.0055

Neumann, K., Sietz, D., Hilderink, H., Janssen, P., Kok, M., & van Dijk, H. (2015). Environmental drivers of human migration in drylands – A spatial picture. *Applied Geography*, *56*, 116–126. https://doi.org/10.1016/j.apgeog.2014.11.021

Neupane, A., Bulbul, I., Wang, Z., Lehman, R. M., Nafziger, E., & Marzano, S. Y. L. (2021). Long term crop rotation effect on subsequent soybean yield explained by soil and root-associated microbiomes and soil health indicators. *Scientific Reports*, *11*(1), 9200. https://doi.org/10.1038/s41598-021-88784-6

Nevermann, H., Aminzadeh, M., Madani, K., & Shokri, N. (2024). Quantifying water evaporation from large reservoirs: Implications for water management in water-stressed regions. *Environmental Research*, *262*(1), 119860. https://doi.org/10.1016/j.envres.2024.119860

Nguyen, A. D., & Savenije, H. H. (2006). Salt intrusion in multi-channel estuaries: A case study in the Mekong Delta, Vietnam. *Hydrology and Earth System Sciences*, *10*(5), 743–754. https://doi.org/10.5194/hess-10-743-2006

Nie, X. J., Zhao, T. Q., & Qiao, X. N. (2013). Impacts of soil erosion on organic carbon and nutrient dynamics in an alpine grassland soil. *Soil Science & Plant Nutrition*, *59*(4), 660–668. https://doi.org/10.1080/00380768.2013.795475

Nissenbaum, H. (2004). Privacy as contextual integrity. *Washington Law Review*, *79*, 119.

Nkonya, E., Anderson, W., Kato, E., Koo, J., Mirzabaev, A., von Braun, J., & Meyer, S. (2016). Global cost of land degradation. In E. Nkonya, A. Mirzabaev, & J. von Braun (Eds.), *Economics of Land Degradation and Improvement – A Global Assessment for Sustainable Development* (pp. 117–165). Springer International Publishing. https://doi.org/10.1007/978-3-319-19168-3_6

Nobel, P. S., Geller, G. N., Kee, S. C., & Zimmerman, A. D. (1986). Temperatures and thermal tolerances for cacti exposed to high temperatures near the soil surface. *Plant, Cell and Environment*, *9*(4), 279–287. https://doi.org/10.1111/1365-3040.ep11611688

Noori, R., Maghrebi, M., Jessen, S., Bateni, S. M., Heggy, E., Javadi, S., et al. (2023). Decline in Iran's groundwater recharge. *Nature Communications*, *14*(1), 6674. https://doi.org/10.1038/s41467-023-42411-2

Nordfjell, T., Öhman, E., Lindroos, O., & Ager, B. (2019). The technical development of forwarders in Sweden between 1962 and 2012 and of sales between 1975 and 2017. *International Journal of Forest Engineering*, *30*, 1–13. https://doi.org/10.1080/14942119.2019.1591074

Nordstrom, K. F., & Hotta, S. (2004). Wind erosion from cropland in the USA: A review of problems, solutions and prospects. *Geoderma*, *121*(3–4), 157–167. https://doi.org/10.1016/j.geoderma.2003.11.012

Nyssen, J., Frankl, A., Zenebe, A., Deckers, J., & Poesen, J. (2015). Land management in the northern Ethiopian highlands: Local and global perspectives; past, present and future. *Land Degradation & Development*, *26*(6), 759–764. https://doi.org/10.1002/ldr.2336

Nyssen, J., Poesen, J., & Deckers, J. (2009). Land degradation and soil and water conservation in tropical highlands. *Soil and Tillage Research*, *103*(2), 197–202. https://doi.org/10.1016/j.still.2008.08.002

Oades, J. M., & Waters, A. G. (1991). Aggregate hierarchy in soils. *Soil Research*, *29*(6), 815–828. https://doi.org/10.1071/sr9910815

Obokata, R., Veronis, L., & McLeman, R. (2014). Empirical research on international environmental migration: A systematic review. *Population and Environment*, *36*(1), 111–135. https://doi.org/10.1007/s11111-014-0210-7

Obour, P. B., & Ugarte, C. M. (2021). A meta-analysis of the impact of traffic-induced compaction on soil physical properties and grain yield. *Soil and Tillage Research*, *211*, 105019. https://doi.org/10.1016/j.still.2021.105019

Oenema O, B. F., Lammel, J., Bascou, P., Billen, G., Dobermann, A., Erisman, J. W., et al. (2016). *Nitrogen Use Efficiency (NUE) - EU Nitrogen Expert Panel, Nitrogen Use Efficiency (NUE) – Guidance document for assessing NUE at farm level*. Wageningen University. Alterra, PO Box 47, NL-6700 Wageningen, Netherlands.

Okin, G. S., Baker, A. R., Tegen, I., Mahowald, N. M., Dentener, F. J., Duce, R. A., et al. (2011). Impacts of atmospheric nutrient deposition on marine productivity: Roles of nitrogen, phosphorus, and iron. *Global Biogeochemical Cycles*, *25*(2), GB2022. https://doi.org/10.1029/2010gb003858

Okin, G. S., Mahowald, N., Chadwick, O. A., & Artaxo, P. (2004). Impact of desert dust on the biogeochemistry of phosphorus in terrestrial ecosystems. *Global Biogeochemical Cycles*, *18*(2). https://doi.org/10.1029/2003gb002145

Oldeman, L. R. (1992). Global extent of soil degradation. In *Bi-annual report 1991-1992/ISRIC* (pp. 19–36). ISRIC.

Oldeman, L. R., Hakkeling, R. T. A., & Sombroek, W. G. (1991). World map of the status of human-induced soil degradation: An explanatory note.

Oldeman, R. L., Hakkeling, R. T. A., & Sombroek, W. G. (1990). *World map of the status of human-induced soil degradation*. International Soil Reference and Information Centre.

Oldfield, E. E., Bradford, M. A., & Wood, S. A. (2019). Global meta-analysis of the relationship between soil organic matter and crop yields. *Soil*, *5*(1), 15–32. https://doi.org/10.5194/soil-5-15-2019

Oliveira, E. M., Wittwer, R., Hartmann, M., Keller, T., Buchmann, N., & van der Heijden, M. G. A. (2024). Effects of conventional, organic and conservation agriculture on soil physical properties, root growth and microbial habitats in a long-term field experiment. *Geoderma*, *447*, 116927. https://doi.org/10.1016/j.geoderma.2024.116927

Olsson, L., Cotrufo, F., Crews, T., Franklin, J., King, A., Mirzabaev, A., et al. (2023). The State of the World's Arable Land. *Annual Review of Environment and Resources*, *48*(1), 451–475. https://doi.org/10.1146/annurev-environ-112320-113741

Omuto, C. T., Vargas, R. R., El Mobarak, A. M., Mohamed, N., Viatkin, K., & Yigini, Y. (2020). Mapping of salt-affected soils–technical manual.

Or, D., Furtak-Cole, E., Berli, M., Shillito, R., Ebrahimian, H., Vahdat-Aboueshagh, H., & McKenna, S. A. (2023). Review of wildfire modeling considering effects on land surfaces. *Earth-Science Reviews*, *245*, 104569. https://doi.org/10.1016/j.earscirev.2023.104569

Or, D., & Ghezzehei, T. A. (2002). Modeling post-tillage soil structural dynamics: A review. *Soil and Tillage Research*, *64*(1–2), 41–59. https://doi.org/10.1016/s0167-1987(01)00256-2

Or, D., Keller, T., & Schlesinger, W. H. (2021). Natural and managed soil structure: On the fragile scaffolding for soil functioning. *Soil and Tillage Research*, *208*, 104912. https://doi.org/10.1016/j.still.2020.104912

Orgiazzi, A., Ballabio, C., Panagos, P., Jones, A., & Fernández-Ugalde, O. (2018). LUCAS soil, the largest expandable soil dataset for Europe: A review. *European Journal of Soil Science*, *69*(1), 140–153. https://doi.org/10.1111/ejss.12499

Ossai, I. C., Ahmed, A., Hassan, A., & Hamid, F. S. (2020). Remediation of soil and water contaminated with petroleum hydrocarbon: A review. *Environmental Technology & Innovation*, *17*, 100526. https://doi.org/10.1016/j.eti.2019.100526

Oster, J. D., & Frenkel, H. (1980). The chemistry of the reclamation of sodic soils with gypsum. *Soil Science Society of America Journal*, *44*(1), 41–45. https://doi.org/10.2136/sssaj1980.03615995004400010010x

O'Sullivan, M. F., & Robertson, E. A. G. (1996). Critical state soil parameters from intact samples of two agricultural topsoils. *Soil and Tillage Research*, *39*, 161–173.

Öttl, L. K., Wilken, F., Hupfer, A., Sommer, M., & Fiener, P. (2022). Non-inversion conservation tillage as an underestimated driver of tillage erosion. *Scientific Reports*, *12*(1), 20704. https://doi.org/10.1038/s41598-022-24749-7

Öttl, L. K., Wilken, F., Juřicová, A., Batista, P. V. G., & Fiener, P. (2024). A millennium of arable land use – The long-term impact of tillage and water erosion on landscape-scale carbon dynamics. *Soil*, *10*(1), 281–305. https://doi.org/10.5194/soil-10-281-2024

Pacheco, P., Beatty, C., & Patel, J. (2024). An economic view on the costs and benefits of forest restoration. In *Restoring Forests and Trees for Sustainable Development:* Policies, Practices, Impacts, and Ways Forward (pp. 238–260). Oxford Academic. https://doi.org/10.1093/9780197683958.003.0009

Pachepsky, Y. A., & Rawls, W. J. (2003). Soil structure and pedotransfer functions. *European Journal of Soil Science*, *54*(3), 443–452. https://doi.org/10.1046/j.1365-2389.2003.00485.x

Padarian, J., Stockmann, U., Minasny, B., & McBratney, A. B. (2022). Monitoring changes in global soil organic carbon stocks from space. *Remote Sensing of Environment*, *281*, 113260. https://doi.org/10.1016/j.rse.2022.113260

Pagenkemper, S. K., Uteau Puschmann, D., Peth, S., & Horn, R. (2013). Investigation of time dependent development of soil structure and formation of macropore networks as affected by various precrop species. *International Soil and Water Conservation Research*, *2*, 51–66. https://doi.org/10.1016/s2095-6339(15)30006-x

Pahalvi, H. N., Rafiya, L., Rashid, S., Nisar, B., & Kamili, A. N. (2021). Chemical fertilizers and their impact on soil health. In *Microbiota and Biofertilizers* (pp. 1–20). Springer. https://doi.org/10.1007/978-3-030-61010-4_1

Pan, G., Smith, P., & Pan, W. (2009). The role of soil organic matter in maintaining the productivity and yield stability of cereals in China. *Agriculture, Ecosystems & Environment*, *129*(1), 344–348. https://doi.org/10.1016/j.agee.2008.10.008

Panagos, P., Ballabio, C., Himics, M., Scarpa, S., Matthews, F., Bogonos, M., et al. (2021). Projections of soil loss by water erosion in Europe by 2050. *Environmental Science & Policy*, *124*(1), 380–392. https://doi.org/10.1016/j.envsci.2021.07.012

Panagos, P., Borrelli, P., Jones, A., & Robinson, D. A. (2024). A 1 billion euro mission: A soil deal for Europe. *European Journal of Soil Science*, *75*(1), e13466. https://doi.org/10.1111/ejss.13466

Panagos, P., Borrelli, P., Matthews, F., Liakos, L., Bezak, N., Diodato, N., & Ballabio, C. (2022). Global rainfall erosivity projections for 2050 and 2070. *Journal of Hydrology*, *610*, 127865. https://doi.org/10.1016/j.jhydrol.2022.127865

Panagos, P., Borrelli, P., Meusburger, K., Yu, B., Klik, A., Jae Lim, K., et al. (2017). Global rainfall erosivity assessment based on high-temporal resolution rainfall records. *Scientific Reports*, *7*(1), 1–12. https://doi.org/10.1038/s41598-017-04282-8

Panagos, P., Borrelli, P., & Poesen, J. (2019). Soil loss due to crop harvesting in the European Union: A first estimation of an underrated geomorphic process. *The Science of the Total Environment*, *664*, 487–498. https://doi.org/10.1016/j.scitotenv.2019.02.009

Panagos, P., Borrelli, P., Poesen, J., Ballabio, C., Lugato, E., Meusburger, K., et al. (2015). The new assessment of soil loss by water erosion in Europe. *Environmental Science & Policy*, *54*, 438–447. https://doi.org/10.1016/j.envsci.2015.08.012

Panagos, P., Broothaerts, N., Ballabio, C., Orgiazzi, A., De Rosa, D., Borrelli, P., et al. (2024). How the EU Soil Observatory is providing solid science for healthy soils. *European Journal of Soil Science*, *75*(3), e13507. https://doi.org/10.1111/ejss.13507

Panagos, P., De Rosa, D., Liakos, L., Labouyrie, M., Borrelli, P., & Ballabio, C. (2024). Soil bulk density assessment in Europe. *Agriculture, Ecosystems & Environment*, *364*, 108907. https://doi.org/10.1016/j.agee.2024.108907

Panagos, P., Hengl, T., Wheeler, I., Marcinkowski, P., Rukeza, M. B., Yu, B., et al. (2023). Global rainfall erosivity database (GloREDa) and monthly R-factor data at 1 km spatial resolution. *Data in Brief*, *50*, 109482. https://doi.org/10.1016/j.dib.2023.109482

Panagos, P., Jones, A., Lugato, E., & Ballabio, C. (2025). A Soil Monitoring Law for Europe. *Global Challenges*, *9*(3), 2400336. https://doi.org/10.1002/gch2.202400336

Panagos, P., Karydas, C. G., Borrellia, P., Ballabio, B., & Meusburger, K. (2014). Advances in soil erosion modelling through remote sensing data availability at European scale. *Proceedings of SPIE*, *9229*, 92290I. https://doi.org/10.1117/12.2066383

Panagos, P., Matthews, F., Patault, E., de Michele, C., Quaranta, E., Bezak, N., et al. (2024). Understanding the cost of soil erosion: An assessment of the sediment removal costs from the reservoirs of the European Union. *Journal of Cleaner Production*, *434*, 140183. https://doi.org/10.1016/j.jclepro.2023.140183

Panagos, P., Van Liedekerke, M., Borrelli, P., Köninger, J., Ballabio, C., Orgiazzi, A., et al. (2022). European Soil Data Centre 2.0: Soil data and knowledge in support of the EU policies. *European Journal of Soil Science*, *73*(6), e13315. https://doi.org/10.1111/ejss.13315

Panagos, P., Van Liedekerke, M., Jones, A., & Montanarella, L. (2012). European Soil Data Centre: Response to European policy support and public data requirements. *Land Use Policy*, *29*(2), 329–338. https://doi.org/10.1016/j.landusepol.2011.07.003

Park, S. J., & Lee, D. K. (2021). Predicting susceptibility to landslides under climate change impacts in metropolitan areas of South Korea using machine learning. *Geomatics, Natural Hazards and Risk*, *12*(1), 2462–2476. https://doi.org/10.1080/19475705.2021.1963328

Parlak, M., & Blanco-Canqui, H. (2015). Soil losses due to potato harvesting: A case study in western Turkey. *Soil Use & Management*, *31*(4), 525–527. https://doi.org/10.1111/sum.12225

Parlak, M., Everest, T., & Blanco-Canqui, H. (2021). Soil Loss due to Sugar Beet Harvesting in Northwestern Turkey. *Journal of Soil Science and Plant Nutrition*, *21*(4), 2992–3001. https://doi.org/10.1007/s42729-021-00584-z

Parlak, M., Everest, T., Tunçay, T., Caballero-Calvo, A., & Rodrigo-Comino, J. (2022). Soil losses due to leek and groundnut root crop harvesting: An unstudied regional problem in Turkey. *Land Degradation & Development*, *33*(11), 1799–1809. https://doi.org/10.1002/ldr.4262

Parlak, M., Palta, Ç., Yokuş, S., Blanco-Canqui, H., & Çarkacı, D. A. (2016). Soil losses due to carrot harvesting in south central Turkey. *Catena*, *140*, 24–30. https://doi.org/10.1016/j.catena.2016.01.012

Pastrana, A. M., Borrero, C., Pérez, A. G., & Avilés, M. (2023). Soilborne pathogens affect strawberry fruit flavor and quality. *Plant Science*, *326*, 111533. https://doi.org/10.1016/j.plantsci.2022.111533

Peralta, A. L., Sun, Y., McDaniel, M. D., & Lennon, J. T. (2018). Crop rotational diversity increases disease suppressive capacity of soil microbiomes. *Ecosphere*, *9*(5), e02235. https://doi.org/10.1002/ecs2.2235

Pereira, J. O., Défossez, P., & Richard, G. (2007). Soil susceptibility to compaction by wheeling as function of some properties of a silty soil as affected by the tillage system. *European Journal of Soil Science*, *58*, 34–44. https://doi.org/10.1111/j.1365-2389.2006.00798.x

Pérez-Lucas, G., Vela, N., El Aatik, A., & Navarro, S. (2019). Environmental risk of groundwater pollution by pesticide leaching through the soil profile. *Pesticides-use and misuse and their impact in the environment*, *17*, 1–28.

Pesch, C., Weber, P. L., de Jonge, L. W., Greve, M. H., Norgaard, T., & Moldrup, P. (2021). Soil–air phase characteristics: Response to texture, density, and land use in Greenland and Denmark. *Soil Science Society of America Journal*, *85*(5), 1534–1554. https://doi.org/10.1002/saj2.20284

Pfahl, S., O'Gorman, P. A., & Fischer, E. M. (2017). Understanding the regional pattern of projected future changes in extreme precipitation. *Nature Climate Change*, *7*(6), 423–427. https://doi.org/10.1038/nclimate3287

Phefadu, K., & Munjonji, L. (2022). The use of Visual Evaluation of Soil Structure (VESS) method to assess soil quality: Review. *SSRN Electronic Journal*. https://doi.org/10.2139/ssrn.4249884

Philippot, L., Chenu, C., Kappler, A., Rillig, M. C., & Fierer, N. (2024). The interplay between microbial communities and soil properties. *Nature Reviews Microbiology*, *22*(4), 226–239. https://doi.org/10.1038/s41579-023-00980-5

Phillips, M. L., McNellis, B. E., Howell, A., Lauria, C. M., Belnap, J., & Reed, S. C. (2022). Biocrusts mediate a new mechanism for land degradation under a changing climate. *Nature Climate Change*, *12*(1), 71–76. https://doi.org/10.1038/s41558-021-01249-6

Phoenix, G. K., Emmett, B. A., Britton, A. J., Caporn, S. J. M., Dise, N. B., Helliwell, R., et al. (2012). Impacts of atmospheric nitrogen deposition: Responses of multiple plant and soil parameters across contrasting ecosystems in long-term field experiments. *Global Change Biology*, *18*(4), 1197–1215. https://doi.org/10.1111/j.1365-2486.2011.02590.x

Pichtel, J. (2016). Oil and gas production wastewater: Soil contamination and pollution prevention. *Applied and Environmental Soil Science*, *2016*, 2707989. https://doi.org/10.1155/2016/2707989

Pitcairn, C. E. R., Leith, I. D., Sheppard, L. J., Sutton, M. A., Fowler, D., Munro, R. C., et al. (1998). The relationship between nitrogen deposition, species composition and foliar nitrogen concentrations in woodland flora in the vicinity of livestock farms. *Environmental Pollution*, *102*(Supplement 1), 41–48. https://doi.org/10.1016/s0269-7491(98)80013-4

Plekhov, D., VanValkenburgh, P., Rojas Vega, C., & Reátegui Díaz, A. (2024). Is terraced agriculture 'sustainable?': A view from the Middle Utcubamba Valley, Peru. *Archaeological and Anthropological Sciences*, *16*(7), 102. https://doi.org/10.1007/s12520-024-02008-2

Poeplau, C., Don, A., Vesterdal, L., Leifeld, J., Van Wesemael, B., Schumacher, J., & Gensior, A. (2011). Temporal dynamics of soil organic carbon after land-use change in the temperate zone – Carbon response functions as a model approach. *Global Change Biology*, *17*(7), 2415–2427. https://doi.org/10.1111/j.1365-2486.2011.02408.x

Poesen, J. (2018). Soil erosion in the Anthropocene: Research needs. *Earth Surface Processes and Landforms*, *43*(1), 64–84. https://doi.org/10.1002/esp.4250

Poesen, J., Nachtergaele, J., Verstraeten, G., & Valentin, C. (2003). Gully erosion and environmental change: Importance and research needs. *Gully Erosion and Global Change*, *50*(2–4), 91–133. https://doi.org/10.1016/S0341-8162(02)00143-1

Poesen, J., Verstraeten, G., Soenens, R., & Seynaeve, L. (2001). Soil losses due to harvesting of chicory roots and sugar beet: An underrated geomorphic process? *Gully Erosion and Global Change*, *43*(1), 35–47. https://doi.org/10.1016/S0341-8162(00)00125-9

Poggio, L., De Sousa, L. M., Batjes, N. H., Heuvelink, G. B., Kempen, B., Ribeiro, E., & Rossiter, D. (2021). SoilGrids 2.0: Producing soil information for the globe with quantified spatial uncertainty. *Soils*, *7*(1), 217–240. https://doi.org/10.5194/soil-7-217-2021

Poppiel, R. R., Cherubin, M. R., Novais, J. J., & Demattê, J. A. (2025). Soil health in Latin America and the Caribbean. *Communications Earth & Environment*, *6*(1), 141. https://doi.org/10.1038/s43247-025-02021-w

Pouyan, S., Pourghasemi, H. R., Bordbar, M., Rahmanian, S., & Clague, J. J. (2021). A multi-hazard map-based flooding, gully erosion, forest fires, and earthquakes in Iran. *Scientific Reports*, *11*(1), 14889. https://doi.org/10.1038/s41598-021-94266-6

Pozza, L. E., & Field, D. J. (2020). The science of Soil Security and Food Security. *Soil Security*, *1*, 100002. https://doi.org/10.1016/j.soisec.2020.100002

Prakash, J., Agrawal, S. B., & Agrawal, M. (2023). Global Trends of Acidity in Rainfall and Its Impact on Plants and Soil. *Journal of Soil Science and Plant Nutrition*, *23*(1), 398–419. https://doi.org/10.1007/s42729-022-01051-z

Prasuhn, V. (2020). Twenty years of soil erosion on-farm measurement: Annual variation, spatial distribution and the impact of conservation programmes for soil loss rates in Switzerland. *Earth Surface Processes and Landforms*, *37*(4), 1539–1554. https://doi.org/10.1002/esp.4829

Prăvălie, R. (2021). Exploring the multiple land degradation pathways across the planet. *Earth-Science Reviews*, *220*, 103689. https://doi.org/10.1016/j.earscirev.2021.103689

Prăvălie, R., Borrelli, P., Panagos, P., Ballabio, C., Lugato, E., Chappell, A., et al. (2024). A unifying modelling of multiple land degradation pathways in Europe. *Nature Communications*, *15*(1), 3862. https://doi.org/10.1038/s41467-024-48252-x

Prăvălie, R., Nita, I. A., Patriche, C., Niculiţă, M., Birsan, M. V., Roşca, B., & Bandoc, G. (2021). Global changes in soil organic carbon and implications for land degradation neutrality and climate stability. *Environmental Research*, *201*, 111580. https://doi.org/10.1016/j.envres.2021.111580

Prăvălie, R., Patriche, C., Borrelli, P., Panagos, P., Roşca, B., Dumitraşcu, M., et al. (2021). Arable lands under the pressure of multiple land degradation processes. A global perspective. *Environmental Research*, *194*, 110697. https://doi.org/10.1016/j.envres.2020.110697

Pretty, J. N., Noble, A. D., Bossio, D., Dixon, J., Hine, R. E., Penning de Vries, F. W. T., & Morison, J. I. L. (2006). Resource-conserving agriculture increases yields in developing countries. *Environmental Science and Technology*, *40*(4), 1114–1119. https://doi.org/10.1021/es051670d

Price, K., Jackson, C. R., & Parker, A. J. (2010). Variation of surficial soil hydraulic properties across land uses in the southern Blue Ridge Mountains, North Carolina, USA. *Journal of Hydrology*, *383*(3–4), 256–268. https://doi.org/10.1016/j.jhydrol.2009.12.041

Pulido-Moncada, M., Munkholm, L. J., & Schjønning, P. (2019). Wheel load, repeated wheeling, and traction effects on subsoil compaction in northern Europe. *Soil and Tillage Research*, *186*, 300–309. https://doi.org/10.1016/j.still.2018.11.005

Qadir, M., Quillérou, E., Nangia, V., Murtaza, G., Singh, M., Thomas, R. J., et al. (2014). Economics of salt-induced land degradation and restoration. *Natural Resources Forum*, *38*(4), 282–295. https://doi.org/10.1111/1477-8947.12054

Qi, S., Degen, A., Wang, W., Huang, M., Li, D., Luo, B., et al. (2024). Systemic review for the use of biochar to mitigate soil degradation. *GCB Bioenergy*, *16*(6), e13147. https://doi.org/10.1111/gcbb.13147

Qian, J., Ji, C., Yang, J., Zhao, H., Wang, Y., Fu, L., & Liu, Q. (2024). The advantage of afforestation using native tree species to enhance soil quality in degraded forest ecosystems. *Scientific Reports*, *14*(1), 20022. https://doi.org/10.1038/s41598-024-71162-3

Quinn, N. W. T. (2020). Policy innovation and governance for irrigation sustainability in the arid, saline San Joaquin River Basin. *Sustainability*, *12*(11), 4733. https://doi.org/10.3390/su12114733

Quinton, J. N., & Fiener, P. (2024). Soil erosion on arable land: An unresolved global environmental threat. Progress in Physical Geography. *Earth and Environment*, *48*(1), 1–27. https://doi.org/10.1177/03091333231216595

Quinton, J. N., Govers, G., van Oost, K., & Bardgett, R. D. (2010). The impact of agricultural soil erosion on biogeochemical cycling. *Nature Geoscience*, *3*(5), 311–314. https://doi.org/10.1038/ngeo838

Quirk, J. P., & Schofield, R. K. (1955). The effect of electrolyte concentration on soil permeability. *Journal of Soil Science*, *6*(2), 163–178. https://doi.org/10.1111/j.1365-2389.1955.tb00841.x

Rabot, E., Wiesmeier, M., Schlüter, S., & Vogel, H. J. (2018). Soil structure as an indicator of soil functions: A review. *Geoderma*, *314*, 122–137. https://doi.org/10.1016/j.geoderma.2017.11.009

Rackow, T., Pedruzo-Bagazgoitia, X., Becker, T., Milinski, S., Sandu, I., Aguridan, R., et al. (2025). Multi-year simulations at kilometre scale with the Integrated Forecasting System coupled to FESOM2.5 and NEMOv3.4. *Geoscientific Model Development*, *18*(1), 33–69. https://doi.org/10.5194/gmd-18-33-2025

Radford, B. J., Yule, D. F., McGarry, D., & Playford, C. (2007). Amelioration of soil compaction can take 5 years on a Vertisol under no till in the semiarid subtropics. *Soil and Tillage Research*, *97*(2), 249–255. https://doi.org/10.1016/j.still.2006.01.005

Raghavendra, M., Sharma, M. P., Ramesh, A., Richa, A., Billore, S. D., & Verma, R. K. (2020). Soil health indicators: Methods and applications. In A. Rakshit, S. Ghosh, S. Chakraborty, V. Philip, & A. Datta (Eds.), *Soil analysis: recent trends and applications*. Springer. https://doi.org/10.1007/978-981-15-2039-6_13

Ramani, R. V. (2012). Surface mining technology: Progress and prospects. *Procedia Engineering*, *46*, 9–21. https://doi.org/10.1016/j.proeng.2012.09.440

Rast, M., Nieke, J., Adams, J., Isola, C., & Gascon, F. (2021). Copernicus hyperspectral imaging mission for the environment (chime). In *2021 IEEE international geoscience and remote sensing symposium IGARSS* (pp. 108–111).

Ravi, S., Breshears, D. D., Huxman, T. E., & D'Odorico, P. (2010). Land degradation in drylands: Interactions among hydrologic–aeolian erosion and vegetation dynamics. *Geomorphology*, *116*(3–4), 236–245. https://doi.org/10.1016/j.geomorph.2009.11.023

Ravi, S., D'Odorico, P., Breshears, D. D., Field, J. P., Goudie, A. S., Huxman, T. E., et al. (2011). Aeolian processes and the biosphere. *Reviews of Geophysics*, *49*, RG3001. https://doi.org/10.1029/2010RG000328

Ravi, S., Law, D. J., Caplan, J. S., Barron-Gafford, G. A., Dontsova, K. M., Espeleta, J. F., et al. (2022). Biological invasions and climate change amplify each other's effects on dryland degradation. *Global Change Biology*, *28*(1), 285–295. https://doi.org/10.1111/gcb.15919

Reed, M. S., Stringer, L. C., Dougill, A. J., Perkins, J. S., Atlhopheng, J. R., Mulale, K., & Favretto, N. (2015). Reorienting land degradation towards sustainable land management: Linking sustainable livelihoods with ecosystem services in rangeland systems. *Journal of Environmental Management*, *151*, 472–485. https://doi.org/10.1016/j.jenvman.2014.11.010

Rehbein, K., van der Meer, M., Grob, U., Wegmann, F., & Keller, A. (2011). Das Nationale Bodeninformationssystem NABODAT in der Schweiz. In *Proceedings of the Jahrestagung Deutschen Bodenkundlichen Gesellschaft* (pp. 3–9).

Reichstein, M., Bahn, M., Ciais, P., Frank, D., Mahecha, M. D., Seneviratne, S. I., et al. (2013). Climate extremes and the carbon cycle. *Nature*, *500*(7462), 287–295. https://doi.org/10.1038/nature12350

Reicosky, D. (1997). Tillage-induced CO2 emission from soil. *Nutrient Cycling in Agroecosystems*, *49*(1–3), 273–285. https://doi.org/10.1023/a:1009766510274

Reinhart, K. O., & Callaway, R. M. (2006). Soil biota and invasive plants. *New Phytologist*, *170*(3), 445–457. https://doi.org/10.1111/j.1469-8137.2006.01715.x

Remund, D., Liebisch, F., Liniger, H. P., Heinimann, A., & Prasuhn, V. (2021). The origin of sediment and particulate phosphorus inputs into water bodies in the Swiss Midlands – A twenty-year field study of soil erosion. *Catena*, *203*, 105290. https://doi.org/10.1016/J.CATENA.2021.105290

Ren, C., Liu, K., Dou, P., Shao, X., Zhang, D., Wang, K., et al. (2022). Soil nutrients drive microbial changes to alter surface soil aggregate stability in typical grasslands. *Journal of Soil Science and Plant Nutrition*, *22*(4), 4943–4959. https://doi.org/10.1007/s42729-022-00972-z

Ren, S., Wang, T., Guenet, B., Liu, D., Cao, Y., Ding, J., et al. (2024). Projected soil carbon loss with warming in constrained Earth system models. *Nature Communications*, *15*(1), 102. https://doi.org/10.1038/s41467-023-44433-2

Rengasamy, P. (2006). World salinization with emphasis on Australia. *Journal of Experimental Botany*, *57*(5), 1017–1023. https://doi.org/10.1093/jxb/erj108

Renschler, C. S. (2003). Designing geo-spatial interfaces to scale process models: The GeoWEPP approach. *Hydrological Processes*, *17*(5), 1005–1017. https://doi.org/10.1002/hyp.1177

Reynolds, B., Chamberlain, P. M., Poskitt, J., Woods, C., Scott, W. A., Rowe, E. C., et al. (2013). Countryside survey: National “soil change” 1978–2007 for topsoils in Great Britain—Acidity, carbon, and total nitrogen status. *Vadose Zone Journal*, *12*(2), 1–15. https://doi.org/10.2136/vzj2012.0114

Ribeiro, E., Batjes, N. H., & Van Oostrum, A. J. M. (2018). World Soil Information Service (WoSIS)-Towards the standardization and harmonization of world soil data. *Procedures manual*, *166*.

Richard, G., Cousin, I., Sillon, J. F., Bruand, A., & Guérif, J. (2001). Effect of compaction on the porosity of a silty soil: Influence on unsaturated hydraulic properties. *European Journal of Soil Science*, *52*(1), 49–58. https://doi.org/10.1046/j.1365-2389.2001.00357.x

Richards, L. A. (1954). *Diagnosis and improvement of saline and alkali soils*. US Government Printing Office.

Rillig, M. C., Lehmann, A., Orr, J. A., & Rongstock, R. (2024). Factors of global change affecting plants act at different levels of the ecological hierarchy. *The Plant Journal*, *117*(6), 1781–1785. https://doi.org/10.1111/tpj.16509

Rillig, M. C., Li, C., Rodríguez del Río, Á., Zhu, Y.-G., & Jin, L. (2024). Elevated levels of antibiotic resistance genes as a factor of human-caused global environmental change. *Global Change Biology*, *30*(7), e17419. https://doi.org/10.1111/gcb.17419

Rillig, M. C., Ryo, M., & Lehmann, A. (2021). Classifying human influences on terrestrial ecosystems. *Global Change Biology*, *27*(11), 2273–2278. https://doi.org/10.1111/gcb.15577

Rillig, M. C., Ryo, M., Lehmann, A., Aguilar-Trigueros, C. A., Buchert, S., Wulf, A., et al. (2019). The role of multiple global change factors in driving soil functions and microbial biodiversity. *Science*, *366*(6467), 886–890. https://doi.org/10.1126/science.aay2832

Rillig, M. C., van der Heijden, M. G. A., Berdugo, M., Liu, Y., Riedo, J., Sanz-Lazaro, C., et al. (2023). Increasing the number of stressors reduces soil ecosystem services worldwide. *Nature Climate Change*, *13*(5), 478–483. https://doi.org/10.1038/s41558-023-01627-2

Rinot, O., Levy, G. J., Steinberger, Y., Svoray, T., & Eshel, G. (2019). Soil health assessment: A critical review of current methodologies and a proposed new approach. *Science of the Total Environment*, *648*, 1484–1491. https://doi.org/10.1016/j.scitotenv.2018.08.259

Ritzema, H. P. (2016). Drain for Gain: Managing salinity in irrigated lands—A review. *Agricultural Water Management*, *176*, 18–28. https://doi.org/10.1016/j.agwat.2016.05.014

Robinson, D. A., Fraser, I., Dominati, E. J., Davíðsdóttir, B., Jónsson, J. O. G., Jones, L., et al. (2014). On the value of soil resources in the context of natural capital and ecosystem service delivery. *Soil Science Society of America Journal*, *78*(3), 685–700. https://doi.org/10.2136/sssaj2014.01.0017

Robinson, D. A., Jones, S. B., Lebron, I., Reinsch, S., Domínguez, M. T., Smith, A. R., et al. (2016). Experimental evidence for drought induced alternative stable states of soil moisture. *Scientific Reports*, *6*(1), 20018. https://doi.org/10.1038/srep20018

Robinson, D. A., Nemes, A., Reinsch, S., Radbourne, A., Bentley, L., & Keith, A. M. (2022). Global meta-analysis of soil hydraulic properties on the same soils with differing land use. *Science of the Total Environment*, *852*, 158506. https://doi.org/10.1016/j.scitotenv.2022.158506

Robinson, D. A., Thomas, A., Reinsch, S., Lebron, I., Feeney, C. J., Maskell, L. C., et al. (2022). Analytical modelling of soil porosity and bulk density across the soil organic matter and land-use continuum. *Scientific Reports*, *12*(1), 7085. https://doi.org/10.1038/s41598-022-11099-7

Roces-Díaz, J. V., Santín, C., Martínez-Vilalta, J., & Doerr, S. H. (2022). *A global synthesis of fire effects on ecosystem services of forests and woodlands*. Frontiers in EcologRogger.

Rodríguez del Río, Á., Scheu, S., & Rillig, M. C. (2025). Soil microbial responses to multiple global change factors as assessed by metagenomics. *Nature Communications*, *16*(1), 5058. https://doi.org/10.1038/s41467-025-60390-4

Rolston, D. E., & Moldrup, P. (2002). Gas diffusivity. In J. H. Dane & G. C. Topp (Eds.), *Methods of soil analysis*, *Part 4. SSSA Book Series* (Vol. 5, pp. 1113–1139). SSSA.

Rosenfeld, D., Rudich, Y., & Lahav, R. (2001). Desert dust suppressing precipitation: A possible desertification feedback loop. *Proceedings of the National Academy of Sciences*, *98*(11), 5975–5980. https://doi.org/10.1073/pnas.101122798

Rothwell, J. J., Futter, M. N., & Dise, N. B. (2008). A classification and regression tree model of controls on dissolved inorganic nitrogen leaching from European forests. *Environment and Pollution*, *156*(2), 544–552. https://doi.org/10.1016/j.envpol.2008.01.007

Rücknagel, J., Hofmann, B., Deumelandt, P., Reinicke, F., Bauhardt, J., Hülsbergen, K.-J., & Christen, O. (2015). Indicator based assessment of the soil compaction risk at arable sites using the model REPRO. *Ecological Indicators*, *52*, 341–352. https://doi.org/10.1016/j.ecolind.2014.12.022

Rüegg, K. (2000). *Development, test and use of an air-pycnometer to measure air-filled porosity on undisturbed soil samples* (M.Sc. thesis). Aalborg University.

Ruehlmann, J., & Körschens, M. (2009). Calculating the effect of soil organic matter concentration on soil bulk density. *Soil Science Society of America Journal*, *73*(3), 876–885. https://doi.org/10.2136/sssaj2007.0149

Rui, L. I., Wenyou, H. U., Zhongjun, J. I. A., Hanqiang, L. I. U., Shunhua, Y. A. N. G., Yuguo, Z. H. A. O., et al. (2025). Soil degradation: A global threat to sustainable use of black soils. *Pedosphere*, *35*(1), 264–279. https://doi.org/10.1016/j.pedsph.2024.06.011

Ruiz, F., Cherubin, M. R., & Ferreira, T. O. (2020). Soil quality assessment of constructed technosols: Towards the validation of a promising strategy for land reclamation, waste management and the recovery of soil functions. *Journal of Environmental Management*, *276*, 111344. https://doi.org/10.1016/j.jenvman.2020.111344

Ruiz, F., Perlatti, F., Oliveira, D. P., & Ferreira, T. O. (2020). Revealing tropical Technosols as an alternative for mine reclamation and waste management. *Minerals*, *10*(2), 110. https://doi.org/10.3390/min10020110

Ruiz, S., Hallett, P. D., & Or, D. (2023). Bioturbation – Physical processes. In M. J. Goss & M. Oliver (Eds.), *Encyclopedia of soils in the environment* (2nd ed.). Academic Press.

Rusanov, V. A. (1994). USSR standards for agricultural mobile machinery: Permissible influences on soils and methods to estimate contact pressure and stress at a depth of 0.5 m. *Soil and Tillage Research*, *29*(2–3), 249–252. https://doi.org/10.1016/0167-1987(94)90063-9

Ruysschaert, G., Poesen, J., Verstraeten, G., & Govers, G. (2004). Soil loss due to crop harvesting: Significance and determining factors. Progress in Physical Geography. *Earth and Environment*, *28*(4), 467–501. https://doi.org/10.1191/0309133304pp421oa

Ruysschaert, G., Poesen, J., Verstraeten, G., & Govers, G. (2007). Soil loss due to harvesting of various crop types in contrasting agro-ecological environments. *Agriculture, Ecosystems & Environment*, *120*(2–4), 153–165. https://doi.org/10.1016/j.agee.2006.08.012

Ruysschaert, G., Poesen, J., Wauters, A., Govers, G., & Verstraeten, G. (2007). Factors controlling soil loss during sugar beet harvesting at the field plot scale in Belgium. *European Journal of Soil Science*, *58*(6), 1400–1409. https://doi.org/10.1111/j.1365-2389.2007.00945.x

Ryken, N., Vanden Nest, T., Al-Barri, B., Blake, W., Taylor, A., Bodé, S., et al. (2018). Soil erosion rates under different tillage practices in central Belgium: New perspectives from a combined approach of rainfall simulations and 7 Be measurements. *Soil and Tillage Research*, *179*, 29–37. https://doi.org/10.1016/j.still.2018.01.010

Sadeghi, M., Shokri, N., & Jones, S. B. (2012). A novel analytical solution to steady-state evaporation from porous media. *Water Resources Research*, *48*(9), W09516. https://doi.org/10.1029/2012WR012060

Saggau, P., Busche, F., Brunotte, J., Duttmann, R., & Kuhwald, M. (2024). Soil loss due to crop harvesting in highly mechanized agriculture: A case study of sugar beet harvest in northern Germany. *Soil and Tillage Research*, *242*, 106144. https://doi.org/10.1016/j.still.2024.106144

Saggau, P., Kuhwald, M., & Duttmann, R. (2023). Effects of contour farming and tillage practices on soil erosion processes in a hummocky watershed. A model-based case study highlighting the role of tramline tracks. *Catena*, *228*, 107126. https://doi.org/10.1016/j.catena.2023.107126

Saggau, P., Kuhwald, M., Hamer, W. B., & Duttmann, R. (2022). Are compacted tramlines underestimated features in soil erosion modeling? A catchment-scale analysis using a process-based soil erosion model. *Land Degradation & Development*, *33*(3), 452–469. https://doi.org/10.1002/ldr.4161

Saha, S., Roy, J., Arabameri, A., Blaschke, T., & Tien Bui, D. (2020). Machine learning-based gully erosion susceptibility mapping: A case study of Eastern India. *Sensors*, *20*(5), 1313. https://doi.org/10.3390/s20051313

Sahbeni, G., Ngabire, M., Musyimi, P. K., & Székely, B. (2023). Challenges and opportunities in remote sensing for soil salinization mapping and monitoring: A review. *Remote Sensing*, *15*(10), 2540. https://doi.org/10.3390/rs15102540

Sanderman, J., & Berhe, A. (2017). The soil carbon erosion paradox. *Nature Climate Change*, *7*(5), 317–319. https://doi.org/10.1038/nclimate3281

Sanderman, J., Hengl, T., & Fiske, G. J. (2017). Soil carbon debt of 12,000 years of human land use. *Proceedings of the National Academy of Sciences*, *114*(36), 9575–9580. https://doi.org/10.1073/pnas.1706103114

Sanfo, S., Fonta, W. M., Diasso, U. J., Nikiéma, M. P., Lamers, J. P. A., & Tondoh, J. E. (2017). Climate- and environment-induced intervillage migration in southwestern Burkina Faso, West Africa. *Weather, Climate, and Society*, *9*(4), 823–837. https://doi.org/10.1175/WCAS-D-16-0065.1

Santini, T. C., & Fey, M. V. (2018). From tailings to soil: Long-term effects of amendments on progress and trajectory of soil formation and in situ remediation in bauxite residue. *Journal of Soils and Sediments*, *18*(5), 1935–1949. https://doi.org/10.1007/s11368-017-1867-1

Sartori, M., Philippidis, G., Ferrari, E., Borrelli, P., Lugato, E., Montanarella, L., & Panagos, P. (2019). A linkage between the biophysical and the economic: Assessing the global market impacts of soil erosion. *Land Use Policy*, *86*, 299–312. https://doi.org/10.1016/j.landusepol.2019.05.014

Schad, P. (2018). Technosols in the World Reference Base for Soil Resources – History and definitions. *Soil Science & Plant Nutrition*, *64*(2), 138–144. https://doi.org/10.1080/00380768.2018.1432973

Schepanski, K. (2018). Transport of mineral dust and its impact on climate. *Geosciences*, *8*(5), 151. https://doi.org/10.3390/geosciences8050151

Schillaci, C., Perego, A., Valkama, E., Märker, M., Saia, S., Veronesi, F., et al. (2021). New pedotransfer approaches to predict soil bulk density using WoSIS soil data and environmental covariates in Mediterranean agro-ecosystems. *Science of the Total Environment*, *780*, 146609. https://doi.org/10.1016/j.scitotenv.2021.146609

Schjønning, P., de Jonge, L. W., Moldrup, P., Christensen, B. T., & Olesen, J. E. (2010). Searching the critical soil organic carbon threshold for satisfactory tilth conditions – Test of the Dexter clay:carbon hypothesis. In *Proceedings of the 1st international conference and exploratory workshop on soil architecture and physico-chemical functions "Cesar"* (pp. 341–346).

Schjønning, P., Eden, M., Moldrup, P., & de Jonge, L. W. (2013). Two-chamber, two-gas and one-chamber, one-gas methods for measuring the soil-gas diffusion coefficient: Validation and inter-calibration. *Soil Science Society of America Journal*. https://doi.org/10.2136/sssaj2012.0379

Schjønning, P., & Lamandé, M. (2018). Models for prediction of soil precompression stress from readily available soil properties. *Geoderma*, *320*, 115–125. https://doi.org/10.1016/j.geoderma.2018.01.028

Schjønning, P., van den Akker, J. J. H., Keller, T., Greve, M. H., Lamandé, M., Simojoki, A., et al. (2015). Chapter Five - Driver-Pressure-State-Impact-Response (DPSIR) Analysis and risk assessment for soil compaction—A European perspective. In D. L. Sparks (Ed.), *Advances in agronomy* (Vol. 133, pp. 183–237). Academic Press. https://doi.org/10.1016/bs.agron.2015.06.001

Schneider, F., & Don, A. (2019). Root-restricting layers in German agricultural soils. Part I: Extent and cause. *Plant and Soil*, *442*(1–2), 433–451. https://doi.org/10.1007/s11104-019-04185-9

Schneider, F., Don, A., Hennings, I., Schmittmann, O., & Seidel, S. J. (2017). The effect of deep tillage on crop yield – What do we really know? *Soil and Tillage Research*, *174*, 193–204. https://doi.org/10.1016/j.still.2017.07.005

Schuur, E. A. G., McGuire, A. D., Schädel, C., Grosse, G., Harden, J. W., Hayes, D. J., et al. (2015). Climate change and the permafrost carbon feedback. *Nature*, *520*(7546), 171–179. https://doi.org/10.1038/nature14338

Seaton, F. M., Robinson, D. A., Monteith, D., Lebron, I., Bürkner, P., Tomlinson, S., et al. (2023). Fifty years of reduction in sulphur deposition drives recovery in soil pH and plant communities. *Journal of Ecology*, *111*(2), 464–478. https://doi.org/10.1111/1365-2745.14039

Sentis, I. (1996). Soil salinization and land desertification. *Soil degradation and desertification in Mediterranean environments*, 105–129.

Sepehrnia, N., Abbasi Teshnizi, F., Hallett, P., Coyne, M., Shokri, N., & Peth, S. (2024). Modeling Bacterial Transport and Fate: Insight into the Cascading Consequences of Soil Water Repellency and Contrasting Hydraulic Conditions. *Science of the Total Environment*, *954*, 176196. https://doi.org/10.1016/j.scitotenv.2024.176196

Shah, A. N., Tanveer, M., Shahzad, B., Yang, G., Fahad, S., Ali, S., et al. (2017). Soil compaction effects on soil health and cropproductivity: An overview. *Environmental Science and Pollution Research*, *24*(11), 10056–10067. https://doi.org/10.1007/s11356-017-8421-y

Shahid, S. A., Zaman, M., & Heng, L. (2018). Soil Salinity: Historical Perspectives and a World Overview of the Problem. In M. Zaman, S. A. Shahid, & L. Heng (Eds.), *Guideline for Salinity Assessment, Mitigation and Adaptation Using Nuclear and Related Techniques* (pp. 43–53). Springer International Publishing. https://doi.org/10.1007/978-3-319-96190-3_2

Shahmohamadloo, R. S., Febria, C. M., Fraser, E. D. G., & Sibley, P. K. (2022). The sustainable agriculture imperative: A perspective on the need for an agrosystem approach to meet the United Nations Sustainable Development Goals by 2030. *Integrated Environmental Assessment and Management*, *18*(5), 1199–1205. https://doi.org/10.1002/ieam.4558

Shao, Y., Wyrwoll, K.-H., Chappell, A., Huang, J., Lin, Z., McTainsh, G. H., et al. (2011). Dust cycle: An emerging core theme in Earth system science. *Aeolian Research*, *2*(4), 181–204. https://doi.org/10.1016/j.aeolia.2011.02.001

Sharma, P., Sharma, P., & Thakur, N. (2024). Sustainable farming practices and soil health: A pathway to achieving SDGs and future prospects. *Discover Sustainability*, *5*(1), 250. https://doi.org/10.1007/s43621-024-00447-4

Sharma, R. C., & Mondal, A. K. (2006). Mapping of soil salinity and sodicity using digital image analysis and GIS in irrigated lands of the Indo-Gangetic plain. *Agropedology*, *16*(2), 71–76.

Sheppard, L. J., Leith, I. D., Crossley, A., van Dijk, N., Cape, J. N., Fowler, D., & Sutton, M. A. (2009). Long-term cumulative exposure exacerbates the effects of atmospheric ammonia on an ombrotrophic Bog: Implications for critical levels. In M. A. Sutton, S. Reis, & S. M. H. Baker (Eds.), *Atmospheric Ammonia: Detecting emission changes and environmental impacts* (pp. 49–58). Springer.

Shi, H., Luo, G., Sutanudjaja, E. H., Hellwich, O., Chen, X., Ding, J., et al. (2023). Recent impacts of water management on dryland's salinization and degradation neutralization. *Science Bulletin*, *68*(24), 3240–3251. https://doi.org/10.1016/j.scib.2023.11.012

Shi, S., Peng, C., Wang, M., Zhu, Q., Yang, G., Yang, Y., et al. (2016). A global meta-analysis of changes in soil carbon, nitrogen, phosphorus and sulfur, and stoichiometric shifts after forestation. *Plant and Soil*, *407*(1–2), 323–340. https://doi.org/10.1007/s11104-016-2889-y

Shi, Z. H., Fang, N. F., Wu, F. Z., Wang, L., Yue, B. J., & Wu, G. L. (2012). Soil erosion processes and sediment sorting associated with transport mechanisms on steep slopes. *Journal of Hydrology*, *454*, 123–130. https://doi.org/10.1016/j.jhydrol.2012.06.004

Shojaeezadeh, S. A., Al-Wardy, M., Nikoo, M. R., Mooselu, M. G., Alizadeh, M. R., Adamowski, J. F., et al. (2024). Soil erosion in the United States: Present and future (2020–2050). *Catena*, *242*, 108074. https://doi.org/10.1016/j.catena.2024.108074

Shojaei, M. J., Or, D., & Shokri, N. (2022). Localized delivery of liquid fertilizer in coarse textured soils using foam as carrier. *Transport in Porous Media*, *143*(3), 787–795. https://doi.org/10.1007/s11242-022-01820-5

Shokri, N. (2014). Pore-scale dynamics of salt transport and distribution in drying porous media. *Physics of Fluids*, *26*(1), 012106. https://doi.org/10.1063/1.4861755

Shokri, N. (2019). Comment on "Analytical estimation show low depth-independent water loss due to vapor flux from deep aquifers by John S. Selker [2017]". *Water Resources Research*, *55*(2), 1730–1733. https://doi.org/10.1029/2018WR023347

Shokri, N., Aminzadeh, M., Flury, M., Jin, Y., Matin, M., Panagos, P., et al. (2025). Sustainability Nexus AID. *Soil Health, Sustainability Nexus Forum*, *33*(1), 3. https://doi.org/10.1007/s00550-025-00560-6

Shokri, N., Hassani, A. H., & Sahimi, M. (2024). Multi-scale soil salinization dynamics from global to pore scale: A Review. *Reviews of Geophysics*, *62*(4), e2023RG000804. https://doi.org/10.1029/2023RG000804

Shokri, N., Lehmann, P., & Or, D. (2009). Characteristics of evaporation from partially-wettable porous media. *Water Resources Research*, *45*(2), W02415. https://doi.org/10.1029/2008WR007185

Shokri, N., & Sahimi, M. (2012). The structure of drying fronts in three-dimensional porous media. *Physical Review E - Statistical Physics, Plasmas, Fluids, and Related Interdisciplinary Topics*, *85*(6), 066312. https://doi.org/10.1103/physreve.85.066312

Shokri, N., Sahimi, M., & Or, D. (2012). Morphology, propagation, dynamics and scaling characteristics of drying fronts in porous media. *Geophysical Research Letters*, *39*(9), L09401. https://doi.org/10.1029/2012GL051506

Shokri, N., Zhou, P., & Keshmiri, A. (2015). Patterns of Desiccation Cracks in Saline Bentonite Layers. *Transport in Porous Media*, *110*(2), 333–344. https://doi.org/10.1007/s11242-015-0521-x

Shokri-Kuehni, S. M. S., Norouzirad, M., Webb, C., & Shokri, N. (2017). Impact of type of salt and ambient conditions on saline water evaporation from porous media. *Advances in Water Resources*, *105*, 154–161. https://doi.org/10.1016/j.advwatres.2017.05.004

Shokri-Kuehni, S. M. S., Raaijmakers, B., Kurz, T., Or, D., Helmig, R., & Shokri, N. (2020). Water Table Depth and Soil Salinization: From Pore-Scale Processes to Field-Scale Responses. *Water Resources Research*, *56*(2), e2019WR026707. https://doi.org/10.1029/2019WR026707

Shokri-Kuehni, S. M. S., Sahimi, M., & Shokri, N. (2022). A personal perspective on prediction of saline water evaporation from porous media. *Drying Technology*, *40*(4), 691–696. https://doi.org/10.1080/07373937.2021.1999256

Shokri-Kuehni, S. M. S., Vetter, T., Webb, C., & Shokri, N. (2017). New insights into saline water evaporation from porous media: Complex interaction between evaporation rates, precipitation and surface temperature. *Geophysical Research Letters*, *44*(11), 5504–5510. https://doi.org/10.1002/2017GL073337

Shuman, J. K., Balch, J. K., Barnes, R. T., Higuera, P. E., Roos, C. I., Schwilk, D. W., et al. (2022). Reimagine fire science for the anthropocene. *PNAS Nexus*, *2022*(1), 1–14. https://doi.org/10.1093/pnasnexus/pgac115

Simberloff, D., Martin, J.-L., Genovesi, P., Maris, V., Wardle, D. A., Aronson, J., et al. (2013). Impacts of biological invasions: What's what and the way forward. *Trends in Ecology & Evolution*, *28*(1), 58–66. https://doi.org/10.1016/j.tree.2012.07.013

Sinclair, A., & Fryxell, J. (1985). The Sahel of Africa: Ecology of a disaster. *Canadian Journal of Zoology*, *63*, 987–994.

Singh, A. (2015). Soil salinization and waterlogging: A threat to environment and agricultural sustainability. *Ecological Indicators*, *57*, 128–130. https://doi.org/10.1016/j.ecolind.2015.04.027

Singh, A., & Agrawal, M. (2008). Acid rain and its ecological consequences. *Journal of Environmental Biology*, *29*(1), 15–24.

Singh, K. (2016). Microbial and enzyme activities of saline and sodic soils. *Land Degradation & Development*, *27*(3), 706–718. https://doi.org/10.1002/ldr.2385

Sishodia, R. P., Ray, R. L., & Singh, S. K. (2020). Applications of remote sensing in precision agriculture: A review. *Remote Sensing*, *12*(19), 3136. https://doi.org/10.3390/rs12193136

Sitters, J., & Andriuzzi, W. S. (2019). Impacts of browsing and grazing ungulates on soil biota and nutrient dynamics. In I. Gordon & H. Prins (Eds.), *The ecology of browsing and grazing II*. *Ecological studies* (Vol. 239, pp. 215–236). Springer. https://doi.org/10.1007/978-3-030-25865-8_9

Slessarev, E. W., Lin, Y., Bingham, N. L., Johnson, J. E., Dai, Y., Schimel, J. P., & andChadwick, O. A. (2016). Water balance creates a threshold in soil pH at the global scale. *Nature*, *540*(7634), 567–569. https://doi.org/10.1038/nature20139

Smith, P., Cotrufo, M. F., Rumpel, C., Paustian, K., Kuikman, P. J., Elliott, J. A., et al. (2015). Biogeochemical cycles and biodiversity as key drivers of ecosystem services provided by soils. *Soils*, *1*(2), 665–685. https://doi.org/10.5194/soil-1-665-2015

Smith, P., House, J. I., Bustamante, M., Sobocká, J., Harper, R., Pan, G., et al. (2016). Global change pressures on soils from land use and management. *Global Change Biology*, *22*(3), 1008–1028. https://doi.org/10.1111/gcb.13068

Smith, P., Poch, R. M., Lobb, D. A., Bhattacharyya, R., Alloush, G., Eudoxie, G. D., et al. (2024). Status of the World's soils. *Annual Review of Environment and Resources*, *49*(1), 73–104. https://doi.org/10.1146/annurev-environ-030323-075629

Sobhi Gollo, V., González, E., Elbracht, J., Fröhle, P., & Shokri, N. (2024). Soil salinization due to saltwater intrusion in coastal regions: The role of soil characteristics and heterogeneity. *InterPore J*, *1*(1), ipj260424-6. https://doi.org/10.69631/ipj.v1i1nr15

Socci, P., Errico, A., Castelli, G., Penna, D., & Preti, F. (2019). Terracing: From agriculture to multiple ecosystem services. *Oxford Research Encyclopedia of Environmental Science*. https://doi.org/10.1093/acrefore/9780199389414.013.206

Solomon, N., Nirea, K., Ghebretinsae, F., & Yihdego, A. G. (2024). Assessing carbon sequestration potential and socio-economic benefits of ficus thonningii in the Tigray Region, Northern Ethiopia. In *Forests and climate change* (pp. 731–750). Springer Nature Singapore.

Sonderegger, T., & Pfister, S. (2021). Global assessment of agricultural productivity losses from soil compaction and water erosion. *Environmental Science & Technology*, *55*(18), 12162–12171. https://doi.org/10.1021/acs.est.1c03774

Soussi, A., Zero, E., Sacile, R., Trinchero, D., & Fossa, M. (2024). Smart sensors and smart data for precision agriculture: A review. *Sensors*, *24*(8), 2647. https://doi.org/10.3390/s24082647

Squires, V. R., & Glenn, E. P. (2011). Salination, desertification and soil erosion. *The role of food, agriculture, forestry and fisheries in human nutrition*, *3*, 102–123.

Stehfest, E., van Zeist, W.-J., Valin, H., Havlik, P., Popp, A., Kyle, P., et al. (2019). Key determinants of global land-use projections. *Nature Communications*, *10*(1), 2166. https://doi.org/10.1038/s41467-019-09945-w

Steinbrenner, E. C. (1959). A portable air permeameter for forest soils. *Soil Science Society of America Journal*, *23*(6), 478–481. https://doi.org/10.2136/sssaj1959.03615995002300060033x

Steinhoff-Knopp, B., & Burkhard, B. (2018). Soil erosion by water in Northern Germany: Long-term monitoring results from Lower Saxony, *165*, 299–309. https://doi.org/10.1016/j.catena.2018.02.017

Stettler, M., Keller, T., Weisskopf, P., Lamandé, M., Lassen, P., & Schjønning, P. (2014). Terranimo® – A web-based tool for evaluating soil compaction. *Landtechnik*, *69*, 132–137.

Stiles, W. A. V., Rowe, E. C., & Dennis, P. (2017). Long-term nitrogen and phosphorus enrichment alters vegetation species composition and reduces carbon storage in upland soil. *Science of the Total Environment*, *593–594*, 688–694. https://doi.org/10.1016/j.scitotenv.2017.03.136

Stinson, K. A., Campbell, S. A., Powell, J. R., Wolfe, B. E., Callaway, R. M., Thelen, G. C., et al. (2006). Invasive Plant Suppresses the Growth of Native Tree Seedlings by Disrupting Belowground Mutualisms. *PLoS Biology*, *4*(5), e140. https://doi.org/10.1371/journal.pbio.0040140

Stockmann, U., Padarian, J., McBratney, A., Minasny, B., de Brogniez, D., Montanarella, L., et al. (2015). Global soil organic carbon assessment. *Global Food Security*, *6*, 9–16. https://doi.org/10.1016/j.gfs.2015.07.001

Straffelini, E., Cucchiaro, S., & Tarolli, P. (2021). Mapping potential surface ponding in agriculture using UAV-SfM. *Earth Surface Processes and Landforms*, *46*(10), 1926–1940. https://doi.org/10.1002/esp.5135

Stumm, W., & Morgan, J. J. (1996). *Aquatic chemistry, chemical equilibria and rates in natural waters*. John Wiley and Sons.

Sun, D., Yang, H., Guan, D., Yang, M., Wu, J., Yuan, F., et al. (2018). The effects of land use change on soil infiltration capacity in China: A meta-analysis. *Science of the Total Environment*, *626*, 1394–1401. https://doi.org/10.1016/j.scitotenv.2018.01.104

Sun, F., Xiao, B., Kidron, G. J., & Heitman, J. L. (2022). Insights about biocrust effects on soil gas transport and aeration in drylands: Permeability, diffusivity, and their connection to hydraulic conductivity. *Geoderma*, *427*, 116137. https://doi.org/10.1016/j.geoderma.2022.116137

Sun, L., Liu, Y. F., Wang, X., Liu, Y., & Wu, G. L. (2022). Soil nutrient loss by gully erosion on sloping alpine steppe in the northern Qinghai-Tibetan Plateau. *Catena*, *208*, 105763. https://doi.org/10.1016/j.catena.2021.105763

Sun, R., He, H., Jing, Y., Leng, S., Yang, G., Lü, Y., et al. (2024). Global wind erosion reduction driven by changing climate and land use. *Earth's Future*, *12*(10), e2024EF004930. https://doi.org/10.1029/2024EF004930

Sun, W., Niu, X., Wang, Y., Yin, X., Teng, H., Gao, P., & Liu, A. (2023). Effects of forest age on soil erosion and nutrient loss in Dianchi watershed, China. *Environmental Monitoring and Assessment*, *195*(2), 340. https://doi.org/10.1007/s10661-023-10920-8

Suz, L. M., Bidartondo, M. I., van der Linde, S., & Kuyper, T. W. (2021). Ectomycorrhizas and tipping points in forest ecosystems. *New Phytologist*, *231*(5), 1700–1707. https://doi.org/10.1111/nph.17547

Swap, R., Garstang, M., Greco, S., Talbot, R., & Kallberg, P. (1992). Saharan dust in the Amazon basin. *Tellus B: Chemical and Physical Meteorology*, *44*(2), 133–149. https://doi.org/10.1034/j.1600-0889.1992.t01-1-00005.x

Szabolcs, I. (1989). *Salt-affected soils* (p. 274). CRC Press.

Tahat, M. M., Alananbeh, K. M., Othman, Y. A., & Leskovar, D. I. (2020). Soil health and sustainable agriculture. *Sustainability*, *12*(12), 4859. https://doi.org/10.3390/su12124859

Tahmasebi, P., Kamrava, S., Bai, T., & Sahimi, M. (2020). Machine learning in geo- and environmental sciences: From small to large scale. *Advances in Water Resources*, *142*, 10361. https://doi.org/10.1016/j.advwatres.2020.103619

Talhelm, A. F., Burton, A. J., Pregitzer, K. S., & Campione, M. A. (2013). Chronic nitrogen deposition reduces the abundance of dominant forest understory and groundcover species. *Forest Ecology and Management*, *293*, 39–48. https://doi.org/10.1016/j.foreco.2012.12.020
Talukder, R., Plaza-Bonilla, D., Cantero-Martinez, C., Wendroth, O., & Castel, J. L. (2022). Soil gas diffusivity and pore continuity dynamics under different tillage and crop sequences in an irrigated Mediterranean area. *Soil and Tillage Research*, *221*, 105409. https://doi.org/10.1016/j.still.2022.105409
Tang, K., & Cai, Q. (1991). Relationship between soil erosional process and eco-environmental evolution on the Loess Plateau. In *Collected papers on environmental evolution and law on water and silt operation in huanghe drainage basin* (pp. 15–35). Geologic Press.
Tang, L., & Werner, T. T. (2023). Global mining footprint mapped from high-resolution satellite imagery. *Communications Earth & Environment*, *4*(1), 134. https://doi.org/10.1038/s43247-023-00805-6
Tanksali, A., & Soraganvi, V. S. (2021). Assessment of impacts of land use/land cover changes upstream of a dam in a semi-arid watershed using QSWAT. *Modeling Earth Systems and Environment*, *7*(4), 2391–2406. https://doi.org/10.1007/s40808-020-00978-5
Tefera, A. S., Siyum, Z. G., Berhe, D. H., & Gebru, B. M. (2024). Impact of climate variability and environmental policies on vegetation dynamics in the semi-arid Tigray. *Discover Environment*, *2*(1), 6. Article 6. https://doi.org/10.1007/s44274-024-00031-7
Terry, T. J., Hardegree, S. P., & Adler, P. B. (2024). Modeling cheatgrass distribution, abundance, and response to climate change as a function of soil microclimate. *Ecological Applications*, *34*(8), e3028. https://doi.org/10.1002/eap.3028
Teshome, A., de Graaff, J., Ritsema, C., & Kassie, M. (2016). Farmers' perceptions about the influence of land quality, land fragmentation and tenure systems on sustainable land management in the North Western Ethiopian highlands. *Land Degradation & Development*, *27*(4), 884–898. https://doi.org/10.1002/ldr.2298
Tetteh, R. N. (2015). Chemical soil degradation as a result of contamination: A review. *Journal of Soil Science and Environmental Management*, *6*(11), 301–308. https://doi.org/10.5897/jssem15.0499
Thackeray, C. W., Hall, A., Norris, J., & Chen, D. (2022). Constraining the increased frequency of global precipitation extremes under warming. *Nature Climate Change*, *12*(5), 441–448. https://doi.org/10.1038/s41558-022-01329-1
Thapa, B., & Dura, R. (2024). A review on tillage system and no-till agriculture and its impact on soil health. *Archives of Agriculture and Environmental Science*, *9*(3), 612–617. https://doi.org/10.26832/24566632.2024.0903028
Thiede, B., Gray, C., & Mueller, V. (2016). Climate variability and inter-provincial migration in South America, 1970–2011. *Global Environmental Change*, *41*, 228–240. https://doi.org/10.1016/j.gloenvcha.2016.10.005
Thiessen, L. D., & Woodward, J. E. (2012). Diseases of Peanut Caused by Soilborne Pathogens in the Southwestern United States. *International Scholarly Research Notices*, *2012*, 517905. https://doi.org/10.5402/2012/517905
Thimonier, A., Graf Pannatier, E., Schmitt, M., Waldner, P., Walthert, L., Schleppi, P., et al. (2010). Does exceeding the critical loads for nitrogen alter nitrate leaching, the nutrient status of trees and their crown condition at Swiss Long-term Forest Ecosystem Research (LWF) sites? *European Journal of Forest Research*, *129*(3), 443–461. https://doi.org/10.1007/s10342-009-0328-9
Thomas, A., Seaton, F., Dhiedt, E., Cosby, B. J., Feeney, C., Lebron, I., et al. (2024). Topsoil porosity prediction across habitats at large scales using environmental variables. *Science of the Total Environment*, *922*, 171158. https://doi.org/10.1016/j.scitotenv.2024.171158
Thomas, J. C., Mueller, E. V., Gallagher, M. R., Clark, K. L., Skowronski, N., Simeoni, A., & Hadden, R. M. (2021). Coupled assessment of fire behavior and firebrand dynamics. *Frontiers of Mechanical Engineering*, *7*, Scopus. https://doi.org/10.3389/fmech.2021.650580
Tibbett, M. (2024). Post-mining ecosystem reconstruction. *Current Biology*, *34*(9), R387–R393. https://doi.org/10.1016/j.cub.2024.03.065
Tikka, T., Makynen, J., & Shimoni, M. (2023). Hyperfield-Hyperspectral small satellites for improving life on Earth. In *2023 IEEE Aerospace Conference* (pp. 1–8). IEEE.
Todd-Brown, K. E., Abramoff, R. Z., Beem-Miller, J., Blair, H. K., Earl, S., Frederick, K. J., et al. (2022). Reviews and syntheses: The promise of big diverse soil data, moving current practices towards future potential. *Biogeosciences*, *19*(14), 3505–3522. https://doi.org/10.5194/bg-19-3505-2022
Togbévi, Q. F., Van Der Ploeg, M., Tohoun, K. A., Agodzo, S. K., & Preko, K. (2022). Assessing the Effects of Anthropogenic Land Use on Soil Infiltration Rate in a Tropical West African Watershed (Ouriyori, Benin). *Applied and Environmental Soil Science*, *2022*(1), 8565571.
Tolon-Becerra, A., Tourn, M., Botta, G. F., & Lastra-Bravo, X. (2011). Effects of different tillage regimes on soil compaction, maize (*Zea mays* L.) seedling emergence and yields in the eastern Argentinean Pampas region. *Soil and Tillage Research*, *117*, 184–190. https://doi.org/10.1016/j.still.2011.10.003
Tornese, I., Matera, A., Rashvand, M., & Genovese, F. (2024). Use of probes and sensors in agriculture—current trends and future prospects on intelligent monitoring of soil moisture and nutrients. *AgriEngineering*, *6*(4), 4154–4181. https://doi.org/10.3390/agriengineering6040234
Torres, L. C., Nemes, A., ten Damme, L., & Keller, T. (2024). Current limitations and future research needs for predicting soil precompression stress: A synthesis of available data. *Soil and Tillage Research*, *244*, 106225.
Torres, N., Herrera, I., Fajardo, L., & Bustamante, R. O. (2021). Meta-analysis of the impact of plant invasions on soil microbial communities. *BMC Ecology and Evolution*, *21*(1), 172. https://doi.org/10.1186/s12862-021-01899-2
Tóth, G., Hermann, T., Da Silva, M. R., & Montanarella, L. (2016). Heavy metals in agricultural soils of the European Union with implications for food safety. *Environment International*, *88*, 299–309. https://doi.org/10.1016/j.envint.2015.12.017
Tóth, G., Hermann, T., Szatmári, G., & Pásztor, L. (2016). Maps of heavy metals in the soils of the European Union and proposed priority areas for detailed assessment. *Science of the Total Environment*, *565*, 1054–1062. https://doi.org/10.1016/j.scitotenv.2016.05.115
Treseder, K. K. (2008). Nitrogen additions and microbial biomass: A meta-analysis of ecosystem studies. *Ecology Letters*, *11*(10), 1111. https://doi.org/10.1111/j.1461-0248.2008.01230.x
Trigo, A., Marta-Costa, A., & Fragoso, R. (2021). Principles of sustainable agriculture: Defining standardized reference points. *Sustainability*, *13*(8), 4086. https://doi.org/10.3390/su13084086
Tripathi, B. P., Timsina, J., Vista, S. P., Gaihre, Y. K., & Sapkota, B. R. (2022). Improving soil health and soil security for food and nutrition security in Nepal. In J. Timsina, T. N. Maraseni, D. Gauchan, J. Adhikari, & H. Ojha (Eds.), *Agriculture, natural resources and food security: Lessons from Nepal* (pp. 121–143). Springer International Publishing. https://doi.org/10.1007/978-3-031-09555-9_8
Tsunekawa, A., Liu, G., Yamanaka, N., & Du, S. (Eds.). (2014). Restoration and development of the degraded Loess Plateau, China, *Ecological Research Monographs*. SpringerLink.
Tullberg, J. N., Yule, D. F., & McGarry, D. (2007). Controlled traffic farming – From research to adoption in Australia. *Soil and Tillage Research*, *97*(2), 272–281. https://doi.org/10.1016/j.still.2007.09.007
Uekoetter, F. (2008). The magic of one: Reflections on the pathologies of monoculture. In *Yale University Agrarian Studies Colloquium*. Yale University.
UNCCD. (2022). United Nations Convention to Combat Desertification. In *The Global Land Outlook* (2nd ed.). UNCCD.
UNCCD. (2025). United Nations Convention to Combat Desertification. UNCCD Data Dashboard, Sustainable Development Goal (SDG) Indicator 15.3.1. Retrieved from https://data.unccd.int/land-degradation

UNEP (U.N. Environment Programme). (2022). *Global peatlands assessment, the state of the world's peatlands: Evidence for action toward the conservation, restoration, and sustainable management of peatlands*. Report of the Global Peatland Initiative. UNEP.

UNEP, WMO, UNCCD. (2016). *Global assessment of sand and dust storms*. United Nations Environment Programme.

U.S. Salinity Laboratory Staff. (1954). Diagnosis and improvement of saline and alkali soils. In *USDA Agric. Handb. 60*. U.S. Gov. Print. Office.

Uteau, D., Pagenkemper, S. K., Perth, S., & Horn, R. (2013). Root and time dependent soil structure formation and its influence on gas transport in the subsoil. *Soil and Tillage Research*, *132*, 69–76. https://doi.org/10.1016/j.still.2013.05.001

Valla, M., Kozak, J., & Ondracek, V. (2000). Vulnerability of aggregates separated from selected anthrosols developed on reclaimed dumpsites. *Rostlinna Vyroba*, *46*(12), 563–568.

Vallejo-Gómez, D., Osorio, M., & Hincapié, C. A. (2023). Smart Irrigation Systems in Agriculture: A Systematic Review. *Agronomy*, *13*(2), 342. https://doi.org/10.3390/agronomy13020342

van Breemen, N., Mulder, J., & Driscoll, C. T. (1983). Acidification and alkalinization of soils. *Plant and Soil*, *75*(3), 283–308. https://doi.org/10.1007/BF02369968

van den Akker, J. J. H. (2004). SOCOMO: A soil compaction model to calculate soil stresses and the subsoil carrying capacity. *Soil and Tillage Research*, *79*(1), 113–127. https://doi.org/10.1016/j.still.2004.03.021

Van den Hoven, J., Vermaas, P. E., & Van de Poel, I. (Eds.). (2015). *Handbook of Ethics, Values, and Technological Design*. Springer.

Van de Poel, I. (2015). Conflicting values in design for values. In *Handbook of ethics, values, and technological design: Sources, theory, values and application domains* (pp. 89–116). Springer. https://doi.org/10.1007/978-94-007-6970-0_5

van der Esch, S., Sewell, A., Bakkenes, M., Berkhout, E., Doelman, J., Stehfest, E., et al. (2021). *The global potential for land restoration: Scenarios for the Global Land Outlook 2*. PBL Netherlands Environmental Assessment Agency. Retrieved from https://www.pbl.nl/en/publications/the-global-potential-for-land-restoration-scenarios-for-the-global-land-outlook-2

Van der Geest, K. (2019). The Impacts of Climate Change on Ecosystem Services and Human Well-being. *Environmental Research Letters*, *14*(3), 034015.

van der Wiel, K., & Bintanja, R. (2021). Contribution of climatic changes in mean and variability to monthly temperature and precipitation extremes. *Communications Earth & Environment*, *2*(1), 1–11. https://doi.org/10.1038/s43247-020-00077-4

Van Eynde, E., Fendrich, A. N., Ballabio, C., & Panagos, P. (2023). Spatial assessment of topsoil zinc concentrations in Europe. *Science of the Total Environment*, *892*, 164512. https://doi.org/10.1016/j.scitotenv.2023.164512

Van Geel, M., Jacquemyn, H., Peeters, G., van Acker, K., Honnay, O., & Ceulemans, T. (2020). Diversity and community structure of ericoid mycorrhizal fungi in European bogs and heathlands across a gradient of nitrogen deposition. *New Phytologist*, *228*(5), 1640–1651. https://doi.org/10.1111/nph.16789

Vanmaercke, M., Panagos, P., Vanwalleghem, T., Hayas, A., Foerster, S., Borrelli, P., et al. (2021). Measuring, modelling and managing gully erosion at large scales: A state of the art. *Earth-Science Reviews*, *218*, 103637. https://doi.org/10.1016/j.earscirev.2021.103637

Van Muysen, W., Govers, G., & van Oost, K. (2002). Identification of important factors in the process of tillage erosion: The case of mouldboard tillage. *Soil and Tillage Research*, *65*(65), 77–93. https://doi.org/10.1016/s0167-1987(01)00282-3

Van Oost, K., Govers, G., de Alba, S., & Quine, T. A. (2006). Tillage erosion: A review of controlling factors and implications for soil quality. *Progress in Physical Geography: Earth and Environment*, *30*(4), 443–466. https://doi.org/10.1191/0309133306pp487ra

Van Oost, K., Govers, G., & Desmet, P. (2000). Evaluating the effects of changes in landscape structure on soil erosion by water and tillage. *Landscape Ecology*, *15*(6), 577–589. https://doi.org/10.1023/A:1008198215674

Van Oost, K., Quine, T. A., Govers, G., de Gryze, S., Six, J., Harden, J. W., et al. (2007). The impact of agricultural soil erosion on the global carbon cycle. *Science*, *318*(5850), 626–629. https://doi.org/10.1126/science.1145724

Van Oost, K., van Muysen, W., Govers, G., Deckers, J., & Quine, T. A. (2005). From water to tillage erosion dominated landform evolution. *Geomorphology*, *72*(1–4), 193–203. https://doi.org/10.1016/j.geomorph.2005.05.010

van Vliet, J. (2019). Direct and indirect loss of natural area from urban expansion. *Nature Sustainability*, *2*(8), 755–763. https://doi.org/10.1038/s41893-019-0340-0

Vanwalleghem, T., Gómez, J. A., Infante Amate, J., González de Molina, M., Vanderlinden, K., Guzmán, G., et al. (2017). Impact of historical land use and soil management change on soil erosion and agricultural sustainability during the Anthropocene. *Anthropocene*, *17*, 13–29. https://doi.org/10.1016/j.ancene.2017.01.002

Veeckman, C., Schuurman, D., Leminen, S., & Westerlund, M. (2013). Linking living lab characteristics and their outcomes: Towards a conceptual framework. *Technology Innovation Management Review*, *3*(12), 6–15. https://doi.org/10.22215/timreview/748

Veerman, C., Teresa, P. C., Catia, B., Biro, B., Bouma, J., Cienciala, E., et al. (2020). *Caring for soil is caring for life: Ensure 75% of soils are healthy by 2030 for food, people, nature and climate*. Report of the Mission Board for Soil health and food. European Commission. https://doi.org/10.2777/821504

Velasco-Muñoz, J. F., Aznar-Sánchez, J. A., Batlles-delaFuente, A., & Fidelibus, M. D. (2018). Sustainable water use in agriculture: A review of worldwide research. *Sustainability*, *10*(4), 1084. https://doi.org/10.3390/su10041084

Velten, S., Leventon, J., Jager, N. W., & Newig, J. (2015). What is sustainable agriculture? A systematic review. *Sustainability*, *7*(6), 7833–7865. https://doi.org/10.3390/su7067833

Verachtert, E., Maetens, W., van den Eeckhaut, M., Poesen, J., & Deckers, J. (2011). Soil loss rates due to piping erosion. *Earth Surface Processes and Landforms*, *36*(13), 1715–1725. https://doi.org/10.1002/esp.2186

Verhoeven, D., Berkhout, E., Sewell, A., & van der Esch, S. (2024). The global cost of international commitments on land restoration. *Land Degradation & Development*, *35*(16), 4864–4874. https://doi.org/10.1002/ldr.5263

Vieira, D. C. S., Yunta, F., Baragaño, D., Evrard, O., Reiff, T., Silva, V., et al. (2024). Soil pollution in the European Union–An outlook. *Environmental Science & Policy*, *161*, 103876.

Virto, I., Imaz, M. J., Fernández-Ugalde, O., Gartzia-Bengoetxea, N., Enrique, A., & Bescansa, P. (2014). Soil degradation and soil quality in Western Europe: Current situation and future perspectives. *Sustainability*, *7*(1), 313–365. https://doi.org/10.3390/su7010313

Vogelbacher, A., Aminzadeh, M., Madani, K., & Shokri, N. (2024). An analytical framework to investigate groundwater- atmosphere interactions influenced by soil properties. *Water Resources Research*, *60*(4), e2023WR036643. https://doi.org/10.1029/2023WR036643

von Uexküll, H. R., & Mutert, E. (1995). Global extent, development and economic impact of acid soils. *Plant and Soil*, *171*, 1–15. https://doi.org/10.1007/bf00009558

Wadoux, A. M. J., Román Dobarco, M., Malone, B., Minasny, B., McBratney, A. B., & Searle, R. (2023). Baseline high-resolution maps of organic carbon content in Australian soils. *Scientific Data*, *10*(1), 181. https://doi.org/10.1038/s41597-023-02056-8

Waidelich, P., Batibeniz, F., Rising, J., Kikstra, J. S., & Seneviratne, S. I. (2024). Climate damage projections beyond annual temperature. *Nature Climate Change*, *14*(6), 592–599. https://doi.org/10.1038/s41558-024-01990-8

Waldner, P., Thimonier, A., Graf Pannatier, E., Etzold, S., Schmitt, M., Marchetto, A., et al. (2015). Exceedance of critical loads and of critical limits impacts tree nutrition across Europe. *Annals of Forest Science*, *72*(7), 929–939. https://doi.org/10.1007/s13595-015-0489-2

Walker, T. W., & Syers, J. K. (1976). The fate of phosphorus during pedogenesis. *Geoderma*, *15*(1), 1–19. https://doi.org/10.1016/0016-7061(76)90066-5

Wang, F., Guan, Q., Tian, J., Lin, J., Yang, Y., Yang, L., & Pan, N. (2020). Contamination characteristics, source apportionment, and health risk assessment of heavy metals in agricultural soil in the Hexi Corridor. *Catena*, *191*, 104573. https://doi.org/10.1016/j.catena.2020.104573

Wang, J., Zhen, J., Hu, W., Chen, S., Lizaga, I., Zeraatpisheh, M., & Yang, X. (2023). Remote sensing of soil degradation: Progress and perspective. *International Soil and Water Conservation Research*, *11*(3), 429–454. https://doi.org/10.1016/j.iswcr.2023.03.002

Wang, J. X. L. (2015). Mapping the global dust storm records: Review of dust data sources in supporting modeling/climate study. *Current Pollution Reports*, *1*(2), 82–94. https://doi.org/10.1007/s40726-015-0008-y

Wang, M., Zhang, S., Guo, X., Xiao, L., Yang, Y., Luo, Y., et al. (2024). Responses of soil organic carbon to climate extremes under warming across global biomes. *Nature Climate Change*, *14*(1), 98–105. https://doi.org/10.1038/s41558-023-01874-3

Wang, Y., Du, Z., Zhang, Y., Chen, S., Lin, S., Hopke, P. K., et al. (2023). Long-term exposure to particulate matter and COPD mortality: Insights from causal inference methods based on a large population cohort in southern China. *Science of the Total Environment*, *863*, 160808. https://doi.org/10.1016/j.scitotenv.2022.160808

Wang, Y., Min, D., Ye, W., Wu, K., & Yang, X. (2023). The impact of rural location on farmers' livelihood in the Loess Plateau: Local, urban–rural, and interconnected multi-spatial perspective. *Land*, *12*(8), 1624. https://doi.org/10.3390/land12081624

Wang, Z., Hoffmann, T., Six, J., Kaplan, J. O., Govers, G., Doetterl, S., & Van Oost, K. (2017). Human-induced erosion has offset one-third of carbon emissions from land cover change. *Nature Climate Change*, *7*(5), 345–349. https://doi.org/10.1038/nclimate3263

Wang, Z., Walker, G. W., Muir, D. C. G., & Nagatani-Yoshida, K. (2020). Toward a global understanding of chemical pollution: A First Comprehensive analysis of National and Regional Chemical Inventories. *Environmental Science and Technology*, *54*(5), 2575–2584. https://doi.org/10.1021/acs.est.9b06379

Webb, R. H. (2002). Recovery of severely compacted soils in the Mojave Desert. *Arid Land Research and Management*, *16*(3), 291–305. https://doi.org/10.1080/153249802760284829

Wedin, D. A., & Tilman, D. (1996). Influence of nitrogen loading and species composition on the carbon balance of grasslands. *Science*, *274*(5293), 1720–1723. https://doi.org/10.1126/science.274.5293.1720

Weersink, A., & Wossink, A. (2005). Lessons from agri-environmental policies in other countries for dealing with salinity in Australia. *Australian Journal of Experimental Agriculture*, *45*(11), 1481–1493. https://doi.org/10.1071/ea04156

Wei, H., Xu, W., Kang, B., Eisner, R., Muleke, A., Rodriguez, D., et al. (2024). Irrigation with Artificial Intelligence: Problems, premises, promises. *Human-Centric Intelligent Systems*, *4*(2), 187–205. https://doi.org/10.1007/s44230-024-00072-4

Weiss, P., Herbert, R., & Stier, P. (2025). ICON-HAM-lite 1.0: Simulating the Earth system with interactive aerosols at kilometer scales. *Geoscientific Model Development*, *18*(12), 3877–3894. https://doi.org/10.5194/gmd-18-3877-2025

Werder, M., Bont, L. G., Schweier, J., & Thees, O. (2025). A comprehensive analysis of time investment in skid trail planning for forest access. *PLoS One*, *20*(2), e0317963. https://doi.org/10.1371/journal.pone.0317963

West, P. C., Gerber, J. S., Engstrom, P. M., Mueller, N. D., Brauman, K. A., Carlson, K. M., et al. (2014). Leverage points for improving global food security and the environment. *Science*, *345*(6194), 325–328. https://doi.org/10.1126/science.1246067

Wicke, B., Smeets, E., Dornburg, V., Vashev, B., Gaiser, T., Turkenburg, W., & Faaij, A. (2011). The global technical and economic potential of bioenergy from salt-affected soils. *Energy & Environmental Science*, *4*(8), 2669–2681. https://doi.org/10.1039/c1ee01029h

Wiedermann, M. M., Nordin, A., Gunnarsson, U., Nilsson, M. B., & Ericson, L. (2007). Global change shifts vegetation and plant-parasite interactions in a boreal mire. *Ecology*, *88*(2), 454–464. https://doi.org/10.1890/05-1823

Wilkinson, M. D., Dumontier, M., Aalbersberg, I. J., Appleton, G., Axton, M., Baak, A., et al. (2016). The FAIR guiding principles for scientific data management and stewardship. *Scientific Data*, *3*(1), 1–9. https://doi.org/10.1038/sdata.2016.18

Williams, D. J. (2022). Review of mine waste capping methodologies for use in semi-arid regions of Queensland, Australia. In M. Tibbett, A. B. Fourie, & G. Boggs (Eds.), *Mine Closure 2022:* Proceedings of the 15th International Conference on Mine Closure. Australian Centre for Geomechanics.

Winkler, K., Fuchs, R., Rounsevell, M., & Herold, M. (2021). Global land use changes are four times greater than previously estimated. *Nature Communications*, *12*(1), 2501. https://doi.org/10.1038/s41467-021-22702-2

Wischmeier, W. H., & Smith, D. D. (1978). *Predicting rainfall erosion losses - a guide to conservaton planning:* Agriculture Handbook. U.S. Department of Agriculture.

Withers, P. J. A., Hodgkinson, R. A., Bates, A., & Withers, C. M. (2006). Some effects of tramlines on surface runoff, sediment and phosphorus mobilization on an erosion-prone soil. *Soil Use & Management*, *22*(3), 245–255. https://doi.org/10.1111/J.1475-2743.2006.00034.X

Wong, V. N. L., Greene, R. S. B., Dalal, R. C., & Murphy, B. W. (2010). Soil carbon dynamics in saline and sodic soils: A review. *Soil Use & Management*, *26*(1), 2–11. https://doi.org/10.1111/j.1475-2743.2009.00251.x

Worlanyo, A. S., & Jiangfeng, L. (2021). Evaluating the environmental and economic impact of mining for post-mined land restoration and land-use: A review. *Journal of Environmental Management*, *279*, 111623. https://doi.org/10.1016/j.jenvman.2020.111623

World Bank. (2006). IEG ICR Review: Second Loess Plateau Watershed Rehabilitation Project. *World Bank Document*.

World Bank. (2007). *Restoring China's Loess Plateau*. World Bank Feature Story.

World Health Organization. (2021). *WHO global air quality guidelines: particulate matter (PM2.5 and PM10), ozone, nitrogen dioxide, sulfur dioxide and carbon monoxide*. World Health Organization.

World Health Organization. (2023). *Burden of disease attributable to unsafe drinking-water, sanitation and hygiene, 2019 update*. World Health Organization.

Wurtsbaugh, W. A., Miller, C., Null, S. E., DeRose, R. J., Wilcock, P., Hahnenberger, M., et al. (2017). Decline of the world's saline lakes. *Nature Geoscience*, *10*(11), 816–821. https://doi.org/10.1038/ngeo3052

Xia, X., & Tang, K. (1992). Study on impact of vegetation on soil properties and soil erosion. *Soil and Water Conservation*, *6*(2), 12–20.

Xie, D., Duan, L., Du, E., & de Vries, W. (2024). Chapter 14 - Indicators and thresholds for nitrogen saturation in forest ecosystems. In E. Du & W. D. Vries (Eds.), *Atmospheric Nitrogen Deposition to Global Forests* (pp. 249–261). Academic Press.

Xie, D. N., Ge, X. D., Duan, L., & Mulder, J. (2024). Effects of acid deposition control in China: A review based on responses of subtropical forests. *Frontiers of Environmental Science & Engineering*, *18*(6), 77. https://doi.org/10.1007/s11783-024-1837-4

Xing, Y., & Wang, X. (2024). Precise application of water and fertilizer to crops: Challenges and opportunities. *Frontiers in Plant Science*, *15*, 1444560. https://doi.org/10.3389/fpls.2024.1444560

Xu, H., Wang, Y., Han, T., Li, R., Ma, J., Qiu, X., et al. (2024). Enhanced assessment of regional impacts from wind erosion by integrating particle size. *Catena*, *239*, 107937. https://doi.org/10.1016/j.catena.2024.107937

Yadav, A., & Singh, R. M. (2024). Spatio-temporal Estimation of Evapotranspiration and Runoff in Sub-Watersheds of a Basin Using ArcSWAT. *National Academy Science Letters*, *47*(6), 1–5. https://doi.org/10.1007/s40009-024-01395-3

Yang, G., Ryo, M., Roy, J., Lammel, D., Ballhausen, M., Jing, X., et al. (2022). Multiple anthropogenic pressures eliminate the effects of soil microbial diversity on ecosystem functions in experimental microcosms. *Nature Communications*, *13*(1), 4260. https://doi.org/10.1038/s41467-022-31936-7

Yang, N., Li, L., Han, L., Gao, K., Qu, S., & Li, J. (2024). Retrieving heavy metal concentrations in urban soil using satellite hyperspectral imagery. *International Journal of Applied Earth Observation and Geoinformation*, *132*, 104079. https://doi.org/10.1016/j.jag.2024.104079

Yang, Q., Chen, Y., Li, X., Yang, J., & Gao, Y. (2024). Livelihood vulnerability and adaptation for households engaged in forestry in ecological restoration areas of the Chinese Loess Plateau. *Chinese Geographical Science*, *34*(5), 849–868. https://doi.org/10.1007/s11769-024-1451-8

Yimer, F., Messing, I., Ledin, S., & Abdelkadir, A. (2008). Effects of different land use types on infiltration capacity in a catchment in the highlands of Ethiopia. *Soil Use & Management*, *24*(4), 344–349. https://doi.org/10.1111/j.1475-2743.2008.00182.x

Yoon, A., & Hohenegger, C. (2025). Muted amazon rainfall response to deforestation in a global storm-resolving model. *Geophysical Research Letters*, *52*(4), e2024GL110503. https://doi.org/10.1029/2024gl110503

Yu, H., Chin, M., Yuan, T., Bian, H., Remer, L. A., Prospero, J. M., et al. (2015). The fertilizing role of African dust in the Amazon rainforest:A first multiyear assessment based on data from Cloud-Aerosol Lidar andInfrared Pathfinder Satellite Observations. *Geophysical Research Letters*, *42*(6), 1984–1991. https://doi.org/10.1002/2015GL063040

Yu, Y., Zhao, W., Martinez-Murillo, J. F., & Pereira, P. (2020). Loess Plateau: From degradation to restoration. *Science of the Total Environment*, *738*, 140206. https://doi.org/10.1016/j.scitotenv.2020.140206

Yulianto, F., Raharjo, P. D., Pramono, I. B., Setiawan, M. A., Chulafak, G. A., Nugroho, G., et al. (2023). Prediction and mapping of land degradation in the Batanghari watershed, Sumatra, Indonesia: Utilizing multi-source geospatial data and machine learning modeling techniques. *Modeling Earth Systems and Environment*, *9*(4), 4383–4404. https://doi.org/10.1007/s40808-023-01761-y

Zerbe, S. (2023). Restoration Economy: Costs and Benefits. In *Restoration of Ecosystems* – Bridging Nature and Humans (pp. 487–505). Springer. https://doi.org/10.1007/978-3-662-65658-7_23

Zhang, B., Horn, R., & Hallett, P. D. (2005). Mechanical resilience of degraded soil amended with organic matter. *Soil Science Society of America Journal*, *69*(3), 864–871. https://doi.org/10.2136/sssaj2003.0256

Zhang, B., Jia, Y., Fan, H., Guo, C., Fu, J., Li, S., et al. (2024). Soil compaction due to agricultural machinery impact: A systematic review. *Land Degradation & Development*, *35*(10), 3256–3273. https://doi.org/10.1002/ldr.5144

Zhang, C., Gou, L., Hu, S., & Lu, N. (2022). A thermodynamic formulation of water potential in soil. *Water Resources Research*, *58*(9), e2022WR032369. https://doi.org/10.1029/2022WR032369

Zhang, D., Li, H., Luo, X.-S., Huang, W., Pang, Y., Yang, J., et al. (2022). Toxicity assessment and heavy metal components of inhalable particulate matters (PM2. 5 and PM10) during a dust storm invading the city. *Process Safety and Environmental Protection*, *162*, 859–866. https://doi.org/10.1016/j.psep.2022.04.065

Zhang, G., Bai, J., Zhai, Y., Jia, J., Zhao, Q., Wang, W., & Hu, X. (2024). Microbial diversity and functions in saline soils: A review from a biogeochemical perspective. *Journal of Advanced Research*, *59*, 129–140. https://doi.org/10.1016/j.jare.2023.06.015

Zhang, H., He, H., Gao, Y., Mady, A., Filipović, V., Dyck, M., et al. (2023). Applications of computed tomography (CT) in environmental soil and plant sciences. *Soil and Tillage Research*, *226*, 105574. https://doi.org/10.1016/j.still.2022.105574

Zhang, H., & Renschler, C. S. (2024). QGeoWEPP: An open-source geospatial interface to enable high-resolution watershed-based soil erosion assessment. *Environmental Modelling & Software*, *179*, 106118. https://doi.org/10.1016/j.envsoft.2024.106118

Zhang, H., Wei, W., Chen, L., & Wang, L. (2017). Effects of terracing on soil water and canopy transpiration of Pinus tabulaeformis in the Loess Plateau of China. *Ecological Engineering*, *102*, 557–564. https://doi.org/10.1016/j.ecoleng.2017.02.044

Zhang, H., & Zhuang, L. (2019). The impact of soil erosion on internal migration in China. *PLoS One*, *14*(4), e0215124. https://doi.org/10.1371/journal.pone.0215124

Zhang, W., Li, H., Pueppke, S. G., Diao, Y., Nie, X., Geng, J., et al. (2020). Nutrient loss is sensitive to land cover changes and slope gradients of agricultural hillsides: Evidence from four contrasting pond systems in a hilly catchment. *Agricultural Water Management*, *237*, 106165. https://doi.org/10.1016/j.agwat.2020.106165

Zhang, X., Lark, T. J., Clark, C. M., Yuan, Y., & LeDuc, S. D. (2021). Grassland-to-cropland conversion increased soil, nutrient, and carbon losses in the US Midwest between 2008 and 2016. *Environmental Research Letters*, *16*(5), 054018. https://doi.org/10.1088/1748-9326/abecbe

Zhang, X., Zhang, W., Sai, X., Chun, F., Li, X., Lu, X., & Wang, H. (2022). Grazing altered soil aggregates, nutrients and enzyme activities in a Stipa kirschnii steppe of Inner Mongolia. *Soil and Tillage Research*, *219*, 105327. https://doi.org/10.1016/j.still.2022.105327

Zhang, Z. (2018). Theoretical and technical analysis of vegetation restoration and construction in Loess Plateau. In *7th International Conference on Social Science, Education and Humanities Research (SSEHR 2018). Web of Proceedings*.

Zhang, Z., & Peng, X. (2021). Bio-tillage: A new perspective for sustainable agriculture. *Soil and Tillage Research*, *206*, 104844. https://doi.org/10.1016/j.still.2020.104844

Zhao, H., Dong, J., Yang, Y., Zhao, J., He, J., & Yue, C. (2024). Vegetation restoration increases the drought risk on the Loess Plateau. *Plants*, *13*(19), 2735. https://doi.org/10.3390/plants13192735

Zhao, H., He, H., Wang, J., Bai, C., & Zhang, C. (2018). Vegetation restoration and its environmental effects on the Loess Plateau. *Sustainability*, *10*(12), 4676. https://doi.org/10.3390/su10124676

Zhao, J., Feng, X., Deng, L., Yang, Y., Zhao, Z., Fu, B., et al. (2020). Quantifying the effects of vegetation restorations on the soil erosion export and nutrient loss on the Loess Plateau. *Frontiers in Plant Science*, *11*, 573126. https://doi.org/10.3389/fpls.2020.573126

Zhao, Q., Yu, L., Li, X., Peng, D., Zhang, Y., & Gong, P. (2021). Progress and trends in the application of Google Earth and Google Earth Engine. *Remote Sensing*, *13*(18), 3778. https://doi.org/10.3390/rs13183778

Zhen, L. (2013). The national census for soil erosion and dynamic analysis in China. *International Soil and Water Conservation Research*, *1*(2), 12–18. https://doi.org/10.1016/S2095-6339(15)30035-6

Zheng, H., Miao, C., Huntingford, C., Tarolli, P., Li, D., Panagos, P., et al. (2025). The impacts of erosion on thecarbon cycle. *Reviews of Geophysics*, *63*(1), e2023RG000829. https://doi.org/10.1029/2023RG000829

Zheng, X.-Y., Guo, S.-J., Hu, J.-X., Meng, R.-L., Xu, Y.-J., Lv, Y.-H., et al. (2024). Long-term associations of PM1 vs. PM2. 5 and PM10 with asthma and asthma-related respiratory symptoms in middle-aged and elderly population. *ERJ Open Research*.

Zhou, Y., Yang, Z., Liu, J., Li, X., Wang, X., Dai, C., et al. (2023). Crop rotation and native microbiome inoculation restore soil capacity to suppress root disease. *Nature Communications*, *14*(1), 8126. https://doi.org/10.1038/s41467-023-43926-4

Zhu, K., Sun, Z., Zhao, F., Yang, T., Tian, Z., Lai, J., et al. (2021). Relating hyperspectral vegetation indices with soil salinity at different depths for the diagnosis of winter wheat salt stress. *Remote Sensing*, *13*(2), 250. https://doi.org/10.3390/rs13020250

Zickgraf, C. (2018). Immobility. In *Routledge Handbook of Environmental Displacement and Migration* (pp. 71–84). Routledge.

10.1029/2025RG000883

Zimmermann, B., Elsenbeer, H., & De Moraes, J. M. (2006). The influence of land-use changes on soil hydraulic properties: Implications for runoff generation. *Forest Ecology and Management*, *222*(1–3), 29–38. https://doi.org/10.1016/j.foreco.2005.10.070
Zimnicki, T., Boring, T., Evenson, G., Kalcic, M., Karlen, D. L., Wilson, R. S., et al. (2020). On quantifying water quality benefits of healthy soils. *BioScience*, *70*(4), 343–352. https://doi.org/10.1093/biosci/biaa011
Zomer, R. J., Bossio, D. A., Sommer, R., & Verchot, L. V. (2017). Global sequestration potential of increased organic carbon in cropland soils. *Scientific Reports*, *7*(1), 15554. https://doi.org/10.1038/s41598-017-15794-8
Zucca, C., Fleiner, R., Bonaiuti, E., & Kang, U. (2022). Land degradation drivers of anthropogenic sand and dust storms. *Catena*, *219*, 106575. https://doi.org/10.1016/j.catena.2022.106575