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Foreword

Soils are the silent foundation of life on Earth. They are at the heart of global food security, biodiversity conservation, climate regulation, and sustainable development. They support 95 percent of our food production, host 59 percent of global biodiversity, and serve as the planet's second largest carbon sink after the oceans. Yet, despite their fundamental functions and services, soil degradation continues to accelerate at an alarming rate.

The *Status of the world's soil resources* 2026 is a timely reminder that we can no longer afford to take soil for granted. Building on the inaugural 2015 edition, this second global assessment provides an authoritative, science-based review of current soil management practices worldwide and their impact on soil threats as well as the trends in adoption of sustainable soil management (SSM) worldwide. It reflects a decade of scientific progress, awareness raising, updated global data, and growing international policy attention to soil health and resilience.

This report was prepared by the Intergovernmental Technical Panel on Soils (ITPS), the scientific and technical advisory body of the Global Soil Partnership (GSP), whose Secretariat is hosted by the Food and Agriculture Organization of the United Nations (FAO). Since 2021, the ITPS has led the development of this edition, coordinating the contributions of more than 290 experts from 75 countries through a broad, inclusive and participatory process. This collaborative spirit underscores the shared global responsibility to care for one of our planet's most precious yet fragile resources.

The findings of the report are both sobering and motivating. Since 2015, global population growth of 0.8 billion has driven a 24 percent increase in harvested areas for major crops, contributing to mounting pressure on soils. Soil erosion remains the most severe and interconnected threat, while other threats – such as the loss of soil organic carbon, nutrient mismanagement, biodiversity loss, salinization, pollution, and soil sealing – are also worsening. Alarmingly, only 10 percent of regional assessments show improving trends in the adoption of SSM practices.



But there is hope. The report shows that soil degradation is not inevitable. Scientific knowledge and proven practices – such as reduced tillage, cover crops, organic amendments, and agroforestry – have expanded greatly. With informed policies, targeted incentives, inclusive governance, and the application of SSM practices, we can reverse degradation, restore soil functions and secure their contributions for future generations.

As we enter the final stretch of the 2030 Agenda for Sustainable Development, and in the context of the UN Decade on Ecosystem Restoration, this report offers a critical resource for policymakers, scientists, land users, and all stakeholders committed to a sustainable future. Healthy soils are essential to restoring ecosystems, enhancing resilience, and achieving land degradation neutrality (LDN). Advancing SSM, strengthening soil governance, enhancing knowledge and literacy, promoting awareness, improving harmonized soil assessment and monitoring systems, and fostering technical cooperation will be essential to support future assessments and reinforce collective action toward healthy soils worldwide.

The SWSR 2026 calls for renewed commitment and collective action to safeguard soils for present and future generations. The evidence presented in this report highlights both the urgency of the challenges ahead and the opportunities to accelerate the transition towards SSM worldwide. Let this report inspire stronger collaboration, increased investment, and decisive action – because without healthy soils, there can be no food security, nor sustainable development.

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Abbreviations

AMP	adaptive multipaddock
AMPA	α-amino-3-hydroxy-5-methyl-4-isoxazolepropionic acid
AU	African Union
BNF	biological nitrogen fixation
BP	before present
BSWM	Bureau of Soils and Water Management
CAAAP	Conservation Agriculture Alliance for Asia-Pacific
CAFO	concentrated animal feed operation
CAP	Common Agricultural Policy
CBD	United Nations Convention on Biological Diversity
CCI	Climate Change Initiative
CE	climate erosivity
CEC	cation exchange capacity
CF	contamination factor
CL	critical level
CTRC	Conservation Tillage Research Centre
COP	Conference of the Parties
DDT	dichlorodiphenyl-trichloroethane
DGVM	dynamic global vegetation model
DOC	dissolved organic carbon
DPSIR	Driver-Pressure-State-Impact-Response framework
EC	electrical conductivity
EEA	European Environmental Agency
EEF	enhanced efficiency fertilizer
ERW	explosive remnants of war
ES	ecosystem services
ESA	European Space Agency
ESDAC	European Soil Data Centre
ESM	earth system model
ESP	exchangeable sodium percentage
FAO	Food and Agriculture Organization of the United Nations
AEZ	Agro-Ecological Zoning
GAEZ	Global Agro-Ecological Zoning
GDP	gross domestic product
GHG	greenhouse gas
GSASmap	Global Map of Salt-affected Soils
GSBI	Global Soil Biodiversity Initiative
GSP	Global Soil Partnership
GML	ground magnesium limestone
GWP	global warming potential
HCH	hexachlorocyclohexane
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
ISAM	Integrated Science Assessment Model
ISFM	integrated soil fertility management
JRC	Joint Research Centre
ITPS	Intergovernmental Technical Panel on Soils
LAC	Latin America and the Caribbean
LDN	land degradation neutrality
LHASA	Landslide Hazard Assessment for Situational Awareness model
LS	length and steepness
LST	land surface temperature
LTPS	long-term productivity study
LUC	land use capability
LUCAS	Land Use/Cover Area frame Survey
LULC	land use and land cover
MA	Millennium Ecosystem Assessment
MAC	maximum allowable concentration
MAPAMA	Spanish Ministry of Agriculture, Fisheries, Food and Environment
MLIT	Ministry of Land, Infrastructure, Transport and Tourism
MSW	municipal solid waste
MNM	manufactured nanomaterial
NBSAP	national biodiversity strategies and action plan
NCP	nature's contributions to people
NENA	Near East and North Africa
NGO	non-governmental organization
NPP	net primary productivity
NUE	nitrogen use efficiency
OC	organic carbon
OECD	Organisation for Economic Co-operation and Development
OTU	operational taxonomic unit
PAH	polycyclic aromatic hydrocarbons
PAE	phthalic acid ester
PCB	polychlorinated biphenyl

PEC predicted environmental concentration

PET potential evapotranspiration

PFAS polyfluoroalkyl substances

PFOA perfluorooctanoic acid

PFOS perfluorooctanesulfonic acid

PLFA phospholipid-derived fatty acid

PNEC predicted no-effect-concentration

POP persistent organic pollutant

POC particulate organic carbon

PPCP pharmaceuticals and personal care products

PUE phosphorus use efficiency

PV photovoltaic

RCP Representative Concentration Pathway

ROC resistant organic carbon

(R)USLE (Revised) Universal Soil Loss Equation

RWEE Revised Wind Erosion Equation model

SDG Sustainable Development Goal

SIC soil inorganic carbon

SIDS Small Island Developing States

SLCH soil loss due to crop harvesting

SOC soil organic carbon

SOM soil organic matter

SPI Science–Policy Interface

SLM sustainable land management

SSM sustainable soil management

STP soil test P

SUITMAs soils of urban, industrial, traffic, mining and military areas

SWAT Soil and Water Assessment Tool

SWSR Status of the world's soil resources

UHI urban heat island

UN United Nations

UNCCD United Nations Convention to Combat Desertification

UNEP United Nations Environment Programme

UNFCCC United Nations Framework Convention on Climate Change

UNITAR United Nations Institute for Training and Research

UNMAS United Nations Mine Action Service

UNOSAT United Nations Satellite Centre

USDA United States Department of Agriculture

US EPA United States Environmental Protection Agency

USLE Universal Soil Loss Equation

UXO unexploded ordnance

VGSSM Voluntary Guidelines for Sustainable Soil Management

WOCAT World Overview of Conservation Approaches and Technologies

WWF World Wide Fund for Nature

Ag silver

Al aluminium

As arsenic

Au gold

B boron

Be beryllium

Br bromine

C carbon

Ca calcium

Cd cadmium

Cl chlorine

Co cobalt

Cr chromium

Cs caesium

Cu copper

Eu europium

F fluorine

Fe iron

H hydrogen

Ha hafnium

Hg mercury

I iodine

K potassium

Li lithium

Mg magnesium

Mn manganese

Mo molybdenum

N nitrogen

Na sodium

Ni nickel

P phosphorus

Pb lead

Po polonium

Ra radium

Rb rubidium

S sulphur

Sb antimony

Se	selenium
Sr	strontium
Tb	terbium
Ti	titanium
Th	thorium
U	uranium
V	vanadium
W	tungsten
Yt	ytterbium
Zn	zinc
Zr	zirconium




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Chapter 1 | Introduction

1.1 Purpose and approach of the report

Over the past few decades, the role of soils in supporting humanity has gained increasing recognition. Facilitated by the Food and Agriculture Organization of the United Nations (FAO) and the global soil science community, the International Year of Soils (IYS) 2015 reached a major public awareness milestone through the collaborative engagement of multiple stakeholders, including governments, civil society, academia, and the private sector.

In the same year, on World Soil Day (5 December), the first *Status of the world's soil resources* (SWSR) report (FAO and ITPS, 2015), led by the Intergovernmental Technical Panel on Soils (ITPS) of the Global Soil Partnership (GSP), was released. The goal of the 2015 SWSR report was to make the essential connections clear between human well-being and soil resources and to provide a benchmark against which we could measure our collective progress in the conservation, sustainable use and restoration of soils towards Sustainable Development Goals (SDGs).

The emphasis on soil health in the 2015 *Revised World Soil Charter* (FAO, 2015) highlights the increasing recognition of soil as a living entity and has been central to the wider uptake of this concept across the resource management and science communities since then. The ITPS defines soil health as "the ability of the soil to sustain the productivity, diversity and environmental services of terrestrial ecosystems" (FAO, 2020, p. 1), emphasizing the essential linkages between life in the soil and contributions that soils make to societies.

Since 2015, besides improving the assessment of the status of soils, there has also been an extraordinary effort made by the soil science community to quantify the ability of specific soil management practices to reduce the effects of harmful processes that threaten soils. Improved management practices contribute to the sustainable management of the soil resource, as the *Revised World Soil Charter* explains:

> *"Soil management is sustainable if the supporting, provisioning, regulating, and cultural services provided by soil are maintained or enhanced without significantly impairing either the soil functions that enable those services or biodiversity." (FAO, 2015, p. 4).*

The suite of practices associated with sustainable soil management (SSM) were identified in the *Voluntary Guidelines for Sustainable Soil Management* (VGSSM) (FAO, 2017).

The 2026 SWSR focuses on assessing the degree to which SSM practices known to prevent, control and mitigate soil threats have been adopted since 2015. This approach reflects the emphasis since 2015 on promoting SSM within the work of the ITPS and the GSP. The methodology for the assessment is further described in Chapter 7 of this report. This methodological approach reflects both significant advances in scientific understanding of SSM and continuing limitations in the availability of globally consistent data needed to reliably assess changes in soil properties and degradation processes over time. Although considerable progress has been made since 2015 in global soil mapping and in modelling processes associated with soil threats, the available information remains insufficient to provide a consistent global assessment of changes in the biophysical condition of soils across all threat categories. Consequently, the ITPS adopted a management-based approach, which provides an indirect, proxy-based assessment of progress in addressing soil threats through improved soil management.

To maintain an actionable scope, the assessment focuses on soil threats that can be directly prevented, controlled or mitigated through SSM practices and therefore concentrates on managed agricultural landscapes such as croplands and pastures. Consequently, this excludes larger environmental shifts driven by climate change; for example, while permafrost thawing or coastal salinization from sea-level rise represent severe soil degradation, addressing them requires system-wide interventions that extend far beyond localized SSM.

The primary audience for the report are stakeholders involved in soil governance in agriculture (soil stewards), including policymakers at international, regional and national scales, land-use planners, members of organizations that support decision-making, investors (public and private) in land and soil management, and individual farmers, forest users and land managers, ranchers and other soil managers who are ultimately responsible for the management that occurs. For FAO, the concept of governance constitutes:

> *"formal and informal rules, organizations, and processes through which public and private actors articulate their interests; frame and prioritize issues; and make, implement, monitor, and enforce decisions." (FAO and UNEP, 2020, p. 28).*

The content presented in the report is consistent with the ITPS's primary mandate to provide scientific and technical advice on global soil issues and to advocate for the inclusion of SSM into various development agendas. It aims to inform two phases of the policy cycle (Bouma and Montanarella, 2016):

1. the signalling phase, in which problems are identified based on a characterization of current conditions; and
2. the design phase, in which options for corrective action are defined based on research using existing and newly acquired information.



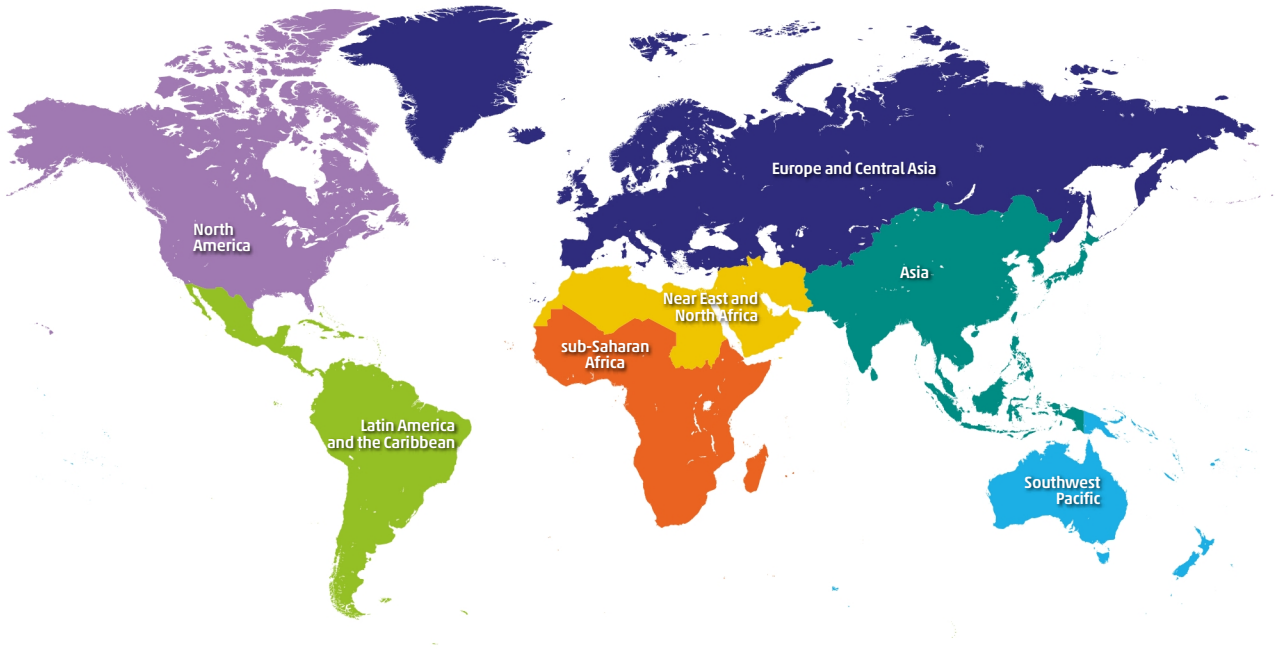
1.2 Structure of the report

The report is organized into sixteen chapters, including this chapter, which introduces the scope and the approach used to produce this report. Chapter 2 through to Chapter 6 synthesize a literature review of the scientific advances made by the soil science community that enhance our ability to assess the status of soils and improve their status, as follows:

- Chapter 2 establishes the connections between soils and the SDGs.
- Chapter 3 reviews the advances made by the soil science community since the preparation of the 2015 SWSR on the specific threats to soil functions, including newly emerging threats. The primary focus is on studies that contribute to a global perspective.
- Chapter 4 identifies the main drivers underlying threats to soil functions.
- Chapter 5 reviews the progress made since the preparation of the 2015 SWSR in understanding the role that SSM plays in reducing the threats to soil functions and hence supporting the achievement of the SDGs.
- Chapter 6 discusses how to create an enabling environment for the adoption of SSM.

Chapter 7 summarizes the methodology and criteria used for the regional assessments of the state and trend of threats to soil functions, informed by a management-based approach and the confidence levels assigned by the assessment panel. Chapter 8 to Chapter 14 present regional summaries of the assessment for the seven GSP regions: Asia, Europe and Central Asia, Latin America and the Caribbean, the Near East and North Africa, North America, Southwest Pacific, and sub-Saharan Africa respectively (Figure 1.1). Chapter 15 provides a global overview of the state and trends organized by threats to soil functions and builds on the findings of the regional assessments. Finally, Chapter 16 presents the main conclusions and recommendations developed by the ITPS to address the main barriers to the broader adoption of SSM.

Figure 1.1. Regions used in the Status of the world's soil resources report



Refer to the disclaimer on page ii for the names and boundaries used in this map.

Source: Authors' own elaboration.

References for Chapter 1

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Chapter 2 | Soils and the Sustainable Development Goals

2.1 Introduction

The 17 SDGs (Figure 2.1) were adopted by the UN in 2015 through a resolution, *Transforming our world: The 2030 Agenda for Sustainable Development* (2030 Agenda) (UN, 2015) with an overall aspiration to end poverty, protect the planet and ensure global peace and prosperity. The announcement of the 2030 Agenda and the associated SDGs immediately sparked in-depth examinations by the soil science community on how soils could contribute to the realization of the SDGs and led to many review articles. These reviews (such as Bouma, Montanarella and Evanylo, 2019) acknowledge that while soils are only specifically mentioned in four targets in the SDGs, importantly, a number of other targets cannot be met without soils playing a large role. This chapter outlines how soil management and health contribute to achieving various SDGs. Across many of these review papers, the contribution of soil science is deemed to be greatest for SDG 2: Zero Hunger, SDG 3: Good Health and Well-being, SDG 6: Clean Water and Sanitation, SDG 13: Climate Action and SDG 15: Life on Land (e.g. Bouma, Montanarella and Evanylo, 2019; Lal *et al.*, 2021). Therefore, this chapter will focus on those SDGs.

Figure 2.1. The seventeen Sustainable Development Goals of the United Nations



Source: **UN.** n.d. SDG poster [poster]. New York, USA. https://www.un.org/sustainabledevelopment/news/communications-material/

Authors such as Smith *et al.* (2021), Keesstra *et al.* (2016) and Lal *et al.* (2021) also draw links between soils and the remaining SDGs, reinforcing the multifaceted contribution of soils to human and planetary well-being.

2.2 Ecosystem services, nature's contributions to people, and soil functions

Due to the limited mention of soils in the SDGs, the link between soils and the SDGs may not be immediately clear to the policymaking community and the broader public. However, connections can be made by first establishing the contributions that soil makes to ecological goods and services that are the foundation of the SDGs, and then demonstrating how the fundamental physical, chemical and biological soil processes (or soil functions) make these contributions possible.

The benefits derived from these soil functions are threatened by harmful processes and actions that impair the soil functions and degrade soils. The main degradation processes are soil erosion, loss of soil organic carbon (SOC), nutrient mismanagement, accumulation of salts, soil pollution, loss of soil biodiversity, soil sealing, urbanization, and physical deterioration. These harmful processes and the capacity of SSM to address them is the focus of this report.

Since 2005, the contributions that nature makes to humanity have been codified as ecosystem services (ES). The highly influential Millennium Ecosystem Assessment (MA) (WRI, 2005) drew upon the literature on ES to illustrate the linkages between ecosystems and human well-being. The assessment defined ES as the benefits that ecosystems provided to people, and identified four main groups of ES: provisioning, regulating, cultural, and supporting services. The definition of ES provided a common means for environmental research to assess the impact of the processes under study on human well-being. The use of ES by the MA has led to their use in many subsequent studies, including the 2015 SWSR report (FAO and ITPS, 2015). The background on the adoption of the ES concept is presented in Section 1.2 of that report.

In 2018, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) introduced the concept of nature's contributions to people (NCP), which was designed to address some of the perceived shortcomings to the ES concept (Díaz *et al.*, 2018). Nature's contributions to people are defined as:

> *The contributions, both positive and negative, of living nature (diversity of organisms, ecosystems, and their associated ecological and evolutionary processes) to people's quality of life. Beneficial contributions include, for example, food provision, water purification, and artistic inspiration, whereas detrimental contributions include disease transmission and predation that damage people or their assets. Many NCP may be perceived as benefits or detriments depending on the cultural, socioeconomic, temporal, or spatial context* (Díaz *et al.*, 2018, pp. 270–271).

The NCP framework was used in a series of papers concerning soil science which are summarized by Smith *et al.* (2021). However, the adoption of NCP in the soil science literature has been uneven and therefore, the ES framework is primarily used throughout this report.

Most of the research conducted by soil scientists focuses on specific physical, chemical and biological processes that occur in the soil. Soil functions are defined for this report as bundles of soil processes that, in whole or in part, deliver ecosystem goods and services (Bünemann *et al.*, 2018). The soil processes included in functions are a subset of all possible soil processes, as there are soil processes (such as clay illuviation in the soil profile) that operate over millennia and are not necessarily involved in the delivery of ecosystem goods and services at timescales that are relevant to human societies. The functions of greatest importance to the delivery of ecosystem goods and services are summarized in Table 2.1.

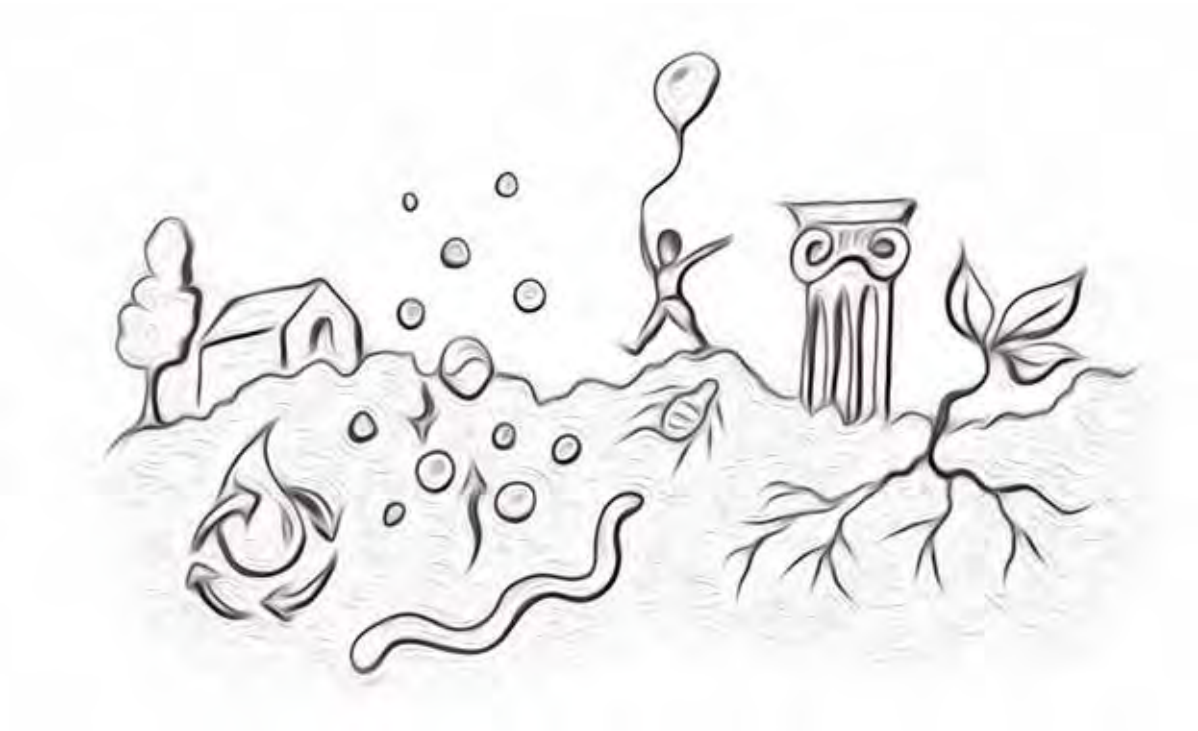
Table 2.1. Soil functions that support the delivery of specific ecosystem goods and services/nature's contributions to people

Material contributions	
Biomass production for provision of food, feed (forage and fodder), fibre and energy	
Ecosystem goods and services/nature's contributions to people	Soil functions.
Water cycling and root growth	Creation and maintenance of soil structure (aggregates and horizons) for gas and water flow, seed germination and root growth. Retention and release of water for plant growth.
Nutrient cycling	Weathering of minerals and release of nutrients. Transformation and accumulation of organic matter. Transformation of organic materials by soil organisms. Retention and release of nutrients on charged surfaces of colloids.
Protection of humans and the environment	
Water supply regulation	Regulation of water infiltration into soil and water flow within the soil. Drainage of excess water out of soil and into groundwater and surface water.
Water, air and soil quality regulation	Filtering and buffering of substances in soil water. Source of particulate matter. Transformation of contaminants.
Climate regulation	Regulation of carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4) emissions. Storage of soil organic carbon (SOC) and carbonates.
Erosion regulation	Maintenance of good soil structure for water infiltration and soil detachment resistance.
Biodiversity reservoir	
Habitat for soil microorganisms, animals, birds, etc.	Providing habitat for soil microorganisms, animals, birds, etc.
Medicinal, biochemical and genetic resources	Preservation of unique biological materials. Production of materials derived from organisms (plants, animals, fungi, and microbes) used for medicinal, veterinary and pharmacological purposes.
Raw earth material supply	
Materials for construction and soil amendments	Provision of topsoil, clay materials, aggregates, peat, etc.
Non-material contributions	
Physical basis of human activities	Maintenance of "soil-as-is".
Preservation of geogenic and archaeological heritage	Maintenance of "soil-as-is".
Aesthetic and spiritual values Learning and inspiration, physical and psychological experiences and supporting identities	Maintenance of "soil-as-is".

◊ *Source:* Adapted from **FAO & ITPS**. 2015. *Status of the world's soil resources: main report.* Rome. https://www.fao.org/documents/card/en/c/c6814873-efc3-41db-b7d3-2081a10ede50/

This report follows the same framework as the 2015 SWSR report, which is built on soil functions and their contributions to ecosystem goods and services, and the threats to those soil functions. The ES concept is explicit in Principle 5 of the *Revised World Soil Charter* (FAO, 2015):

> *Soil management is sustainable if the supporting, provisioning, regulating, and cultural services provided by soil are maintained or enhanced without significantly impairing either the soil functions that enable those services or biodiversity. The balance between the supporting and provisioning services for plant production and the regulating services the soil provides for water quality and availability and for atmospheric greenhouse gas composition is a particular concern* (FAO, 2015, p. 4).



2.3 Links between soils and specific Sustainable Development Goals

2.3.1 Food security and sustainable agriculture (SDG 2: Zero Hunger)



Target 2.4 is of greatest relevance for soil, which states:

> *By 2030, ensure sustainable food production systems and implement resilient agricultural practices that increase productivity and production, that help maintain ecosystems, that strengthen capacity for adaptation to climate change, extreme weather, drought, flooding and other disasters and that progressively improve land and soil quality* (UN, 2022, p. 2).

The specific indicator linked to this target is Indicator 2.4.1, which is the proportion of agricultural area under productive and sustainable agriculture. Sustainable soil management is an integral part of sustainable agriculture and is the primary focus of the work of the ITPS and FAO's GSP.

Soil functions play a key role in sustainable food production systems – as well as biomass production in general – by providing the essential ecological services of water and nutrient cycling (Table 2.1) (FAO, 2022). These are major contributors to plant growth (including the growth of crops in agricultural systems).

Ninety-four percent of the global average daily protein intake by humans is from the soil (58 percent directly from plants and 36 percent indirectly from animals raised on soil-grown foods) (FAO, 2024). The contribution of soil to daily food energy intake is even greater, with 98.8 percent of the daily average food energy intake being from soil-grown foods (83 percent from plants and 15.8 percent from animal products) (Our World in Data, 2024).

The solid soil phase (such as soil minerals and soil organic matter [SOM]) (Figure 2.2) is the main reservoir of essential nutrients. The soil supplies 14 mineral elements required for the nutrition of higher plants, which are taken up by plants, primarily in mineral form from the soil solution (FAO, 2022a). Soil-derived nutrients which are required in large amounts by plants, especially nitrogen (N), phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg) and sulphur (S) are classified as major nutrients (FAO, 2022a) (Table 2.2). Elements such as iron (Fe), manganese (Mn), and zinc (Zn) which are required in lesser quantities, are classified as micronutrients.

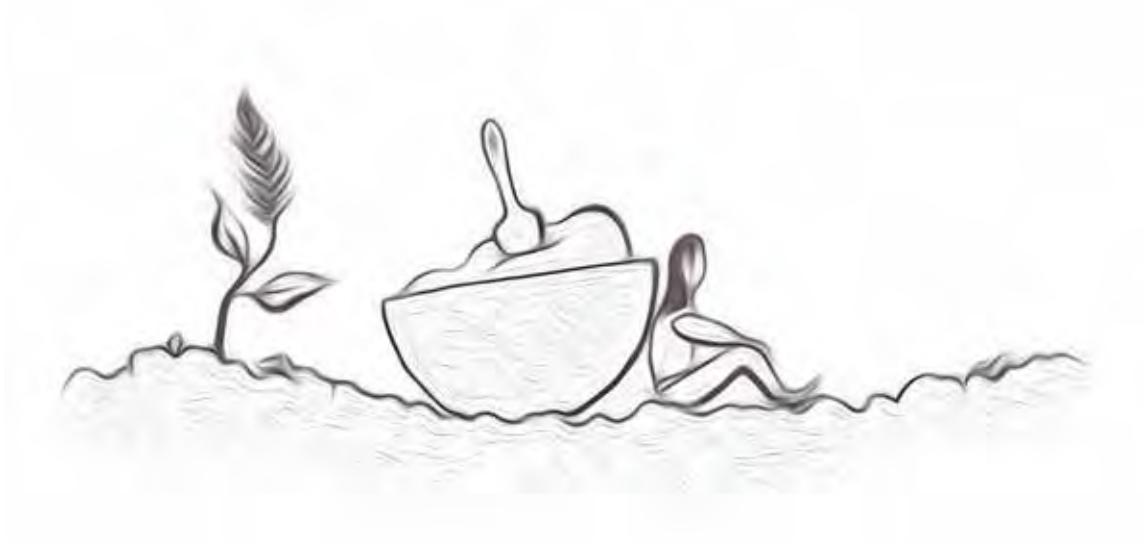
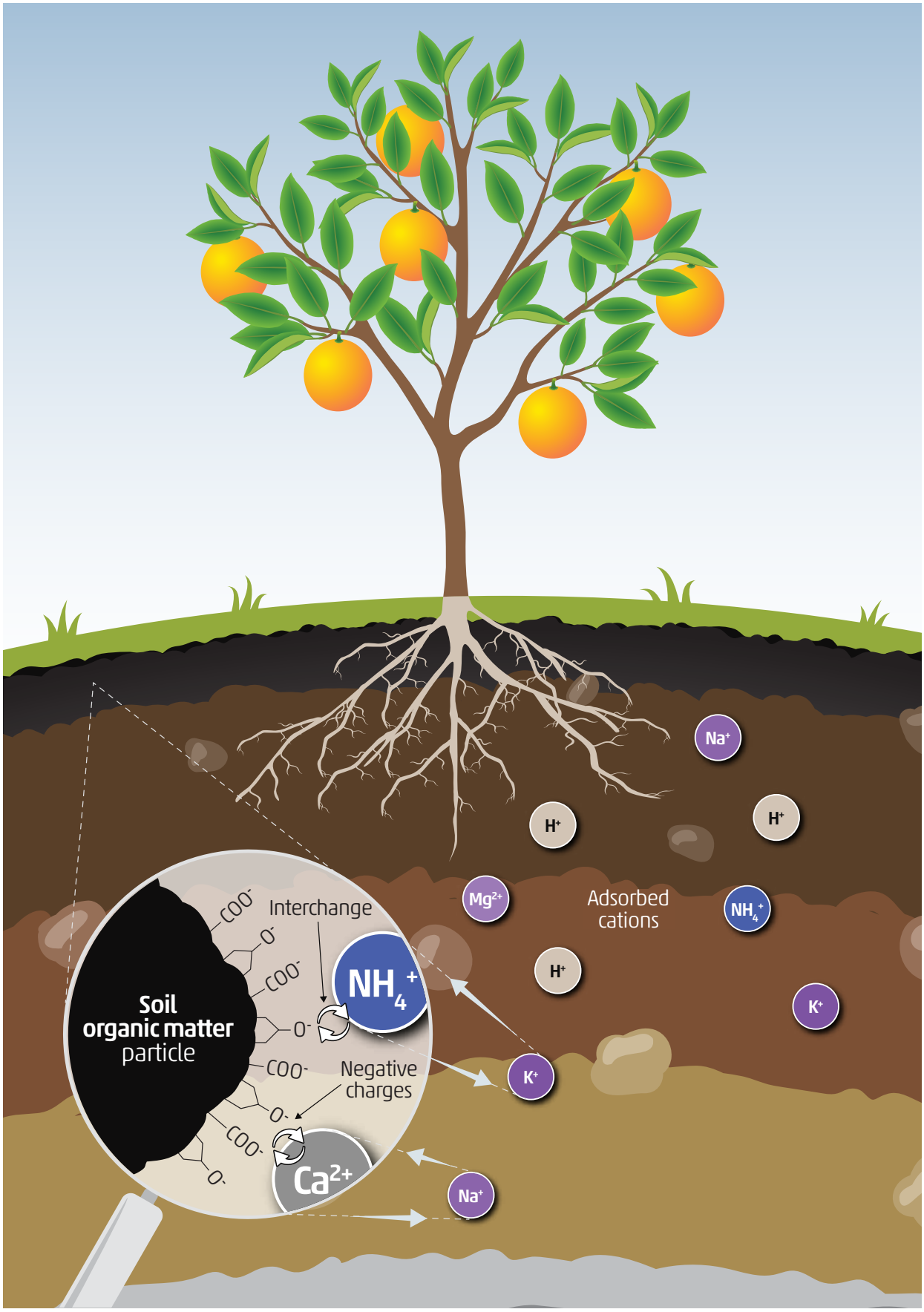


Figure 2.2. Soil solid phase (mineral and soil organic matter) is the major source of nutrients for plant growth



Source: **FAO.** 2022. *Soils for nutrition: state of the art*. Rome. https://doi.org/10.4060/cc0900en

Table 2.2. Essential nutrients used by plants and their concentration in plant tissue

Major nutrients			
Element	**Symbol**	**Primary forms used by plants**	**Plant content (%, range percentage)**
Nitrogen	N	NH_4^+, NO_3^-	0.5–5
Phosphorus	P	HPO_4^{2-}, $H_2PO_4^-$	0.1–5
Potassium	K	K^+	0.5–5
Calcium	Ca	Ca^{2+}	0.05–5
Magnesium	Mg	Mg^{2+}	0.1–1
Sulphur	S	SO_4^{2-}	0.05–0.5
Micronutrients			
			ppm
Iron	Fe	Fe^{3+}, Fe^{2+}	50–1 000
Manganese	Mn	Mn^{4+}, Mn^{2+}	20–200
Zinc	Zn	Zn^{2+}	10–100
Copper	Cu	Cu^{2+}	2–20
Boron	B	H_3BO_3 (boric acid)	2–100
Molybdenum	Mo	Mo, MoO_4^{2-}	0.1–10
Chlorine	Cl	Cl^-	100–10 000

◊ *Source:* **FAO.** 2022. *Soils for nutrition: state of the art.* Rome. https://doi.org/10.4060/cc0900en

The elements are made available through the mineralization of SOM, biological cycles and chemical and physical processes, or can be added as fertilizers. The role of soil organisms is critical to transform organic materials into plant-available forms and to increase availability of some major nutrients (e.g. N from biological fixation and P from mycorrhizal interactions).

Nutrient budgets are an essential tool to ensure the sustainable use of nutrients (FAO, 2022a) (Figure 2.3). Ideally, inputs of nutrients should be roughly in balance with outputs of nutrients in crops and animal products. If outputs are greater than inputs, the mining of soil nutrient reserves will occur, and plant yields will drop below the maximum levels possible in a given location. If inputs are greater than outputs, the surplus nutrients can cause harm to air and water quality.

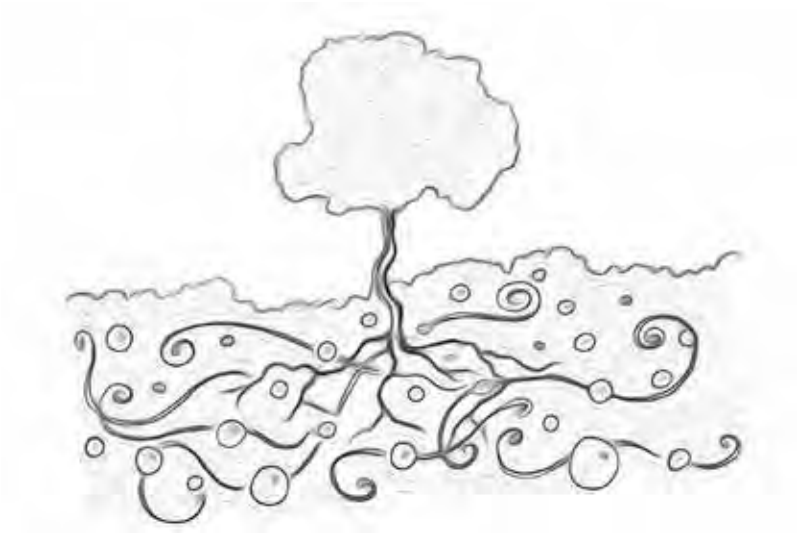
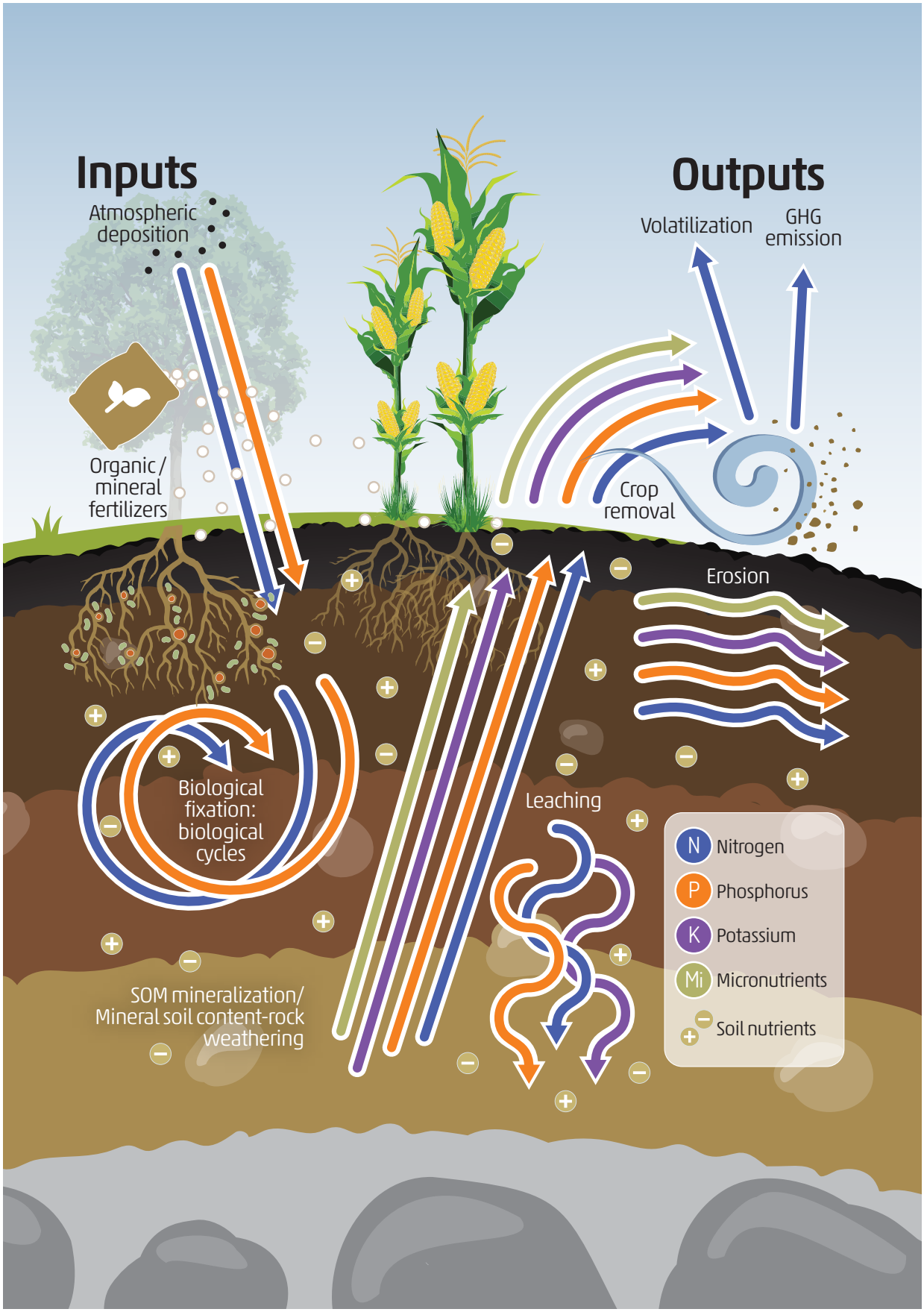


Figure 2.3. A schematic of nutrient budgets for a system



Note: The left side of the figure shows the inputs and the right side shows the outputs. The arrows indicate inputs and output fluxes.

Source: **FAO.** 2022. *Soils for nutrition: state of the art*. Rome. https://doi.org/10.4060/cc0900en

The challenges that are faced in nutrient management have been highlighted by Kopittke *et al.* (2021) and Rockström *et al.* (2023), through their work on safe boundaries for maintaining and enhancing the stability and resilience of the Earth system over time, thus safeguarding its functions and ability to support humans and all other living organisms. Both articles stress that currently, both N and P inputs are considerably over the global Earth system boundaries laid out in the report, and hence, a substantial reduction in N and P additions needs to occur. At the same time, however, regions like sub-Saharan Africa and in the Brazilian Cerrado biome require higher levels of N and P inputs to increase food security. This problem of global surplus due to excess N and P applications in some regions and regional deficits in other regions is the overarching challenge in the global management of nutrient inputs.

2.3.2 Water security and resources (SDG 6: Clean Water and Sanitation)



Two SDG 6 targets are relevant to soil. The first is Target 6.4: By 2030, substantially increase water-use efficiency across all sectors and ensure sustainable withdrawals and supply of freshwater to address water scarcity and substantially reduce the number of people suffering from water scarcity.

The indicators for Target 6.4 are: Indicator 6.4.1: Change in water-use efficiency over time, and Indicator 6.4.2: Level of water stress: freshwater withdrawal as a proportion of available freshwater resources.

Water scarcity is a growing threat to global well-being. Mekonnen and Hoekstra (2016) estimated that 4 billion people, or two-thirds of the world population, experienced water scarcity during at least part of the year. One approach to address the scarcity issue would be to improve the efficiency with which water is used in human activities, which is the focus of Target 6.4.

The greatest portion of freshwater-use is agricultural, with the soil water reservoir being a critical water source for crop production, as shown by Mialyk *et al.* (2024). Soils play an essential role in the regulation and provision of water for crop growth and for human use in general. Keesstra *et al.* (2021) distinguished between green water (the plant-available water in the soil) and blue water (the water in rivers, lakes and ponds, and groundwater available to be pumped to the surface).

In 2019, an estimated 5 912 billion m^3 of green water was used in crop production, accounting for 87 percent of total water used in crop production. Blue water-use for crop production in 2019 was estimated at 977 billion m^3 (or 12.6 percent of total water used in crop production). Net irrigation (including blue water withdrawals, plus water volumes remaining in the soil, returned to blue water, or lost during conveyance) was estimated at 977 billion m^3, or 14.3 percent of total water-use (Keesstra *et al.*, 2021).

The second soil-relevant target is Target 6.6: By 2020, protect and restore water-related ecosystems, including mountains, forests, wetlands, rivers, aquifers and lakes. The key indicator is Indicator 6.6.1: Change in the extent of water-related ecosystems over time.

Of the water-related ecosystems referenced in Indicator 6.6, wetlands are of particular interest in soil science due to their high SOC storage and high biodiversity (Ramsar Convention on Wetlands, 2018). As demonstrated by Fluet-Chouinard *et al.* (2023), the major driver of the loss of wetlands through time was their conversion to cropland, with the global area of natural wetlands declining by an estimated 3.4 million km^2 since 1700. Wetland drainage for upland croplands was the most common cause of loss of natural wetlands (2.0 million km^2 or 61.7 percent of total loss), followed by conversion to flooded rice (*Oryza sativa*) (18.2 percent), urban areas (8 percent), forestry (4.7 percent), non-rice wetland cultivation (4.3 percent), pasture (2 percent) and peat extraction (0.9 percent). The drivers of wetland drainage are discussed in greater detail in Section 4.2.1 of this report.

Conversion to agriculture has also been a major driver of the loss of mangrove forests. Brazil, Indonesia, Malaysia and Papua New Guinea contain more than 50 percent of the global mangrove carbon (C) stock, with Indonesia alone accounting for 30 percent of the global mangrove C stock (Hamilton and Friess, 2018). Mangrove deforestation is associated with five major drivers: conversion to aquaculture or agriculture, coastal erosion, extreme climate events, mangrove clearing due to logging and other activities, and human settlements (Adame *et al.*, 2021).

Peatlands are SOC-rich and are considered a hotspot in many UN documents (UNEP, 2024). The two largest peatlands in the tropical zone are in the Congo Basin (Dargie *et al.*, 2019) and in Indonesia (Warren *et al.*, 2017). These peatlands, as well as those in Brazil, Malaysia and Peru, are juxtaposed with densely populated urban areas and are subject to multiple land use pressures, including agriculture (Turetsky *et al.,* 2015). The same factors are threatening the flooded forests, rivers, lakes and other wetlands that cover about 14 percent of the Amazon Basin (Amazon Waters Alliance, n.d.). Peatlands in boreal forest and northern regions face lower pressures from conversion to agriculture than tropical and subtropical peatlands (UNEP, 2024).

Agriculture poses a threat to the quality of blue water through pollution by agrochemicals and sediment eroded from agricultural fields (Keesstra *et al.*, 2021). Agriculturally derived pollution occurs when nutrients and pesticides dissolved in surface runoff water are bound to eroded soil carried in the runoff water or dissolved in groundwater that discharges to surface waterbodies. Such pollution is termed non-point source pollution as it derives from many sources, but its cumulative impact is huge (FAO and UNEP, 2021a).

2.3.3 Anthropogenic climate change (SDG 13: Climate Action)



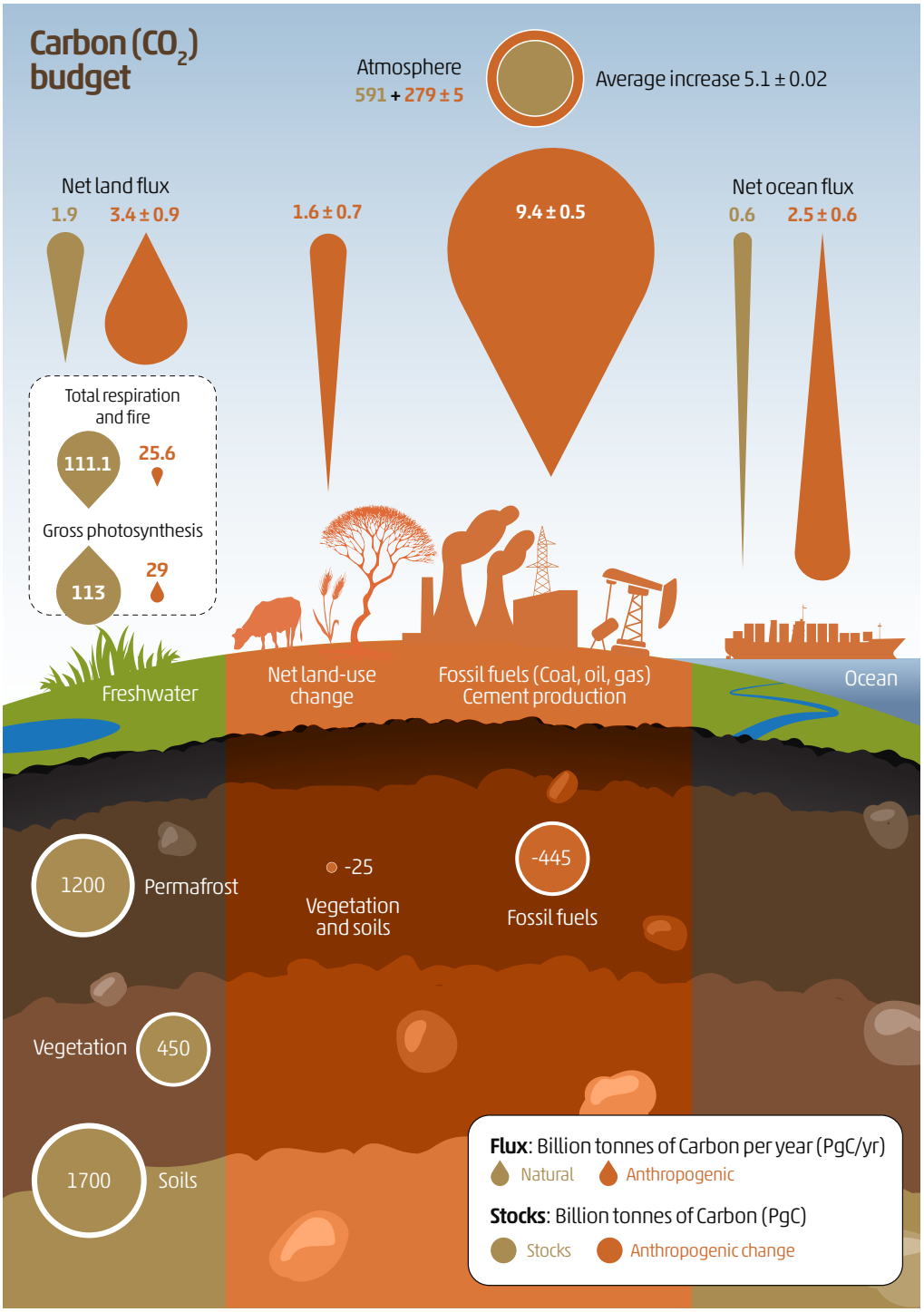
Of particular relevance to soil is Target 13.2: Integrate climate change measures into national policies, strategies and planning. The specific indicator is Indicator 13.2.2: Total greenhouse gas (GHG) emissions per year (UN, 2022).

Soils are both a major source of GHG emissions to the atmosphere and a potential sink for GHGs removed from the atmosphere, as both are important for total GHG emissions per year (Canadell *et al.*, 2021). National reporting on GHGs sinks is required under articles of the United Nations Framework Convention on Climate Change (UNFCCC), the Kyoto Protocol and the Paris Agreement (FAO, 2017).

The three GHG emissions of relevance to soils are carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). For the period 1960–2019, the relative contribution to the change in the energy balance in the Earth's atmosphere (i.e. total effective radiative forcing) was 63 percent for CO_2, 11 percent for CH_4 and 6 percent for N_2O, with the remaining 17 percent due to halogenated species unconnected to soils (Canadell *et al.,* 2021). The effect of the three gases on radiative forcing differs due to their atmospheric chemistry and lifetime in the atmosphere. For the purposes of GHG emissions, a unit of N_2O is judged to have an equivalent global warming potential to 273 units of CO_2 and CH_4: an equivalent of 27 units of CO_2 (Canadell *et al.*, 2021).

Soil is the largest land pool, at approximately 1 700 billion tonnes of C (Figure 2.4). Respiration releases C from the soil pool as CO_2, while photosynthesis by plants withdraws C from the atmospheric pool. Including CO_2 release from fires, the net land balance in the 2010–2019 period between outputs of CO_2 from land and uptake of CO_2 by plants is a net increase in the land C pool of approximately 3.4 billion tonnes per year (Canadell *et al.*, 2021).

Figure 2.4. Global carbon budget



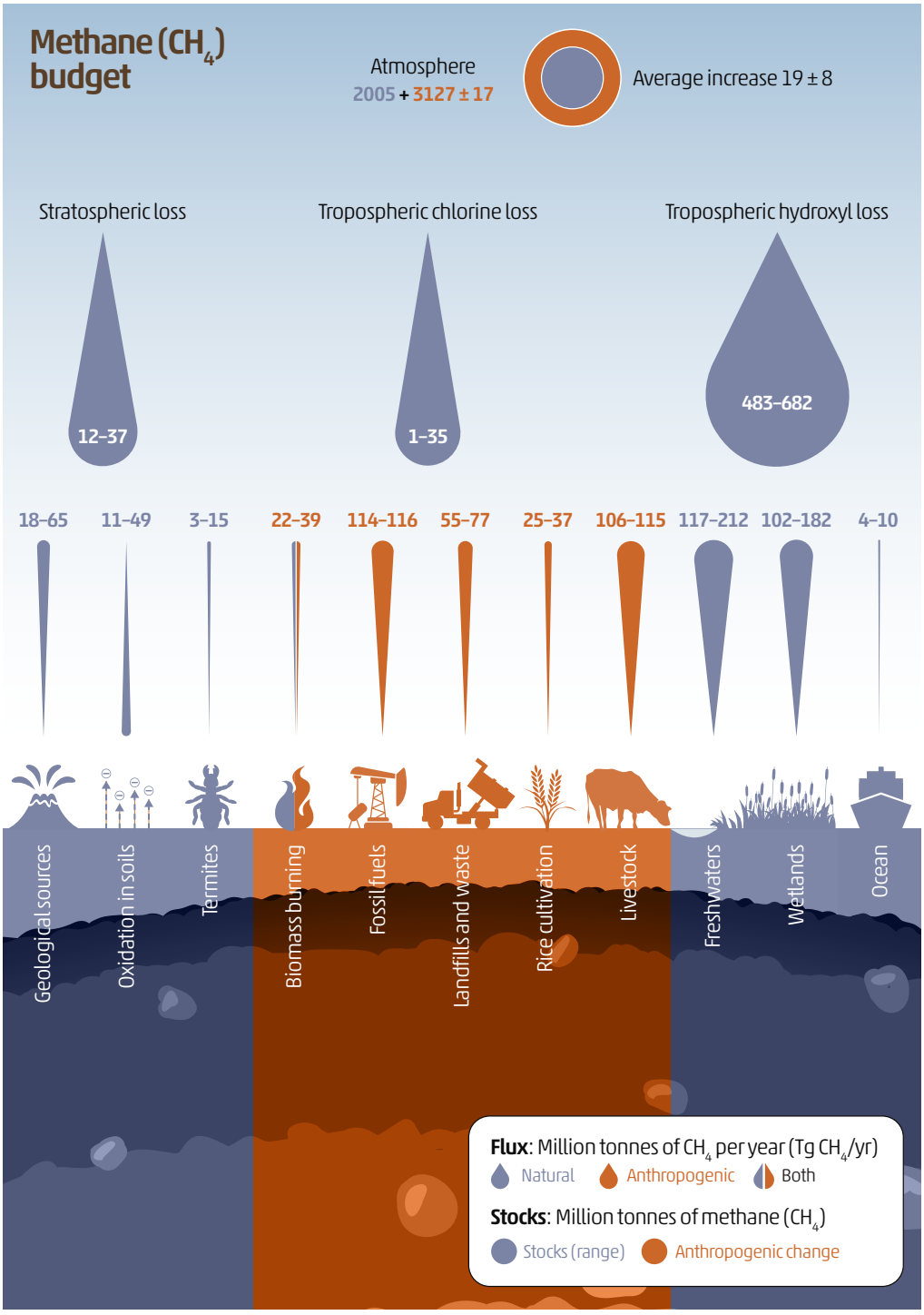
Note: Brown arrows represent annual carbon flows in billion tonnes of carbon. Dark orange arrows represent annual anthropogenic fluxes in billion tonnes of carbon averaged over the time period 2010–2019. Numbers in brown circles represent pre-industrial carbon stocks in billion tonnes of carbon. Numbers in dark orange circles represent cumulative anthropogenic changes to these stocks since 1750.

Source: Adapted from **Canadell, J.G., Monteiro, P.M.S., Costa, M.H., Cotrim da Cunha, L., Cox, P.M., Eliseev, A.V., Henson, S.** *et al.* 2021. Global Carbon and Other Biogeochemical Cycles and Feedbacks. In: V. Masson-Delmotte, P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud *et al.*, eds. Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change, p. 144. Cambridge, United Kingdom, Cambridge University Press. doi:10.1017/9781009157896.007

There is also a net release of C from land use change (Figure 2.4) of approximately 1.6 billion tonnes per year (Canadell *et al.*, 2021). The historical release of C from the soil pool has been immense, as shown by Sanderman, Hengl and Fiske (2017), who estimated a total loss of C due to agriculture of 133 billion tonnes for the top 2 m of soil since the initial adoption of agriculture. These losses are discussed in greater detail in Section 3.3.

Methane emissions from soils are principally associated with oxygen-poor conditions in the soil profile. Emissions of CH_4 from wetlands soils are a major natural source of emissions (Figure 2.5) and non-wetland soils also consume atmospheric CH_4 through oxidation (Canadell *et al.*, 2021). The major anthropogenic emissions of CH_4 related to soils are through rice production and from livestock production, the second depending largely on management of pastures in extensive and intensive livestock systems (FAO, 2023).

Figure 2.5. Global methane budget for the 2008–2017 period



Note: Grey arrows represent natural annual methane flux in million tonnes of methane. Dark orange arrows represent annual anthropogenic fluxes in million tonnes of methane averaged over the time period 2008–2017.

◊ *Source:* Adapted from **Canadell, J.G., Monteiro, P.M.S., Costa, M.H., Cotrim da Cunha, L., Cox, P.M., Eliseev, A.V., Henson, S.** *et al.* 2021. Global Carbon and Other Biogeochemical Cycles and Feedbacks. In: V. Masson-Delmotte, P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud *et al.*, eds. Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change, p. 144. Cambridge, United Kingdom, Cambridge University Press. doi:10.1017/9781009157896.007

Nitrous oxide differs from the other two GHGs in so far as agriculture is the dominant source of anthropogenic emissions from the land surface (Figure 2.6). These emissions also occur naturally from soils, but anthropogenic emissions are almost as large as those from natural soils. Tian *et al.* (2019) used a suite of process-based terrestrial biosphere models to estimate the contribution of different sources of the increase in global N_2O levels. Overall emissions increased from 6.3 ± 1.1 million tonnes N_2O-N per year in the preindustrial period (the 1860s) to 10.0 ± 2.0 million tonnes N_2O-N per year in the 2007–2016 period. Increases in cropland emissions accounted for 82 percent of the total increase. It was estimated that in the 2007–2016 period, N fertilizer application was responsible for 54 percent of the increase, followed by N deposition from the atmosphere (26 percent), climate change (24 percent) and manure N application (15 percent). Cui *et al.* (2021) calculated crop-specific emissions factors of 1.02 percent for maize *(Zea mays*), 0.58 percent for wheat (*Triticum*), 0.52 percent for rice and 1.20 percent for other crops, which were broadly in agreement with the Intergovernmental Panel on Climate Change (IPCC) default values of 1 percent, calculated using an emission factor (i.e. the percentage of applied N emitted directly as N_2O-N).

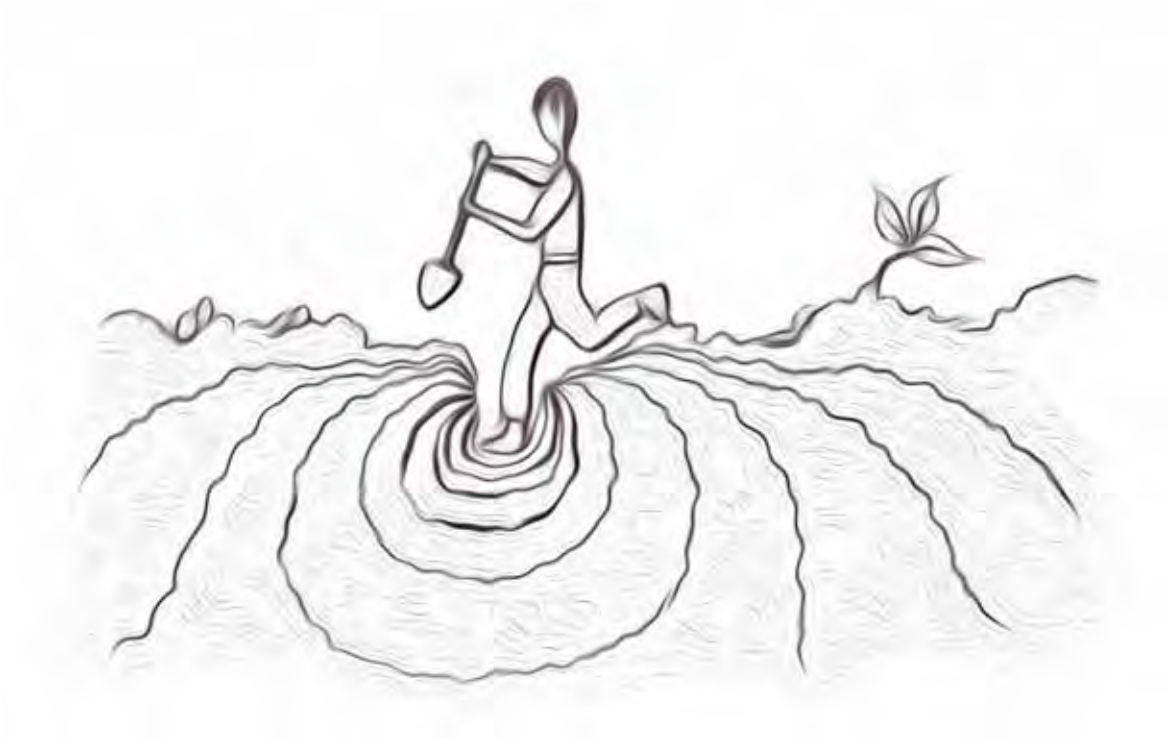
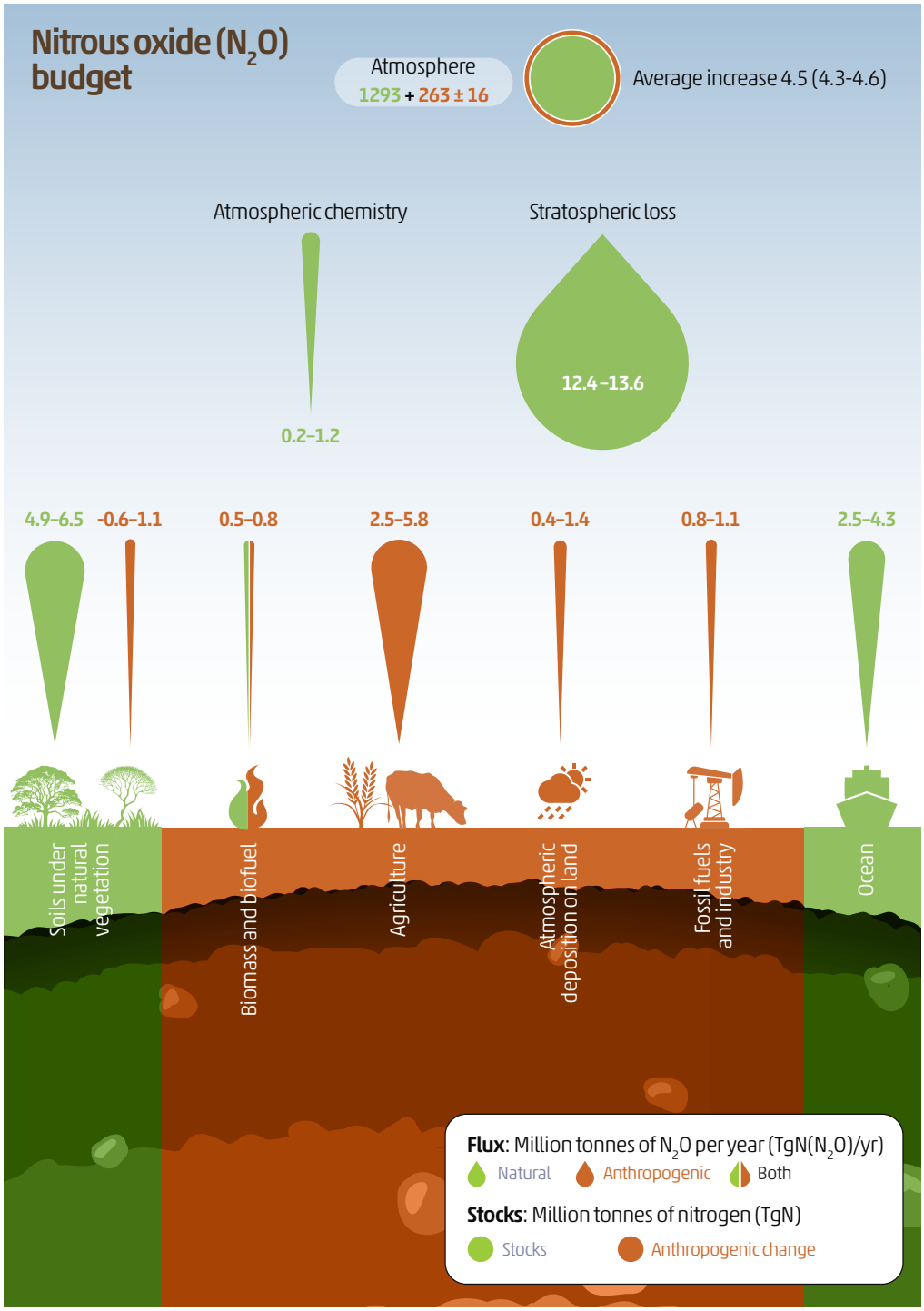


Figure 2.6. Global nitrous oxide budget for the period 2007–2016



Note: Green arrows represent annual nitrous oxide flux in million tonnes of nitrous oxide. Dark orange arrows represent annual anthropogenic fluxes in million tonnes of nitrous oxide averaged over the time period 2007–2016. Circles with green numbers represent pre-industrial nitrous oxide stocks in million tonnes of nitrous oxide. Circles with dark orange numbers represent cumulative anthropogenic changes to these stocks since 1750.

Source: Adapted from **Canadell, J.G., Monteiro, P.M.S., Costa, M.H., Cotrim da Cunha, L., Cox, P.M., Eliseev, A.V., Henson, S.** *et al.* 2021. Global Carbon and Other Biogeochemical Cycles and Feedbacks. In: V. Masson-Delmotte, P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud *et al.*, eds. Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change. p. 144. Cambridge, United Kingdom, Cambridge University Press. doi:10.1017/9781009157896.007

As soil management practices can reduce emissions of CO_2, CH_4 and N_2O from the soil and, in some cases, withdraw C and N from the atmospheric pool, quantifying and clarifying the contribution of soils to GHG dynamics have been of increasing interest as a focus of research in soil science. The specific impact of various threats to soils related to GHG emissions is examined in Section 3.3

and Section 3.5, while the contribution of specific management practices to these total mitigation potentials is examined in Section 5.5.

2.3.4 Soil pollution and human and ecosystem health

According to the *Global Assessment of Soil Pollution* (FAO and UNEP, 2021a, 2021b), soil pollution hinders the achievement of many of the SDGs, including those related to poverty elimination, especially SDG 1: No Poverty, SDG 2: Zero Hunger and SDG 3: Good Health and Well-being. Soil pollution hits the most vulnerable hardest, especially children and women (SDG 5: Gender Equality). The supply of safe drinking water is threatened by the leaching of contaminants into groundwater and runoff (SDG 6: Clean Water and Sanitation). CO_2 and N_2O emissions from unsustainably managed soils accelerate climate change (SDG 13: Climate Action). Soil pollution contributes to land degradation and the loss of terrestrial and aquatic biodiversity (SDG 15: Life on Land and SDG 14: Life below Water respectively), and decreased security and resilience of cities (SDG 11: Sustainable Cities and Communities), among others.

Soils perform a range of specific functions related to pollution (Sarker *et al.*, 2024) and can serve as a source of contaminants. Weathering of a soil's parent material can be a source of contaminants, releasing potentially harmful trace elements such as cadmium (Cd) (Sarker *et al.*, 2024). Human alterations of the soil chemical environment (such as reducing SOM content or altering the acidity of the soils) can also make these elements more available to plants or more mobile in groundwater.

In addition to the natural sources of potentially toxic elements, contaminants are added to the soil through human activities such as mining, dumping of wastes, deposition of atmospheric contaminants, addition of organic material and agrochemicals, irrigation with contaminated water and spills of toxic chemicals. Some soil pollution is due to accidental events at specific points, whereas agricultural sources (such as agrochemicals and contaminated irrigation water) can be widespread and are termed non-point sources.

Soil functions play a wide range of roles in minimizing the potentially harmful impacts of contaminants that are added to soil (Sarker *et al.*, 2024). First, soils can retain contaminants on charged surfaces on soil colloids such as minerals and SOM, which prevents the movement of the contaminants into plants, the atmosphere or surface water and groundwater. Second, the mobility of some potential contaminants may be slowed or stopped in response to changes in the soil chemical environment (such as through acidity levels or the availability of oxygen). Third, soil functions that are primarily related to the soil microbial community can transform some contaminants into less toxic forms through functions such as biodegradation, biotransformation and biomineralization (Song *et al.*, 2019). Biological processes are important "nature-based" solutions to soil pollution and are examined in Section 5.6 of this report.

While the links between potentially toxic elements (Table 2.3) and human health have been well-established (Figure 2.7), other threats to soil functions may also directly impact human health. For example, Froude and Petley (2018) documented that landslides were responsible for thousands of human fatalities between 2004 and 2016, with the toll likely to increase with changes in global climatic patterns. Emissions of ammonia (NH_3) from soils are associated with presence of fine particulate matter (particulate matter with an aerodynamic diameter smaller than 2.5 µm) in the atmosphere, which has been linked to many health impacts, including acute lower respiratory infections, chronic obstructive pulmonary disease (COPD) and lung cancer (Pozzer *et al.*, 2017). Dust particles in the atmosphere due to wind erosion have also been linked to a number of respiratory health issues (Middleton, Tozer and Tozer, 2019).

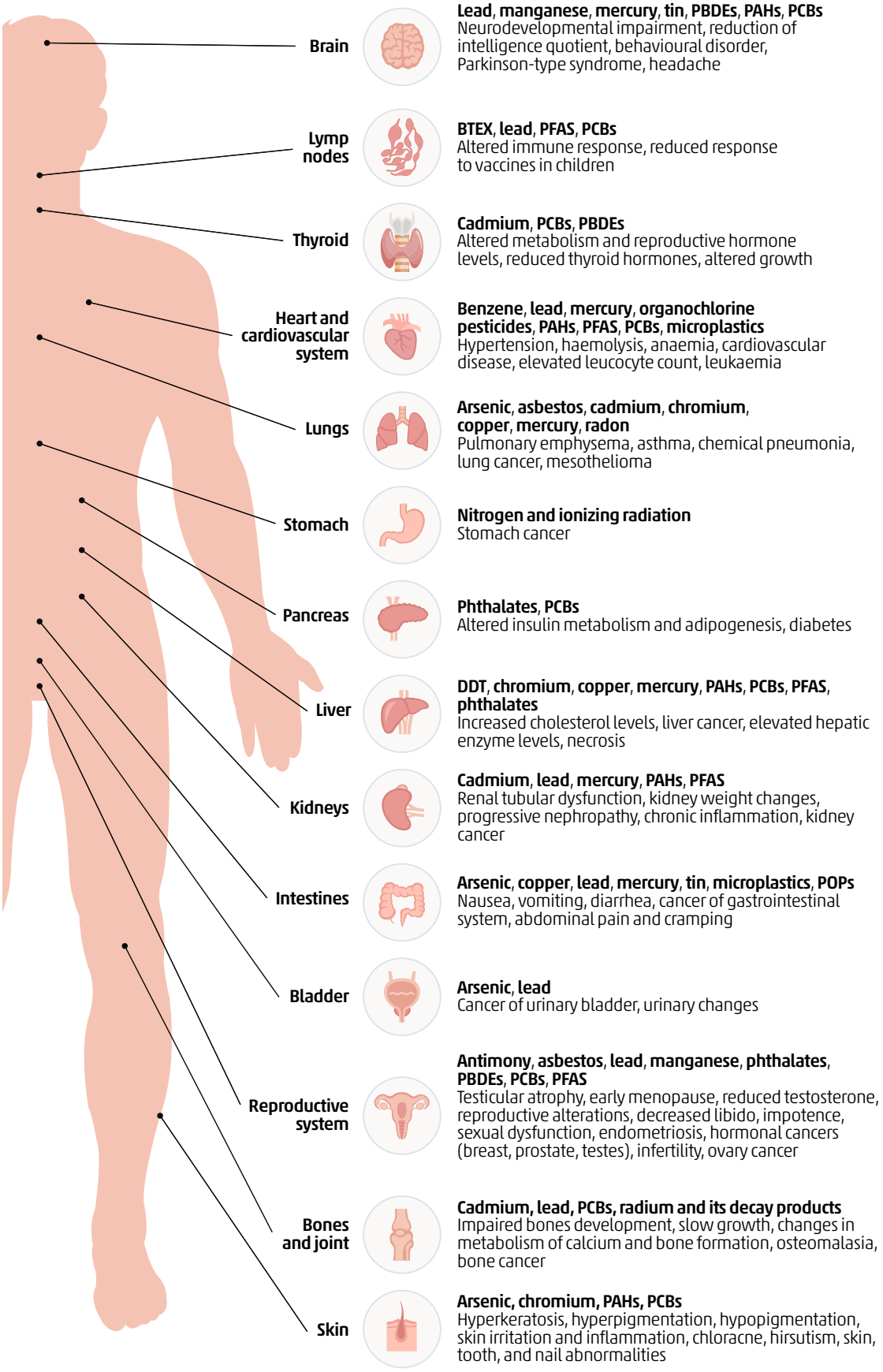
Table 2.3. List of elements and symbols

Element	Symbol	Main effects
Manganese	Mn	Readily taken up by plants but low risk to humans
Zinc	Zn	Readily taken up by plants but low risk to humans
Copper	Cu	Readily taken up by plants but low risk to humans
Nickel	Ni	Readily taken up by plants but low risk to humans
Cadmium	Cd	Significant risks to human and animal health, most toxic element (with Pb) for humans
Lead	Pb	Significant risks to human and animal health; most toxic element (with Cd) for humans
Chromium	Cr	Low uptake risk by plants
Vanadium	V	Associated with petroleum processing and use
Selenium	Se	High risk for food chain contamination and poses risks to human and animal health
Cobalt	Co	High risk for food chain contamination and poses risks to human and animal health
Zirconium	Zr	Low uptake risk by plants
Rubidium	Rb	Low human health risk
Arsenic	As	High risk for food chain contamination and poses risks to human and animal health
Titanium	Ti	Low uptake risk by plants
Beryllium	Be	Low human health risk
Tungsten	W	Limited environmental impact
Mercury	Hg	Toxic to animals and humans

◊ *Source:* **FAO.** 2018. *Soil pollution: a hidden reality*. Rome. https://openknowledge.fao.org/server/api/core/bitstreams/3f7e6959-db0b-44d3-971e-109bcfe78195/content



Figure 2.7. Main effects of soil contamination on human health, including the organs or systems affected and the contaminants causing them



◊ *Source:* **FAO & UNEP (United Nations Environment Programme).** 2021. Global Assessment of Soil Pollution. Main Report. Rome. https://doi.org/10.4060/cb4894en

2.3.5 Soil biodiversity (SDG 15: Life on Land)

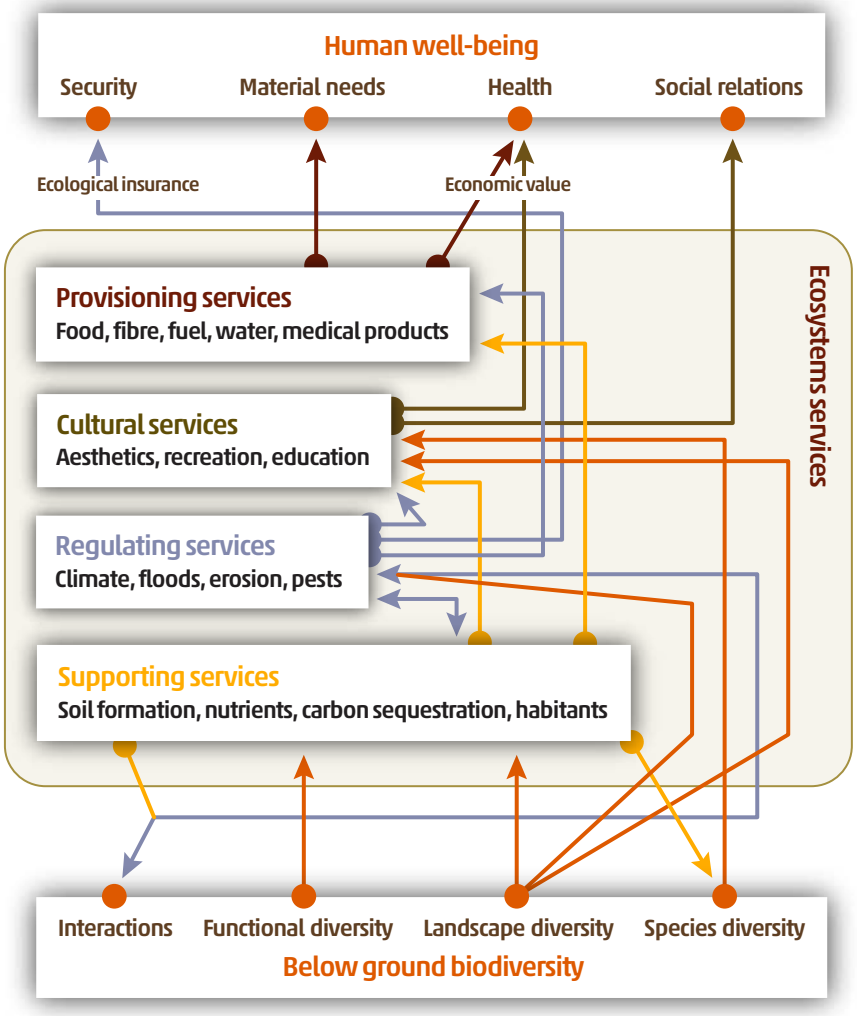
Soil biodiversity is defined as:

> *the variety of life belowground, from genes and species to the communities they form, as well as the ecological complexes to which they contribute and to which they belong, from soil micro-habitats to landscapes* (FAO *et al.*, 2020, p. 7).

Soil is the most biodiverse system on earth, containing an estimated 59 percent of the biodiversity on the planet (Anthony, Bender and Van der Heijden, 2023). This estimate, however, has wide confidence limits of ± 15 percent, indicating how much still remains to learn about the organisms that live in the soil.

The importance of soil biodiversity to a host of soil functions and ES was highlighted in the landmark study on the *State of Knowledge of Soil Biodiversity*, published jointly by FAO, ITPS, the Global Soil Biodiversity Initiative (GSBI), the Secretariat of the Convention on Biological Diversity (SCBD) and the European Commission (EC) in 2020 (FAO *et al.*, 2020). The contribution of soil biodiversity to ecosystem functions and services is summarized in Figure 2.8.

Figure 2.8. Relationship between soil biodiversity and ecosystem services



◊ *Source:* **FAO, ITPS, GSBI, CBD & EC**. 2020. State of knowledge of soil biodiversity - Status, challenges and potentialities. Report 2020. Rome, FAO. https://doi.org/10.4060/cb1928en

Soil biodiversity occupies a unique status in the *Revised World Soil Charter* (FAO, 2015). The preamble of the charter states that:

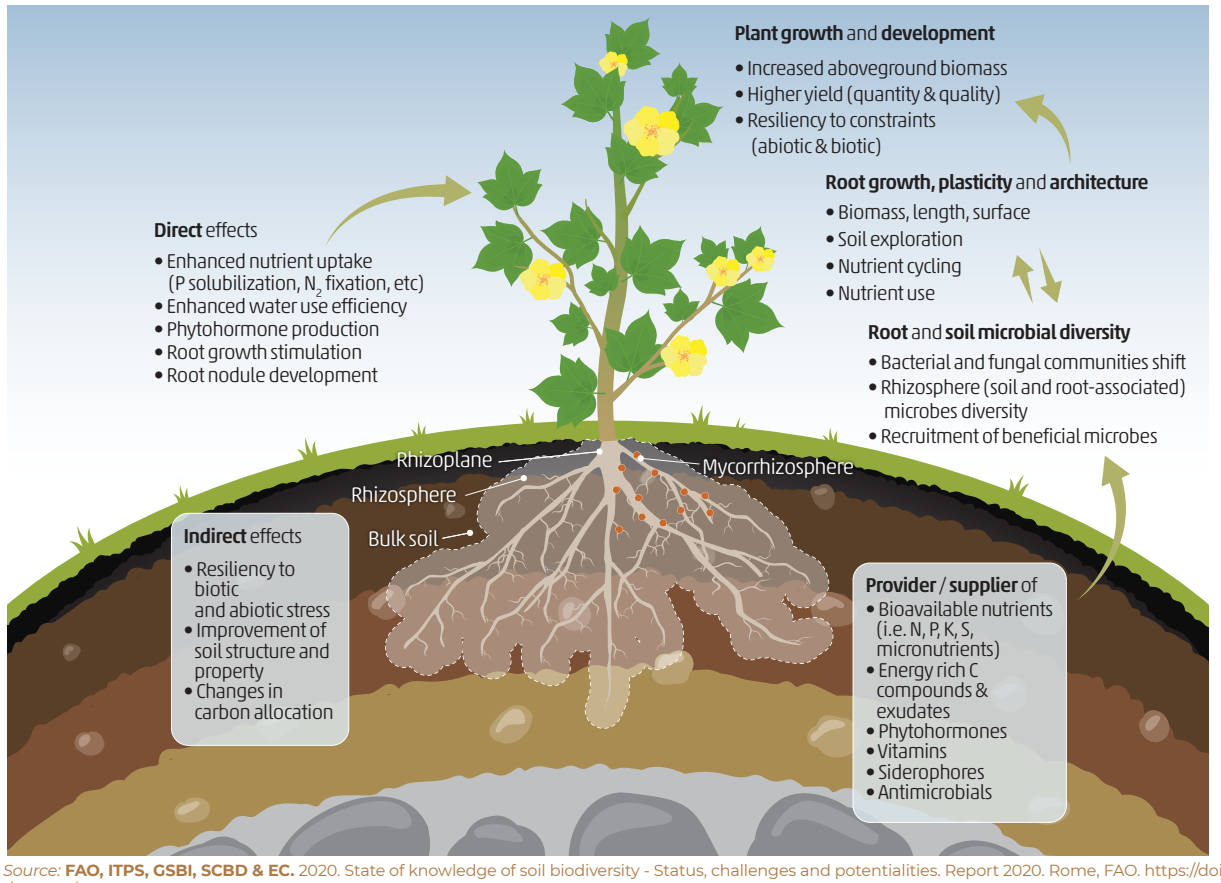
> *Soils are fundamental to life on Earth but human pressures on soil resources are reaching critical limits. Careful soil management is one essential element of sustainable agriculture and provides a valuable lever for climate regulation and a pathway for safeguarding ecosystem services and biodiversity* (FAO, 2015, p. 4).

The word biodiversity was added at the end of the final sentence at the request of delegates at the GSP Plenary Assembly in 2015 when the charter was approved, reflecting a desire that biodiversity should be valued on its own, not simply as a contributor to ES.

There have been major advances in documenting soil biodiversity since 2015, in part because of the advances in technology available to researchers. As documented in Section 3.4 of this report, global maps of the soil microbiome (such as bacteria and fungi) have been produced along with studies on larger organisms such as nematodes (Van den Hoogen *et al.*, 2019, 2020) and earthworms (Phillips *et al.*, 2019). These studies built upon the tremendous foundation provided by the *Global Soil Biodiversity Atlas* (Orgiazzi *et al.*, 2016).

The actions of soil organisms vividly illustrate the concept of nature's contributions to people. Although the roles of soil organisms generally have positive consequences for people and the environment (Figure 2.9), soil organisms also contain a host of human and plant pathogens (i.e. organisms that can cause disease in humans or plants) (Samaddar *et al.*, 2021). Pathogens include pests, predators, competitors, parasites and other potentially harmful organisms, as well as microbial pathogens.

Figure 2.9. Main elements and processes involved in the microbial consortium concept



◊ *Source:* **FAO, ITPS, GSBI, SCBD & EC.** 2020. State of knowledge of soil biodiversity - Status, challenges and potentialities. Report 2020. Rome, FAO. https://doi.org/10.4060/cb1928en

The communities of organisms in the soil play a critical role in determining whether a pathogen can colonize a soil and infect a potential host (Samaddar *et al.*, 2021). In some cases, the entire microbial community can exert competitive and antagonistic pressures on the pathogen, suppressing its effect. Other organisms such as arthropods like Collembola and mites, protists and other taxa also play important roles in suppressing pathogens. Although many of the specific mechanisms by which suppression occurs remain to be identified, experimental studies have established that reductions in bacterial or fungal biodiversity cause increased vulnerability to invasions by pathogens (Samaddar *et al.*, 2021).

The threats to soil biodiversity are examined in Section 3.4 of this report, and the role of SSM in addressing threats to soil biodiversity is discussed in Section 5.7.

2.3.6 Terrestrial ecosystems (SDG 15: Life on Land)



The objective of SDG 15 is to protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss. The emphasis on SSM in this report is inherently related to sustainable use of terrestrial ecosystems and is the focus of Chapter 5.

The specific indicator of greatest relevance to soil is SDG Target 15.3: By 2030, combat desertification, restore degraded land and soil, including land affected by desertification, drought and floods, and strive to achieve a land degradation neutral world.

Indicator 15.3.1 is the proportion of land that is degraded over the total land area. This indicator is assigned to the United Nations Convention to Combat Desertification (UNCCD) and is linked to the achievement of the UNCCD's commitment to land degradation neutrality (LDN):

> *The LDN programme of the UNCCD strives to achieve the primary objective of SDG Target 15.3, which is '... to achieve a land degradation-neutral world'. LDN is "a state whereby the amount and quality of land resources, necessary to support ecosystem functions and services and enhance food security, remains stable or increases within specified temporal and spatial scales and ecosystems"* (Sims *et al.*, 2021, p. 4).

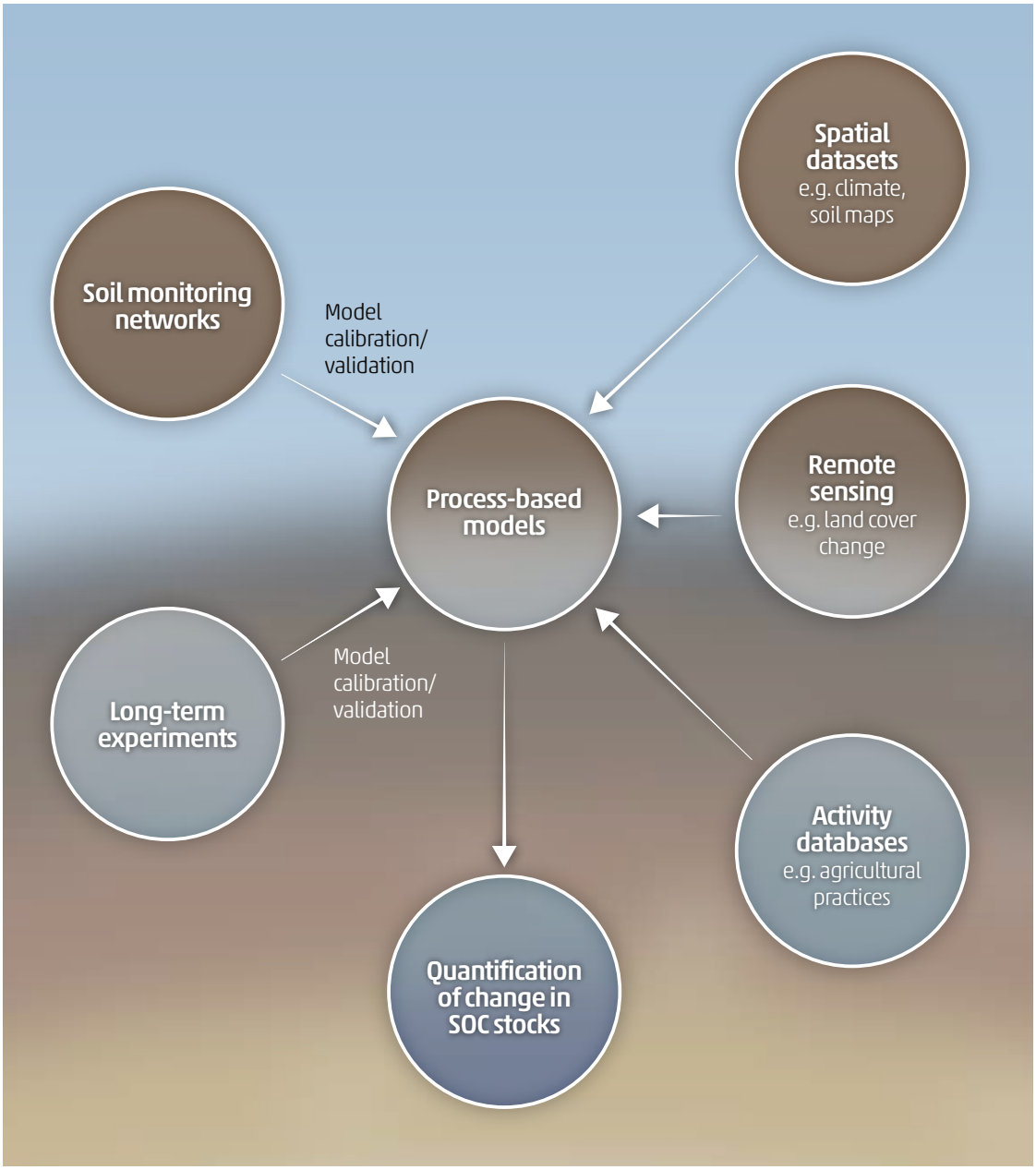
The approach used by UNCCD for Indicator 15.3.1, and as laid out in UNCCD *Good Practice Guidance* (Sims *et al.*, 2021), is to use three subindicators: trends in land cover, trends in land productivity and trends in C stocks (above and below ground), currently represented by SOC stocks. The inclusion of SOC is the only soil indicator explicitly used in the SDGs.

Work is currently underway to develop the global map defining the proportion of land that is degraded over the total land area (FAO, n.d.).

The process that will be used to quantify changes in SOC stocks is to establish a baseline of SOC stocks globally and then quantify C stock changes over time (likely to be ten years) (Sims *et al.*, 2021). The Good Practice Guidance state that it is good practice to identify the key degradation processes relevant to a country and to ensure that they are accounted for in the stock change calculation. These key degradation processes include urban expansion, deforestation and land clearing, overgrazing, unsustainable cropping practices, including tillage and burning or irrigation, and natural events such as intense fires or drought that substantially or persistently reduce vegetation cover and increase soil erosion. The threats associated with changes in C stock are discussed in detail in Section 3.3 of this report.

Sims *et al.* (2021) suggested a Tier 3 approach for estimating changes in SOC, where process-based models are used to quantify changes using inputs from various sources (Figure 2.10) and an arbitrary 10 percent average change in SOC stocks over the assessment period used to distinguish between "degraded" and "not degraded".

Figure 2.10. Schematic of possible national Tier 3 approach for estimating change in soil organic carbon stock



◊ *Source:* Adapted from **Sims, N.C., Newnham, G.J., England, J.R., Guerschman, J., Cox, S.J.D., Roxburgh, S.H., Viscarra Rossel, R.A., Fritz, S. & Wheeler, I.** 2021. Good Practice Guidance. SDG Indicator 5.3.1, Proportion of Land That Is Degraded Over Total Land Area. Version 2.0. Bonn, Germany, UNCCD.

Beyond the role of SOC in SDG 15, all the threats to soil functions covered in this report contribute to soil degradation and hence must be addressed if SDG 15 is to be realized. Salinization and sodification, the loss of soil biodiversity, nutrient mismanagement, soil pollution, soil sealing and soil erosion in all its forms are well established in the literature and the policy realm as major threats to soil functions (Montanarella, Scholes and Brainich, 2018; Olsson *et al.*, 2019; FAO, 2022b; Gibbs and Solomon, 2015; Prăvălie *et al.*, 2021). The threats to soil functions and their impact on ecosystems and societies are discussed in Chapter 3 of this report.

2.3.7 SDG 11: Sustainable Cities and Communities



The objective of SDG 11: Sustainable Cities and Communities is to call for inclusive, safe, resilient and sustainable cities, and covers the spatial aspect of urbanization by Indicator 11.3.1: Ratio of land consumption rate to population growth rate. Marquard *et al.* (2020) suggested that the land consumption indicator allowed the identification of areas where land conversions to urban development were disproportionately high compared to population dynamics.

In the 2015 SWSR report, urbanization and soil sealing were identified as the number one threat to soil functions in Europe and Eurasia, but they were not identified as a top four threat in any other region, where threats such as erosion and salinization were more prominent (FAO and ITPS, 2015).

The term "land consumption" used in Indicator 11.3.1 is one of several overlapping terms used to discuss the conversion of natural and agricultural land to urban uses. Morel, Chenu and Lorenz (2015) examined ES provided by a broad range of urban areas, including soils of urban, industrial, traffic, mining and military areas (SUITMAs). Urbanization, as defined in ITPS (2022), is the conversion of agricultural, natural or semi-natural areas to an artificial land use, including sparse settlements, urban fringes, industrial estates and transport infrastructure. This definition will be used throughout this report unless otherwise noted.

Soil sealing is defined by the ITPS as the permanent covering of the soil surface with impervious materials such as concrete or asphalt, tar seal, and buildings or other structures that cannot be easily removed (ITPS, 2022). The proportion of urban land that is sealed differs across the land uses included in the definition of urban lands. For example, Decoville and Feltgen (2023) found that in the centre of major cities in Europe, the average rate of imperviousness of artificial areas was approximately 52 percent, but that substantial differences occurred between cities. Other land uses include artificial grass surfaces installed in sports fields, parks, golf courses, and home yards and gardens.

Studies on the effect of urbanization on soil functions and ES (Morel, Chenu and Lorenz, 2015; Minixhofer and Stangl, 2021; O'Riordan *et al.* 2021; Gardi, Florczyk and Scalenghe, 2021) identified clear differences in the impact on ES between the various land uses in urban areas. A key concern was the loss of food production with the conversion of agricultural land to urban uses. While some authors use the term "land take" to identify the conversion of land to urban purposes, "urbanization" will be used here to avoid any confusion with the different meanings of the other term. Gardi, Florczyk and Scalenghe (2021) estimated that the area of global urbanization between 2000 and 2014 represented only 0.1 percent of the global land surface but disproportionately affected 1 percent of the naturally fertile soils. Van Vliet, Eitelberg and Verburg (2017) forecast that urbanization from 2000 to 2040 could cause a substantial conversion of natural lands to agricultural uses to replace the land lost to urbanization, with all the attendant consequences for ES.

All the studies on the effects of urbanization identify that the imposition of impervious soil surface cover has the greatest effect on ES, particularly on provisioning services, such as food and biomass production, the regulation of hydrological flows, tC sequestration potential and biodiversity loss (ITPS, 2022). The impacts can be minimized through policies that prevent soil sealing (Naumann *et al.*, 2018), as well as through unsealing measures such as the adoption of green infrastructure (Minixhofer and Stangl, 2021), or through unsealing (Pannicke-Prochnow and Albrecht, 2024). The measures are examined in greater detail in Section 5.3.6 and Section 5.4.1 of this report.

2.4 Soils and human societies

One of the main arguments for the introduction of the NCP approach is that the ES approach has undervalued the cultural contributions that nature makes to human society (Díaz *et al.*, 2018). Although the ES approach includes non-economic valuations, economic valuation has become inextricably linked to the use of ES and research on the non-economic aspects has languished.

To raise the importance of the cultural contributions of nature, the NCP approach includes non-material contributions: "Nonmaterial contributions are nature's effects on subjective or psychological aspects underpinning people's quality of life, both individually and collectively." (Díaz *et al.*, 2018, p. 271).

The specific non-material NCPs are learning and inspiration (NCP 15), physical and psychological experiences (NCP 16), and supporting identities (NCP 17) (McElwee, 2021). In many cases, soil is an inseparable part of the whole landscape that provides artistic inspiration, essential mental health benefits and a range of religious, spiritual, symbolic and cultural experiences that can be derived from nature. Such experiences often combine to create distinct identities formed by the relations between humans and soil. The closely related concepts of "care" and "stewardship" are of immediate relevance for soil management, and both can be readily integrated into the emerging overarching concept of soil health (McElwee, 2021).

McElwee (2021) suggested that including non-material NCPs would lead to a realization that soil is a co-constructed entity, integrating nature and culture together. The role of soil in supporting identities has been a central theme in the research on ethnopedology, which aims to document and understand the local approaches to soil perception, classification, appraisal use and management (Barrera-Bassols and Zinck, 2003; Hunyth *et al.*, 2020). In the literature, various terms such as traditional, folk, local, Indigenous Peoples, and farmers' and peoples' soil knowledge systems are used interchangeably to refer to ethnopedology, although Barrera-Bassols and Zinck (2003) pointed out that they were not strictly synonymous. These authors and many others in this field are emphatic that the relationship between people and soil in many cultures cannot be understood solely by western scientific approaches. Scientific approaches typically do not seek to understand the cultural framework of soil knowledge which is often localized and runs counter to the desire for universally applicable findings in western science (Teuber *et al.*, 2022).

The learning and inspiration and physical and psychological experiences aspects of the non-material NCPs can also be present in urban societies, for example, the impact that urban gardening has on the gardeners (Teuber, 2021). Urban agriculture plays an important role in cities by contributing to local food production and maintaining soil functions within urban landscapes (FAO, Rikolto and RUAF, 2022; FAO, 2025). The act of gardening involves continual learning on the part of the gardener and provides physical exercise and a high level of satisfaction to those who tend the garden. These non-material contributions are in addition to the provision of food or regulation of the local water balance provided by the garden plots themselves. Moreover, although there is a potential disconnect with soils and food production on the part of urbanites, gardens and gardening provide people with direct experience with soil and its essential contributions to food production.

Another aspect of supporting identities is the preservation of the natural and archaeological heritage by the maintenance of the soil in the form that developed through time: the "soil-as-is". Soils are an archive of the environments that formed the soils through time (Makeev and Rusakov, 2023). The use of soils to unravel the effects of environmental change is the primary focus of paleopedology (the study of ancient soils). The mixing of soils and the burial of soils in landscape positions where deposition of eroded soil occurs also leads to the inclusion of human artifacts in the soil profile. This role of soils is the focus of archaeopedology, where soil scientists and archaeologists, for example, have collaborated for many decades to understand how past societies affected the soils and landscapes around settlements (Teuber *et al.*, 2022).

The importance of linking soils and human society, as laid out in this chapter, emphasizes the need for an interdisciplinary approach to the development of SSM approaches when addressing the many threats that soils face (Brevik *et al.*, 2015). Anthropology, archaeology, economics, sociology, human geography and other social sciences and humanities all have a part to play if the discipline-specific findings of soil scientists are to be taken up and successfully implemented by wider society, although so far, this has only been generally discussed in the soil science literature (e.g. Brevik *et al.*, 2015; Teuber *et al.*, 2022). The relationship between soils and humans is further discussed in Chapter 6 of this report.

2.5 Summary

The adoption of the 17 SDGs by the UN in 2015 set both action goals for the global community and a set of associated targets by which progress towards the goals would be measured. Soils are explicitly mentioned in four SDGs: SDG 2: Zero Hunger; SDG 3: Good Health and Well-being; SDG 12: Responsible Consumption and Production; and SDG 15: Life on Land. Well-functioning soils are also critical for the achievement of other SDGs such as SDG 6: Clean Water and Sanitation, and SDG 13: Climate Action.

The importance of soils in realization of the SDGs can be illustrated with some of the key statistics, as follows:



SDG 2: Zero Hunger. Ninety-four percent of the global average daily protein intake by humans is from the soil (FAO, 2024) and 98.8 percent of daily average calorie intake is from soil-grown foods (Our World in Data, 2024).



SDG 6: Clean Water and Sanitation. Eighty-seven percent of total water-used in crop production comes from the soil (Mialyk *et al.*, 2024).



SDG 13: Climate Action. Soils are the largest land pool of C at approximately 1 700 billion tonnes of C stored in the soils of the world (Canadell *et al.*, 2021).



SDG 15: Life on Land. Soil is the most biodiverse system on earth, containing an estimated 59 percent of the biodiversity on the planet (Anthony, Bender and Van der Heijden, 2023).

The goal of SSM, as stated in the Revised World Soil Charter (FAO, 2015) is the maintenance or enhancement of the ES provided by soil and hence SSM is critical to the achievement of many of the SDGs previously discussed.

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Chapter 3 | Threats to soil functions

3.1 Introduction and scope of chapter

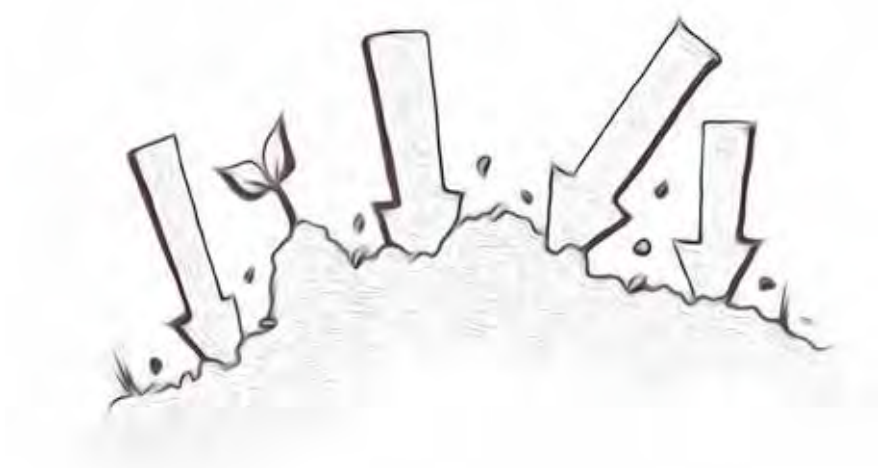
As indicated in Chapter 1, the main focus of the 2026 SWSR report is the assessment of the status of the main threats to soil functions through a management-based approach and the ability of SSM to address these threats. The threats chosen for inclusion are soil erosion, nutrient mismanagement, the loss of SOC, salt accumulation, soil pollution, the loss of soil biodiversity, soil physical deterioration, and urbanization and soil sealing.

The selection of threats for inclusion in the 2026 report began with the list of threats included in the 2015 SWSR report (FAO and ITPS, 2015). The list of threats was finalized after a review of reports from other UN organizations and regional organizations, such as UNCCD (2022), Olsson *et al.* (2019), the Organisation for Economic Co-operation and Development (OECD) (OECD, 2019), and IPBES (2018), followed by discussions and adoption by ITPS at the November 2021 ITPS meeting.

Soil waterlogging was considered in the 2015 SWSR report but was judged to be of minor importance at the regional scale in that report (FAO and ITPS, 2015) and it was not included at the global level. Waterlogging is discussed in the current report as a separate threat in the savannah and highlands subregion for sub-Saharan Africa. It is one of a suite of soil hydrological conditions that can affect soils, and it is explored in the context of the other soil threats, such as soil physical deterioration and salt accumulation.

A parallel decision was also made to focus primarily on threats that can be addressed through the implementation of SSM practices. Sustainable soil management is a foundational principle of the work of the ITPS, as established in the *Revised World Soil Charter* (FAO, 2015). Subsequently, the ITPS established the VGSSM (FAO, 2017) and the *International Code of Conduct for the Sustainable Use and Management of Fertilizers* (Fertilizer Code) (FAO, 2019a) along with studies on specific soil threats and their management. Practically speaking, this focus on SSM means that specific examples of threats that require a broader societal response are allocated less space in the report than those that can be principally addressed through implementation of SSM practices. For example, permafrost melting or the salinization of coastal areas through sea level rise are grave threats caused by global-scale climate change, but the measures required to address them extend far beyond SSM alone.

The content of this chapter focuses on the review of global-scale studies on the extent and severity of threats to soil functions. Examples of regional studies are considered only when no global-scale studies are available. Advances since 2015 in our understanding of the underlying physical, chemical or biological controls on these threats are only considered if they are relevant for the global-scale studies.



3.2 Soil erosion

3.2.1 Introduction

Soil erosion has long been a focus of research and a major societal concern, yet it continues to be a major threat to soil functions, with the 2015 SWSR report judging soil erosion to be the number one threat to soil functions in majority of regions (Africa, Asia, Latin America, the Near East and North Africa [NENA], and North America) (FAO and ITPS, 2015). Soil erosion research was the focus of the Global Symposium on Soil Erosion, held in Rome in 2019 and co-organized by FAO, its GSP and ITPS, together with the Science–Policy Interface (SPI) of the UNCCD and the FAO/IAEA Joint Centre of Nuclear Techniques in Food and Agriculture. The report by FAO (2019b) summarized the research up to 2019.

Soil erosion is defined as the net long-term balance of all processes that detach soil and move it from its original location. The 2015 SWSR report examined erosion by water, wind and tillage. The scope of erosion is expanded in this report to include erosion by shallow landslides and crop harvesting. Water erosion by gully formation is also examined in greater detail than in the 2015 SWSR report.

3.2.2 Thresholds for erosion

As soil erosion occurs naturally over geologic timescales, a goal of reducing erosion to zero is not a realistic objective. Therefore, researchers have instead sought to establish acceptable or tolerable rates of accelerated soil erosion and to design measures to keep erosion within this tolerable range.

The two main impacts of erosion on crop productivity are the reduction in soil depth and the removal of nutrients (Quinton and Fiener, 2024). The most widely used definition of tolerable soil loss using the T2 approach is that of Wischmeier and Smith (1978, p. 12), who stated that it was “the maximum level of soil erosion that will permit a high level of crop productivity to be sustained economically and indefinitely”.

The OECD estimated the range of tolerable soil loss to be from 1 t/ha annually on shallow sandy soils to 6 t/ha annually on deeper, well-developed soils (OECD, 2008). The 6 t/ha annual threshold is also used in Canada, with soils below this threshold considered capable of sustaining long term crop production and maintaining agri-environmental health under current conditions (Lobb, McConkey and Martin, 2016). The environmental assessment system in Europe also considers that tolerable soil loss varies with soil depth, soil type and climatic conditions, suggesting a range between 1 t/ha annually for shallow sandy soils and 5 t/ha annually for deep, well-developed soils (Di Stefano *et al.*, 2023)

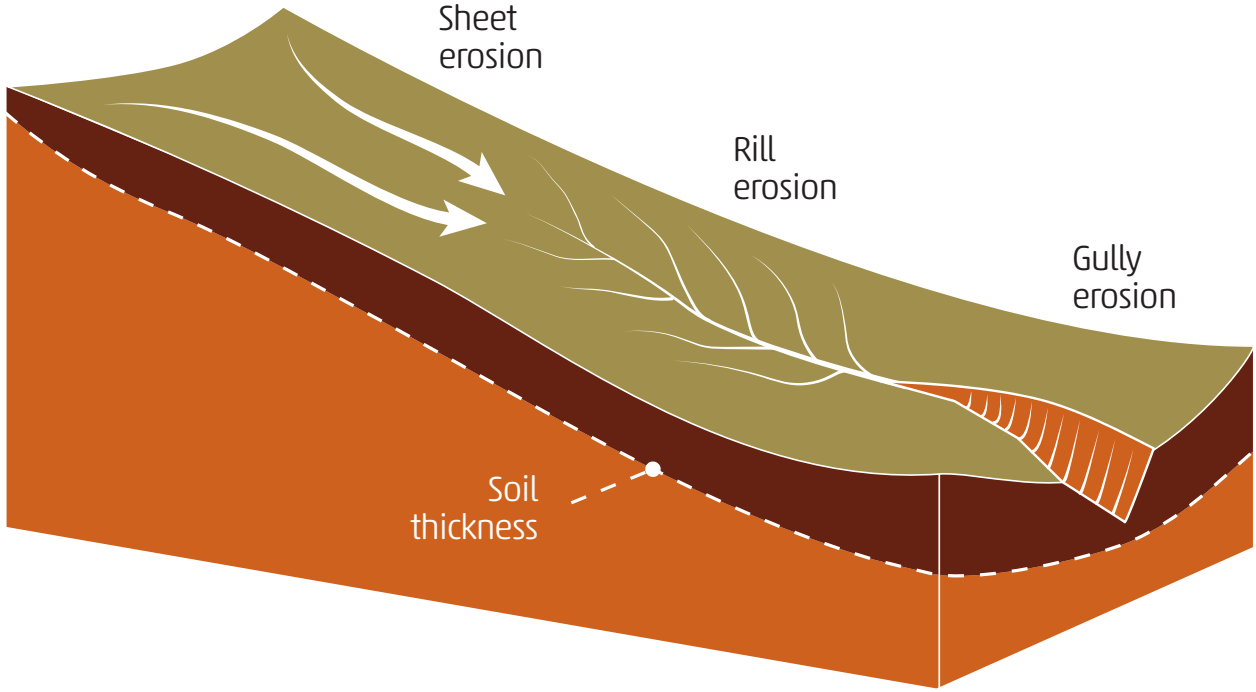
In subsequent sections of this report, erosion rates from any of the erosion processes greater than 6 t/ha annually are deemed to be unsustainable. The lower limit of 1 t/ha annually is used where shallow soils such as Leptosols or soils with growth-limiting subsurface horizons are present (and the relevant information is available).

3.2.3 Water erosion

Overview

Water erosion is the most widely studied form of erosion and affects the greatest land area (FAO, 2019b). Water erosion begins with detachment of soil from the soil mass, which occurs due to both raindrop splash and from forces exerted by water flowing across the soil surface (runoff). Transport of detached soil occurs in the sheet of runoff flowing over the soil surface (sheet or interrill erosion). Often, the surface runoff concentrates and begins to incise shallow channels (rills) or deeper incisions (gullies) (Figure 3.1). The soil carried in the runoff (sediment) settles out of the water and is deposited when either the depth or the velocity of runoff is reduced (for example, when the runoff contacts a vegetative barrier at the base of a field). In other cases, the sediment is exported from the source landscape into a stream system.

Figure 3.1. Schematic diagram of the main water erosion processes



◊ *Source:* **FAO.** 2019. *Soil erosion: the greatest challenge for sustainable soil management.* Rome. https://openknowledge.fao.org/server/api/core/bitstreams/a268e967-a373-4a56-adac-ae1d7ab3567f/content

Global and regional modelling

Modelling has been widely used to inform policymaking with regards to soil erosion (Box 3.1). The review of the soil erosion modelling literature by Borrelli *et al.* (2021) listed 20 citations for global-scale soil erosion modelling, 18 of which specifically modelled water erosion. Of the water erosion-modelling studies, 17 used models based on the Universal Soil Loss Equation (USLE) model (Wischmeier and Smith, 1978) and revised versions of the USLE (generally abbreviated as RUSLE models). These models are empirical and derived from standardized erosion plots (Alewell *et al.*, 2019; Benavidez *et al.*, 2018). Although the original data for the USLE was derived from plots in the United States of America, it has been adapted for use in many other countries. Alewell *et al.* (2019) cited 1 556 papers that estimated soil loss by water erosion using the RUSLE models.

Box 3.1. Opportunities for the effective use of global soil erosion modelling to inform policy development

Global and continental soil erosion models (Borrelli et al., 2023a, 2023b) offer predictions and projections, such as due to changes in land use (Borrelli et al., 2017) or climate change (Borrelli et al., 2020). These can be effectively used by policymakers to understand the potential extent and magnitude of soil erosion and develop mitigation strategies. Models at this scale provide a robust overview of the problem most suitable for informing global, continental, and national policy development.

The United Nations Convention to Combat Desertification (UNCCD) specifically targets land degradation and desertification, with soil erosion recognised as a major land degradation pathway (Prăvălie, 2021). Signatory countries to the UNCCD are required to develop and implement national action programmes (NAPs) to combat desertification, which often involves assessing and addressing soil erosion. This supports the UNCCD's land degradation neutrality (LDN) framework, which aims to achieve a no net loss of healthy and productive land by 2030.

The Paris Agreement under the United Nations Framework Convention on Climate Change (UNFCCC) is primarily focused on climate change mitigation and adaptation (UNFCCC, 2016). However, it also emphasizes the importance of sustainable land management, where soil erosion data is essential for understanding and managing land-based carbon (C) sinks (Borrelli et al., 2018) and for designing climate-resilient agricultural practices. The Intergovernmental Panel on Climate Change (IPCC) recognizes soil erosion future projections as a serious issue. In the IPCC Special Report on Climate Change and Land (SRCCL) (IPCC, 2019), soil erosion is highlighted as a significant concern side by side with climate change and land degradation. As soil is the basis of all terrestrial ecosystems, soil erosion can severely impact biodiversity (Orgiazzi and Panagos, 2018). Moreover, soil erosion data can be used to develop strategies for habitat conservation and restoration under the Convention on Biological Diversity (CBD), or to assist with the development of wetland conservation and management plans developed under the Ramsar Convention on Wetlands (Ramsar, n.d.). Excessive river sediment discharge to the ocean can adversely affect reef ecosystems (Suárez-Castro et al., 2021), whose protection and restoration is central to the United Nations Decade on Ecosystem Restoration 2021–2030 (UNEP and FAO, n.d.).

At the continental scale, soil erosion modelling is important for large-scale restoration programmes such as Africa's Great Green Wall Initiative, and which aim to combat desertification and restore degraded lands and improve production of agrosilvopastoral systems across the drylands. Such programmes require data and information including soil erosion to guide reforestation and land restoration efforts. In Europe, soil erosion assessments and models have been adequately developed to inform policymakers about the extent of the problem (Panagos et al., 2015; Borrelli et al., 2023). In addition, soil erosion is a key indicator to be monitored with the European Union Soil Monitoring Law and the Soil Mission implementation plan (Panagos et al., 2024). An example of national-level downscaling of the global modelling approach is provided by the Welsh government's Environment and Rural Affairs Monitoring and Modelling Programme (ERAMMP), where output is combined with field survey and river sediment monitoring for the assessment of agrienvironmental vulnerability (ERAMMP, n.d.).

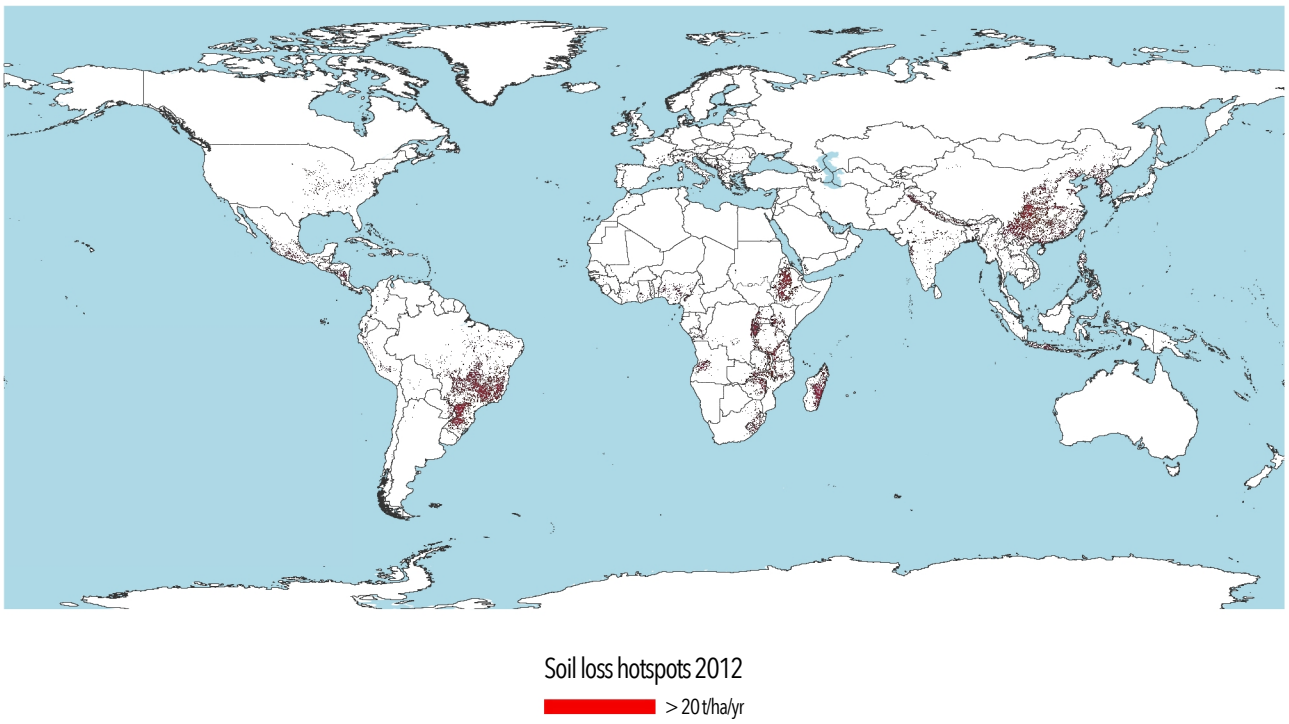
Global soil erosion models provide a versatile, internally consistent approach to determining the extent and magnitude of soil erosion that can be effectively used by policymakers to develop global to local prevention and remediation strategies. Together with other major land degradation indicators (such as loss of C stocks, aridity and salinization), erosion modelling will be hugely important to track progress towards LDN (Wuepper et al., 2021).

◊ Sources: See References.

Borrelli *et al.* (2017) modelled global water erosion using a RUSLE-based modelling approach with a 250-by-250-m grid cell resolution. The study provided results for water erosion based on data for 2001 and 2012, and used information on land cover type, vegetation cover dynamics and the farming systems of each cell, allowing for the effect of land cover change on erosion to be assessed in the two time periods. A global map of hotspots of water erosion was also provided, defined as cells with annual water erosion rates greater than 20 t/ha annually (FAO, 2019b).

The authors found that 6.1 percent of the land area (or about 7.5 million km^2 in total) had water erosion rates greater than 10 t/ha annually. The average annual soil loss from cropland was estimated at 12.7 t/ha, more than 77 times the estimated annual rate for forests (0.16 t/ha) and 7 times higher than the average for other forms of natural vegetation (1.8 t/ha) (Figure 3.2).

Figure 3.2. Water erosion hotspots



Refer to the disclaimer on page ii for the names and boundaries used in this map. Dotted line represents approximately the Line of Control in Jammu and Kashmir agreed upon by India and Pakistan. The final status of Jammu and Kashmir has not yet been agreed upon by the parties. Final boundary between the Republic of Sudan and the Republic of South Sudan has not yet been determined. Final status of the Abyei area is not yet determined.

◊ *Sources:* **Borrelli, P., Robinson, D.A., Fleischer, L.R., Lugato, E., Ballabio, C., Alewell, C., Meusburger, K.** *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

◊ **FAO**. 2019. *Soil erosion: the greatest challenge for sustainable soil management*. Rome. https://openknowledge.fao.org/server/api/core/bitstreams/a268e967-a373-4a56-adac-ae1d7ab3567f/content

Carr *et al.* (2020, 2021) modelled water erosion for global wheat and maize cropland using an Environmental Policy Integrated Climate (EPIC)-based model (Carr *et al.*, 2020). Water erosion was modelled over a 31-year period, between 1980 and 2010. The authors estimated that in the 2000–2010 period, water erosion reduced maize yields in 58 percent of the land area used to produce maize, and that the median annual soil loss of this area was estimated to be 11 t/ha. For wheat, 46 percent of the land area used for wheat production experienced yield loss due to erosion and had estimated median annual losses of 6 t/ha.

The limitations of the RUSLE models have been widely discussed (Borrelli *et al.*, 2017; Carr *et al.*, 2020; Alewell *et al.*, 2019; Benavidez *et al.*, 2018). First, the RUSLE models were not designed to simulate gully or streambank erosion, or deposition in floodplains and hence they offer an incomplete picture of water erosion. Second, most of the RUSLE models only calculate soil loss and do not model the deposition of sediment within the field and broader landscape. Batista *et al.* (2019) concluded that in general, model-based estimates of erosion and deposition did not compare well with independent spatial data in part due to the difficulty of measuring soil erosion and deposition across landscapes. Third, the USLE was based on a limited range of data geographically (i.e. data from the United States) and physiographically, making model applications beyond the source region problematic

(Carr *et al.*, 2020; Quine and Van Oost, 2020). Fourth, the area of individual grid cells used in the models is much larger than the area of the erosion plots used to assess the accuracy of the models (Alewell *et al.*, 2019) which limits the ability to validate the model results. Given the limitations of the RUSLE modelling framework, it is desirable to develop alternative modelling approaches that do not have its inherent problems.

Tan *et al.* (2018, 2020, 2022) developed a physically based model to predict soil erosion and sediment yield (i.e. the amount of eroded soil exported from a basin in a given time period). The model was based on the Morgan-Morgan-Finney model (Morgan, 2001), which considers detachment by both raindrop impact and runoff and transport by runoff. The model was implemented within an earth system model (ESM) framework, which allowed the various components of the model to interact with each other during model runs. The model provided a global-scale 0.5-degree resolution (an approximately 56-km-by-56-km cell) map of soil erosion and the sediment delivery ratio (Tan *et al.* 2022). The coarse grid size precludes any reliable comparison with the other global models but indicates the potential of integration of erosion models into larger ESMs.

Climate change impact on water erosion

The impacts of climate change on water erosion have been examined using scenarios of future climate change under various socioeconomic pathways (such as shared socioeconomic pathways [SSPs]) to provide future climate and land-use data for the soil erosion models discussed previously. The erosion rates under these future scenarios are then compared to baseline erosion rates for the present day to assess changes in rates through time.

Using the RUSLE model, Borrelli *et al.* (2020) modelled sheet and rill erosion rates in 2015 (using the year as a baseline) and 2070. The RUSLE model was linked to three SSPs and representative concentration pathways (RCPs), with the three RCPs differing based on the increase in radiative forcing by 2100.

For the baseline year of 2015, Borrelli *et al.* (2020) estimated the global erosion from sheet and rill erosion at 43 billion tonnes annually. The effect of land-use changes alone by 2070 showed divergent trends, depending on the RCP used (Borrelli *et al.*, 2020). Decreases in the area of agricultural land were estimated under SSP1-RCP2.6 (i.e. a best-case scenario), which lowered erosion rates (compared to the baseline of 2015) by about 4.5 billion tonnes annually, whereas SSP5-RCP8.5 (a worst-case scenario) resulted in a 10 percent increase in wind erosion to 47.3 billion tonnes annually. The combined land-use and climate simulations showed a substantial potential increase in average soil erosion totalling +30 percent (to 56.1 billion tonnes annually for the SSP3-RCP2.6 and +66 percent (to 71.6 billion tonnes annually) for the SSP5-RCP8.5 scenarios (with wide confidence intervals for each estimate).

Rather than applying a single model to global modelling, Eekhout and De Vente (2022) conducted a systematic review of 224 modelling studies that examined the climate change–water erosion relationship. While the studies were generally using basin or hillslope scales, when considered as an ensemble, they were able to provide a more globally relevant result.

Soil erosion was projected to increase by 10 percent overall (weighted median) in the twenty-first century, although the variation in the original erosion estimates was very high (from −76 percent to 605 percent). The projected increase was mostly in semi-arid climate zones (at +17.3 percent) and least in the subarctic/Arctic (−2.8 percent), while other climate zones projected close to the median estimate of 10 percent. Of the land-use changes studied, agricultural expansion under climate change led to the largest increase (with a weighted median of 78.6 percent) whereas agricultural land abandonment and reforestation led to a median decrease of −52.5 percent (Eekhout and De Vente (2022).

Economic impact

The impact of soil erosion on crop productivity is the most studied aspect of the economic impact of erosion. Based on a review of the available literature, the 2015 SWSR report (FAO and ITPS, 2015) found that estimates of annual crop yield loss due to erosion ranged from 0.1 to 0.4 percent and suggested an average annual crop yield loss of 0.3 percent due to erosion. Based on a literature survey, Panagos *et al.* (2018) assumed that a total crop productivity loss of 8 percent had occurred over the past 25 to 30 years prior to the 2018 publication date (approximately 0.3 percent annually) for intensively cultivated fields with erosion rates greater than 11 t/ha annually. No crop productivity loss for fields with lower erosion rates was assumed (in part because intensive fertilization compensates for soil loss).

The global assessment carried out by Sartori *et al.* (2019) used the Panagos *et al.* (2018) assumption about the crop productivity loss of 8 percent on land with severe erosion (>11 t/ha annually) and the long-term soil erosion rates estimated by Borrelli *et al.* (2017) using the RUSLE model. The economic impact was simulated with the Modular Applied GeNeral Equilibrium Tool (MAGNET) (Woltjer *et al.*, 2014). The global erosion simulation predicted that more than 3.4 million km^2 of arable land worldwide (24 percent) was suffering from severe erosion. Modelling showed that soil erosion was harmful to the real gross domestic product (GDP), but that the effect was small. India and Indonesia had the largest losses due to erosion, at 0.1 percent of GDP, followed by Nigeria and Central America and the Caribbean at closer to 0.04 percent. For Europe, the United States and Oceania, marginal gains due to erosion were predicted, with those regions ending up in a more favourable production and trade position. The study estimated that global food production had decreased by approximately 33.7 million tonnes due to severe soil erosion (equivalent to 0.41 percent of global agricultural production).

Carr *et al.* (2021) simulated 31 years of water erosion on the global productivity of maize and wheat (from 1980 to 2010), using the gridded crop model, EPIC-IIASA. Cells with high yield reductions were associated with settings with low fertilizer inputs on steep slopes exposed to intensive precipitation. In total, water erosion reduced the annual global production of maize and wheat by 9 million tonnes and 6 million tonnes respectively, although this was less than 1 percent of the annual global average maize and wheat production between 2013 and 2018 (Carr *et al.*, 2021). The simulated yield losses would cause an annual global loss of approximately USD 2 billion in maize and USD 1.3 billion in wheat. The nitrogen (N) runoff due to erosion was also estimated, with the annual N fertilizer replacement costs calculated to be USD 642 million for maize and USD 255 million for wheat.

Carr *et al.* (2021) concluded that overall, water erosion had a small impact on annual global production losses in absolute terms. Most maize and wheat production occurs on relatively flat terrain, which has lower rainfall erosivity than the global average and which receives high fertilizer inputs that offsets losses of nutrients due to erosion. Likewise, Panagos *et al.* (2018) concluded that soil erosion by water was not a threat to food security in the European Union.

There are no global studies on the estimated costs for the impacts of water erosion beyond decreases in crop productivity, although costs of lost production have been estimated at national and regional scales. For example, Graves *et al.* (2015) estimated that the total cost of water erosion in England and Wales (countries in the United Kingdom of Great Britain and Northern Ireland) was about GBP 150 million annually, of which 27 percent being the costs of onsite losses due to decreased productivity and nutrient export, and 73 percent being offsite costs. These offsite costs include the treatment cost of nutrient removal for drinking water, the damage costs of nutrients passing to the water environment, sediment removal from rivers, lakes and urban drainage systems and greenhouse gas (GHG) losses due to erosion events.

Gully erosion

According to Bartley *et al.* (2020), the most common types of gully erosion include ephemeral gullies in croplands, permanent gullies in uncultivated land (also known as colluvial or hillslope gullies), alluvial, bank or valley bottom gullies (often found in flatter areas of the landscape adjacent to larger drainage lines), urban gullies, and badlands or ravine lands. In landscapes prone to gully erosion, gullies typically occupy less than 1 to 5 percent of the total area, but their contribution to sediment production often exceeds 5 percent and may exceed 80 percent in some landscapes (Poesen, 2018).

There are no global estimates of gully erosion. Existing models have concentrated on individual gullies or local scales and typically require considerable local calibration to operate, which limits their regional or global application. A successful global model would need to account for where and why gullies occur, when and how gullies expand, and the contribution of gullies to catchment sediment yields (Vanmaercke *et al.*, 2021).

Borrelli *et al.* (2022) assessed gully erosion in the European Union and drew upon field assessments of gullies conducted at 24 759 sites across the European Union, as part of the 2018 Land Use/Cover Area frame survey (LUCAS) topsoil survey (d'Andrimont *et al.*, 2020). The field surveys were supplemented with Google Earth surveys of the type carried out by Vanmaercke *et al.* (2020) in the Horn of Africa. Surveyors recorded gullies at 211 of the visited sites (equal to approximately 1 percent of the total sites with field assessments). Sites with gullies were most common in Spain (61 percent of the total) and the remainder were primarily found in the Mediterranean zone. Gullies were observed in croplands (33 percent), woodlands (28 percent), grasslands (17 percent), and shrublands (16 percent). Borrelli *et al.* (2022) distinguished between ephemeral and permanent gullies, slightly blurring the distinction between rills and gullies.

Despite the lack of currently available regional and global-scale studies on gully erosion, the extensive review by Vanmaercke *et al.* (2021) summarized the increasing availability of geospatial data sets for key controls such as vegetation cover and topography. The lack of research on the effects of soil and water conservation measures on gully erosion was also highlighted by the authors as an impediment to modelling efforts.

The economic costs of gullying include the loss of planting area, increases in agricultural costs and workloads due to field fragmentation and access issues, increases in the cost of maintaining roads or transport routes or costs with the paralysis of roads, impacts on infrastructure such as bridges, dykes and water pipes, and costs associated with sedimentation off-field (Kuhn *et al.*, 2023). These authors state that the socioeconomic impacts and related costs should be better documented in literature.

3.2.4 Wind erosion

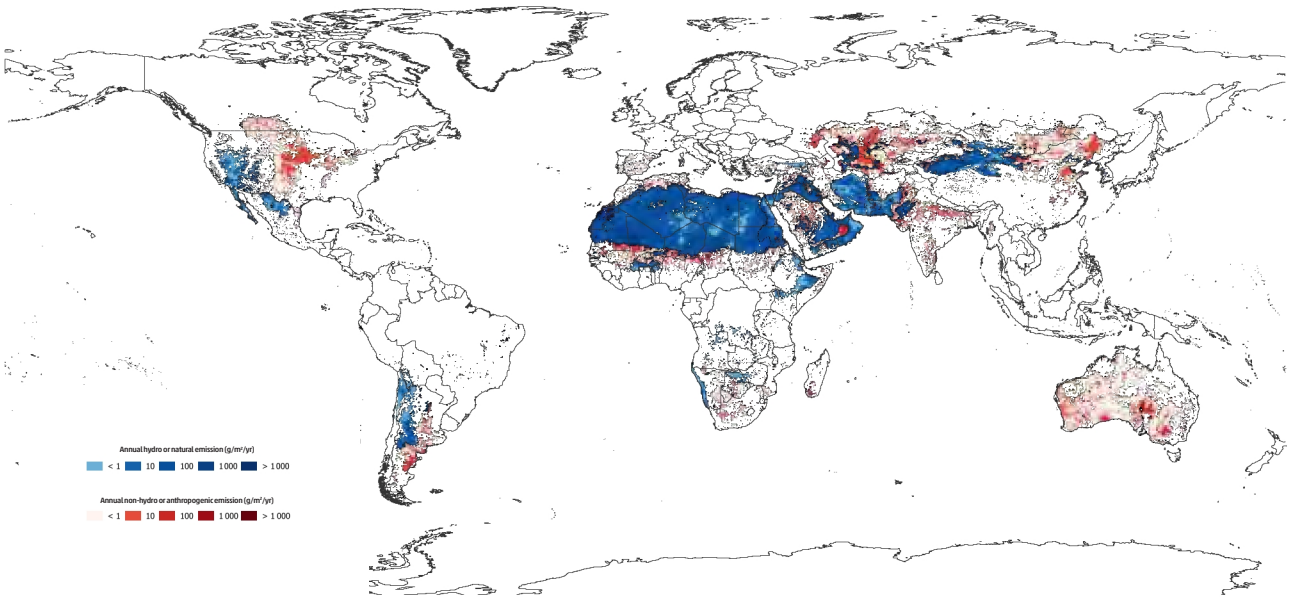
Wind erosion begins with the transfer of momentum from the wind to the soil surface, which exerts a drag or shear stress on the surface (FAO, 2019). This shear stress can detach soil from the surface, which is then transported through creep (rolling along the surface), saltation (bouncing along the surface in short hops) or through suspension in the wind column. Coarser particles such as medium to coarse sand typically move through creep or saltation and are deposited close to the point of detachment. However, smaller particles and organic matter can be transported as dust over greater distances, including between continents. For example, the deposition in the Amazon rainforest of dust from North Africa is a critical source of P for the rainforest region (Kok *et al.*, 2021).

Two different approaches have been used to develop global estimates of wind erosion. First, global studies on wind erosion can draw upon satellite-based observations of dust in the atmosphere to develop spatial and time-series data (Ginoux *et al.*, 2012; Kok *et al.*, 2021; Chappell *et al.*, 2023). This technique is not able to capture transport by creep and saltation, and if used alone, will

underestimate the total wind erosion. Satellite data can be used to readily identify the source areas and seasonal variations of major dust emissions. The second approach is to use the Revised Wind Erosion Equation (RWEQ) model (Jarrah *et al.*, 2020) or other physically based models to simulate global wind erosion.

Ginoux *et al.* (2012) used the MODIS Deep Blue satellite and a physically based model based on wind speed to identify dust sources and has been widely cited in the period since 2015. Ginoux *et al.* (2012) estimated total annual global dust emissions of 1.54 billon tonnes, averaged over the period 2003–2009, of which 75 percent were from natural sources and 25 percent from anthropogenic (primarily agricultural) sources. The greatest sources of anthropogenic emissions were in the Sahel, the highlands of Saudi Arabia, the Ganges basin and Indus basin in the Indian subcontinent, northern China and Mongolia, the high plains of the United States and the Sonoran Desert in Mexico, river valleys in Patagonia, southeast Africa, and areas in northern central Australia (Figure 3.3).

Figure 3.3. Annual mean dust emission from land use (2003 to 2009)



Refer to the disclaimer on page ii for the names and boundaries used in this map. Dotted line represents approximately the Line of Control in Jammu and Kashmir agreed upon by India and Pakistan. The final status of Jammu and Kashmir has not yet been agreed upon by the parties. Final boundary between the Republic of Sudan and the Republic of South Sudan has not yet been determined. Final status of the Abyei area is not yet determined.

Note: Hydrological and natural sources are in blue and anthropogenic sources are in red.

◊ *Sources:* **Ginoux, P., Prospero, J.M., Gill, T.E., Hsu, N.C. & Zhao, M.** 2012. Global-scale attribution of anthropogenic and natural dust sources and their emission rates based on MODIS Deep Blue aerosol products. *Reviews of Geophysics*, 50(3). https://doi.org/10.1029/2012RG000388

◊ **Cherlet, M., Hutchinson, C., Reynolds, J., Hill, J., Sommer, S. & Von Maltitz, G.**, eds. 2018. *World Atlas of Desertification*. Luxembourg, Publications Office of the European Union.

Chappell *et al.* (2023) also used a MODIS-based model to show the global differences in seasonal dust sources annually from 2001 to 2020 which showed that the Earth's predominant dust sources (in rank order) were the NENA deserts, Australian shrublands and East Asian deserts and grasslands. The Patagonia region of Argentina was also identified as a significant dust source especially, in the December to February period. The average global dust emissions were estimated to be 29.1 million tonnes per year.

In an earlier study, Chappell *et al.* (2019) modelled global wind erosion and the associated loss of SOC. They used the physically-based wind erosion model of Chappell and Webb (2016) and a satellite-based surrogate for vegetation cover to estimate wind erosion in the 2001 to 2016 period.

The Chappell *et al.* (2019) global map of mean wind erosion shows broad similarity to that of Ginoux *et al.* (2012), but significant differences occur. For example, the Ganges and Indus basins in the Indian subcontinent do not show high rates of wind erosion in the Chappell *et al.* (2019) map, and the area of wind erosion in Australia is mostly in the southern part of the country. The most severe rates for wind erosion shown on the Chappell *et al.* (2019) global map are between 1 and 7 t/ha annually and these occur only in small areas in North Africa, the Islamic Republic of Iran and Central Asia. These rates are considerably lower than the maximum rates for sheet and rill erosion discussed earlier.

Yang *et al.* (2022a) used a spatially distributed version of Revised Wind Erosion Equation (RWEQ) model to simulate global wind erosion from 2001 to 2010 on a 10-km grid. It was estimated that on average, 17.6 percent of the modelled global area had annual wind erosion rates greater than 5 t/ha annually, with 10.5 percent of the area with rates greater than 15 t/ha annually. Africa had the greatest share of global wind erosion (70 percent) with areas of severe loss in North Africa and the Namib and Kalahari deserts of southwest Africa. Asia had the second-highest proportion (21 percent) with the highest losses in the Arabian Peninsula, central areas of the Islamic Republic of Iran and the Taklimakan desert in Central Asia. A final area with high rates is in Patagonia.

The RWEQ was also used by Sun *et al.* (2024) to model global wind erosion from 1982 to 2019. Annual wind erosion rates were estimated that were substantially higher than the other global studies, with large areas in North Africa, southeast Africa, the Arabian Peninsula, central Asia, southeast and south central Australia, the Islamic Republic of Iran, Patagonia, and west central United States, with annual rates in excess of 100 t/ha. An average soil loss of 312.5 billion tonnes was estimated due to wind erosion. Their estimate shows a decreasing trend of 1.04 billion tonnes per year due to wind stilling and higher vegetation cover.

However, the estimates of annual dust emissions and wind erosion differ widely across these studies. Using a physically based wind erosion model Chappell *et al.* (2019) predicted maximum annual rates of 1 to 7 t/ha for small areas of the globe, while using the RWEQ, Yang *et al.* (2022a) estimated high losses of over 15 t/ha per year in 10.5 percent of the global modelled area. Sun *et al.* (2024) estimated that large areas of the globe had annual rates greater than 100 t/ha per year, while Borrelli *et al.* (2020) estimated that 43 billion tonnes of soil annually were lost due to sheet and rill erosion. The wind erosion estimate of Sun *et al.* (2024) was approximately seven times this value and far higher than the other studies.

While there are wide differences in the estimates of total wind erosion and rates on a per hectare basis across the various modelling studies of global wind erosion, there is more agreement about the locations with high dust emissions. Most dust emissions due to wind erosion are from natural sources, principally in North Africa, Saudi Arabia, the Taklamakan Desert of East Asia, southwest United States and northern Mexico, the deserts of Chile and Peru in South America and the Namib desert in Southeast Africa. The greatest sources of anthropogenic emissions were in the Sahel in sub-Saharan Africa, the highlands of Saudi Arabia, northern China and Mongolia, the high plains of the United States and the Sonoran Desert in Mexico, river valleys in Patagonia, southeast Africa, and areas in Australia.

Economic impact of wind erosion

There are no global studies on the economic impact of wind erosion alone, and comparisons across the existing regional studies are difficult due to the lack of common methods and time periods (Middleton, Tozer and Tozer, 2019).

Climate change impact on wind erosion

According to Mirzabaev *et al.* (2019) in the IPCC's *Special Report on Climate Change and Land*, the impacts of climate change on dust storm activity were not yet well understood and global circulation models were unable to capture recent observed dust emissions and transport, limiting the confidence in any future projections.

Recent studies on global terrestrial stilling (the decades-long decrease in mean global surface wind speed) also highlight the complexity of the factors in the climate–wind–erosion relationship. Zeng *et al.* (2019) reported that a clear turning point in mean global surface wind speed occurred in 2010. Prior to this, the mean global surface wind speed decreased at a rate of −0.08 m/s every decade between 1978 and 2010. Between 2010 and 2019, the mean global surface wind speed had increased at a decadal rate of 0.24 m/s. The trends in mean global surface wind speed are most likely due to decadal variability of large-scale ocean–atmospheric circulations, but the linkage of these to climate change is unclear.

3.2.5 Tillage erosion

Tillage erosion was not recognized as a major concern in soil science until the 1990s, and it remains less prominent in the scientific literature than wind and water erosion (FAO, 2019b). In tillage erosion, mechanical tillage implements used in field operations (such as ploughs, harrows and cultivators) cause soil detachment and transport. Tillage operations on sloping land cause the net downslope movement of soil and, through time, result in the thinning of soils on convex-shaped shoulder slopes and often leading to depositional soils in concave-shaped lower slope positions. The loose, redeposited soil is often vulnerable to subsequent transport by rill and gully erosion by water. Unlike wind and water erosion, tillage cannot transport soil beyond the field boundary and hence is associated with the redistribution of soil within the field, rather than a net loss from the field. However, tillage erosion is indirectly responsible for the transport of eroded soil beyond field boundaries through its role as a major delivery mechanism for water erosion.

Global and regional modelling

Global and regional modelling of tillage erosion has lagged behind that of water and wind erosion. Van Oost *et al.* (2007) modelled water and tillage erosion in the context of their study on erosion impacts on the C cycle, and although dated, it remains the only global modelling estimate available for tillage erosion. The modelling was carried out for water and tillage sediment flux (in billion tonnes annually) on arable land. This provided an estimated 17.1 billion tonnes of sediment flux annually for water erosion and 5.1 billion tonnes of sediment flux annually for tillage erosion, on an area-averaged basis equivalent to 12 t/ha annually for water erosion and 3.5 t/ha annually for tillage erosion (Van Oost *et al.*, 2007). Tillage erosion was estimated as the product of slope curvature and a tillage (soil) transport coefficient, although the coefficient varies for different tillage implements. Water erosion was estimated using an equation based on RUSLE, and no global map for tillage erosion was produced. As the only available estimate, the annual tillage erosion estimate of 5.1 billion tonnes has subsequently been cited by Quinton *et al.* (2010) and in Borrelli *et al.* (2021).

Van Oost, Cerdan and Quine (2009) simulated tillage erosion at 1 km^2 resolution at the regional scale in the European Union, using a slope curvature and tillage coefficient model, and simulated water erosion using the PESERA model. The area weighted mean for tillage erosion was 3.3 t / ha annually, with the highest rates of >5 t/ha annually in Italy (Sicily and Apennine Mountains), Spain (Andalusia) and the hilly foot slopes of mountain ranges in Central Europe. There was an estimated net soil removal by tillage erosion of 0.35 billion tonnes annually for the study region. The average water erosion rate was 3.9 t/ha annually, corresponding to a net soil loss of approximately 0.41 billion tonnes annually. The ratio of tillage to water erosion was highest for Albania (1.67), Greece (2.14) and Spain (1.54).

Borrelli *et al.* (2021) summarized the results on tillage erosion from subnational scale studies. Only 37 tillage erosion modelling studies were cited by Borrelli *et al.* (2021), totalling only 1.8 percent of entries in their global database of erosion modelling studies. The median rate for tillage erosion across the modelling studies was 30.1 t/ha annually compared to 6.3 t/ha annually for wind erosion models and 7.4 t/ha annually for water erosion (primarily for sheet and rill erosion).

Economic impact

The economic impact of soil loss in a tillage erosion-dominated area was estimated in the subnational study of the loss of A-horizon soil (topsoil) by Thaler, Larsen and Yu (2021) from agricultural soils in the corn belt of the United States, with the spatial pattern of soil loss (loss of soil from hilltops and hillslopes, depositional in concave lower slopes) being consistent with tillage erosion. Using a combination of satellite imagery and field data, it was estimated that approximately 35 percent of the cultivated area in the study region had lost A-horizon soil, decreasing crop yields in the corn belt by 6 percent (±2 percent), and leading to annual losses of USD 2.8 billion (± USD 0.9 billion).

3.2.6 Shallow landslides

While most types of erosion involve individual particles or small aggregates of soil, mass movements such as landslides, mudflows and debris flows transport large masses of material downslope under the influence of gravity. They are triggered by intense or prolonged rainfall, seismic activities, rapid temperature changes, and anthropogenic activities such as mining, construction, improper drainage, Land-use change, and deforestation (Kirschbaum and Stanley, 2018). The different types of mass movement are often referred to as landslides in the literature, and therefore this term will be followed in this section.

Landslides are directly responsible for human fatalities. Data presented by Froude and Petley (2018) showed that 55 997 people were killed in landslides unrelated to earthquakes in 4 862 distinct events between January 2004 to December 2016. The majority (79 percent) of these fatal landslide events are triggered by rainfall (Froude and Petley, 2018), with 16 percent of fatal landslides associated with mines or quarries (423 events) and roads (568 events) (usually during road construction). Other causes of anthropogenically caused landslides include illegal hill cutting, conflict and explosions, leaking pipes, garbage collapse, recreation, and fire (either natural or due to anthropogenic activities).

Fatal landslides were widely distributed across the globe (Froude and Petley, 2018). Asia had the highest incidence (75 percent of the total) with substantial numbers along the Himalayan Arc, Bangladesh, southeastern China, in states across India, the Lao People's Democratic Republic, and Myanmar, and southwards on islands through Indonesia and the Philippines. Other areas of high incidence were found in Central America between Costa Rica and the south of Mexico, the Caribbean islands, along the Andes mountain range in South America from the Bolivarian Republic of Venezuela to the Plurinational State of Bolivia (and to a lesser extent, Chile), with another cluster of events on the east coast of Brazil (around the states of São Paulo and Rio de Janeiro), around the borders between Burundi, the Democratic Republic of the Congo, Kenya, Rwanda, the United Republic of Tanzania and Uganda in East Africa, and the Islamic Republic of Iran, Türkiye and the European Alps.

Landslides also have a significant economic effect beyond the death toll associated with them. Although data on global economic impacts are limited, Klose, Maurischat and Damm (2016) estimated a total global annual loss caused by landslides of approximately USD 20 billion.

The economic effects of landslides are different than the other forms of erosion as their impact includes a loss of infrastructure (such as roads, bridges and terraces) rather than a loss of productivity (Blaschke, Trustrum and Hicks, 2000). Impacts of mass movement on cropland productivity are

anecdotal at best, even in countries such as Indonesia where the incidence of mass movement is high. The best documented example is for mudstone hill country pastureland in New Zealand, where rates of mass movement increased following clearance of forest since 1840, with lower pasture production on landslide scars detectable for decades (Blaschke, Trustrum and Hicks, 2000).

Operational landslide monitoring systems have been implemented at the country level in Italy, Japan, Norway and the United States (Kirschbaum and Stanley, 2018). Stanley *et al.* (2021) reported on a global-scale system (the Landslide Hazard Assessment for Situational Awareness [LHASA] model) that combined seven predictive variables (snow mass, soil moisture, slope, distance to faults, lithologic strength, antecedent rainfall and daily rainfall). All variables are globally extensive and publicly available with very short delays in data availability.

Stanley *et al.* (2021) compared predictions from the model with 2 global and 17 national landslide inventories. The shift from version 1 (Kirschbaum and Stanley, 2018) to version 2 doubled the accuracy of the "nowcast" for the inventories evaluated but the model still had mixed success. It was suggested that this was due to limitations in the precipitation variable and to the effect of anthropogenic factors, which were not considered. Both roads and mines were associated with enhanced landslide activity, with far less rainfall required to trigger landslides on modified terrain than natural surfaces. Deforestation was a second factor known to enhance landslide susceptibility.

The importance of precipitation also provides an obvious link to the impact of climate change on landslides. Gariano and Guzzetti (2016) reviewed 103 studies between 1983 and 2016 that investigated the impact of climate change on landslides. Four climate factors that impacted slope stability were examined: total precipitation, rainfall intensity, air temperature, and increases in wind speed and duration. The global pattern of the effects of predicted climate change corresponded to the predicted spatial patterns of change in climate variables (for example, annual mean precipitation was projected to increase at the high latitudes and in the equatorial Pacific, under the RCP8.5 scenario). In these areas, and particularly in mountainous and hilly terrains, deep-seated landslides, rockslides, earthflows, and mudflows were expected to occur more frequently, or to increase their seasonal activity.

Gariano and Guzzetti (2016) also stressed that the effects of climate change on land use might be as important as the change in the climate factors themselves. For example, in areas of Southeast Asia and elsewhere, in response to climate and economic drivers, forests may be cut down (for example, to obtain wood) or be completely converted to other land uses (such as from native vegetation to oil palms or other agricultural uses). In these cases, landslide activity and the frequency of landslide occurrences were expected to increase, although these changes are difficult to predict or model.

3.2.7 Soil loss due to crop harvesting

Soil loss due to crop harvesting (SLCH) occurs when fine earth adhering to the crop and loose soil and rock fragments are "harvested" and removed from the field together with the crop (Ruysschaert *et al.* 2004). Soil loss due to crop harvesting is associated with crops that are in direct contact with the soil, such as tuber and root crops including sugar beet (*Beta vulgaris* subsp. *vulgaris* [*var. saccharifera*]) and potato (*Solanum tunerosum*). Soil losses through harvesting were first recognized by soil scientists in the 2000s, but as of 2025, there are still limited data available.

The literature available as of 2022 was summarized by Kuhwald *et al.* (2022). Of the 55 studies summarized, 5 had rates of SLCH between 10 and 22.4 t/ha per harvest, 42 had rates between 1 and 10 t/ha per harvest, 3 had rates between 0.1 and 1 t/ha per harvest, and 5 had rates less than 0.1 t/ha per harvest. This range reinforces the statement made by Ruysschaert *et al.* (2004) that losses due to root crop harvests are comparable to those caused by other erosion processes. Kuhwald *et al.* (2022) stressed that most of these studies were conducted in the 2000s and might not reflect current agronomic practices. The studies were also biased towards European studies, with few studies from

other regions. This gap is important to highlight due to the large areas of cassava (*Manihot esculenta*) and groundnut (*Arachis hypogaea*) crops, cultivated in Africa and some Latin American countries. Kuhwald *et al.* (2022) summarized the differences between SLCH and other forms of erosion as follows:

- Soil loss due to crop harvesting only occurs through the harvesting of tuber and root crops.
- There are no dominant climate controls on SLCH.
- It can occur at any relief position and parent material.
- It always affects entire fields that are harvested.
- It is not selective by particle size.

Panagos, Borrelli and Poesen (2019) provided an estimate of SLCH for the European Union using results from field studies and the total areas planted to sugarbeet and potato crops in European Union countries. The estimates suggested that the erosion rate for sugar beets ranged from 2.3 t/ha per harvest for Croatia to 10 t/ha per harvest for Denmark and France. For potatoes, a single value of 3 t/ha per harvest was used across European Union countries.

3.3 Soil organic carbon loss

3.3.1 Introduction

Soils are the largest land-based store of C. In their Sixth Assessment report, the IPCC (Canadell *et al.*, 2021) estimated permafrost soils stored approximately 1 200 billion tonnes of C and that mineral and wetland soils outside of the permafrost region stored 1 700 billion tonnes, with terrestrial vegetation storing a further 450 billion tonnes of C.

The SOC content has been the focus of an extraordinary research effort since 2015. This focus stems in part from the possibility of using increased SOC storage to offset anthropogenic emissions of CO_2 to the atmosphere (C sequestration) fostered in particular by the "4 per 1000" Initiative (n.d.). This research effort has led to progress in almost all areas of SOC research, including fundamental reappraisals of the nature of SOC, the global and regional mapping of SOC distribution (Figure 3.4) and of its sequestration potential, the relationship between erosion and SOC and the interaction between climate change and SOC (the research on C sequestration is presented in Section 5.3.1 of this report).

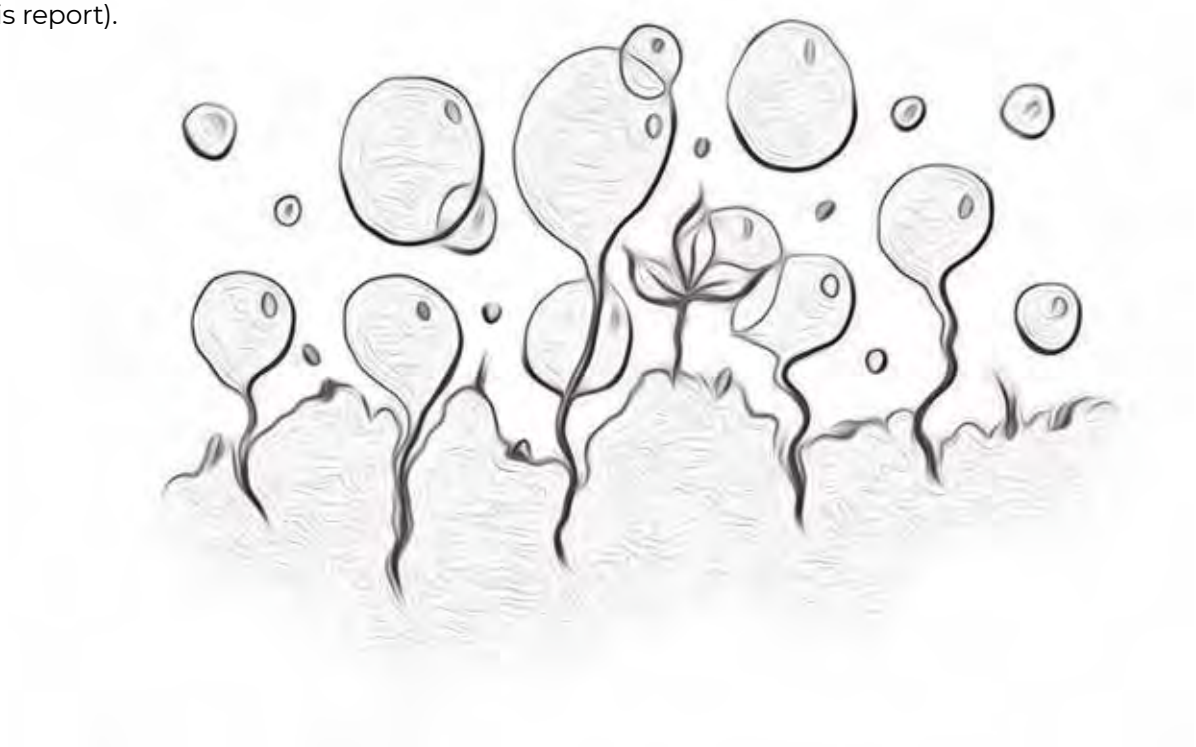
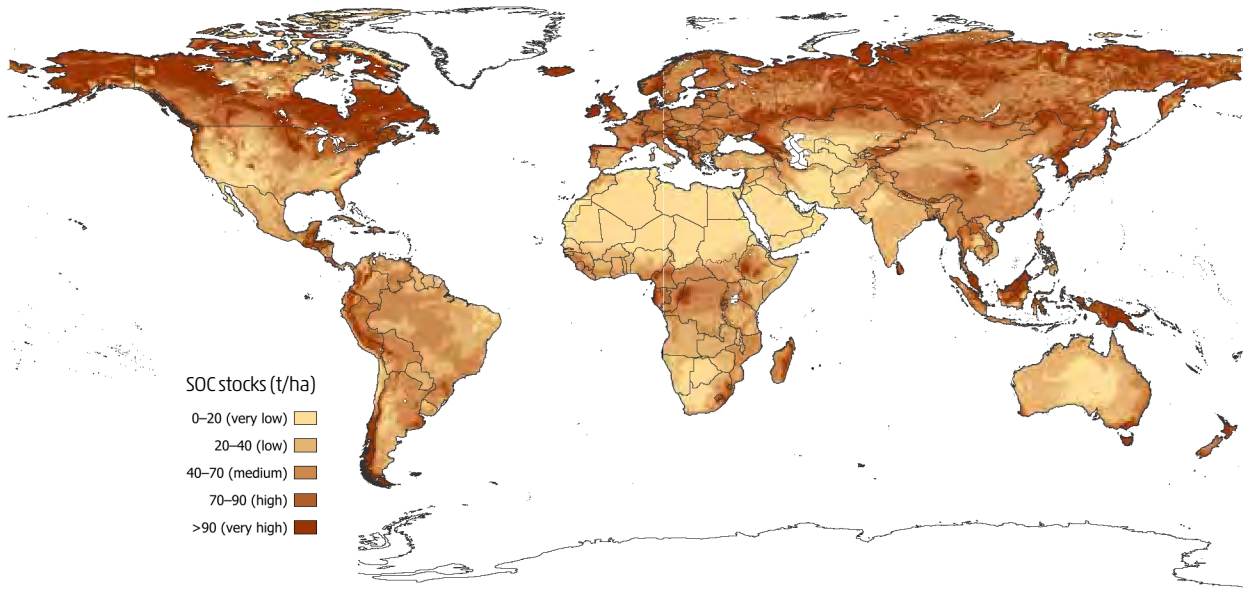


Figure 3.4. Global soil organic carbon stock (t/ha) at 1 km resolution



Refer to the disclaimer on page ii for the names and boundaries used in this map. Dotted line represents approximately the Line of Control in Jammu and Kashmir agreed upon by India and Pakistan. The final status of Jammu and Kashmir has not yet been agreed upon by the parties. Final boundary between the Republic of Sudan and the Republic of South Sudan has not yet been determined. Final status of the Abyei area is not yet determined.

◊ *Source:* **FAO.** 2023. *GLOSIS: Harmonized World Soil Database*. [Accessed 15 April 2025] https://data.apps.fao.org/glosis/?lang=en. License: CC-BY-4.0.

Soil organic carbon losses in four distinct kinds of soil will be examined in the following sections:

Mineral soils, where SOC is intermixed with mineral soil particles to form organically-enriched mineral soil horizons (Section 3.3.2).

Wetland soils, where SOC occurs as organic layers (or peat) overlying mineral subsoils (peat soils are found in wetlands where oxygen-poor conditions retard the decomposition of organic matter) (Section 3.3.3).

Permafrost soils, where internal temperatures remain below 0 °C in cold regions throughout the year in at least part of the soil (such permafrost soils are of particular concern in a warming climate) (Section 3.3.4).

The **soil inorganic carbon (SIC) pool** is a potential C source and sink especially in arid and semi-arid environments (Section 3.3.5).

Although the focus is on the C content of soil, C is only one component of the overall SOM. On average, C comprises 58 percent of SOM, while other important chemical constituents include nutrients such as N and P as well as other elements such as K and S. Soil organic matter occurs in a variety of forms, from plant litter lying on the soil surface through to free organic particles formed by decomposition. The organisms responsible for SOM transformations (from microorganisms through to larger soil fauna) are themselves part of the SOM pool.

Since the nineteenth century, the scientific understanding of SOM has been based on a humification model (Lehmann and Kleber, 2015). In this view, fresh plant residues decomposition through the activity of soil organisms and most of the C is lost as CO_2. However, a small proportion of the SOM is transformed into compounds that resist further decomposition (recalcitrance), with the most recalcitrant substances persisting in soils for hundreds or thousands of years. These resistant compounds are called humic substances. At that time, the dominant simulation models, such as Century (Parton *et al.*, 1987), were based on C transfer between pools with different levels of recalcitrance. The modelled relationship between the varying recalcitrant pools and rising

temperatures (due to global warming) was based on the carbon quality temperature (CQT) theory that predicted that the more recalcitrant pools would be more sensitive to rising temperatures than the simpler pools (Lehmann and Kleber, 2015).

However, since 2015, this understanding of SOM has been challenged (e.g. Lehmann and Kleber, 2015; Lehmann *et al.*, 2020; Georgiou *et al.,* 2022). An alternative model has been explored that focuses on the ability of decomposer organisms to access SOM. In this model, SOM persists when protected from decomposers through physical isolation in soil aggregates, or through the association of SOM with charged mineral surfaces in the soil. Importantly, the biochemical nature of the SOM itself is generally not altered during decomposition. Instead, the particle size of the SOM is made progressively finer by the various decomposing organisms (although some structural changes may also occur). Differences in the biochemical nature of the SOM particles are largely inherited from the source organic material, and not through microbial transformation.

The controversy about the fundamental nature of SOM remains unresolved but does not alter well-established relationships that are critical for SSM, although it does impact our ability to simulate future changes in SOM and SOC levels in soils. For example, Wieder *et al.* (2019) examined the effect of climate change on SOC levels and found wide discrepancies between models, especially for Arctic regions. It was concluded that uncertainties around how the physical and chemical properties of soil limit microbial access to organic material released during the decomposition of thawed permafrost limit the usefulness of the current generation of global soil biogeochemical models.

3.3.2 Soil organic carbon loss in mineral soils: Definition and processes responsible for loss

The conversion of land from native vegetation to a managed state is the dominant driver of losses of SOC. Under native vegetation or permanent pastures, SOC reaches an equilibrium with regional environmental factors, such as climate and vegetation, and local factors, such as water redistribution by topography. A balance is created between the inputs of organic material and the outputs by decomposition of the input organic materials (as well as the native SOM), by soil organisms and losses to soil erosion. Conversion from native vegetation to arable agriculture upsets the balance and typically depletes SOC, due to:

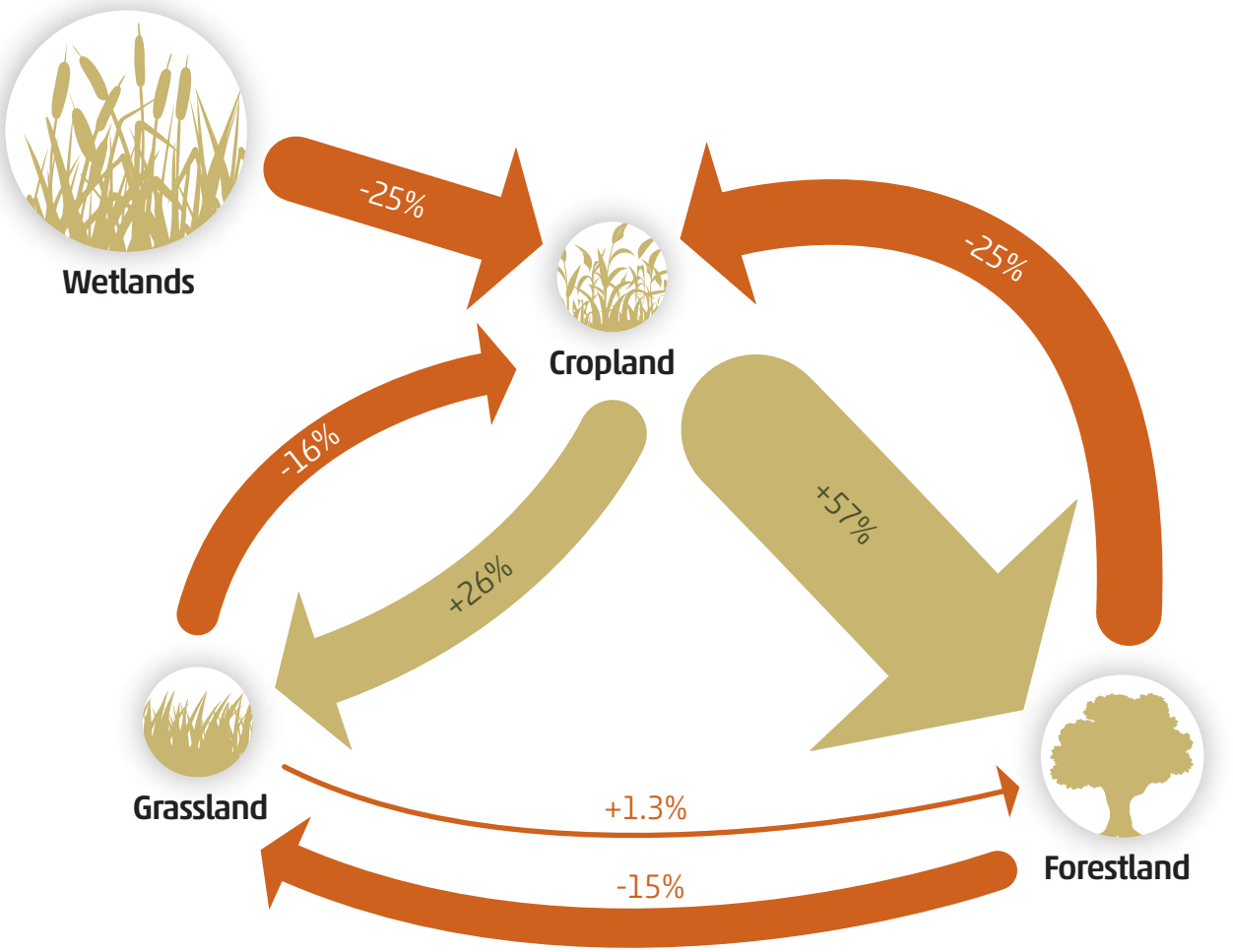
- lower returns of plant litter to the soil after conversion;
- the breakup of soil aggregates and increased aeration (due to tillage in agriculture or site preparation in forestry), which makes locked-up organic materials more accessible to decomposition;
- conversion-induced increases in erosion (which may preferentially remove organic-rich material from the soil) (Janzen, 2006; Haddaway *et al.*, 2017; Mayer *et al.*, 2020); and
- the addition of fertilizers that provide nutrients which enhance microbial activity.

Land-use conversion from a highly disturbed state, such as arable agriculture, to a land use with semi-permanent to perennial vegetation, such as plantation forestry, has the capacity to increase SOC by increasing plant litter inputs, restoring stable aggregates or reducing erosion. These conversions are summarized in Section 5.3.1.

The magnitude of the loss of SOC has been examined in several meta-analyses. The highest relative losses are for conversion of primary forest to cropland, which typically result in a reduction of about 25 percent of the original SOC (summarized in Smith *et al.* 2016). A subsequent meta-analysis of paired, native-cultivated sites by Sanderman, Hengl and Fiske (2017) found median losses of 26 percent of SOC in the top 0–30 cm of soil (140 sites) and 18 percent losses in the 30–100 cm soil

depth range (62 sites). The second-order meta-analysis of Beillouin *et al.* (2023) estimated that there were average losses of 25 percent from the conversion of forestland to cropland and average losses of 16 percent from the conversion of grassland to cropland (averaged over a range of soil depths) (Figure 3.5). The losses from conversion of forestland to grassland were not statistically significant.

Figure 3.5. Percentage change in soil organic carbon due to land-use change

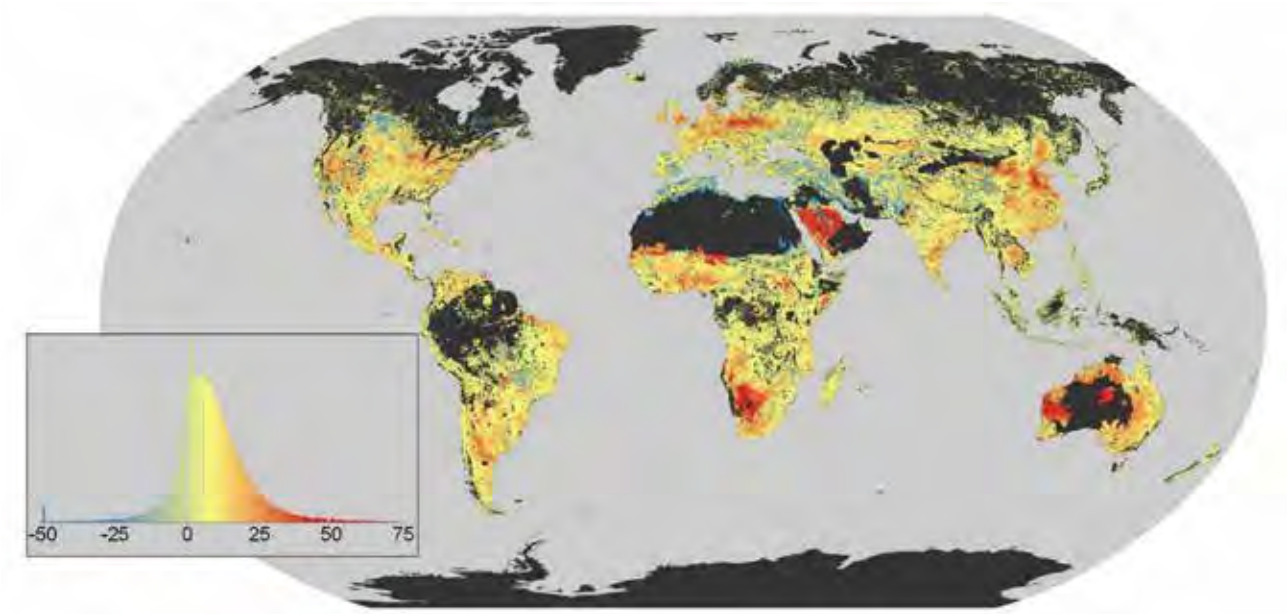


Note: The arrow sizes are proportional to the magnitude of SOC change in reference to the initial land use, with losses in orange and gains in green. NS (by the percentage value) indicates a non-significant change and an asterisk indicates a statistically significant change. The size of the land-use pictograms is proportional to their respective mean SOC stock values (t/ha).

◊ *Source:* **Beillouin, D., Corbeels, M., Demenois, J., Berre, D., Boyer, A., Fallot, A., Feder, F. & Cardinael, R.** 2023. A global meta-analysis of soil organic carbon in the Anthropocene. *Nature Communications*, 14(1): 3700. https://doi.org/10.1038/s41467-023-39338-z

The changes in stock values are important for estimates of the potential for SOC gain, as the gain potential is generally considered to be limited to some fraction of the amount of SOC lost (Janzen, 2006). Sanderman, Hengl and Fiske (2017) estimated this C debt (due to historical losses of C) at 133 billion tonnes C (Figure 3.6). A later simulation study using the Lund–Potsdam–Jena managed Land (LPJmL) biophysical model by Herzfeld *et al.* (2021) put the C debt at 215 billion tonnes C.

Figure 3.6. Percentage loss of soil organic carbon stocks at** 2 m depth **over 12 000 years of human land use



◊ *Source:* **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

Refer to the disclaimer on page ii for the names and boundaries used in this map.

Smith *et al.* (2016) modelled SOC loss due to land-use change from 1860 to 2010 based on results from three dynamic global vegetation models (DVGMs). The results across the three DVGMs differ greatly due to differences in land-use change data, land cover definitions and the processes included in the models. For example, only the Integrated Science Assessment Model (ISAM) includes wood harvest, and it shows substantial SOC loss in the southern boreal forest zone, while the other two models do not. The estimates of total SOC loss due to land-use change from 1860 to 2010 from the three models also differ substantially, from 109 billion tonnes of CO_2 (29.8 billion tonnes of C) to 221 billion tonnes of CO_2 (60 billion tonnes of C).

In a 2016 meta-analysis of absolute changes in C stocks, Deng *et al.* (2016) found that after conversion from primary forest to farmland, SOC stocks had declined on average by 21.05 t/ha (at a rate of 1.74 t/ha annually). The conversion from grassland to farmland had caused an average loss of 12.45 t/ha (at a rate of 0.89 t/ha annually). The conversion from primary forest to secondary or plantation forest had caused an average loss of 12.28 t/ha (at a rate of 0.63 t/ha annually).

Losses of SOC from forest soils have been less studied than losses from agricultural soils. Mayer *et al.* (2020) examined the effects of 13 management techniques on forest SOC levels. They summarized several meta-analyses on the afforestation of former cropland (Laganiere, Angers and Pare, 2010; Don, Schumacher and Freibauer, 2011; Poeplau and Don, 2013; Li, Niu and Luo, 2012; Bárcena *et al.*, 2014) and observed that although afforestation might result in increases in soil C stocks over 100 years, there were also examples where a loss of SOC occurred. Afforestation of former grasslands also showed variable results, with some studies reporting that up to 75 percent of all grassland-to-forest conversions caused SOC losses even after ten years of forestland use. The direction and magnitude of these changes depend on factors such as previous land use, tree species, forest age and management history, making global generalizations challenging.

Beyond land-use conversion, a major cause of SOC change in forests is due to the harvest of wood products. James *et al.* (2021) used results from a literature meta-analysis and data from the North American long-term productivity study (LTSP) to examine the effects of forest harvest and biomass removal on soil C. For the literature-based meta-analysis, SOC changes for individual and combined

soil depths (organic [O]-horizon and 0–30 cm of mineral soil) were assessed for bole-only harvest (where tree limbs and tops are left on the site), bole-only with partial or complete removal of logging slash or the O-horizon, whole-tree harvest, and whole-tree harvest with the partial or complete removal of logging slash or the O-horizon. These SOC changes were then compared to unharvested controls. Bole-only showed no significant difference from the controls. Significant differences to the controls for SOC of the combined soil depths were found for bole-only with removals (a decrease of SOC of 19.2 percent compared to control), whole-tree harvest (a decrease of 15.4 percent) and whole-tree harvest with removals (a decrease of 24.9 percent). Similar but less severe effects were seen for similar treatments in the LTSP studies.

Losses of SOC and SOM from forest soils are also associated with fires. Wildfires have an impact on several soil functions and are an increasing concern due to human-induced climate change (Box 3.2).

Box 3.2 Soil resilience in a fire-prone era

Wildfires are a natural component of the Earth's system, but climate change is driving their increased frequency and severity, profoundly impacting societies and ecosystems. Extended droughts and fire season have led to larger fires, exacerbated by human activities and land-use change. Recent years have witnessed the increasing of extreme events, including Australia's "Black Summer" (2019–2020), record-breaking fires in Canada's boreal forests (2023), and catastrophic events in Siberia, western Amazon and central Chile (Cunningham, Williamson and Bowman, 2024). While the global burned area has declined due to land use-driven reductions in the frequency of fires in African savannas, the rising wildfire activity in peatlands and less fire-adapted ecosystems overshadow these trends (Jones et al., 2024; Cunningham, Williamson and Bowman, 2024). This escalation threatens critical ecosystem services, such as carbon (C) storage and biodiversity preservation, both of which depend on healthy, resilient soils.

Although fire is a natural disturbance in many ecosystems, not all ecosystems are equally adapted to its effects (McLauchlan et al., 2020). These impacts emerge both directly during the fire, and indirectly in the post-fire period. Direct effects are primarily associated with heating and combustion, while indirect effects involve changes such as the temporary loss of vegetation, ash deposition and increased erosion rates. These processes can result in diverse outcomes, which are often context-dependent and influenced by factors such as fire severity, frequency, the specific ecosystem affected or soil type (Negri et al., 2024).

Wildfires have complex and often contrasting impacts on soils (Bowd et al., 2019; Pelligrini et al., 2018). In the immediate aftermath of a fire, ash deposits temporarily enrich soils with essential nutrients such as phosphorus (P), potassium (K) and calcium (Ca), providing a short-term boost in soil fertility. However, this benefit may be offset by post-fire challenges. On steep vegetation-deprived slopes, erosion processes can lead to a net loss of soil quality despite the initial nutrient influx. Fire severity is a critical aspect. Low-severity fires, such as controlled burns used in land management, generally have minimal long-term effects, enabling soil C storage while mitigating fire risks. Conversely, high-severity fires, especially when occurring more frequently than an ecosystem natural fire-return interval, can cause severe and lasting damage. These impacts include shifts in soil hydrophobicity, increased susceptibility to erosion and the alteration of soil C and nutrient storage, all of which degrade soil fertility and structure over time (Doerr et al., 2023; Pellegrini et al., 2018). Increased fire frequency and severity disrupt soil microbial communities and microbially mediated biogeochemical processes (Nelson et al., 2022), which compromises soil recovery and leads to long-term declines in soil productivity and ecosystem resilience.

The increasing impacts of wildfires demand the adoption of effective land management strategies that prioritize soil conservation. This is especially relevant when common post-fire interventions, such as salvage logging, can exacerbate long-term degradation in vulnerable soils (Leverkus et al., 2018; Bowd et al., 2019). Among emergency strategies, mulching has proven to be one of the most cost-effective approaches for reducing soil erosion (García-Girona et al., 2021). The promotion of nature-based solutions is key for sustainable soil recovery, with biological soil crusts recently recognized by their role as ecosystem engineers that enhance soil stability and resilience in fire-affected systems (Muñoz-Rojas et al., 2021). Further efforts should focus on testing the effectiveness of diverse post-fire management aimed at improving soil health (such as organic amendments, soil chemical improvers, herbaceous sowing, afforestation, and dry stone walls restoration)

expanding research in less studied and highly vulnerable regions, such as South America and newly fire-prone areas (Armenteras and de la Barrera, 2023; Wilkinson et al., 2023).

Fire is both a threat and a catalyst for soil processes, with impacts ranging from nutrient enrichment in low-severity events to severe degradation and nutrient loss following high-severity fires. Efforts to manage fire-prone landscapes should consider the dual role of fire: that of balancing the need to prevent severe degradation with the recognition of fire as a natural and often beneficial process. To achieve this, understanding the intricate relationships among fire, soil and vegetation is crucial. As the magnitude of fires increases, a shift toward proactive, soil-centred strategies is needed, recognizing that maintaining soil health is fundamental to ecosystem recovery and long-term resilience (McWethy et al., 2019).

◊ Sources: See References.

However, the response of forest soils to wildfire is complex. Fire can lead to SOC losses from the combustion of plant biomass and organic soil layers, and from subsequent increases in erosion and leaching, but it can also cause transformations in the ecosystem that can offset immediate C losses and ultimately stabilize or even increase ecosystem C storage (Pelligrini *et al.*, 2022; Certini, 2005; Mayer *et al.*, 2020).

Nave *et al.* (2011) completed a meta-analysis on the effects of fire on C and N storage in forest soils in temperate regions. It was found that fires significantly reduced the soil storage of C by 35 percent and N by 28 percent, principally through effects on the organic layer of fresh and decomposing organic material overlying the surface of the underlying mineral soil. While the organic layer lost 59 percent of C and 50 percent of N, the mineral soil horizons showed no significant effect of fire on C or N storage. On average, forest floor C and N storage in burned forests returned to pre-burn levels within 128 and 103 years respectively, although there was wide variation between sites in terms of time taken to return to pre-burn levels.

Soil organic carbon losses due to wildfire are also dependent on fire severity. Substantial combustion of SOM begins in the 200 to 250 °C range with complete combustion occurring at about 460 °C (Certini, 2005), although the response across biomes can differ greatly. In boreal forest soils with a prominent organic horizon, combustion of SOM by fire is a large loss pathway and can take decades to centuries to reaccumulate. However, the effects of fire on the turnover of SOC in mineral soil horizons (which have a higher stabilization potential) will be smaller (Nave *et al.*, 2011; Pelligrini *et al.*, 2022).

Overall research since 2015 has firmly established that land-use change is a major driver of SOC change and that substantial losses of approximately 15 to 25 percent of SOC occur when forests or grasslands are converted to cropland. Soil organic carbon change associated with afforestation of former cropland is uneven and can take many decades for any increases to become evident. Soil organic carbon loss from forestland from wood harvest also occurs, as well as decadal-scale losses of SOC due to wildfire.

Global and regional mapping

There has been tremendous growth in the development of regional and global maps of SOC (Hengl *et al.*, 2017; Endsley *et al.,* 2020; Stoorvogel and Mulder, 2021; Poggio *et al.*, 2021; FAO and ITPS, 2022) (see Figure 3.4), meaning that comparisons can now be made across the different maps (and the models that underlie them) at regional and global scales (e.g. Dai *et al.*, 2019; Tifafi, Guenet and Hatté, 2018; Feeney *et al.*, 2022; Georgiou *et al.*, 2021). In addition, there are global maps of SOC losses associated with land cover change (e.g. Smith *et al.*, 2016; Stoorvogel *et al.*, 2017), with erosion-induced losses of C (e.g. Carr *et al.*, 2021; Chappell *et al.*, 2019; Naipal *et al.*, 2020; Borrelli *et al.*, 2017) and the combined effects of management and climate change (Karstens *et al.*, 2022). Maps of the potential C change through either the implementation of C-sequestering management options

(Peralta *et al*., 2020) or from the effects of global climate change (Wieder *et al*., 2019; Koven *et al*., 2022) have also been produced.

Several generalizations can be made across the map products. First, the same general latitudinal and biome distribution of SOC occurs across all map products, although there are major differences across maps in the specific amount of SOC in different latitudinal zones (Tifafi, Guenet and Hatté, 2018).

Second, the different modelling approaches lead to major differences in the estimates of global C storage (Tifafi, Guenet and Hatté, 2018). These discrepancies are important in terms of the role of soils in the global C cycle but are of lesser importance from the perspective of SSM.

Third, the models generally underestimate SOC stocks when compared to field observations (Tifafi, Guenet and Hatté, 2018) especially for organic and boreal forest soils (Feeney *et al*., 2022). Stoorvogel and Mulder (2021) found that (compared to results from an ensemble of other products) the deviations of their S-World methodology were greatest in the boreal forest region, which was attributed in part to a lack of observations in the boreal zone (for example, there were only 200 sites for boreal forest taiga and 193 sites for tundra, versus 101 401 sites for tropical and subtropical grasslands, savannas and shrublands). Similar wide discrepancies in SOC stock estimates for high-latitude regions also occur for the current generation of soil biogeochemical models (Wieder *et al*., 2019).

Fourth, across the models, the estimates of SOC stores in agricultural soils are generally more consistent than for non-agricultural soils, and are in better agreement with the observational data than estimates for other environments (Stoorvogel and Mulder, 2021; Feeney *et al.* 2022; Tifafi, Guenet and Hatté, 2018). Reliable models are important for estimating the impact of SSM initiatives on agricultural soils and for estimating their C sequestration potential. In part, the better model performance stems from the greater number of observations in more intensively studied agricultural soils.

Fifth and finally, the global estimates of SOC produced by SoilsGrids (SoilGrids250m [Hengl *et al*., 2017] and SoilGrids 2.0 [Poggio *et al.* 2021]) have been shown in comparison studies of global estimates to have the closest fit to the underlying observational data (Tifafi, Guenet and Hatté, 2018; Dai *et al*., 2019). They are also readily accessible and have been used in related studies such as the global wind erosion study of Chappell *et al.* (2019).

Poggio *et al.* (2021) emphasized that the SoilsGrids predictions were not meant to be used at the subnational or local level, as national soil data providers might have access to more detailed profile data or covariate data. This is consistent with the findings at the national scale of Feeney *et al.* (2022), where estimates produced using country-specific mapping approaches had a better fit to the observational data than the global SoilGrids products. Hence the bottom-up approach used in the creation of the Global Soil Organic Carbon Sequestration Potential Map (GSOCseq) (FAO, 2022b), which draws upon the synthesis of nationally developed maps, may be better suited for national-level modelling of responses to changes in management if combined with new data collection through field observations and measurements.

Global maps of SOC losses through time have been produced using several of the approaches outlined previously, and these studies have estimated a substantial loss of SOC through time. Sanderman, Hengl and Fiske (2017) calculated SOC at a time prior to the start of agriculture (12 000 years Before Present [BP]) and then calculated losses at six time points up to 2010 based on shifts in land cover. Sanderman, Hengl and Fiske (2017) estimated a cumulative loss of 133 billion tonnes of C for the upper 2 m of soil, due to agricultural land use through time. Stoorvogel *et al.* (2017) calculated losses from a completely native state up to the 2006–2010 period. Stoorvogel *et al.* (2017) presented a global map of losses but did not provide an estimate of overall loss of SOC.

Karstens *et al.* (2022) used a modelling approach based on the IPCC methods to estimate soil loss

from 1510 to 2010 and to simulate SOC change between 1975 and 2100. They projected that most future losses would be driven by land-use change, with hotpots of losses in the northern Great Plains in Canada, the Cerrado in South America, Central Africa, Central Asia, India, northeastern China and southeastern Asia. Overall, the results reinforce that the expansion of croplands had been the major source of CO_2 emissions from soil due to the removal of vegetation, the lower return of plant litter and the slow depletion of SOC stocks due to soil disturbance as a result of conversion of forests or grasslands to cropland.

The effect of soil erosion alone on SOC has been modelled by several groups involved in wind and water erosion research (see Section 3.2). By combining global maps of erosion with a global map of SOC, it is possible to make estimates of the SOC eroded from the soil surface. The two estimates of SOC loss from water erosion provided by Borrelli *et al.* (2017) and Carr *et al.* (2021) differ in coverage, with Borrelli *et al.* (2017) providing an overall global estimate, and Carr *et al.* (2021) providing global estimates for maize and wheat-growing areas only. Chappell *et al*. (2021) also presented global estimates for SOC loss through wind erosion, as well as water erosion estimates.

It is possible to compare the hotspots (i.e. areas of highest SOC loss) across the maps (Table 3.1) and the previously discussed estimates of SOC loss. Hotspots are defined for this report as locations where three or more of the studies identified high rates of SOC loss. Hotspots of SOC loss were identified in five of the seven regions used in this report (sub-Saharan Africa, Asia, Europe and Central Asia, Latin America and the Caribbean, and North America) (Karstens *et al*., 2022; Stoorvogel *et al*., 2017; Sanderman, Hengl and Fiske, 2017; Carr *et al*., 2021; Borrelli *et al*., 2017; Chappell *et al*., 2021) (Table 3.1).

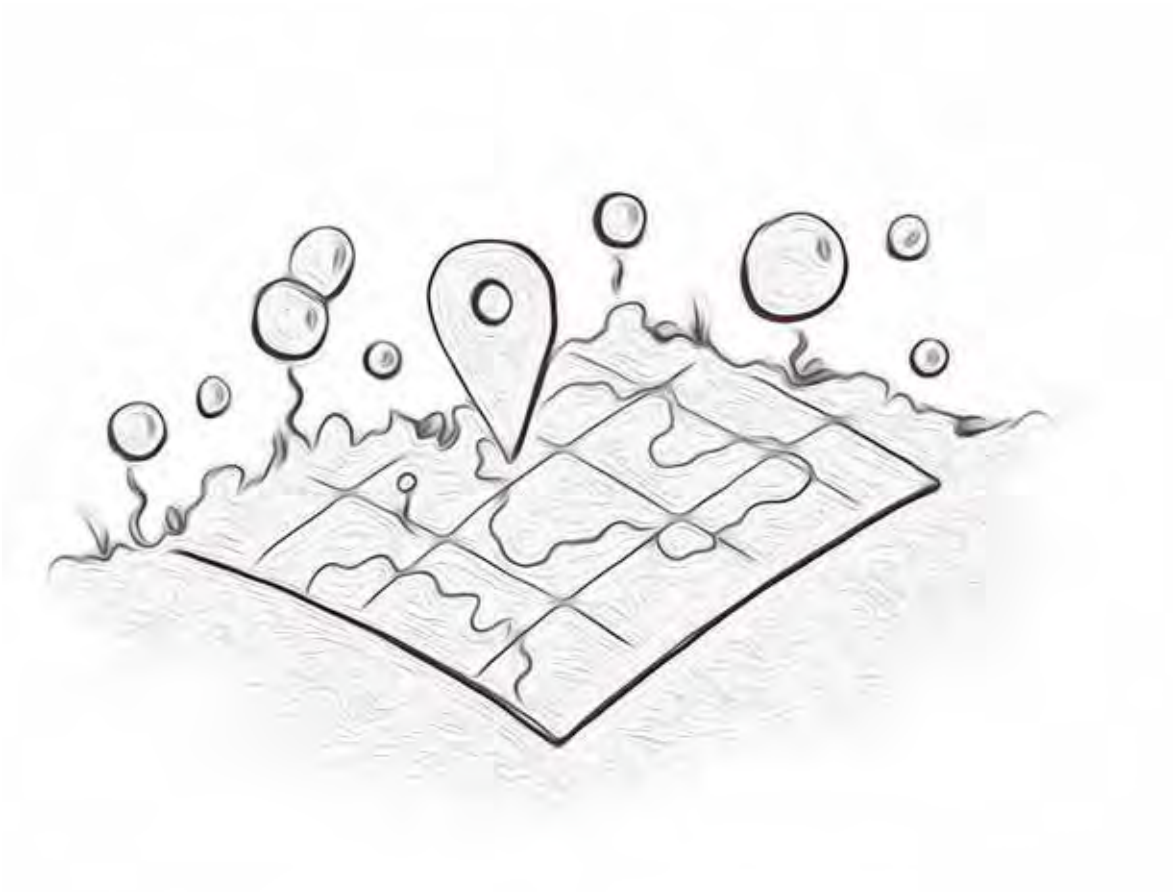


Table 3.1. Locations of high losses of soil organic carbon across studies

Region	Land-use change			Water erosion		Wind erosion
	Karstens *et al.* (2022)	Stoorvogel *et al.* (2017)	Sanderman, Hengl and Fiske (2017)	Carr *et al.* (2021)	Borrelli *et al.* (2017)	Chappell *et al.* (2019)
Sub-Saharan Africa						
Ethiopian Highlands	High (A)*	–	High	High	High	–
Rwanda and Burundi	High (R, A)	–	–	High	High	–
Madagascar	High (A)	–	–	High	High	–
Asia						
Nepal, north India and north Pakistan	–	High	–	High	High	–
Eastern Afghanistan	–	–	–	High	High	High
South central China	–	High	–	High	High	–
Northeast China	High (A)	High	High	–	–	–
Europe and Central Asia						
North central and northwest Spain	High (A)	High	–	High	–	–
Central Asia	High (A)	–	High	–	–	High
Latin America and the Caribbean						
Cordilleran Peru, Columbia and Ecuador	High (R, A)	High	–	High	High	–
Southern Brazil and Uruguay	–	–	High	High	High	–
Southern Mexico, Guatemala, Honduras and Salvador	High (A)	–	High	High	–	–
North America						
Mississippi Valley (United States)	–	–	High	High	High	–

*The (R) and (A) beside the region names for Karstens et al. (2022) indicate whether the loss is absolute (A) or relative (R). All other losses refer to absolute losses.

◊ *Sources:* **Karstens, K., Bodirsky, B.L., Dietrich, J.P., Dondini, M., Heinke, J., Kuhnert, M., Müller, C.** *et al.* 2022. Management-induced changes in soil organic carbon on global croplands. Biogeosciences, 19(21): 5125–5149. https://doi.org/10.5194/bg-19-5125-2022

◊ **Stoorvogel, J.J., Bakkenes, M., Brink, B.J.E. & Temme, A.J.A.M.** 2017. To What Extent Did We Change Our Soils? A Global Comparison of Natural and Current Conditions. Land Degradation & Development, 28(7): 1982–1991. https://doi.org/10.1002/ldr.2721

◊ **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

◊ **Carr, T.W., Balkovič, J., Dodds, P.E., Folberth, C. & Skalský, R.** 2021. The impact of water erosion on global maize and wheat productivity. Agriculture, Ecosystems & Environment, 322(3): 107655. https://doi.org/10.1016/j.agee.2021.107655

Pelligrini *et al.* (2022) mapped the vulnerability of C stocks to fire based on the distribution of topsoil SOC (at a soil depth of 0–30 cm) and fire frequency. It was found that globally, topsoil in fire-prone areas stored 460 billion tonnes of C, with 19.2 billion tonnes of C in regions that burnt annually. A global total of SOC storage was estimated at 653 billion tonnes of C (at a soil depth of 0–30 cm) with 70 percent being exposed to fire. Soil C exposed to fire was highest in the subtropics where a high frequency of fires (a return interval of 6 to 45 years) and high total burned area correlated with lower stocks of SOC.

Climate change effects on soil organic carbon change

The IPCC Sixth Assessment report (Canadell *et al.*, 2021) highlighted that SOC dynamics are critical for understanding the response of the terrestrial C stores to anthropogenic climate change. The

second largest C flux is from the soil to the atmosphere in the process of soil respiration (Bond-Lamberty *et al.*, 2024), which is an order of magnitude greater than anthropogenic C emissions (Bond-Lamberty *et al.*, 2024). Despite its importance, soil respiration remains one of the most uncertain global fluxes (Bond-Lamberty *et al.*, 2024).

Canadell *et al.* (2021) found high agreement and multiple lines of evidence indicating that warming increases decomposition rates. There was high confidence that warming would result in soil C losses relative to a constant climate and contribute to the positive C–climate feedback. There was, however, a wide spread of estimates from earth system models (ESMs) leading to low confidence in the magnitude of global soil C losses with warming.

There is considerable evidence that soil respiration has been increasing in the past few decades. Hashimoto, Ito and Nishina (2023) summarized nine data-driven studies that estimated the trend of soil respiration from 1980 to 2010, finding that eight studies showed an increasing trend through time. The trends ranged from 0.038 billion tonnes of C per year to 0.23 billion tonnes of C per year, with a mean of 0.11 billion tonnes of C per year. These values are small relative to the total annual emissions from respiration and fire of about 130 billion tonnes of C per year (Hashimoto, Ito and Nishina, 2023).

The causes of observed increases in soil respiration have been explored using modelling and experimental approaches. Ren *et al.* (2024) used observational data from long-term soil incubation experiments to constrain SOC submodels of ESMs. The constrained model was run forward to 2100 using projections of net primary productivity (NPP) and temperature and precipitation, showing that global soils would lose between 19.1 and 45.2 billion tonnes of C depending on the SSP used. These projections translate into an estimated annual soil C loss rate of 0.22 to 0.53 billion tonnes of C until the end of the twenty-first century under the different emissions scenarios.

The two main contributors to soil respiration are root and rhizospheric respiration (termed as below ground autotrophic respiration) and from organisms that decompose SOM (termed as heterotrophic respiration). Nissan *et al.* (2023) developed a mechanistic model to explore how changes in soil water content and temperature affected soil heterotrophic respiration. They estimated that from 1980 onwards, soil heterotrophic respiration showed a 0.2 percent per year increase, except in the Arctic, where increases were at approximately 0.4 percent per year.

At the experimental level, the temperature sensitivity of soil heterotrophic respiration is expressed as Q10, the factor by which soil microbial respiration increases with a 10 °C rise in temperature. Sáez-Sandino *et al.* (2023) conducted a laboratory warming experiment using soils from 332 sites across 29 countries. They found that the most important factor regulating global patterns of Q10 was the nature of the soil microbiome (microbial richness, biomass and community composition). This research links soil biodiversity directly to the response of soils to warming climates.

Overall research published since the release of the IPCC Sixth Assessment report has strengthened the finding that warming climates will lead to higher rates of soil respiration and losses of soil C from the soil to the atmosphere. The increased loss from respiration is small, with observed increases in emissions averaging about 0.1 percent per year (Hashimoto, Ito and Nishina, 2023) and modelled increases of approximately 0.2 to 0.3 percent per year (Ren *et al.*, 2024; Nissan *et al.*, 2023). The adoption of C sequestrating practices in soil (as discussed in Chapter 5 of this report) could significantly alter the magnitude of this temperature-induced trend.

Economic impact of soil organic carbon loss

The economic impact of SOM or SOC loss has not been estimated at the global or regional scale. Two costs of SOM or SOC loss need to be calculated for a complete estimate: the onsite costs associated with loss of soil productivity, and the offsite cost associated with GHG emissions and global warming (Graves *et al.*, 2015).

The onsite costs are primarily associated with two soil functions of SOM (rather than SOC alone): the release of nutrients from SOM and the water-holding capacity of soils. Soil organic matter contains nutrients such as N, P and K in relatively fixed proportions that are made available for plant growth via the microbial decomposition of SOM. Graves *et al.* (2015) calculated the N, P and K content of SOM lost in erosion and then used the price of fertilizer containing N, P and K to calculate the cost of the nutrient loss.

Soil organic matter is highly reactive and contributes to the formation of soil aggregates. As both SOM and soil aggregates contribute to the soils' ability to hold and release water (Kane *et al.,* 2021), the loss of SOM can lower the water-holding capacity and hence lower the ability of soils to supply water to crops during drought conditions. This "insurance" capacity of SOM has been demonstrated in studies in countries such as Sweden (Droste *et al.*, 2020) and the United States (Kane *et al.,* 2021).

The offsite cost associated with emissions of CO_2 during the decomposition of SOM is typically calculated by converting the loss of a unit of SOC to the equivalent in atmospheric CO_2 and then assigning a value to the emitted C. The ratio of soil C to atmospheric CO_2 is 1 to 3.67, assuming complete decomposition but Graves *et al.* (2015) used a ratio of 1 to 2.11 for soil C to CO_2 to account for the incomplete decomposition of some SOM components. Graves *et al.* (2015) cited three approaches to estimate the economic cost of the emissions:

1. using the social (damage) cost of C;
2. the marginal abatement cost of reducing emissions; and
3. the price of an emissions trading permit.

Following United Kingdom government advice, the marginal abatement cost of reducing emissions (published in 2009) was used to arrive at a price of GBP 51 per tonne of $CO_{2eq.}$ The annual cost of GHG emissions from SOC loss from England and Wales (in the United Kingdom) was estimated to be GBP 566 million, based on an annual SOC loss of 5.3 million tonnes of C. The economic loss represented 47 percent of the estimated total annual cost of soil degradation for England and Wales (Graves *et al.* 2015). While Graves *et al.* (2015) offers an interesting exercise in calculating the cost of SOC loss, the fluctuations in the price of C at national and regional levels make longer term and larger scale calculations very difficult.

3.3.3 Soil organic carbon from wetlands, peatlands and mangrove forests

Wetland soils

In every region, organic (i.e. peatlands and mangrove forests) and mineral wetland soils contain far greater stores of SOC than other soils. As with mineral soils, land-use conversion is a major source of loss of C from wetland soils. Beillouin *et al.* (2023) estimated the average losses of SOC from conversion of wetlands to cropland of 25 percent, the same as average losses from forestland to cropland (Figure 3.5).

Jackson *et al.* (2017) estimated that peatlands held 543 billion tonnes of SOC despite only covering 3 to 4 percent of the global total land area (Xu *et al.,* 2018; Minasny *et al.,* 2019). They estimated that 427 billion tonnes of SOC (or 79 percent) were in peatlands outside of the permafrost zone (Jackson *et al.,* 2017). Of the non-permafrost peatlands, Jackson *et al. (*2017) estimated that the highest levels of C storage occurred in boreal forest biomes (153 billion tonnes of C to a depth of 2 m), followed by temperate broadleaf and mixed forest biomes (132 billion tonnes of C to a depth of 2 m), tropical and subtropical moist broadleaf forest biomes (103 billion tonnes of C to a depth of 2 m), and temperate conifer forest biomes (28 billion tonnes of C to a depth of 2 m).

Hugelius *et al.* (2020) used peat core data from over 7 000 cores to estimate storage of C in peatlands of the northern latitudes. Total stores of 415 billion tonnes of C were estimated in all northern peatlands, with 239 billion tonnes in permafrost-free peatlands and 185 billion tonnes in permafrost-affected peatlands. In contrast, SOC stores in the tropics were estimated at 105 billion tonnes of C and at 15 billion tonnes of C for the extratropical Southern Hemisphere.

There are no global estimates for SOC storage in mineral wetlands. Based on a review of several regional studies, FAO and ITPS (2021) estimated that wetlands covered roughly 5 to 8 percent of the global land area but represented 20 to 30 percent of the organic C stored in mineral soils. For North America (the United States, Canada and Mexico), Kolka *et al.* (2018) estimated 2.2 million km^2 of wetlands, with 1.3 million km^2 (59 percent) as peatlands and 0.9 million km^2 (41 percent) as mineral wetlands. The great majority of C (both plant and soil) was stored in peatlands (130 billion tonnes C or 81 percent) with 19 percent (31.5 billion tonnes C) in mineral wetlands. Although C storage in mineral wetlands is considerably less than in peatlands, mineral wetlands are more actively managed (such as through drainage and cropping) than peatlands in the North America region and hence are more vulnerable to C loss than peatlands.

During the decomposition of organic matter in well-aerated soils (mineral soils with oxygen present), C is released in the form of CO_2. In oxygen-poor freshwater wetland soils, C is also released in the form of CH_4, which is a much more potent GHG (CH_4 has a global warming potential of 28 over a 100-year time horizon and 84 over a 20-year time horizon [Dean *et al.*, 2018]). According to figures summarized by Dean *et al.* (2018), natural wetlands are the source of 30 percent of total CH_4 emissions (167 million tonnes CH_4 annually out of a total of 558 million tonnes CH_4 annually). Other natural sources contribute a further 11 percent, with the remainder coming from anthropogenic sources (agriculture and waste, biomass burning and fossil fuels).

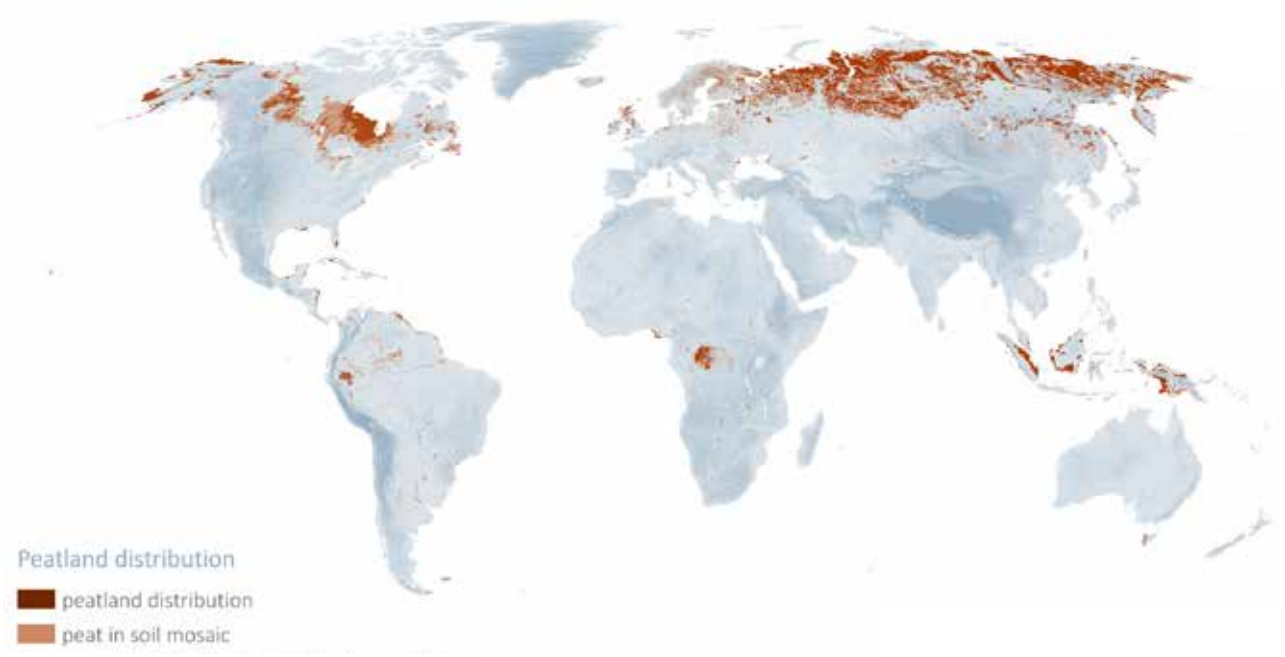
Peatlands

The greatest threats to peatlands stem from changes to the hydrological regime that cause water levels to fall. Lowering the water table allows oxygen into peat layers that were previously saturated with water and microbial decomposition of the peat accelerates greatly (Turetsky *et al.,* 2015). The drying out of the peat also greatly increases the probability of fire in the dry peat layer and the loss of C to the atmosphere. The changes to the hydrological regime that trigger the losses can result from overall global warming, but the greatest current losses are associated with deforestation and drainage from land-use conversions in tropical peatlands (Warren *et al.,* 2017; Kiely *et al.,* 2021).

The threats to peatlands differ greatly between the boreal zone and the peatlands of lower latitudes (Figure 3.7). Boreal zone peatlands are largely unmanaged, with only limited drainage due to forestry, road or infrastructure development, and oil extraction (Nelson *et al.,* 2021). In the tropical zone, the two largest peatlands of the Congo Basin (Dargie *et al*., 2019) and Indonesia (Warren *et al.*, 2017), as well as those in Brazil, Malaysia and Peru, are juxtaposed with densely populated urban areas and are subject to multiple land-use pressures (Turetsky *et al.,* 2015). Direct extraction of peat from peatland is also a significant driver of SOC loss from peatlands and accounts for about 10 percent of the loss of non-tropical peatlands (Räsänen *et al.*, 2023).



Figure 3.7. Peat distribution map



Refer to the disclaimer on page ii for the names and boundaries used in this map. Dotted line represents approximately the Line of Control in Jammu and Kashmir agreed upon by India and Pakistan. The final status of Jammu and Kashmir has not yet been agreed upon by the parties. Final boundary between the Republic of Sudan and the Republic of South Sudan has not yet been determined. Final status of the Abyei area is not yet determined.

◊ *Source:* **UNEP.** 2021. *The Global Peatland Map 2.0.* Nairobi. https://wedocs.unep.org/20.500.11822/37571

All peatlands are at considerable risk from fire. Peat fires are typically of a smouldering (flameless) nature and can persist under low temperatures, high moisture and low oxygen concentrations, able to burn for weeks or months (Turetsky *et al.,* 2015). Normally, burning is confined to the upper peat surface layers, but with falling water tables, the depth of burning is greater. In the past, tropical peatlands have generally been resistant to burning, due to high and frequent rainfall events, but drainage and disturbance have increased their vulnerability (Turetsky *et al.,* 2015; Kiely *et al.,* 2021).

Climate change and fire effects on peatlands

In peatlands the warming climate has two effects. First, the higher temperatures in the upper peat layers will increase decomposition and the release of C through microbial respiration. Second, the higher land surface temperatures will increase the incidence of wildfire and the release of CO_2 through the combustion of peat. Although there are some positive gains in C stocks that result from fire in peatlands, on balance, the cumulative effects of climatic warming, anthropogenic peatland fragmentation and subsequent peatland drying will increase C loss to the atmosphere (Nelson *et al*., 2021). Lin, Liu and Huang (2021) estimated that in 2015, the C loss from wildfires in Arctic boreal peat was about 143 million tonnes C, equivalent to 15 to 40 percent of the average C loss from global peat fires (0.35 to 1 billion tonnes C annually). Under the current boreal warming rate of 0.44 °C every decade, it was projected that fire in a warmer Arctic boreal peatland would release approximately 256 million tonnes of C annually by 2050 and 544 million tonnes of C annually by 2100.

Fire losses have been extensively documented for the tropical peatlands of Indonesia. Indonesian peatlands store about 65 percent of the tropical peatland C reservoir (Page, Rieley and Banks, 2011) with an estimated area of between 14.9 million ha and 20.9 million ha (Warren *et al*., 2017). However, extensive land clearing has occurred to cultivate oil palm or produce timber and pulp, usually with prior drainage of the peat and often with the use of fire as a land-clearing agent (Warren *et al*., 2017;

Nechita-Banda *et al.*, 2018). Warren *et al.* (2017) estimated that between 2000 to 2010, 41 percent of peat swamp forest cover was lost in Sumatra and 24.8 percent lost from the island of Borneo.

Major fire seasons occurred in Indonesia in 1997 and 1998 and in 2015 (Kiely *et al.*, 2021) The 1997 and 1998 fires released approximately 0.95 billion tonnes of C, equivalent to approximately 15 percent of global fossil fuel emissions at that time (Turetsky *et al.,* 2015). Using satellite monitoring, Nechita-Banda *et al.* (2018) estimated a total C release of 0.35 to 0.60 billion tonnes from the 2015 Indonesian fires. Kiely *et al.* (2021) estimated economic losses totalling USD 28 billion from the 2015 fires, composed of a USD 9.4 billion loss due to damages to agriculture, plantations, natural forests and other land covers, a loss of USD 11.3 billion associated with the CO_2 emissions (962 million tonnes of C) from the fire, and a loss of USD 7.3 billion from the health care costs of exposure to the haze from fires.

The vast peatlands of the Cuvette Centrale in the central Congo Basin span both the Congo and the Democratic Republic of the Congo. The extent of the peatlands has been recently estimated at approximately 145 500 km^2, with estimated C stores of approximately 31 billion tonnes C (Dargie *et al.,* 2017). Unlike the Indonesian wetlands, the development pressures (as of 2019) are limited to small-scale forestry and agricultural activities that are largely confined to *terra firma* forest (Dargie *et al.,* 2019).

Mangrove forests

Mangrove forests occur on shallow, tidally inundated coastal margins throughout the tropics, extending to latitudes of about 35 degrees north and south. Brazil, Indonesia, Malaysia and Papua New Guinea contain more than 50 percent of the world's mangrove C stock, with Indonesia accounting for more than 30 percent of the world's mangrove C stock (Hamilton and Friess, 2018).

Mangrove forests are dominated by tree and shrub species that can survive under conditions of high salinity and are important habitats for marine and terrestrial species. The soils under the mangroves are composed of sediments with high levels of C storage. Mangroves contain approximately four times as much C density as tropical, temperate or boreal forests (Taillardat, Friess and Lupascu, 2018; Hamilton and Friess, 2018). Due to complex root structures, high sedimentation rates, waterlogged soils that are free from risk of fires, and oxygen-poor soils, soil C turnover rates are a thousand times slower than those in terrestrial forests (Atwood *et al.*, 2017). There have been many estimates of C storage and rates of C accumulation in mangrove forests. For example, Adame *et al.* (2021) synthesized a range of studies and estimated a mean SOC in the top 1 m of soil of 331 ± 75 tonnes of C/ha and 647 ± 151 tonnes of C/ha in the top 2 m of soil. The mean above ground C was estimated at 101 ± 94 tonnes of C/ha.

Soil organic carbon loss from mangrove deforestation is associated with five major drivers (Adame *et al.*, 2021), as follows:

1. conversion to aquaculture and agriculture;
2. coastal erosion;
3. extreme climate events;
4. mangrove clearing (due to logging and other activities); and
5. human settlements.

The dominant drivers differ between countries. For example, in Southeast Asia, the major driver of deforestation in the 1980s and 1990s was due to the expansion of aquaculture (Richards and Friess, 2016), a trend that has continued, particularly in Kalimantan and Sulawesi. The expansion of rice cultivation is a major driver in Myanmar (which has the fastest rate of mangrove deforestation in Southeast Asia), while the rise of oil palm plantations is the major driver in Borneo, Malaysia and

Sumatra (Richards and Friess, 2016).

Hamilton and Friess (2018) estimated that mangrove C stocks (to a 1 m depth) lost 2 percent of their C stores between 2000 to 2012, equivalent to a maximum potential emission loss of 0.9 million tonnes of C. Adame *et al.* (2021) estimated total cumulative emissions from mangrove loss from 2010 to 2100 of 653 million tonnes of C.

3.3.4 Soil organic carbon loss in permafrost soils

Permafrost is soil, sediment or rock material that is permanently exposed to subzero temperatures for at least two consecutive years (Tarnocai *et al.*, 2009). Permafrost soils occur in mineral (<20 percent C) and organic (or peat) (>20 percent C) soils (there is overlap with the peatlands discussed in Section 3.3.3 and organic permafrost soils). Permafrost is found in Antarctica and mountain regions, but the vast majority occurs in the Northern Hemisphere (Figure 3.8).

Soil organic carbon storage in both mineral and organic permafrost soils is high (1 200 billion tonnes of C (Canadell *et al.*, 2021), as both the freezing temperatures and the lack of liquid water suppress the microbial decomposition of organic material. Soil organic carbon storage in permafrost soils can also occur to greater depths than in non-permafrost soils.

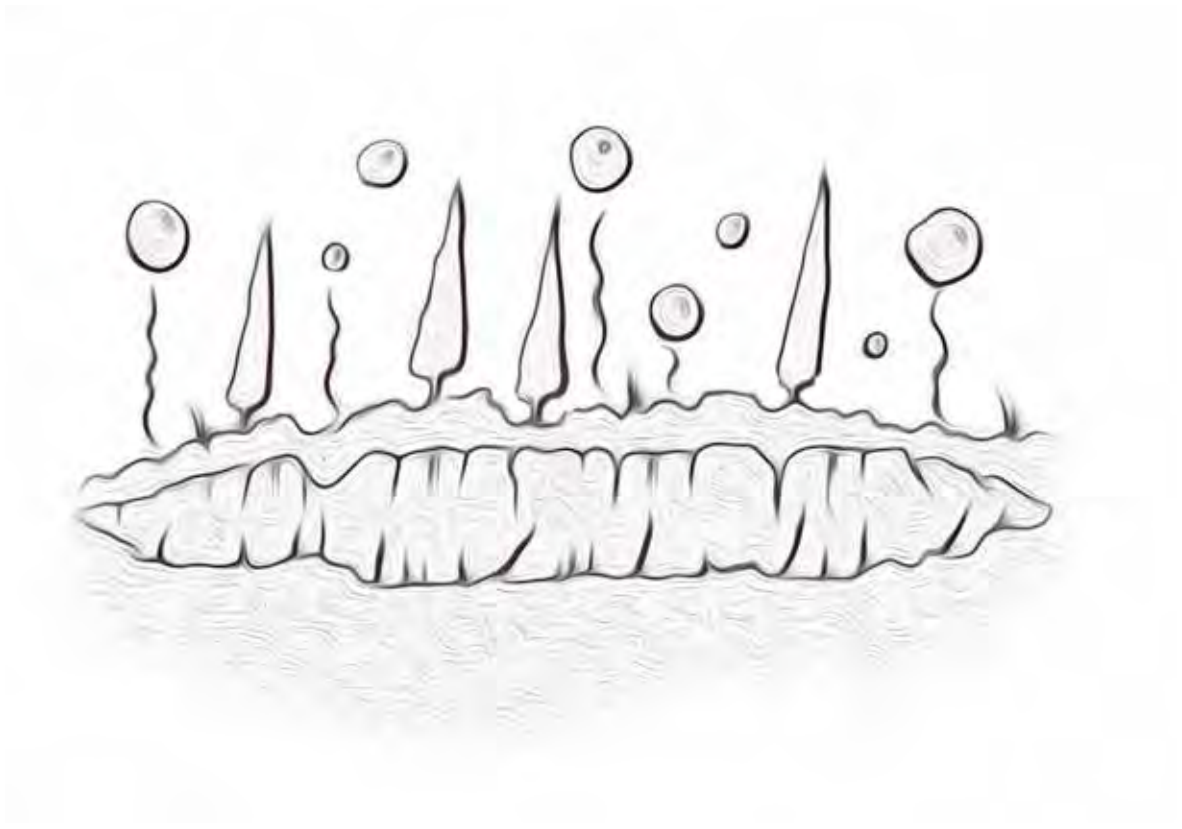
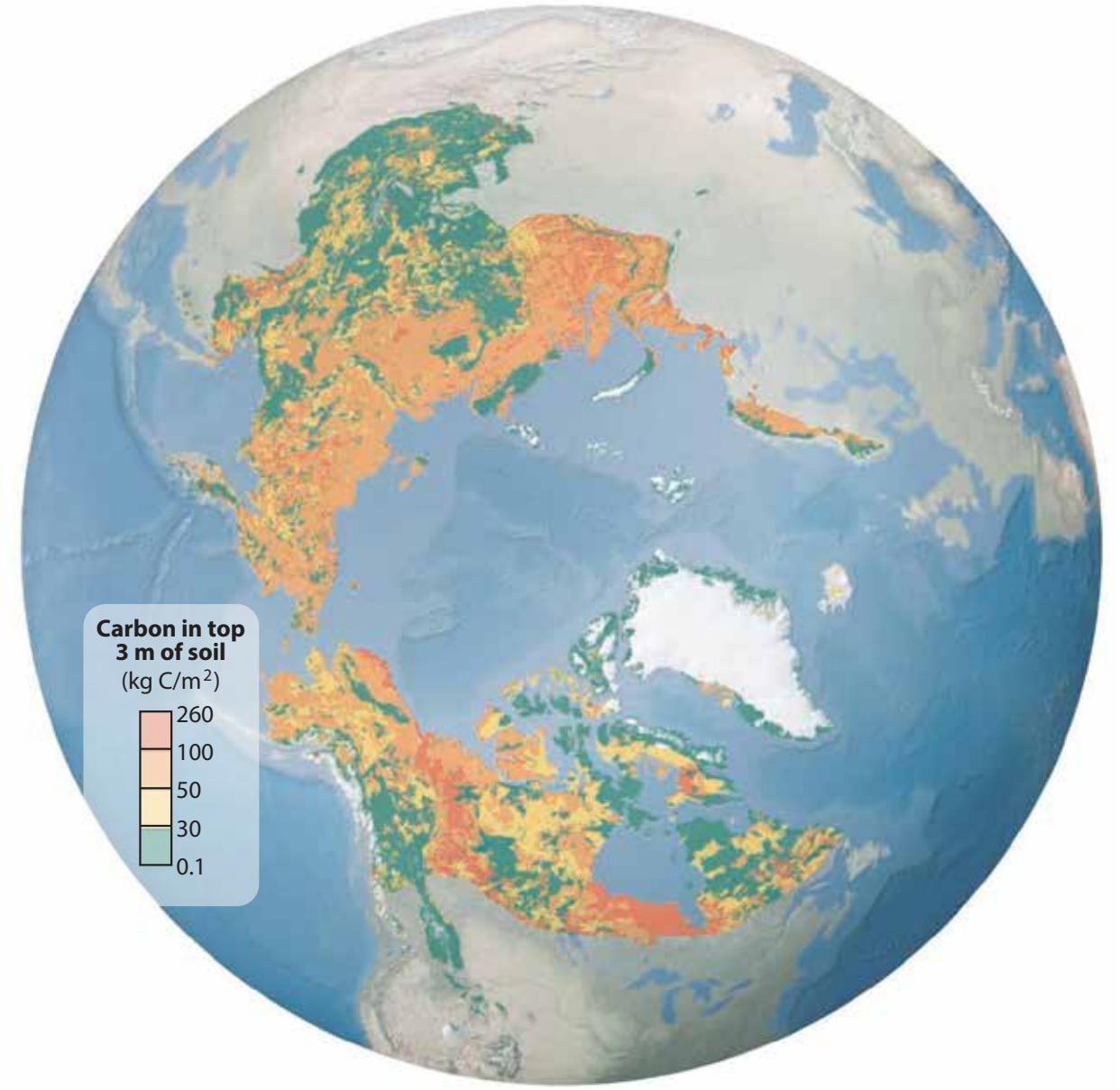


Figure 3.8. Organic carbon content (0–3 m depth) of permafrost soils in the northern circumpolar permafrost region



Refer to the disclaimer on page ii for the names and boundaries used in this map. Dotted line represents approximately the Line of Control in Jammu and Kashmir agreed upon by India and Pakistan. The final status of Jammu and Kashmir has not yet been agreed upon by the parties. Final boundary between the Republic of Sudan and the Republic of South Sudan has not yet been determined. Final status of the Abyei area is not yet determined.

◊ *Source:* **Schuur, E.A.G., Abbott, B.W., Commane, R., Ernakovich, J., Euskirchen, E., Hugelius, G., Grosse, G.** *et al.* 2022. Permafrost and Climate Change: Carbon Cycle Feedbacks From the Warming Arctic. *Annual Review of Environment and Resources*, 47(1): 343–371. https://doi.org/10.1146/annurev-environ-012220-011847

Soil organic carbon storage in permafrost soils is at risk due to the effects of global warming caused by anthropogenic climate change. The surface layer of permafrost soils thaws seasonally and the decomposition of organic materials occurs. However, increasing temperatures cause the depth of surface thaw to increase and hence a greater volume of SOM is subject to decomposition (Dean *et al.*, 2018). Increased thawing can also cause land surface collapse and the creation of water-filled hollows in the peatlands (called thermokarst processes) (Schuur *et al.*, 2022). Thermokarst collapse exposes deeper layers of permafrost to thaw and can lead to the decomposition of otherwise inaccessible SOM.

Given the high C storage and sensitivity to global warming, C dynamics in permafrost soils are a major focus of the IPCC. However, permafrost soils are not directly managed by soil users in the

same way as agricultural or forest soils and the risks to permafrost soils stem from the changing climate and not from unsustainable soil management practices. Therefore, while this section will briefly summarize the information available on the threats to permafrost soils, the brevity reflects the lack of a role for SSM, not the gravity of the threat itself.

A second threat stemming from thawing of permafrost is on human infrastructure. The loss of ground ice through melt destabilizes the near-surface layer of the soil and reduces the structural support for infrastructure (Hjort *et al.*, 2018; Liu *et al.*, 2021). Hjort *et al.* (2018) estimated that by 2050, 3.6 million people (three-quarters of the current population of the Northern Hemisphere permafrost area) might be affected by the damage to infrastructure caused by permafrost melt. Approximately 70 percent of the current pan-Arctic infrastructure is in areas where near-surface permafrost was projected to thaw by the mid-twenty-first century. Railways such as the Qinghai–Tibet and Obskaya–Bovanenkovo lines would be especially at risk along with oil and gas pipelines in Alaska and Siberia.

Most permafrost soils occur in northern Canada, the Russian Federation and Alaska (the United States) (Figure 3.8). Hegelius *et al.* (2014) estimated the stocks of circumpolar permafrost C, based on the upscaling of regional soil maps. The storage of SOC in the top 0–3 m of soils was estimated to 1 035 ± 150 billion tonnes, with 217 ± 12 billion tonnes for the 0–30 cm soil depth and 472 ± 27 billion tonnes for the 0–100 cm soil depth (with ± 95 percent confidence intervals). Estimates of SOC storage below 3 m have greater uncertainty associated with them, so therefore it was estimated that the total SOC storage for the permafrost region was ~1 300 billion tonnes with an uncertainty range of ~1 100 to 1 500 billion tonnes. Schuur *et al.* (2015) stated that the 0–3 m storage accounted for approximately 34 percent of global SOC stores although comprising only 15 percent of the global soil area. Jackson *et al.* (2017) calculated global SOC storage to a 2 m depth and estimated that in permafrost areas, 80 percent of SOC storage was in mineral permafrost soils and 20 percent was in peat permafrost soils.

The threat of the warming global atmosphere to permafrost has been well documented. Biskaborn *et al.* (2019) drew upon borehole data from the Global Terrestrial Network for Permafrost (GTN-P) (Biskaborn *et al.*, 2015) to estimate changes in permafrost temperatures during the 2005–2016 period. It was estimated that, on average, permafrost warmed by 0.33 ± 0.16 °C in northern Asia over the period, by 0.23 ± 0.11 °C per decade in North America, and the Antarctic permafrost warmed by 0.37 ± 0.10 °C per decade. Mountain permafrost in the data set was mainly represented by boreholes in the European Alps, the Nordic countries and central Asia. Using these data, it was estimated that mountain permafrost temperature increased by 0.19 ± 0.05 °C per decade. The decadal change rates of air temperature were estimated to 0.86 ± 0.84 °C during 2005 and 2016 in the Arctic continuous permafrost zone, 0.63 ± 0.91 °C per decade in the Arctic discontinuous permafrost zone, and 0.1 ± 0.50 °C per decade in mountain permafrost.

The impact of warming temperatures on future permafrost thawing has also been estimated through models. For example, Guo and Wang (2017) used air and soil temperature data from ten coupled climate models to estimate permafrost loss associated with a 2 °C global warming. The models predicted that the average soil temperature at a 6 m depth would increase by between 2.34 to 2.67 °C relative to 1990–2000, with the increase intensifying with increasing latitude. The southern boundary of the permafrost zone would retreat northwards (Guo and Wang, 2017), with the total area of permafrost decreasing by 24 to 26 percent, and the ground settlement (owing to permafrost thaw) estimated at between 3.8 and 15 cm on area average (ground settling is the major cause of infrastructure disturbance for transport, pipelines and human settlements). Liu *et al.* (2021) estimated that under the highest emissions scenario (RCP8.5), only 24.1 ± 15.2 percent of the current permafrost area would persist at the end of the twenty-first century.

Schuur *et al.* (2022) compiled a comprehensive range of modelling and expert assessment estimates of cumulative net soil C pool change published since 2015. The results were used to present a

summary table of estimated emissions under different scenarios based on the Representative Concentration Pathways (RCPs) and of different rates of plant and soil response (Table 3.2). These were compared to national-level emissions over the same period (2000–2099). The highest level of cumulative permafrost emissions was equivalent to 100 years of 2019-level emissions from the OECD countries in Europe and the United States combined. The authors also stated that the idea of an abrupt "methane bomb" of massive levels of CH_4 emissions over a few years was not supported by current evidence.

Table 3.2. Low, medium, and high levels of net carbon dioxide and net additional methane emissions to the atmosphere in the 2000–2099 period under various global warming scenarios

Arctic carbon emissions scenarios	Annual emissions			Cumulative emissions
	2021	2049	2099	2000–2099
	Billion tonnes C annually			Billion tonnes C
Low CO_2	0.332	0.374	0.449	37
Medium CO_2	0.344	0.736	1.436	74
High CO_2	0.630	1.470	2.970	149
	Million tonnes C annually			Million tonnes C
Low CH_4	5	11	21	1 090
Medium CH_4	12	26	51	2 575
High CH_4	22	50	100	5 050

◊ *Source:* **Schuur, E.A.G., Abbott, B.W., Commane, R., Ernakovich, J., Euskirchen, E., Hugelius, G., Grosse, G.** *et al.* 2022. Permafrost and Climate Change: Carbon Cycle Feedbacks From the Warming Arctic. Annual Review of Environment and Resources, 47(1): 343–371. https://doi.org/10.1146/annurev-environ-012220-011847

Biogeochemical modelling of ecosystem C dynamics (i.e. soil plus vegetation) has yielded a range of outcomes. For example, Wieder *et al.* (2019) cited previous studies showing that with warming and elevated concentrations of CO_2 in the atmosphere, high-latitude ecosystems would sequester additional C through the end of the twenty-first century due to increases in plant productivity offsetting accelerated decomposition rates. They emphasized that the major discrepancies between the models highlighted knowledge gaps, the need to experimentally evaluate the individual and combined effects of warming and elevated CO_2 on plant productivity and the persistence and decomposition of SOC in the permafrost regions.

Recent modelling of C trends using an ensemble of ESMs by Liu *et al.* (2022) supports the contention of higher plant productivity offsetting the loss of SOC. The outputs of 13 ESMs from phase 6 of the Coupled Model Intercomparison Project (CMIP6) were used to evaluate the dynamics of terrestrial C storage across the northern Asian permafrost region (containing approximately 53 percent of Northern Hemisphere permafrost). Four shared socioeconomic pathway scenarios were used for the 1900–2100 period. Across the 13 models, there was a predicted mean loss of SOC of −3.9 billion tonnes of C (or 1.1 percent of SOC stores averaged over the 1995–2014 period) by the end of the twenty-first century, and mean litter losses of 0.03 billion tonnes C (or 0.2 percent of stores in litter). The C losses are, however, more than offset by mean gains 21.1 billion tonnes of C (95.3 percent of current stores) due to the enhanced growth of vegetation.

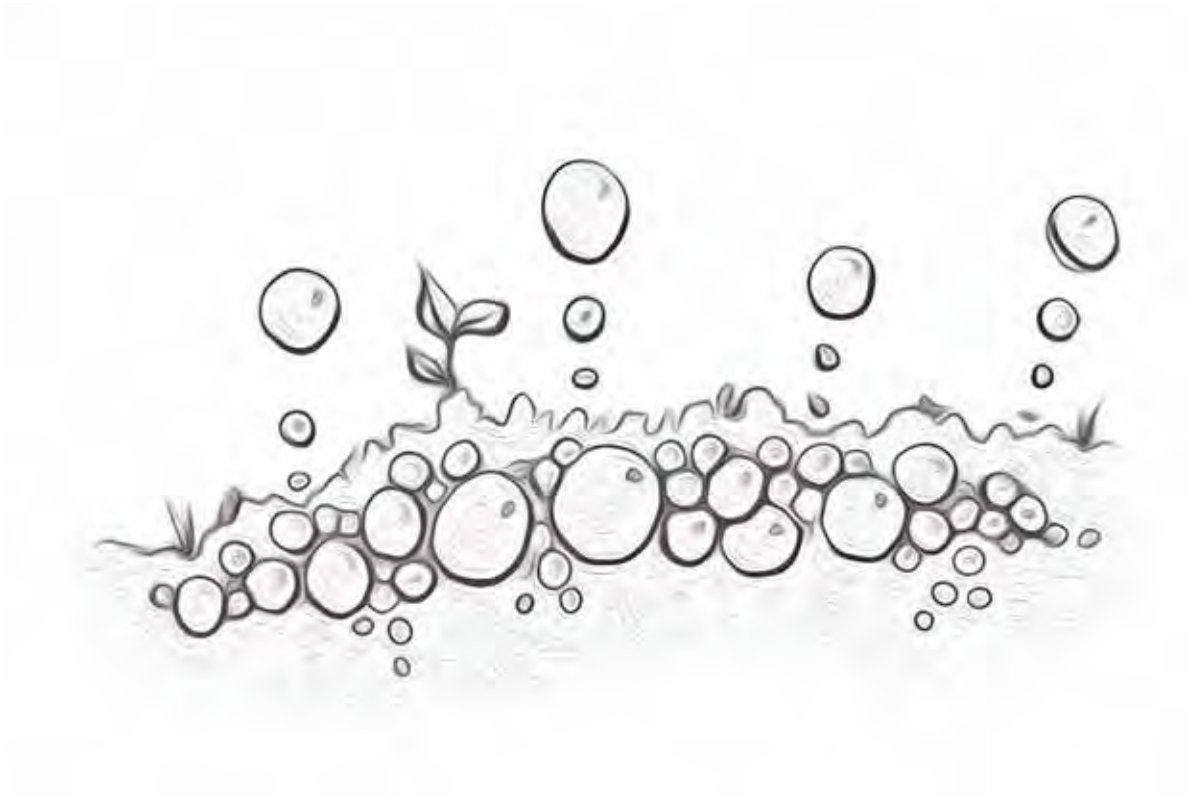
A final threat to SOC stores in permafrost soils is an increased incidence of wildfire in the active layer. Currently, tundra biomes have a long fire return interval (a mean of 212 years) (Pellegrini *et al.*, 2022). Warming temperatures (especially if combined with decreased precipitation) can create drier active layers and could also lead to the northern expansion of the boreal forest into the tundra, increasing the possibility of C release through combustion (Senande-Rivera, Insua-Costa and Miguez-Macho, 2022).

3.3.5 Soil inorganic carbon loss

Research since 2015 has also begun to examine the role of loss of C from the SIC pool. The pool of C in the SIC pool in the upper 2 m of soil is roughly equal in size to the organic C pool (Zamanian, Zhou and Kuzyakov, 2021). Most SIC occurs in calcium carbonate ($CaCO_3$) and magnesium carbonate ($MgCO_3$) (Naorem *et al.,* 2022). Traditionally, this pool has been viewed as relatively inert and only subject to change over long time scales. The consideration of SIC is especially important in arid and semi-arid regions as SIC is the main form of C in these regions (Naorem *et al.,* 2022).

Soil inorganic carbon is, however, subject to release to the atmosphere as CO_2 due to acidification, although the processes leading to release are complex and may be considerably delayed in time. In cropland, the addition of N fertilizers is a major source of acidification. Nitrogen is commonly bound to hydrogen (H) in fertilizers (such as ammonium [NH_4]), and when fertilizer is solubilized in the soil, when the H^+ ions cause acidification and release of C from carbonates. Zamanian, Zhou and Kuzyakov (2021) estimated that about 40 percent of the dissolved SIC-reformed carbonate minerals in the soil, with the remainder being lost as CO_2. It was estimated that approximately 0.41 billion tonnes of C had been lost from soils since 1970 and that a further 1.13 billion tonnes of C would be lost by 2050 due to fertilization-induced acidity.

Carbon dioxide release from non-agricultural soils may also occur due to atmospheric deposition of N compounds in non-agricultural landscapes. Song *et al.* (2022) used datasets from China to estimate the loss of SIC stocks from China in the period from the 1980s to the 2010s and found that total SIC stocks had decreased by 1.37 ± 0.37 billion tonnes of C. The losses of SIC were similar across China's major land uses. China's estimated annual CO_2 emissions associated with the SIC loss were 58.7 million tonnes annually in cropland, 28.1 million tonnes annually in forestland and 2.4 million tonnes annually in grassland. Song *et al.* (2022) and Naorem *et al.* (2022) concluded that the feedback of SIC to the atmospheric C pool should be included in the considerations of the C sequestration potential of soils.



3.4 Soil biodiversity change

3.4.1 Introduction

The period since 2015 has seen a tremendous expansion of the literature on soil biodiversity. Two major global syntheses were produced: the *Global Soil Biodiversity Atlas* (Orgiazzi *et al.,* 2016) and the *State of knowledge of soil biodiversity* (FAO *et al*., 2020). These works present thorough reviews of the nature and taxonomy of soil organisms, the roles they play in the delivery of ecosystem services and the threats that endanger life in the soil.

Calls for the adoption of soil health as a focus for both scientific research and applied science have been increasing since 2015. Although no single definition of soil health has been widely adopted, an emphasis on life in the soil is common to all.

Changes to the biodiversity of a class of organisms are usually assessed by measuring the range of taxonomic units of the class that are present (richness), and, in some cases, then assessing the proportional representation of the different species (evenness). Alpha diversity refers to the richness or evenness within a single site or habitat whereas beta diversity refers to the richness or evenness assessed across two or more sites or habitats (Morris *et al*., 2014). Generally, in biodiversity, the taxa being assessed are described down to species level, but for microorganisms, the species concept is not relevant, so operational taxonomic units (OTUs) based on genomic characterization are used instead. Operational taxonomic units are an example of a phylotype, where an organism is classified on the basis of its genetic makeup rather than its physical traits (as with higher-order organisms). Many studies use a composite diversity index such as the Shannon Diversity index that incorporates information about both richness and evenness (Morris *et al*., 2014).

Threats to organisms can also be assessed using other measures beyond changes in soil taxonomic diversity. Biomass studies focus on the mass of specific groups of organisms present in the soil and can be used for all soil organisms, from bacteria to larger soil-dwelling animals. A reduction in biomass due to a particular threat is viewed as a negative impact of that threat. In toxicological testing, toxicological endpoints are measurements made during or at the end of a test that measure the effect of the added potential toxin. Commonly used endpoints include measures of survival, growth and reproduction, as well as the community effect, such as respiration or photosynthesis. Endpoints are commonly used to assess toxic effects on soil invertebrates (Gunstone *et al*., 2021).

The focus of this section is on global assessments of soil biodiversity and the drivers that control the distribution of organisms. The effects of agriculture on soil biodiversity loss are the most thoroughly studied of the major threats to soil biodiversity.

3.4.2 Global assessments of soil biodiversity

Soil is the most biodiverse system on earth, containing an estimated 59 percent of all the biodiversity on the planet (Anthony, Bender and Van de Heijen, 2023). There have been several initiatives to map the global distribution of specific soil organisms and assess the factors that control their distribution since 2015. For example, Bahram *et al.* (2018) assessed the global distribution of bacteria and fungi (known as the microbiome), finding that the diversity of bacteria was greatest in the mid-latitudes, whereas the latitudinal pattern of fungi was weaker, with fungal diversity being lowest in the mid-latitudes. The global distributions of soil fungi and bacteria were most closely associated with soil pH and precipitation, respectively. Delgado-Baquerizo *et al.* (2018) focused on dominant (i.e. the most abundant and ubiquitous) bacterial phylotypes from 237 locations across six continents and found that only 53 percent of the dominant bacteria showed strong habitat preferences.

Global fungal diversity was examined in a sampling study from over 4 000 global topsoil samples (Mikryukov *et al.*, 2023). The total number of fungal OTUs (alpha diversity) was highest in tropical and temperate ecosystems, with diversity hotspots in the African highlands, tropical forests of the Gulf of Guinea, Appalachian forests in the United States, central American dry forests, and the Himalayan and New Guinean forests. "Coldspots" of low diversity occurred in deserts and Arctic and Antarctic ecosystems. Agricultural expansion and urbanization have led to pronounced declines in diversity.

Global maps of larger soil organisms such as nematodes and earthworms have been produced since 2015. Van den Hoogen *et al.* (2019, 2020) mapped the global distribution of nematodes and showed that nematode abundance was highest in subarctic regions, including the boreal forest and tundra ecological zones. Nematode abundance was strongly positively related to SOC contents and cation exchange capacity (CEC) and negatively related to pH. Phillips *et al.* (2019) assessed the global distribution of the number of earthworm species, their abundance and biomass. High species richness and abundance were reported at locations across the temperate regions of the mid-latitudes but there were legacy effects from the extirpation of earthworms from some regions in the Northern Hemisphere by the continental glaciations. There was no strong association found between soil properties and earthworm communities, but it was suggested this could be due to scale mismatches between the local earthworm data and global soil databases.

The collective understanding of the distribution of soil organisms and the controls on that distribution has improved substantially since 2015. Studies such as the pan-European examination of effects of pH and land cover on soil microbial diversity by Labouyrie *et al.* (2024) need to be repeated in many locations to provide the foundations of good policymaking to address soil biodiversity protection.

3.4.3 Global threats to soil biodiversity

As with above ground organisms, habitat modification or loss is the greatest threat to soil organisms (Bach *et al.*, 2020). The largest driver of habitat loss is the conversion of forests and grasslands to agriculture, and many studies have shown that agricultural fields support smaller and less diverse soil communities than grasslands and forested areas (Bach *et al.*, 2020; Tsiafouli *et al.*, 2015; FAO *et al.*, 2020) (see Section 4.2.1). Wildfires also have an impact on soil biodiversity (Box 3.2).

The only major attempt to map threats to soil biodiversity was produced as part of the *Global Soil Biodiversity Atlas* (Orgiazzi *et al.,* 2016). The map of potential threats (Figure 3.9) was based on a survey of experts in the field that identified potential threats to soil biodiversity before using simple proxies to spatially represent each of the selected potential threats. The potential threats were the loss of above ground biodiversity, pollution and nutrient overloading, agricultural use, overgrazing, fire, soil erosion, land degradation (in this case desertification), and climate change (using a global aridity index). All of these datasets were harmonized on a scale of 0 to 1 and the total scores were categorized as very low, low, moderate, high or very high (Figure 3.9).

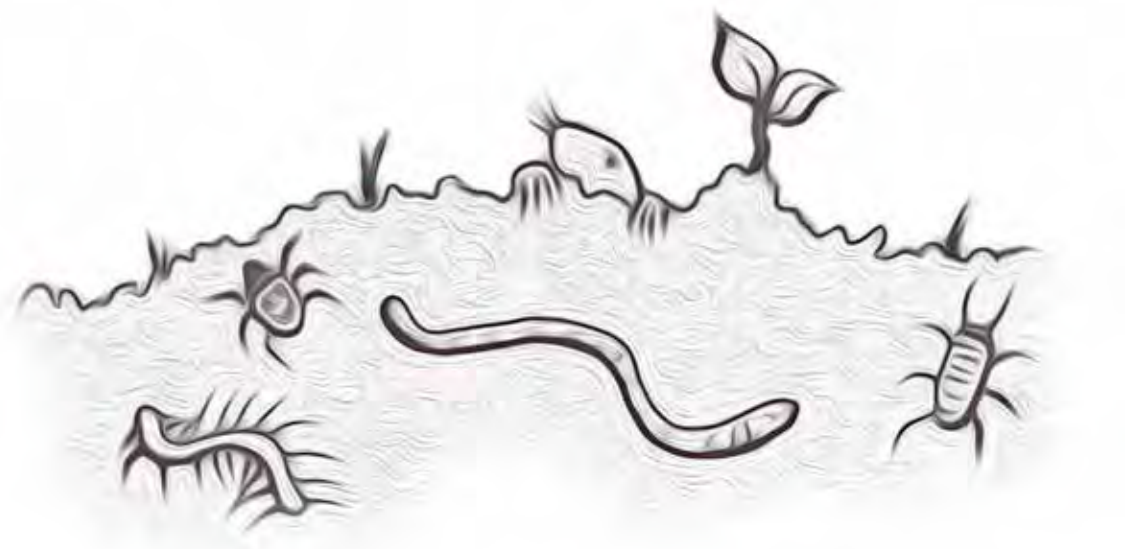
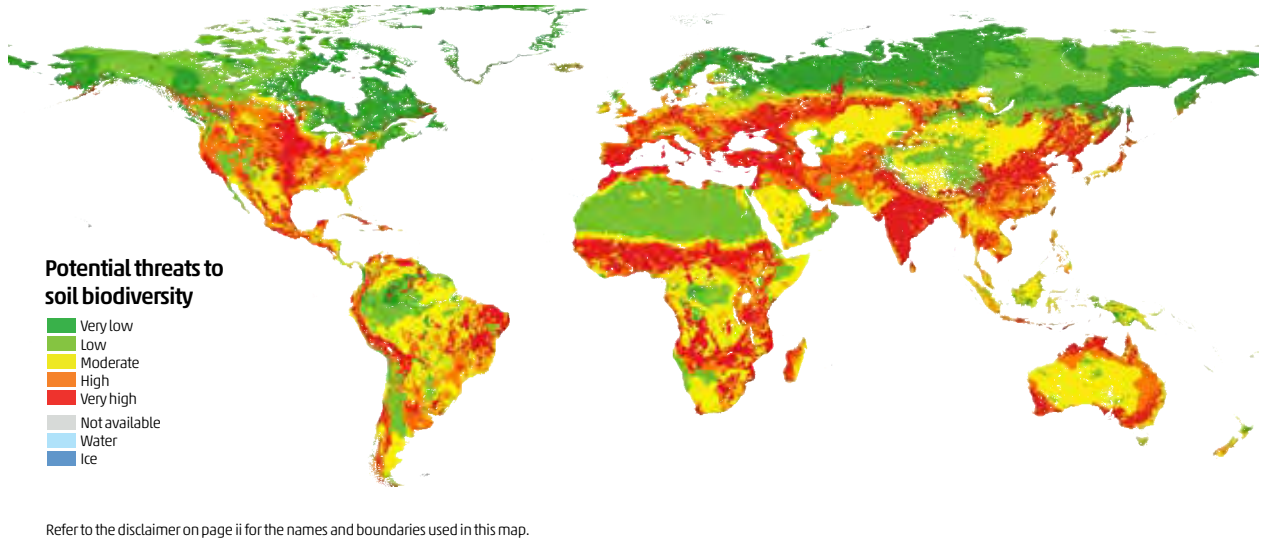


Figure 3.9. Global map of potential threats to soil biodiversity



Refer to the disclaimer on page ii for the names and boundaries used in this map.

◊ *Source:* **Orgiazzi, A., Bardgett, R.D., Barrios, E., Behan-Pelletier, V., Briones, M.J.I., Chotte, J.-L., De Deyn, G. & Eggleton, P.,** eds. 2016. *Global Soil Biodiversity Atlas*. Luxembourg, Publications Office of the European Union.

Deforestation

The effects of deforestation on bacterial and fungal communities were examined in the meta-analysis of Qu *et al.* (2024). They found that the conversion of forest to cropland, grassland or plantation generally led to ecosystems that were dominated by bacteria rather than fungi and where the fungal communities were more homogenous than in native soils. Conversion of forest to agriculture favoured the growth of bacteria with fast turnover times and caused a shift from fungi that had a symbiotic relationship with plants to fungi that were plant pathogen-dominated. The key non-living (abiotic) factor controlling these shifts was an increase in soil pH (i.e. a decrease in soil acidity) caused by deforestation and the decreased vegetation biomass and increased chemical weathering of soil and bedrock, as well as by adding crushed $CaCO_3$ to the soil (an agricultural practice known as liming). The increase in pH also increased the availability of P, decreasing the need for symbiotic relationships between fungi and roots that supplied P in more P-limited soils. The shifts were more pronounced in conversion from forest to cropland and for tropical regions (Qu *et al*., 2024).

The changes in fungal diversity associated with deforestation may also have especially important effects on soil functions such as organic matter decomposition and nutrient availability in low N ecosystems (Dong *et al*., 2024). In their meta-analysis, Dong *et al*. (2024) found that a high diversity of fungi, especially mycorrhizal fungi, played an important role in maintaining soil functions in soils with low total soil N (defined as soils with total soil N less than 1 g of N/kg).

Agriculture

Within agricultural landscapes, the intensity of production can further reduce soil biodiversity, with the dependence of intensive agriculture on irrigation, heavy machinery and the increasing use of agrochemicals such as fertilizer and pesticides all causing significant habitat modification (Geisen, Wall and Van der Putten, 2019). Such modifications cause immediate changes to soil structure and the physical and chemical properties of soil which (along with the changes in plant biodiversity) commonly reduce soil biodiversity.

Tillage

The effects of tillage on soil organisms are greater for larger-sized soil fauna than for soil microorganisms (de Graaf *et al.,* 2019; Tsiafouli *et al.,* 2015). In the meta-analysis of De Graff *et al.* (2019), a 22 percent decrease was found in soil faunal biodiversity and a 7 percent in bacterial biodiversity due to tillage. Two studies (de Vries *et al.*, 2013; Tsiafouli *et al.*, 2015) evaluated the effects of land use (permanent grassland, extensive rotation and intensive wheat rotation) on soil faunal biodiversity at sites in Czechia, Greece, Sweden, and the United Kingdom. They found that the intensive wheat rotation consistently reduced the biomass of all soil organisms and that soil animals with larger body sizes and relatively long lifespans. Earthworms, Collembola and mites were most affected, with only nematodes seeming to be largely unaffected by the increased intensity.

The effects of tillage on soil microorganisms are less consistent. Chen *et al*. (2020) analysed the effects of conventional vs conservation tillage on bacterial and fungal biomass across 60 studies and found that overall soil microbial biomass was approximately 37 percent higher under conservation tillage. However, the positive effect only occurred in the 0–20 cm depth range, and significant decreases of fungal biomass (−20 percent), and bacterial biomass (−24 percent) were observed at depths greater than 20 cm.

De Graaf *et al.* (2019) reported significant negative effects of tillage on bacterial diversity (a mean effect of −7 percent) but not for overall fungal or arbuscular mycorrhizal fungal diversity, where there was large variation between studies. The meta-analysis of Morugán-Coronado *et al.* (2022) focused on bacterial and fungal diversity using a phospholipid-derived fatty acids (PLFAs) assay showed that conservation tillage had a significant and positive effect for bacterial and fungal biomass compared to conventional tillage. However, there were insufficient studies for the effects on microbial diversity to be reliably assessed.

Fertilization

The impact of fertilization on soil biodiversity is complex and depends on the type of fertilizer used and the C, N and P content of the soil (Dincă *et al.*, 2022). In their review of the literature, Dincă *et al.* (2022) found that synthetic fertilizer had no significant influence on the richness and diversity of bacteria and fungi, but the abundance of individual bacteria or fungal species was sensitive to fertilization. This sensitivity was mainly attributed to changes in the soil chemical properties caused by both synthetic and organic fertilization.

Soil acidification due to fertilization is believed to affect soil biodiversity (De Graaf *et al.,* 2019; Cui *et al*., 2022; Hu *et al*., 2024). High rates of NH_4^+ and urea fertilization can inhibit soil microorganisms due to direct NH_4 toxicity and by changes in ionic balance of the soil. Ammonium fertilizers also release H^+ ions upon oxidation and thereby increase soil acidity, which can reduce soil microbial diversity.

Soil pH has been shown to be a major control on soil biodiversity. For example, Fierer and Jackson (2006) found that soil pH had the highest correlation with global bacterial diversity. The general pattern was that bacterial diversity showed a sharp decrease below an approximate pH threshold of 5.5, while above this threshold, diversity did not increase as pH increased. Lower pH values were also associated with a lower diversity of earthworms, attributed to reduced Ca availability at more acidic pH levels (Phillips *et al.,* 2020).

Erosion

Orgiazzi and Panagos (2018) suggested that there was possibly a link between soil loss and organisms populating the soil. Soil erosion could cause the loss of organisms from some landscape positions through the physical transfer of organisms. In turn, the action of organisms would then influence the rate of erosion by controlling functions such as the creation of soil aggregates. Subsequently, Guerra *et al.* (2020) modelled soil erosion by water globally and overlaid a global map

of soil biodiversity on the erosion map to determine if high erosion risk corresponded with areas of high soil biodiversity. Globally, high erosion risk areas overlapped with 6.4 percent of areas with high biodiversity for soil macrofauna and 7.6 percent for areas of high fungal biodiversity. However, Quinton and Fiener (2024) concluded in their review of soil erosion that there were so far too few studies on the effect of erosion on soil biodiversity to draw any conclusions.

Pesticides

The effect of pesticides on soil organisms was examined by FAO and ITPS (2017). The report concluded that there were measurable and statistically significant effects of pesticides on soil microorganisms. The effects led to both significant decreases and significant increases in attributes of soil organisms such as biomass, enzyme activity, respiration, and species composition and diversity. However, there was limited evidence that the observed effects of pesticides on soil organisms had led to significant and long-lasting decreases in soil functions.

FAO and ITPS (2017) reported evidence for significant harmful effects of some pesticides on earthworms. Specifically, the negative effects of copper (Cu)-based fungicides were well-established, and evidence was emerging that neonicotinoids were particularly toxic to earthworms. A subsequent review by Mamy *et al.* (2023) found that neonicotinoids had negative effects on various earthworm species from sub-individual to community levels. Nematodes were less sensitive to neonicotinoids, and studies on terrestrial microorganisms showed little or no effect of neonicotinoids at realistic agricultural conditions in the field (Mamy *et al.*, 2023)

A meta-analysis by Gunstone *et al.* (2021) assessed the effects of pesticides on non-target invertebrates across 394 individual studies. The effects of pesticides were considered using nine categories of endpoints including mortality, abundance, biomass, behaviour, reproduction, richness and diversity, and structural changes. Across the studies examined, negative effects dominated, with 70.5 percent of 2 842 total tested parameters identifying a negative impact to soil organisms from pesticide exposure. Insecticides had the largest negative effects (between 60 and 85 percent for the organisms tested) while specific herbicides and fungicides varied greatly in their effects but were dominantly negative. They concluded that pesticides posed a clear threat to soil organisms.

Assessing of the effects of pesticides on earthworms at the field or community level are complicated by the negative effects of tillage on larger soil organisms, and there is a need for experimental designs that clearly separate pesticide effects from tillage effects. Studies on other soil fauna such as Collembola, mites and potworms (*Enchytraeidae*) indicate a general similarity to earthworms but are less studied.

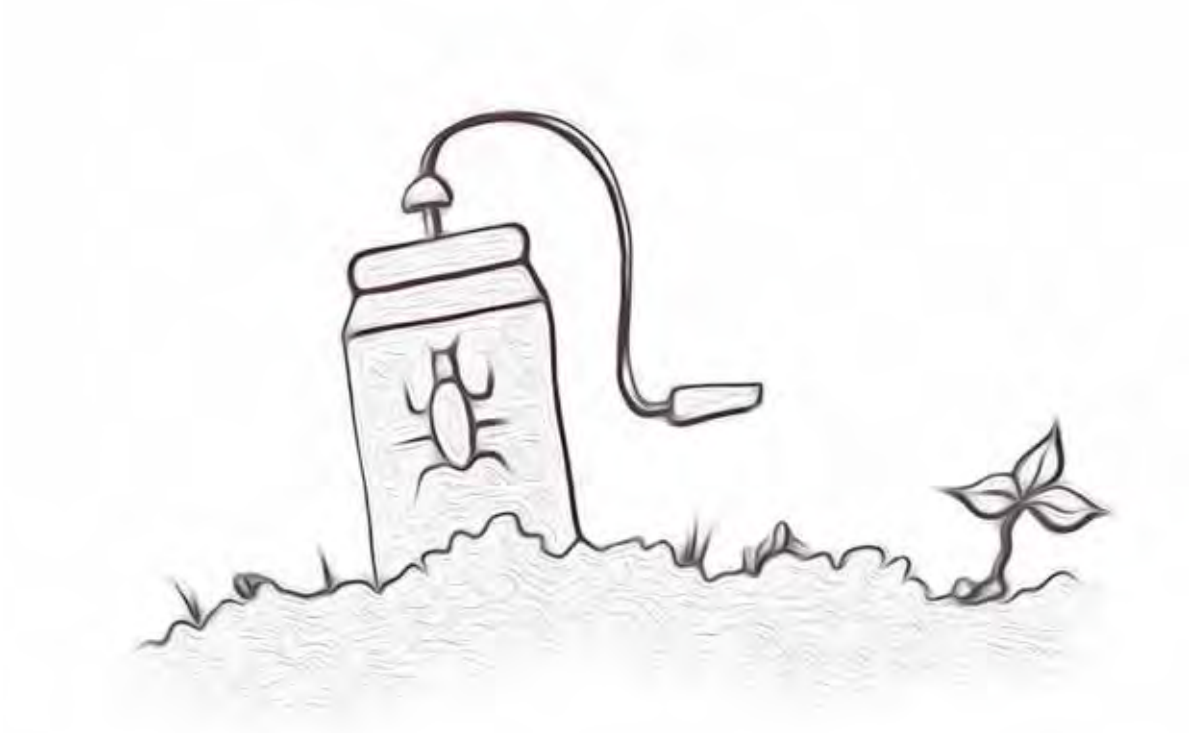
Studies since 2017 have highlighted the ecotoxicological risk that pesticide levels and especially pesticide mixtures pose for soil organisms. For example, Tang *et al.* (2021) produced a global map of potential pesticide pollution in soil, surface water, groundwater and the atmosphere and predicted that 74.8 percent of global agricultural land (or approximately 28.8 million km^2) was at some risk of pesticide pollution, with 31.4 percent judged to be at high risk. New research in Finland has also highlighted that the risks posed by pesticide residues and mixtures of residues may be higher in northern agricultural systems where cold temperatures probably slow the degradation of pesticides (Hagner *et al.*, 2024). These authors found that despite low application rates of pesticides in Finnish soils, the residue concentrations detected were comparable to other European soils with much higher application rates.

Climate change

The effects of human-induced global change factors such as higher global temperatures, altered precipitation, CO_2 enrichment, atmospheric N deposition, nutrient fertilization, land-use change and their combinations on soil organisms have been widely studied since 2015.

In an influential study, Rillig *et al.* (2019) used a laboratory experiment to examine the effect of multiple global change factors on soil functions and microbial biodiversity. Global change factors examined included changes in temperature, water, fertilization, and CO_2 as well as less commonly studied factors such as land-use change, agrochemicals, salinity, invasive species and ozone. It was found that while the response of indicators of ecosystem functions to single factors could be quite variable, a more consistent directional response emerged as the number of factors imposed increased.

The meta-analysis of Zhou, Wang and Luo (2020) used 1 235 observations from 341 publications on the effect of global change factors on measures of microbial diversity and community structure. It was found that the microbial community structure was sensitive to the global change factors, but that the changes in microbial diversity caused by the factors were inconsistent. Soil pH was the most important factor in predicting the effects of global change factors on the number of species within a local site (such as alpha diversity). If a factor increased the pH, the diversity would increase, if a factor decreased pH, the diversity would decrease, and if a factor did not change pH, then the diversity would also not change. It was also found that land-use change caused a significant increase in alpha diversity, possibly due to the increases in pH after deforestation discussed earlier in this section.



3.5 Nutrient mismanagement

Three principal forms of nutrient mismanagement are examined in this report: nutrient mining, nutrient excess and acidification. Nutrient mining is the chronic undersupply of plant nutrients relative to nutrient export in plant products and depletes the store of nutrients and organic matter through time. Nutrient excess is the chronic oversupply of plant nutrients relative to nutrient export in plant products. Excess P and N contribute to nutrient pollution of waterways and, in the case of nitrate, is of harm to human health. Excess N contributes to the emission of the potent GHG, N_2O and to the formation of NH_3, which is harmful to human health. Acidification is principally caused by inputs of acidic fertilizers to soils. The subsequent increases in acidity can make soils less fertile and harm the soil microbiota and larger soil organisms.

3.5.1 Overview of nutrient balances

Nutrient mismanagement includes both insufficient and excess nutrient application. Nutrients are essential for life in the soil. The organisms that live in the soil transform chemical elements from the atmosphere and the mineral soil reservoir into forms that are integral to the growth of organisms and plants. In managed agricultural and forestland – while nutrients are often added to increase plant growth past the limits of natural soil nutrient levels – environmental problems, including eutrophication in waterways and loss of GHGs to the atmosphere, are linked to excess nutrient additions.

Nutrient budgets are an essential tool to ensure sustainable use of nutrients. Ideally, inputs of nutrients should be roughly in balance with outputs of nutrients in crops and animal products. If outputs are greater than inputs, mining of soil nutrient reserves occurs and plant yields will drop below the maximum levels possible in a given location. If inputs are greater than outputs the excess nutrients can cause harm to air and water quality.

The major focus of this section is on N and P, widely regarded as the two nutrients of greatest concern globally (Kopittke *et al.*, 2021; Rockström *et al.*, 2023). Since 2015, there have been many studies that calculate nutrient balances, notably related to those nutrients (e.g. Kaltenegger and Winiwarter, 2020); Zou *et al.*, 2022; Bouwman *et al.*, 2017).

An essential step for sustainable nutrient management is the quantification of a nutrient balance (the balance of the inputs and outputs of nutrients at a farm, catchment or other well-defined scale [Zhang *et al.*, 2020]). Nutrient balances can be used to identify countries or areas within countries down to field level that are experiencing potentially harmful nutrient surpluses or nutrient mining. The four key measures in nutrient balance calculations are:

1. **nutrient balance**: nutrient inputs − (nutrient outputs in products + nutrient losses to the environment);
2. **nutrient use efficiency**: 100 × nutrient outputs in products/nutrient inputs;
3. **nutrient outputs**: including the nutrients contained in harvested plant and animal products; and
4. **nutrient losses to the environment**: including losses in leaching, erosion and gaseous loss to the atmosphere.

Nutrient inputs into cropland mainly originate from synthetic fertilizers, animal manure, and, for N and S, through atmospheric deposition (Figure 2.3). In addition, specific nutrients may have additional sources, such as the fixation of atmospheric N by N-fixing plant species, known as biological nitrogen fixation (BNF). Biofertilizers are a small but growing segment of the fertilizer

market, and they are examined in greater detail in Section 5.2.6. Some losses, such as through runoff and leaching, are common to all nutrients, while others, such as gaseous losses, are important for N but are negligible for nutrients such as P (Mekonnen and Hoekstra, 2018; Zhang *et al.,* 2020).

Global nitrogen use efficiency (NUE) values (calculated from different sources) range between 45 and 49 percent. Kaltenegger and Winiwarter (2020) calculated a NUE of 47 percent and 44 percent from two different cropping maps. These values are close to two other widely-cited estimates of 47 percent from Lassaletta *et al.* (2014) and 49 percent from Bouwman *et al.* (2017).

The global phosphorus use efficiency (PUE) value shows considerable interyear variability but in recent years has been estimated at 60 percent by Bouwman *et al.* (2017) and 66 percent in 2019 by Zou *et al.* (2022). Bouwman *et al.* (2017) documented distinct patterns of P use associated with different regions for all annual and perennial crops and fruits. In high-income countries P surpluses have been decreasing steadily towards a small deficit in recent years with a steady increase in PUE.

The EU Nitrogen Expert Panel (2015) developed a two-dimensional N input-N output diagram that can be used to identify problematic N balances for cropping systems. It was suggested that an NUE greater than 90 percent indicated a high risk of N mining and an NUE greater than 100 indicated N mining was occurring. These values for NUE were often also associated with negative annual N balances. Annual N surpluses greater than 80 kg/ha and NUE less than 50 percent indicated a high potential for environmental harm (and likely economically inefficient N surpluses). It was also suggested that a very low annual N surplus of <20 kg/ha globally could be linked to soil N mining but for this report countries with average N balances less than 0 are identified as nutrient mining.

Phosphorus in soil originates from SOM mineralisation, atmospheric deposition and from inputs of P fertilizers, both inorganic (such as superphosphate) and organic (animal manures and human waste). The relative contribution of organic and inorganic P sources differs between agricultural systems and has shifted through time (Figure 3.10). Chen and Graedel (2016) reported that in 1961, 56 percent of agricultural inputs were from organic sources (principally manure) and 29 percent from inorganic fertilizers. In the 2017–2021 period the average P inputs across countries was 48 percent from synthetic fertilizer and 52 percent from manure (FAOSTAT, 2024).

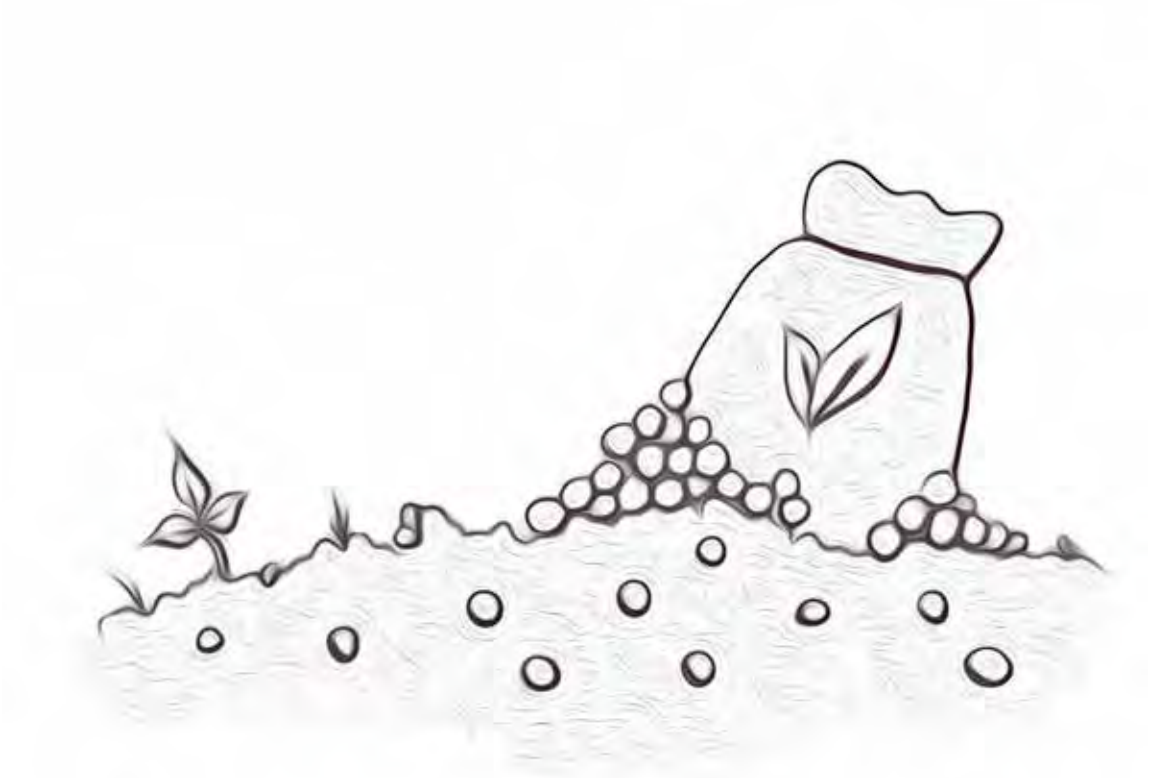
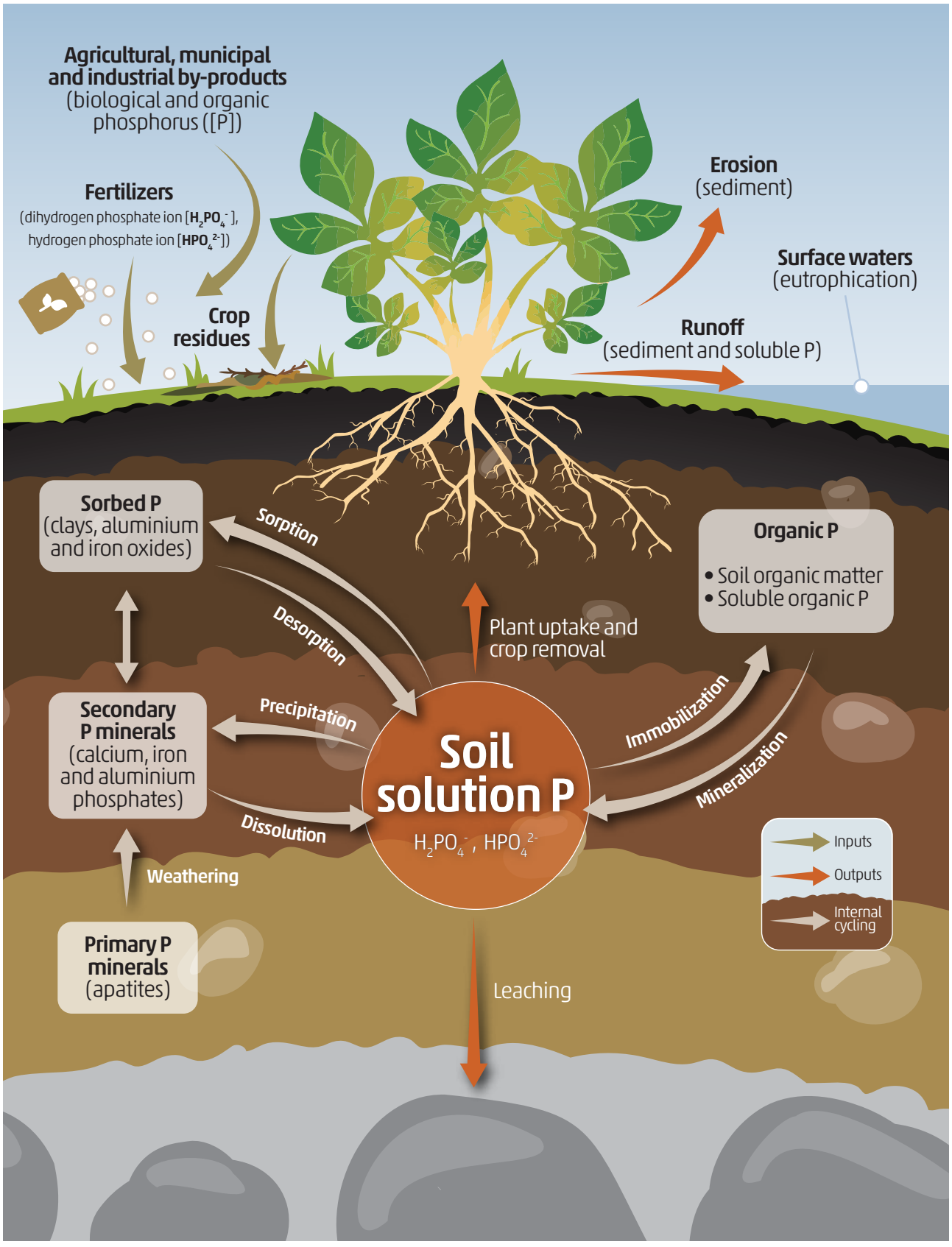


Figure 3.10. Phosphorus cycling in the plant–soil system showing the principal stores, retention and transformation processes in the soil



◊ *Source:* **FAO**. 2022. *Soils for nutrition: state of the art*. Rome. https://doi.org/10.4060/cc0900en

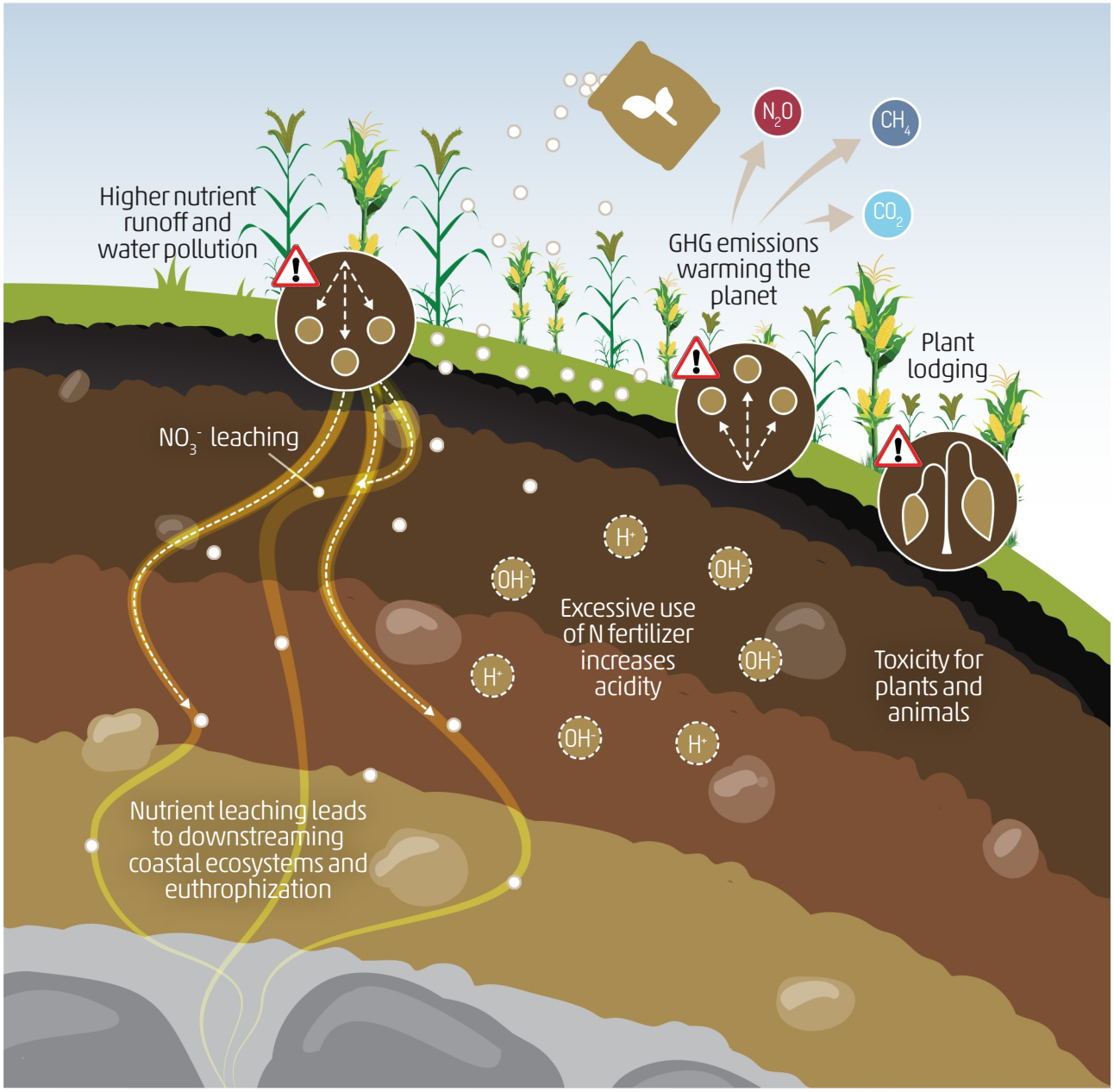
Most of the added P that is not taken up by the crop is retained by the soil (legacy P), or stored in sediments when the eroded soil is deposited on alluvial flood plains or lake and ocean floors.

Phosphorus that has accumulated in the soil can persist for decades (Zhang *et al.*, 2020). In temperate regions, the accumulation of soil P is usually associated with periods of over-fertilization. However, in tropical regions with heavily weathered soils and in volcanic regions where ash deposits occur, the P can be tightly bound to iron and aluminium oxides and is largely unavailable to plants. The release of P from the P legacy pools may mask the effects of management actions that have been taken to reduce P inputs to aquatic systems.

3.5.2 Environmental consequences of nutrient excess

There are several well-established harmful impacts of nutrient excess, as shown in Figure 3.11.

Figure 3.11. Schematic diagram of harmful effects of nutrient excess on soil functions and ecosystem services



Source: **FAO**. 2022. *Soils for nutrition: state of the art*. Rome. https://doi.org/10.4060/cc0900en

Nitrous oxide emissions

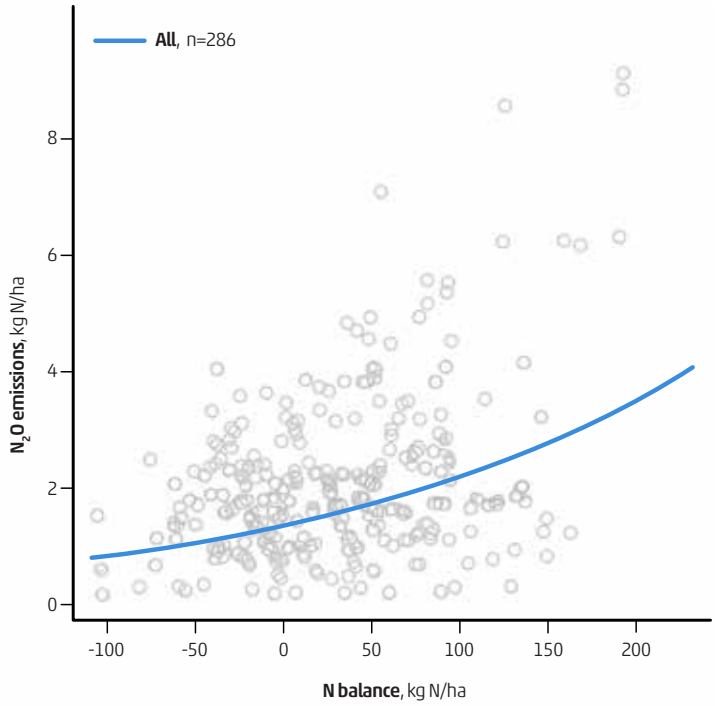
Nitrous oxide is a potent GHG, with a global warming potential of 265 over a 100-year time horizon (Jia *et al.* 2019). Soil processes are the major source of atmospheric N_2O, with more than 60 percent coming from microbial denitrification and nitrification processes in soils (Tian *et al.*, 2019) (Figure 2.6).

Agriculture is responsible for approximately two-thirds of the global N_2O emissions into the atmosphere, driven primarily by increases in manure production and synthetic N fertilizer use from the mid-twentieth century on (Jia *et al.,* 2019). Nitrous oxide emissions from pasturelands and rangelands are responsible for approximately one half of agricultural emissions and primarily result from manure production and deposition on managed pastures (Jia *et al.*, 2019). Emissions from croplands are primarily caused by additions of synthetic N fertilizer.

Tian *et al.* (2019) reported results from the Global N_2O Model Intercomparison Project, which produced simulations from six N_2O models run under structured scenarios for the 1861–2016 period. Between 2007 and 2016, global soil N_2O emissions were estimated to be 10 ± 2.0 billion tonnes of N_2O-N annually according to the "best estimate" of model simulations, which is a 59 percent increase from preindustrial times. Global emissions from croplands increased by 3.0 ± 1.1 billion tonnes of N_2O-N annually and accounted for 82 percent of the global increase. Tian *et al.* (2019) estimated that 27 percent of emissions were from China, 15 percent from South Asia (principally India), 11 percent from the United States, 9 percent from Europe, and 8 percent from Southeast Asia.

The global association between areas with high N surpluses and high N_2O emissions is also apparent for farm-scale studies. Eagle *et al.* (2020) examined the relationship between a simplified N balance (fertilizer N minus crop N) and direct N_2O emissions from fields at study sites drawn mainly from the North American corn belt supplemented with sites from southern England and northern Europe. They were able to identify a generalized relationship between this simplified N balance and N_2O losses for a wide variety of cropping systems and subregions (Figure 3.12). The rate of N loss accelerates as applied N exceeds crop N demand (i.e. once N balance exceeds a threshold value).

Figure 3.12. Generalized relationship between nitrogen balance and nitrous oxide emissions



◊ *Source:* Adapted from **Eagle, A.J., McLellan, E.L., Brawner, E.M., Chantigny, M.H., Davidson, E.A., Dickey, J.B., Linquist, B.A.** *et al.* 2020. Quantifying On-Farm Nitrous Oxide Emission Reductions in Food Supply Chains. *Earth's Future*, 8(10). https://doi.org/10.1029/2020EF001504

Ammonia emissions

Emissions of NH_3 from soils are linked to environmental and human health concerns. Ammonia has many sources, including gaseous release from animal waste (including human), synthetic fertilizer, biomass burning, losses from soils under both native vegetation and agricultural crops, and fossil fuel combustion (Behera *et al.*, 2013). Bauer, Tsigaridis and Miller (2016) cited studies showing that in the past 70 years, global NH_3 emissions have more than doubled (from 23 million tonnes annually to 60 million tonnes annually). This increase was wholly attributable to agriculture, with fertilizer N use accounting for 33 percent of the increase, and livestock production accounting for 66 percent. Ammonia emissions from livestock production came from animal housing and storage systems, animal manure, and grazing.

Regional changes in NH_3 emissions between 2008 and 2018 were estimated by Van Damme *et al.* (2021) using satellite data from the Infrared Atmospheric Sounding Interferometer (IASI). East Asia had the largest increase in emissions over the decade, mostly due to increases in the North China Plain and the Chengdu area in China, as well as high relative increases in the Democratic People's Republic of Korea, the Republic of Korea and Japan. In South Asia, Pakistan had the highest absolute growth trend, which was attributed to the overuse of synthetic fertilizers. In India there was a high upward trend in the Indo-Gangetic Plain but decreasing values in the southeastern part of the country. The increases (both absolute and relative) in West and Central Africa were largely driven by biomass burning emissions associated with agricultural practices.

Nitrogen leaching and nitrogen pollution of waterways

Nitrate leaching is the dominant cause of rising nitrate concentrations in groundwater and contributes to ecosystem eutrophication and water quality degradation. The IPCC (2006) used a fertilizer-induced emission factor (EF) to evaluate nitrate leaching intensity and they assumed that nitrate loss by leaching was 30 percent of applied fertilizer N. Wang *et al.* 2019 reported that fertilizer-induced EFs were not constant (as in the IPCC EF) but instead increased along with the increasing N application rates. Nitrate leaching with increasing N inputs also rose rapidly when the N input exceeded the optimum rate for plant uptake.

Tamagno *et al.* (2022) examined the relationship between N balance and N leaching losses using a meta-analysis of 82 studies (1 110 observations) for rainfed cereal crops in temperate regions. The studies were heavily concentrated in the United States and Europe with some studies from China. It was found that the relationship between nitrate leaching and N balance was exponential for data both from the United States and elsewhere. Besides the N balance, hydrology was a key factor influencing nitrate transport, with excessive precipitation causing downward movement of water, particularly in coarse-textured soils.

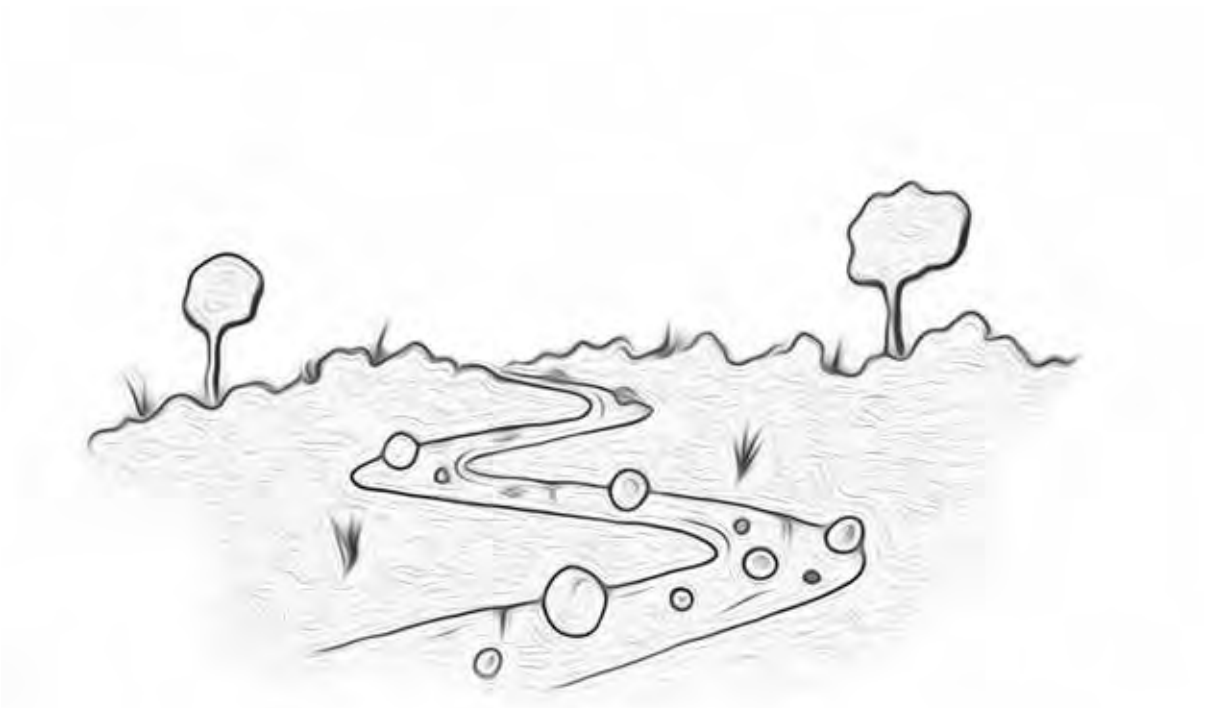
Phosphorus pollution of waterways

The greatest concern with excess P applications is the risk of water pollution. Although agriculture is an important contributor to water pollution by P, its proportional contribution is lower than other sources. Mekonnen and Hoekstra (2018) estimated that in the 2002–2010 period, the average annual global P load to freshwater systems from all sources was 1.47 million tonnes annually. Of this, 62 percent was from domestic and industrial wastewater sources and 38 percent was from agriculture. The contribution of the different sources varied between watersheds but generally, the watersheds with high P loads closely followed high population densities and the intensity of fertilizer and manure application (Mekonnen and Hoekstra, 2018).

Globally, the largest portion of P is in particulate form (attached to eroded soil) that is transported with runoff waters from agricultural fields. In regions where fertilization rates are high, a larger percentage of P is transported in dissolved form. For example, Bian *et al.* (2022) estimated that in

the Mississippi River basin, dissolved inorganic P export (associated with mineral P fertilizer use) accounted for between 30 to 35 percent of exports, and particulate inorganic P (associated with soil erosion) accounted for between 48 to 54 percent of exports.

In some regions, the P legacy pool in soils can be a major contributor to P export. For example, McCrackin *et al.* (2018) estimated that the P loosely bound in soils and surface sediments and dissolved or particulate forms in surface waters (mobile pools of legacy P) were the largest (46 percent) source of P in the Baltic Sea in the period from 2000 to 2013.



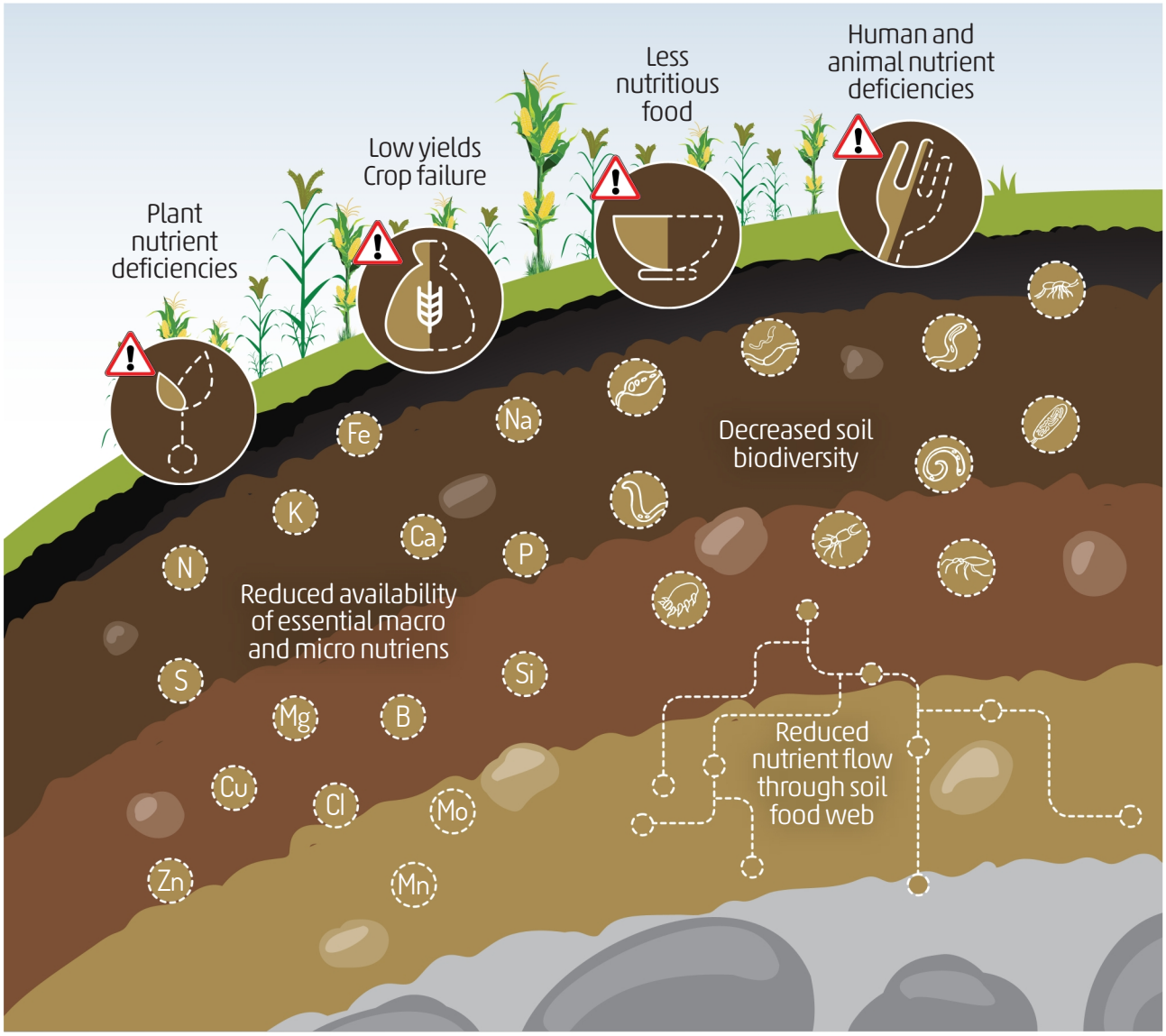
3.5.3 Nutrient mining and yield gaps

Nutrient mining is indicated by negative nutrient budgets (i.e. where the export of nutrients in agricultural products is greater than the inputs of nutrients from inorganic and organic fertilizers, atmospheric deposition, and N fixation). It is a major cause of soil degradation as it causes the ability of the soil to supply nutrients to decline (Figure 3.13). A yield gap is the difference between the actual yield achieved and the potential yield that could be achieved under optimal conditions. Fischer (2015) defines the potential yield as:

> *The measured yield of the best cultivar, grown with optimal agronomy and without manageable biotic and abiotic stresses, under natural resource and cropping system conditions representative of the target area* (Fischer, 2015, p. 9).

A yield gap can occur due to inadequate water supply, pressure from pests, weeds and disease or from incomplete nutrient supply that fails meet all crop nutritional requirements. Nutrient mining is a threat to soil functions whereas yield gaps impact food security but do not threaten soil functions.

Figure 3.13. Schematic diagram of harmful effects of nutrient mining on soil functions and ecosystem services



Source: **FAO.** 2022. *Soils for nutrition: state of the art.* Rome. https://doi.org/10.4060/cc0900en

The two main soil fertility factors that contribute to yield gaps are deficiencies in nutrients and unbalanced nutrient inputs. There are two main categories of nutrients (FAO, 2022a): major nutrients (i.e. nutrients required in large amounts and micronutrients (i.e. nutrients required in very small or trace amounts).

The six major nutrients are N, P, K, sulphur (S), Ca, and Mg, with N and P having been discussed previously. Potassium is added to soils through synthetic fertilizers and manures. It is also made available from the weathering of soil minerals in younger soils or from feldspar-rich parent materials in other regions. Younger soils are common in the Northern Hemisphere in regions that were glaciated during the ice ages. Sandy soils in every region are also often deficient in K. Nutrient balances of S, Ca and Mg are less commonly available. In many cases, the soil reserves of S, Ca and Mg are sufficient to support growth or the requisite amounts are added through organic sources such as manure. Sulphur deficiency is common in soils with low organic matter contents.

The soil micronutrients are chlorine (as chloride: Cl^-), boron (B), Cu, cobalt (Co), Fe, Mn, nickel (N), and Zn. Moreno-Jiménez *et al*. (2022) used data from >1 300 topsoil samples worldwide to create a global map for the distribution of metallic micronutrients (Cu, Fe, Mn, Zn, Co and Ni). The areas most likely to be deficient in these elements are in sub-Saharan Africa and large parts of South America, while areas of lower contents also occur in Australia and South Asia. Boron is also commonly deficient in humid regions with well-drained soils or in sandy soils and B-deficient soils have been reported in more than 80 countries (Das and Purkait, 2020). Chloride in soils is commonly associated with salinity in soils and with high inputs of animal wastes and is also added to soils through rainwater (Geilfus, 2018).

Optimum plant nutrition requires that all nutrients be available in the required forms and amounts. Liebig's Law of the Minimum states that plant growth is determined not by the total of all available resources but by the scarcest resource (the limiting nutrient). The absence of a required nutrient can severely limit plant growth and can cause a soil to be unresponsive when, for example, high amounts of N or P are added. The problem of non-responsive soils is common in sub-Saharan Africa (Sileshi *et al.*, 2022).

3.5.4 Acidification

The level of acidity in the soil is expressed on the pH scale – the lower the pH, the higher the acidity. Increasing soil acidity (acidification, decreased pH) can reduce the availability of important plant nutrients and hence reduce plant growth while increasing the availability of other elements (e.g. aluminium [Al], Cd and Mn) to toxic levels. Generally, nutrients are most available in the soil pH range from 5.5 to 7 (Goulding, 2016). Plants are also affected by changes in the soil chemical environment caused by increasing acidity, and optimum ranges for the growth of specific crops are widely available (e.g. FAO, 2019c). Soil pH is also the most important soil control on the global distribution of soil bacteria, fungi and larger soil animals such as earthworms.

The main anthropogenic-induced causes of acidification are from the deposition of acidifying gases and particles from the atmosphere and the addition of acidifying fertilizers in cropping systems or the cultivation of legumes (Goulding, 2016). In some areas, the draining of S-rich sediments can lead to localized acidification and formation of acid sulphate soils.

Atmospheric deposition of acidifying N (mainly in the forms of nitrogen oxides [NOx], and NH_3) and S, mainly in the form of SO_2 gases and particles, is a global driver of soil acidification. This acidification occurs when urbanization, industrialization, mining, construction or dredging release acidifying substances into the air or water. Chen, Xiao and Chen (2023) summarized global data on soil acidification due to N deposition and stated that there had been a global decline in soil pH of −0.16 pH units due to N deposition. They found that the eastern United States, southern Brazil, Europe, South Asia and East Asia were the main hotspots of soil acidification due to N deposition.

The most important sources of human-induced acidification on agricultural land are the application of NH_4^+-based fertilizers and urea, elemental S fertilizer, the uptake and export of base cations by crops, and from the growth of legumes (Goulding, 2016).

Soil acidification is of the greatest concern in soils that have a low capacity to buffer the decrease in pH and in soils that already have a low pH (Figure 3.14). The 1995 study by Von Uexküll and Mutert (1995) on the extent of acid soils is still widely cited. They report that approximately 30 percent or 3 950 million ha of the world's ice-free surface has acid soils (which they define as having a pH (measured in water) less than 5.5. At the time (1995) only 4.5 percent of the acid soil area (179 million ha) was used for arable crops and a further 33 million ha for perennial tropical crops.

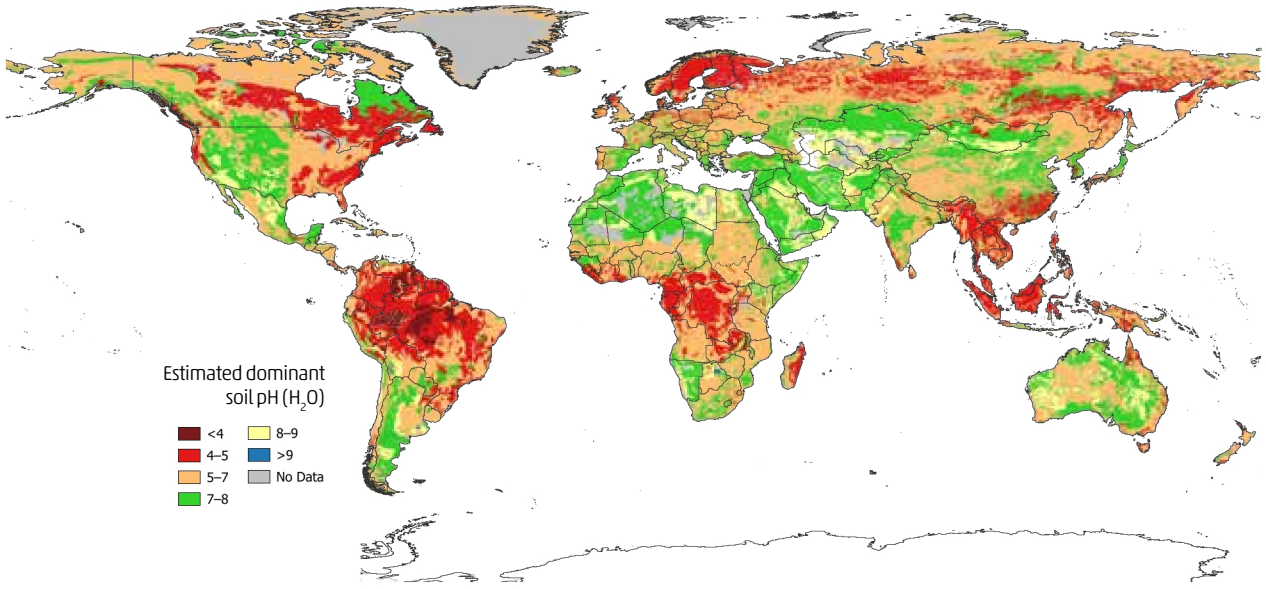
Soils vary in their ability to neutralize (buffer) additions of acids. The differences are closely related to the natural pH range of soils (Figure 3.14). At the global level, soils can be grouped into three main pH groups:

Soils formed in sediments or rocks with $CaCO_3$ or $MgCO_3$ present can buffer added acids from any source and typically have pH values in the pH 6.5 or higher classes.

Soils formed in non-calcareous sediments or rocks (for example, soils in regions with igneous rocks) naturally have acidic pH levels below 6.5. The most acidic soils (for example, those found in northern Canada, the Scandinavian countries of Finland, Norway and Sweden and northern parts of the Russian Federation have pH levels below 4.5.

Although the soils of the subtropical and tropical regions are highly variable, many old, highly-weathered soils in these regions have lost their mobile base cations through leaching and are dominated by Fe and Al oxides that cannot significantly buffer added acids. The pH of strongly weathered and leached soils is often 5.5 or less.

Figure 3.14. Estimated dominant soil pH (H_2O)



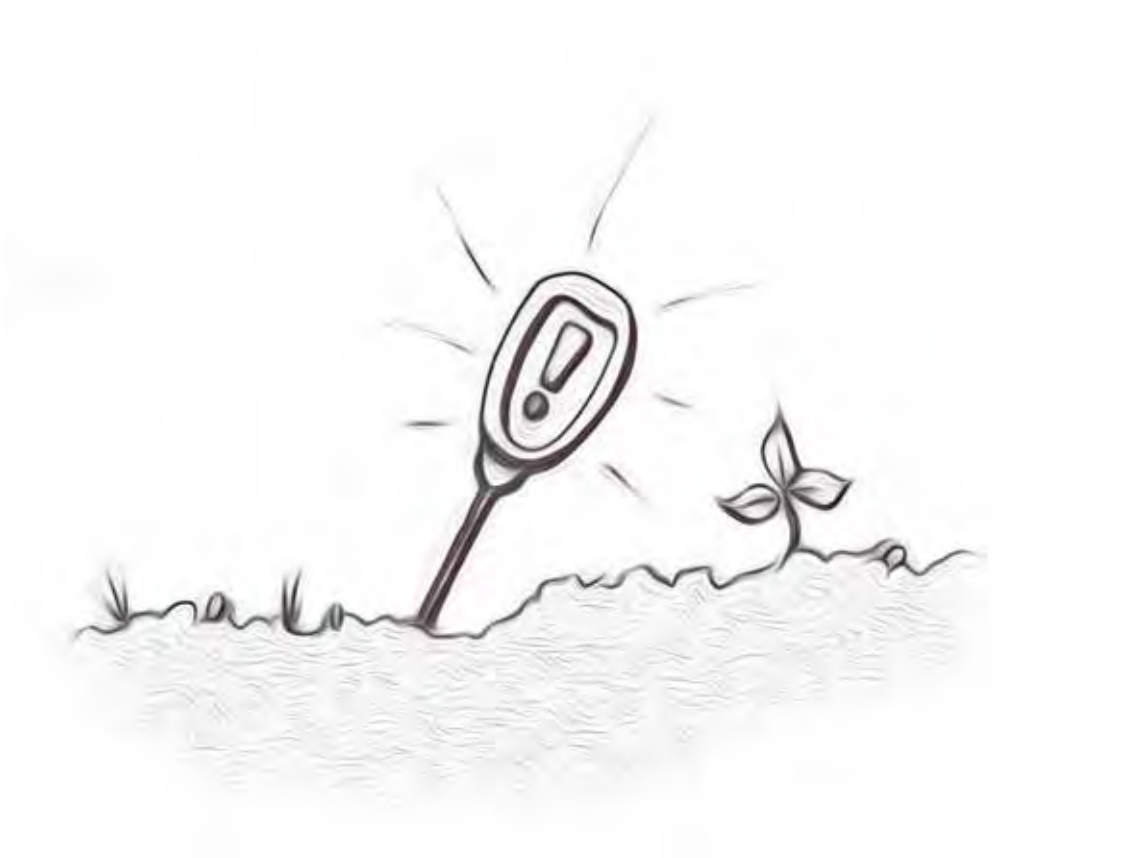
The boundaries and names shown and the designations used on these map(s) do not imply the expression of any opinion whatsoever on the part of FAO concerning the legal status of any country, territory, city or area or of its authorities, or concerning the delimitation of its frontiers and boundaries. Dotted line represents approximately the Line of Control in Jammu and Kashmir agreed upon by India and Pakistan. The final status of Jammu and Kashmir has not yet been agreed upon by the parties. Final boundary between the Republic of Sudan and the Republic of South Sudan has not yet been determined. Final status of the Abyei area is not yet determined.

◊ *Source:* **FAO.** 2023. *GLOSIS: Harmonized World Soil Database.* [Accessed 15 April 2025] https://data.apps.fao.org/glosis/?lang=en. License: CC-BY-4.0.

The buffering capacity of soil is also related to soil texture and organic matter. Generally sandy soils and soils low in organic matter have a low ability to buffer acidic inputs.

Sustained inputs of acidifying N fertilizers can also transform the buffering capacity of soils. Raza *et al.* (2020) documented losses of carbonate (measured as SIC) from Chinese croplands due to N fertilizer added between 1980 and 2020. Seven percent of the total SIC was lost (an average of 16.3 tonnes of C/ha) and 7 million ha of cropland had become carbonate-free in the forty-year period. It was estimated that by 2100, a total of 30 million ha could be carbonate free under a business-as-usual fertilization scenario. As well as the loss of buffering capacity, the release of C from inorganic carbonates represented a significant input of C into the atmosphere.

The impact of acidification in agricultural landscapes is a concern in all poorly buffered soils. It is a particular concern in Australia and is known to affect about half of Australia's agriculturally productive soils. In 2001, the estimated annual value of lost agricultural production due to soil acidity was AUD 1 585 billion, about eight times the estimated cost of soil salinity at that time (McKenzie, 2015). Acidification was judged to be the number one threat to soil functions in the Southwest Pacific region in the 2015 *Status of the world's soil resources* report (Montanarella *et al.*, 2016).

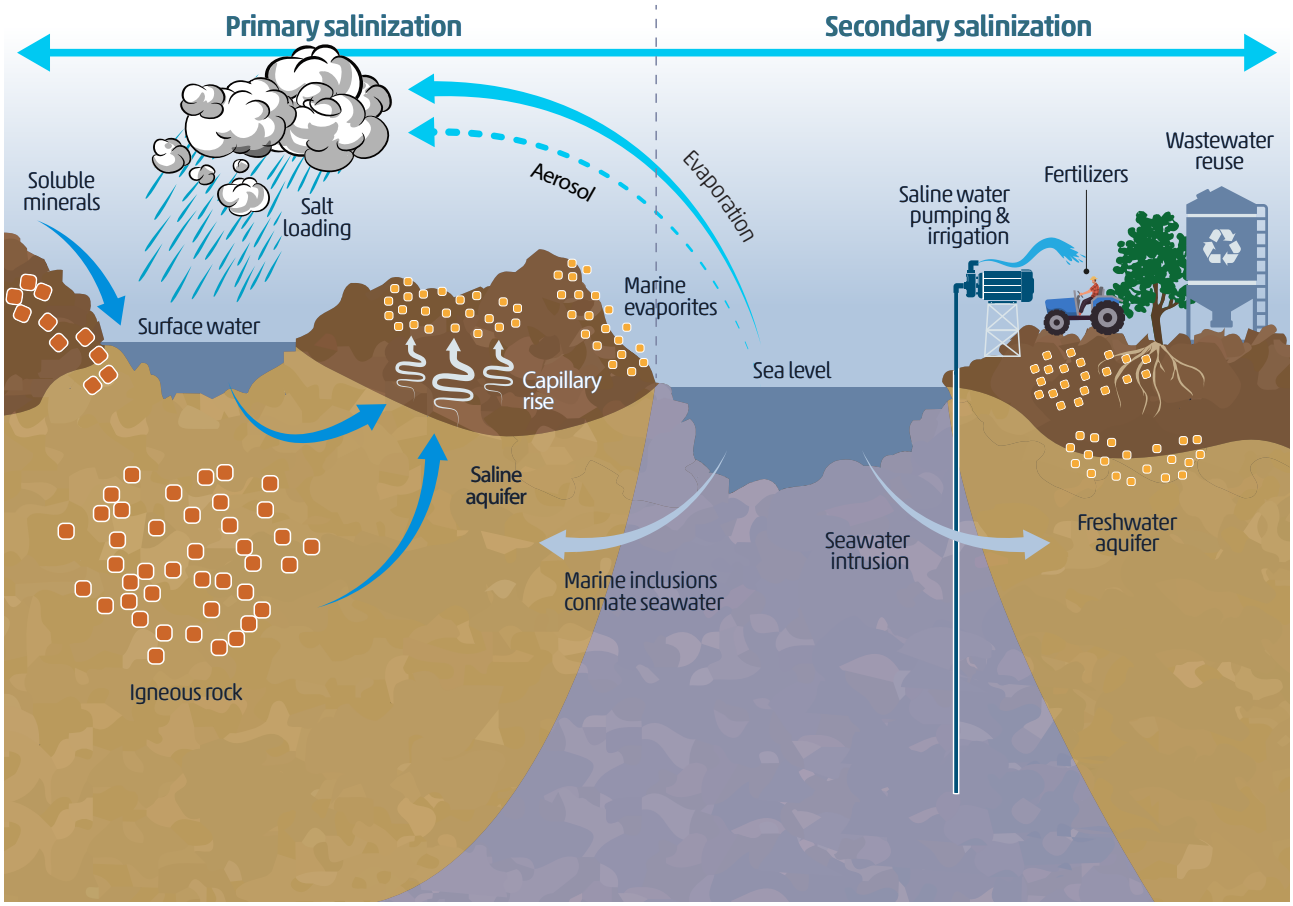


3.6 Salt accumulation

3.6.1 Overview of salt accumulation

Soils that contain salts often occur naturally in semi-arid and arid areas, as well as on coastal margins. However, anthropogenic activities, particularly associated with land clearance and irrigation in drier regions may lead to accumulation of salts in soils which may inhibit plant growth (Figure 3.15). Salt-affected soils include saline, sodic and saline-sodic soils (FAO, 2024). Saline soils are caused by the process of salinization and occur in regions where evaporation exceeds rainfall and irrigation (so that salts are not leached down through the soil), where irrigation is undertaken using salty water, or where seawater or saline groundwater intrudes into the soil. Sodic soils are caused by sodification and contain a high proportion of Na^{+} ions relative to Ca^{+2} and Mg^{+2} ions and are prone to clay dispersion and damage to soil structure. Sodic soils are also known as solonetz, black alkali and dispersive soils (FAO 2024).

Figure 3.15. Causes of primary and secondary salinization



Source: **FAO.** 2024. *Global status of salt-affected soils – Main report.* Rome. https://openknowledge.fao.org/handle/20.500.14283/cd3044en

Salt-affected soils are generally defined in terms of the soil electrical conductivity (EC) of the saturated extract (EC_e) for saline soils and the exchangeable sodium percentage (ESP) for sodic soils. FAO (2024) defines salt-affected soils as follows:

- A **saline soil** has an ECe of a saturated paste extract >4 dS/m at 25 °C, and an ESP <15.
- A **sodic soil** has an ESP >15, and an ECe <4 dS/m at 25 °C (i.e. slight or no salinity).
- A **saline-sodic soil** has both saline and sodic conditions, an ECe >4 dS/m at 25 °C and an ESP >15.

3.6.2 Global distribution of salt-affected soils

A complication in reporting on salt-affected soils is that the threshold used by different authors to identify saline and sodic soils differs between studies. The differences in definitions, and in methods of estimation and mapping have led to marked differences in the estimates of areas of salt-affected soils (Table 3.3). The estimates in Table 3.3 include salinity and sodicity induced by both natural (primary) and anthropogenic (secondary) salinization. It remains difficult to estimate the salinization induced by human activity alone and there are still no comprehensive estimates at the global level. In addition, soil salinity also varies with seasonal changes depending on rainfall and evaporation patterns.

Table 3.3. Estimates of global extent of salt-affected soils from literature sources, both natural and exacerbated by anthropogenic activity

Source	Extent of estimate	Indicator	Criteria*	Area (million ha)	Sub-item	Area (million ha)
Wicke *et al.* (2011)	Global land (average of top 1 m soil)	Sodic (ESP)	ESP ≥15	288	–	–
		Saline (EC)	EC ≥2	683	EC ≥4	77
		Saline and sodic	–	157	–	–
		Salt-affected soils	–	**1 128**	–	–
Prăvălie *et al.* (2021)	Global arable land	Saline soils and sodic soils	Solonetz, solonchak and salt-affected soils	26.8	–	–
Ivushkin *et al.* (2019)	Global land (top layer depth varies from 0–5 cm to 0–60 cm)	Saline (EC) in 1986	EC ≥2	**916**	EC ≥4	38
		Saline (EC) in 2016	EC ≥2	**1 069**	EC ≥4	33
Hassani, Azapagic and Shokri (2020)	Global land (0–30 cm soil) (in >75% of years between 1980 and 2018)	Saline (EC)	EC ≥2	762	EC ≥4	590
		Sodic (ESP)	ESP ≥6	918	ESP ≥15	13
	Global total salt-affected area, (0–30 cm) (1980–2018)	Saline soils and sodic soils	EC >4 and ESP >6	**1 173**	Affected cropland	16
FAO (2024)	Global land	0 to 100 cm	EC ≥4 or ESP ≥15 or pH ≥8.2	**1 381**	–	–
					–	–

Note: Values in bold are summary totals from each study for the global extent of salt-affected soils.
*Criteria for saline soils reported as electrical conductivity of the saturation extract (ECe) (dS/m at 25 °C) and sodic soils as exchangeable sodium percentage (ESP).

◊ *Sources:* **Wicke, B., Smeets, E., Dornburg, V., Vashev, B., Gaiser, T. & Turkenburg, W.** 2011. The global technical and economic potential of bioenergy from salt-affected soils. *Energy and Environmental Science*, 4: 2669–2681.

◊ **Prăvălie, R., Patriche, C., Borrelli, P., Panagos, P., Roșca, B., Dumitrașcu, M., Nita, I.-A.** *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

◊ **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

◊ **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

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

The studies summarized in Table 3.3 provide estimates of the total area of salt-affected land (with an EC ≥2) at about 1 100 million ha, although the specific type of salt-affected soils included in this value differs across the studies. FAO (2024) reported that the global area of salt-affected soils amounted to 1 381 million ha, or 10.7 percent of the total land area, with a further 1 038 million ha potentially at risk of salinization (i.e. where the EC was between 0.75 and 2 dS/m). If the areas of salt-affected soils with an EC >4 dS/m are considered in isolation, the area affected is considerably smaller. In the top 30 cm of salt-affected soils, FAO (2021) estimated that 85 percent were saline, 10 percent were sodic and 5 percent were saline-sodic. In the 30–100 cm depth, 62 percent were saline, 24 percent were sodic and 14 percent were saline-sodic. Hassani, Azapagic and Shokri (2020) noted that the area of salt-affected soils was both spatially and temporally variable in the period of their study (1980 to 2018), with some areas showing increases and others decreases in salinity national to continental scales. For example, salt-affected areas in Australia were varied in response to droughts and wet periods associated with El Niño and La Niña weather patterns (FAO 2024).

There is broad agreement across the three global maps of Wicke *et al*. (2011), Hassani, Azapagic and Shokri (2020), and FAO (2021) of the major areas of salinity and sodicity (Figure 3.16 and Table 3.4). The area with the highest proportion of salt-affected soils was in the Near East, from the Arabian Peninsula through the Zagros Mountains and Central Iranian Plateau and through the Hindu Kush to the Indo-Gangetic Plain. The salt-affected area extended north through the Turan Plain and Kazakh Plateau of Central Asia, with a significant section of sodic soils in the southern part of the Western Siberian Lowland. The extent of combined saline-sodic soils was greatest in the Turan Plain, the Kazakh Uplands, the Indo-Gangetic Plain, the Mesopotamian Plain and in coastal areas of the Zagros Mountains.

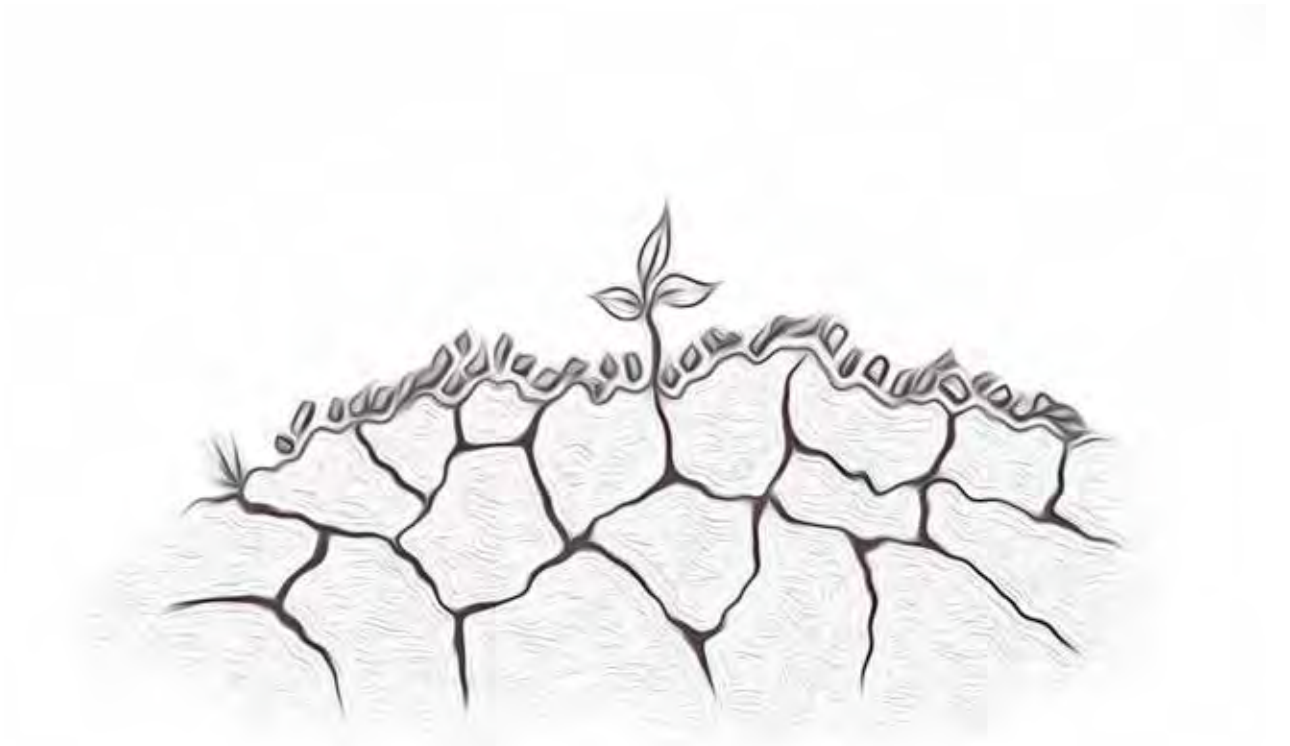
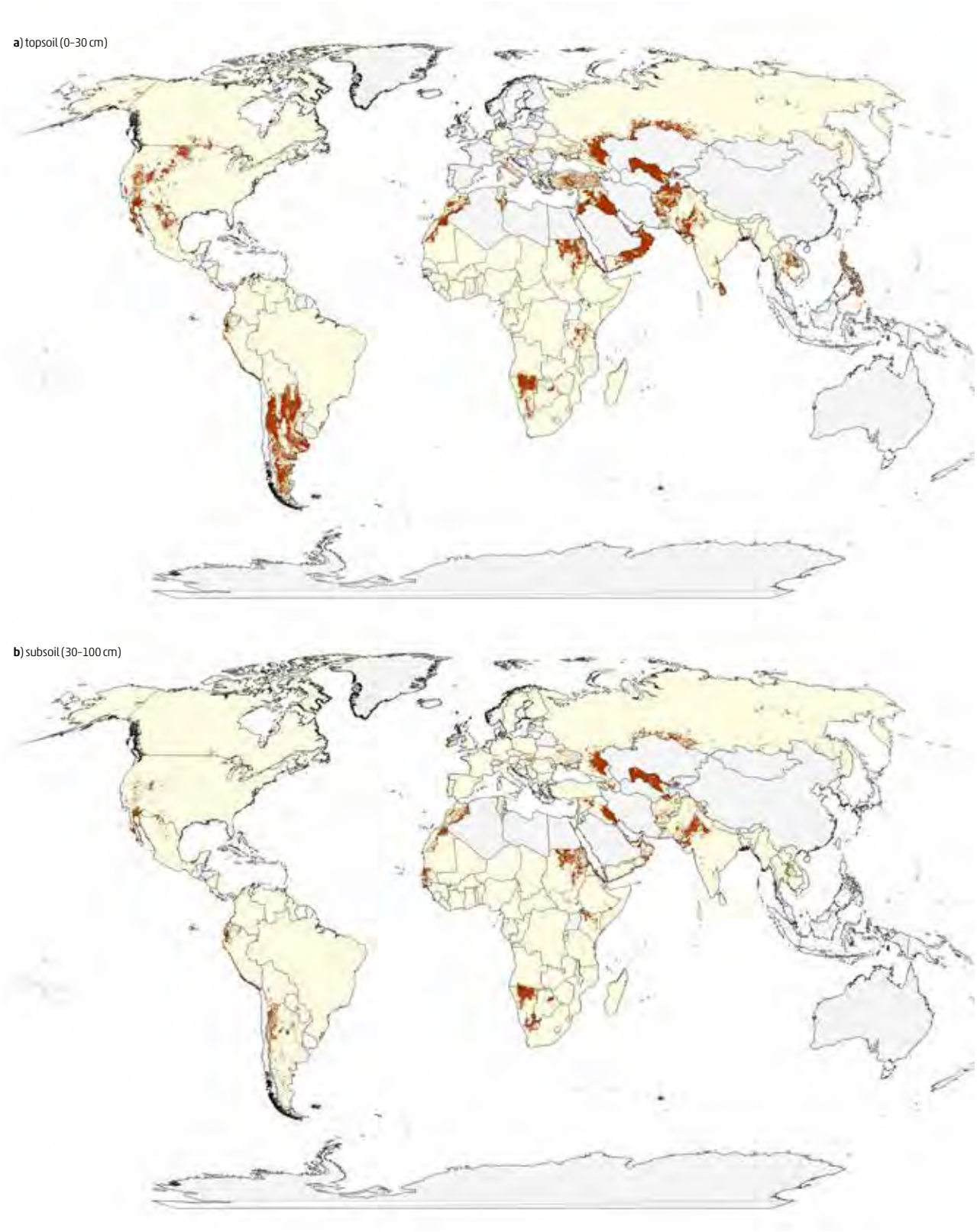


Figure 3.16. Global map of salt-affected soils: a) topsoil (0–30 cm) and b) subsoil (30–100 cm)



◊ *Source:* **FAO**. 2022. *The State of the World's Land and Water Resources for Food and Agriculture – Systems at breaking point. Main report*. Rome. https://doi.org/10.4060/cb9910en

Table 3.4. Areas of salt-affected soils at the regional level

Region*	Area of salt-affected soils** (km^2)	Land area*** (km^2)	%
Pacific	3 570 000	8 472 605	42.1
Europe and Central Asia	2 378 209	27 049 956	8.8
Latin America and the Caribbean	2 352 857	20 026 933	11.7
Near East and North Africa	2 303 461	13 033 953	17.7
Asia	1 543 269	20 722 790	7.4
Africa	883 795	22 046 043	4
North America	779 912	17 936 120	4.3
Total	**13 811 503**	**129 288 400**	**10.7**

* As per the GSP regions laid out in FAO (2023).
** The total areas of salt-affected soils in the countries are listed in Annex 2 and Annex 4, as per the GSP regions.
*** Land area of the whole GSP region.

◊ *Sources:* **FAO.** 2024. *Global status of salt-affected soils* – Main report. Rome. https://doi.org/10.4060/cd3044en

◊ **FAO**. 2023. Global Soil Partnership: Regional Soil Partnerships. In: *FAO*. Rome. [Cited 2023]. https://www.fao.org/global-soil-partnership/regional-partnerships/en/

The three global maps (Wicke *et al.*, 2011; Hassani, Azapagic and Shokri, 2020; FAO, 2021) also show that sodic soils are widely distributed in Latin America and the Caribbean, especially in the Chaco, Pampas and portions of central Patagonia in Argentina, with saline-sodic soils in the Altiplano region. The maps differ in the extent of salinity in northern Mexico, but consistently high levels are shown for the Northern Plateau region of Mexico, extending into the Basin and Range Province and the Intermontane Plateau of the southwestern United States. There are also pockets of salinity and sodicity in the Interior Plains of the United States and Canada. Australia has a very large extent of both saline and sodic soils with sodic soils predominantly found in the east and saline soils in the west. In Africa, the greatest extent of saline soils occurs throughout the Sahara region, the coastal areas of North Africa and in the Great Rift area of East Africa. Combined saline-sodic soils occur in the Chad Basin and in portions of the Kalahari Desert in southern Africa.

The salinity map of Ivushkin *et al.* (2019) differs substantially from the other maps with a greater extent of salinity throughout Mexico and the southwestern United States, eastern portions of the Indo-Gangetic Plain in India, and in the Eastern African Plateaus and marginal areas with generally lower levels of salinity in Central Asia, Australia and Argentina.

A recent global salinity map based on remote sensing by Wang *et al.* (2024) contained saline area estimates that differ greatly from the other major studies. For example, the paper estimated that 65 percent of the land mass of North America had soils with EC values of ≥2 dS/m, including large areas of the boreal forest in western Canada, despite salinity not having been mapped in the soils of this region. Similarly, large areas of the humid tropical forests in the Amazon basin and Central Africa were mapped showing highly saline soils. Therefore, salinity estimates from this paper are not considered further in this section.

Estimates of the proportion of cropland that is affected by salt differ greatly. FAO (2024) estimated the area of salt-affected soils under different land uses for the 120 countries included in the Global Map of Salt-affected Soils (GSASmap) (FAO, 2021), which together account for 73 percent of the total land area. For the combined profile depth of 0–100 cm it was estimated that around 10 percent of irrigated cropland, 10 percent of rainfed cropland and 8 percent of agricultural land worldwide had salt-affected soils. Prăvălie *et al.* (2021) estimated that, globally, 26.7 million ha of arable land (or 1.9 percent of total arable land) was either saline or sodic based on the mapped extent of soils. Hassani, Azapagic and Shokri (2020) estimated that 16 million ha of agricultural land (or 0.88 percent of the global cultivated area) was either saline or sodic based on a digital soil mapping estimate.

One of the regions with the greatest extent of salinization was Central Asia. Measho *et al.* (2022) estimated that in 2018, approximately 110 million ha (about 25 percent) of land in Central Asia was saline (EC >4 dS/m), with marked decadal variability. However, only 1.6 million ha of saline land (EC >4 dS/m) was cropland (or 5.4 percent of total cropland in Central Asia). Over the 1990–2018 period, Measho *et al.* (2022) reported that 6.8 percent of the Central Asia region had an overall increase in soil salinity, although salinity decreased in the croplands.

The modelling studies of Hassani, Azapagic and Shokri (2020) and Ivushkin *et al.* (2019) estimated changes in the area of salt-affected soils during the 1980–2018 period. They reported that only South America (with an increase of approximately 0.95 million ha annually) had a statistically significant increasing trend in the total area of saline soils. Asia and North America had statistically significant decreasing trend for saline land. For sodicity, three regions showed statistically significant increasing trends: Asia (0.56 million ha annually), South America (0.18 million ha annually) and North America (0.17 million ha annually) with no regions showing a significant decreasing trend.

Ivushkin *et al.* (2019) estimated that the total land area with levels of salinity of EC ≥2 dS/m had increased from 916 million ha in May 1986 to 1 069 million ha in May 2016, an average increase of 5.1 million ha annually. Ivushkin *et al.* (2019) also reported that the total land area with EC ≥4 dS/m (the threshold used by Hassani, Azapagic and Shokri [2020]) was 37.6 million ha in 1986 and 33.1 million ha in 2016, with an average decrease of 0.15 million ha annually. Abbas *et al.* (2013) estimated that global salinity was increasing by 2 million ha annually, but no supporting evidence was provided in the paper to verify this widely cited figure.

The estimates of Hassani, Azapagic and Shokri (2020) and Ivushkin *et al.* (2019) provide strong evidence that the area of slightly saline (2 to 4 dS/m) land increased at a rate of approximately 5 million ha annually in the period between the mid 1980s and the mid 2010s. The proportion of land with EC of >4 dS/m) slightly decreased over the same time but with varying trends across regions, including major increases in South America and decreases in Asia and North America. Both studies highlight the dynamic nature of changes in salinity through time and across regions.

3.6.3 Drivers of changes in salt accumulation

Salt accumulation can occur from natural sources (termed primary sources) and from anthropogenic sources (secondary sources) (Hopmans *et al.*, 2021). Primary salinization occurs by release of salts from weathering of rock and sediments, atmospheric deposition and from inundation or seepage from coastal marine water.

The main sources of secondary salinization are the irrigation of crops under poor drainage conditions, the use of marginal irrigation waters, changes to groundwater recharge caused by removal of deep-rooted vegetation, the addition of agrochemicals and wastewater, and the cultivation of more marginal lands due to prime agricultural land being degraded or subject to urban development. Other contributing factors are rising temperatures causing increased evaporation and increased crop water requirements, rising sea levels, and over-pumping (all leading to an increased risk of saltwater intrusion in coastal aquifers) (Hopmans *et al.*, 2021).

Poorly managed irrigation is the major global source of anthropogenic salt accumulation through irrigation without adequate drainage leading to salty groundwater rising into the plant root zone, the use of salty irrigation water and the application of insufficient water to leach salts from the plant root zone. There is a lack of recent estimates of salinization due to poor irrigation and drainage practices (Hopmans *et al.*, 2021). The estimates of Ghassemi, Jakeman and Nix (1995) and Oldeman *et al.* (1991) have been widely cited but are based on data from the 1980s. FAO (2011) estimated that 34 million ha (or 11 percent of the total irrigated area) were affected by some level of salinity and that a further 60 to 80 million ha were affected to some extent by waterlogging and related salt

accumulation. A map of the areas salinized due to irrigation in the FAO (2011) report show hotspots in the central midwest United States, southwest areas of the Islamic Republic of Iran, eastern Pakistan and northeast China. Hopmans *et al.* (2021) cited studies showing a range in secondary salinization overall between 45 to 80 million ha, comprising around 20 to 30 percent of all irrigated land and 5 to 10 percent of the global salinized area, with about half in China, India, Pakistan, and the United States. The authors cautioned that the evidence for these estimates was dated and often unclear.

A second global source of anthropogenic-induced salt accumulation is the inundation of coastal wetlands by seawater. Herbert *et al.* (2015) listed five mechanisms that caused the salinization of coastal wetlands: surface or subsurface intrusion linked to sea level rise; reductions of riverine freshwater flow; alterations of subsurface freshwater; anthropogenic alteration of coastline geomorphology; and storm surges.

Sea level rise and storm surges both push seawater further inland and can cause salinization of low-lying coastal soils. Both sea level rise and storm surges are affected by anthropogenic global climate change (Chen and Mueller, 2018). Over-pumping of coastal aquifers can lead to saltwater incursion in groundwater and if such water is being used for irrigation that in turn leads to the salinization of soils (FAO, 2004).

Sea level rise may impact soil biogeochemical cycles in coastal areas due to the combined effects of submergence and salinity on plant and microbial communities (Mazhar *et al.*, 2022). The effects of sea level rise on C storage in coastal wetlands are complex, in part because salinity and submergence have differing and sometimes contradictory effects on C cycling (Herbert *et al.*, 2015; Mazhar *et al.*, 2022). An increase in salinity reduces CH_4 emissions from coastal wetlands due to the increased availability of sulphate (which reduces CH_4 production). Changes in plant productivity in low-lying coastal environments such as Bangladesh are likely to impact agricultural production and thus human populations (Chen and Mueller, 2018).

The third major source of secondary salinization occurs due to clearance of deep-rooted, perennial vegetation such as grasses, trees and shrubs from soils overlying shallow saline aquifers (Herbert *et al.*, 2015). The resulting decrease in evapotranspiration allows more water to reach the groundwater table, which then rises until the saline groundwater is in contact with the roots of plants and the upper soil profile. This is referred to as dryland salinity, since the land is not irrigated. The best documented example of dryland salinity is in the Murray-Darling Basin of eastern Australia where the combined effects of vegetation clearance, water extraction and river regulation have led to widespread salinization (FAO and ITPS, 2015).

The most widely cited global estimate of the costs of salinity is from Qadir *et al.* (2014). It was reported that the global annual loss in 2013 was USD 27.3 billion due to the salinization of irrigated land, although it was emphasized that the costs were highly variable. It was stated that:

> *the costs of "no action" on salt-affected lands may result in 15–69% losses depending on the crop grown, land degradation type and its intensity, irrigation water quality, provision and capacity of drainage system, and water distribution and on-farm soil and water management* (Qadir *et al.*, 2014, p. 12).

However, the estimates are based on data originally presented by Ghassemi, Jakeman and Nix (1995), based on data from the 1980s and hence may not reflect current conditions. The absence of up-to-date data on the effect of salt-affected soils on agricultural and food production is a major limitation for assessing the cost of salinization and sodification. Regardless of the cost of salinization and sodification, there is a compelling case for action to reclaim salt-affected land.

FAO (2024) estimated yield losses due to salt accumulation on a land area of 644 million ha and found that the total relative yield loss was 0.5 percent, ranging from 0.03 percent for soybean (*Glycine*

max) to 5.05 percent for bean yields. Relative yield losses for other major crops were 0.28 percent for wheat, 0.32 percent for maize, 0.69 percent for rice and 0.21 percent for cotton (*Gossypium* sp.).

Hopmans *et al.* (2021) noted that most global information on soil salinity dated back to the 1980s and though there had been recent modelling attempts, calibration was largely based on the historic data. They suggested that there was a need to improve global mapping to better determine changes in soil salinity along with prioritizing the development of future satellite monitoring to improve observation of spatial and temporal changes in land degradation, including soil erosion and salinity at a global scale.

3.7 Soil pollution

3.7.1 Introduction

Soil pollution has been defined as "the presence in the soil of a chemical or substance out of place and/or present at a higher-than-normal concentration that has adverse effects on any non-targeted organism" (Rodriguez-Eugenio, McLaughlin and Pennock, 2018, p. vi). It is also "invisible to the human eye, but it compromises the quality of the food we eat, the water we drink and the air we breathe and puts human and environmental health at risk" (FAO and UNEP, 2021a, p. 11) (Figure 3.17).

Sources of soil pollutants (Table 3.5) include industrial activities, mining, waste treatment, agriculture, fossil fuel extraction and processing, transport emissions and military activity. Natural (geogenic) sources also occur.

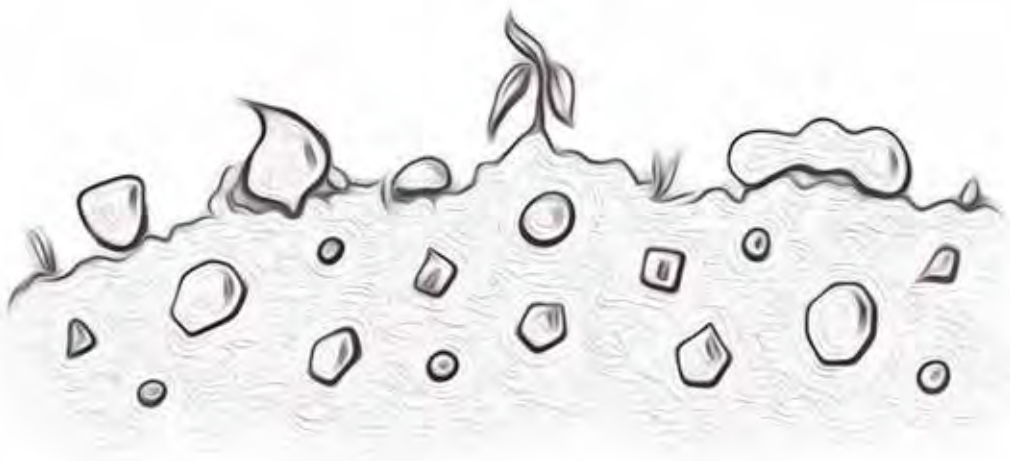


Figure 3.17. Effects of soil pollution on soil functions and ecosystem services



Note: GHG = greenhouse gas.

Source: **FAO & UNEP**. 2021. *Global Assessment of Soil Pollution: Report*. Rome, FAO. https://doi.org/10.4060/cb4894en

Table 3.5. Summary of major sources of soil pollution

Activity	Soil contaminant sources
Industrial activities	Trace elements (such as arsenic [As], cadmium [Cd], lead [Pb], nickel [Ni], or zinc [Zn]) and organic contaminants such as halogenated and non-halogenated solvents, polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs) are released from industrial activity through dust, smoke, and leaching from materials deposited in the landscape. There are emerging concerns about contamination from industrial waste (polyfluoroalkyl substances [PFAS] or PCBs).
Mining	Trace metal pollution and direct surface disturbance comes from resource extraction and ore processing industries. Mercury (Hg), As and cyanide are from gold (Au) extraction and copper (Cu), Pb, Zn and others are from other metal extraction operations and acid mine drainage. There is a wide range of other contaminants specific to different material extractions and processing (for example B from some coal mining operations).
Urban settings, transport and municipal waste	Landfill leachates from the unregulated disposal of end-of-life vehicles and ships, and legacy Pb from leaded fuel and leaded paints. Mismanagement or poorly regulated dumping of urban wastes, poor treatment of wastewater and solid waste, and contamination from electronic waste and used tires.
Oil industry	Soil pollution due to petroleum spills, oil and chemical processing industries, and soil and water contamination from oil sands and pipeline ruptures.
Agriculture	Trace element and organic contaminants from agrichemicals, contaminated irrigation water, impurities in fertilizers and manures, the prior application of obsolete pesticides and persistent organic pesticides, leakage and contamination from stores of obsolete pesticides, contaminants in sediment deposited on floodplains, and the use of plastics in agriculture.
Nuclear activity	Radionuclide pollution from the Chernobyl (Ukraine), and the Fukushima Daiichi nuclear power plant (Japan) failures, legacy pollution from uranium (U) mining and processing, and nuclear weapon testing and use.
Military activities	Legacy contamination from world wars, the Kosovo conflict, pollution from military action and damage to infrastructure in Ukraine, Kuwait, and other regions of military conflict, and Pb contamination from small arms firing ranges.
Geogenic (natural) sources	Arsenic in soils, water and groundwater, particularly in volcanically and geothermally-active regions, and radioactive elements and metals in some geological units. Fallout from wildfires includes trace elements and PAHs.

◊ *Sources:* Adapted from **FAO & UNEP.** 2021. *Global assessment of soil pollution* – Summary for policymakers. Rome, FAO. https://doi.org/10.4060/cb4827en

◊ **FAO & UNEP**. 2021. *Global Assessment of Soil Pollution:* Report. Rome, FAO. https://doi.org/10.4060/cb4894en

Soil pollutants can be broadly subdivided into legacy organic and inorganic chemicals and emerging contaminants (Table 3.6). Potential soil contaminants are comprised of a wide range of compounds, numbering in the thousands. Organic compounds are generally biodegradable and will gradually break down in the soil, while other pollutants, such as the metal elements, are not biodegradable and adhere strongly to the soil, so are able to gradually accumulate and eventually reach concentrations that may be harmful to organisms. Some soil pollutants are the result of legacy activities that are no longer allowed, and so concentrations are stable or gradually declining, while others are continuing to accumulate due to ongoing anthropogenic activities. Soil pollutants may also be recognised as from *point sources* with limited distribution (such as from a fuel storage spill) or from *diffuse* sources (also known as non-point sources), which occur over a wide area and often lack a single easily identified source (Rodriguez-Eugenio, McLaughlin and Pennock, 2018). Runoff from agricultural fields, stormwater water runoff from urban areas and dust emissions are common types of diffuse or non-point emissions.

Table 3.6. Common soil pollutants

Potential contaminant	Symbol	Main effects/comments
Inorganic pollutants: non-carbon compounds		
Asbestos		Hydrated silicate fibres, with high risk if inhaled. The use is now banned in some countries, although legacy issues remain.
Zinc	Zn	Readily taken up by plants but with low risk to humans
Copper	Cu	Readily taken up by plants but with low risk to humans
Nickel	Ni	Readily taken up by plants but with low risk to humans
Cadmium	Cd	Poses significant risks to human and animal health and is the most toxic element (with lead) for humans
Lead	Pb	Risks to human and animal health and is the most toxic element (with cadmium) for humans
Chromium	Cr	Low uptake risk by plants
Vanadium	V	Associated with petroleum processing and use
Selenium	Se	High risk for food chain contamination and poses risks to human and animal health
Cobalt	Co	High risk for food chain contamination and poses risks to human and animal health
Arsenic	As	High risk for food chain contamination and poses risks to human and animal health
Mercury	Hg	Toxic to animals and humans and accumulates in food chain
Radionuclides		Many have a long half-life and are easily taken up in food chain. Carcinogenic with impacts on reproduction
Organic pollutants: carbon compounds		
Polychlorinated biphenyls	PCBs	Has environmental and human health impacts, with the use and production mostly discontinued by the end of the 1970s, although legacy issues remain
Polycyclic aromatic hydrocarbons	PAHs	There are several thousand PAHs, with many being carcinogenic, mutagenic and/or detrimental for reproduction
Perfluoroalkyl chemicals	PFAs	Has a potential impact on human health and the environment even at low concentrations
Persistent organic pollutants	POPs	A wide range of chemicals which may be transported long distances and may cause adverse effects on the environment and on human health
Dichlorodiphenyl-trichloroethane	DDT	The use is now largely discontinued as it concentrates up the food chain, and impacts reproduction
Emerging pollutants		
Pharmaceuticals and personal care products	PPCP	With more than 5 000 products, there are concerns for potential persistence, mobility, toxicity, bioactivity and bioaccumulation
Manufactured nanomaterials	MNMs	The potential bioavailability and toxicity effects are not yet well understood
Microplastics		Easily dispersed into the environment, which can absorb and subsequently leach toxic compounds

◊ *Sources:* **Rodriguez-Eugenio, N., McLaughlin, M. & Pennock, D.** 2018. *Soil Pollution: a hidden reality.* Rome, FAO. https://openknowledge.fao.org/server/api/core/bitstreams/3f7e6959-db0b-44d3-971e-109bcfe78195/content

◊ **FAO & UNEP.** 2021. *Global assessment of soil pollution* – Summary for policymakers. Rome, FAO. https://doi.org/10.4060/cb4827en

◊ **FAO & UNEP**. 2021. *Global assessment of soil pollution:* Report. Rome, FAO. https://doi.org/10.4060/cb4894en

A major advance in our understanding of soil pollution since 2015 was the publication of a global assessment by FAO and UNEP (2021a, 2021b) which reported the rapid growth in the use of potential pollutants and in the contamination of soils that resulted from their use and misuse.

In an extensive global survey, Liu *et al*. (2023) sampled 56 paired urban green spaces and adjacent, unmanaged and semi-natural ecosystems across six continents, including Antarctica. Eight heavy metal(loid)s, 46 pesticide residues, the shape and polymer type of microplastics and 285 antibiotic resistant genes were measured. Similar levels of soil contamination in urban green spaces and adjacent natural areas were reported. Many of the soil contaminants were detected in Antarctica, thus confirming reports of global atmospheric transport of contaminants, including pesticides and microplastics. Anthropogenic influence was evident, with positive associations between population density and levels of metal(loid)s and microplastics. The potential influence of soil contamination on key microbial traits including stress resistance, nutrient cycling and pathogenesis was also identified (Liu *et al*., 2023).

3.7.2 Inorganic pollutants

Inorganic pollutants include trace metals, asbestos and radioactive elements (Table 3.6). Trace elements, (including trace metals) are a group of ubiquitous elements that occur naturally at low levels. Some elements such as Co, Cu and Se, are essential nutrients for microbial, plant and animal health: however, some trace elements can also be toxic to organisms at higher concentrations. Trace metals include elements such as Pb, Cd, Co, Cu, chromium (Cr), mercury (Hg), tin (Sn), Ni, Zn, arsenic (As), antimony (Sb) and selenium (Se). Some such as Zn, Ni, Cu and Co are more toxic to plants while As, Cd, Pb, Cr and Hg are relatively more toxic to higher animals including humans (FAO and UNEP, 2021a, 2021b).

Khan *et al*. (2021) reviewed global soil pollution by trace elements and reported that the soils in and surrounding urban areas in older industrialized regions generally had high levels of trace element pollution. For example, Chicago in the United States, and Moscow in the Russian Federation had particularly high levels. High levels of trace metals in older industrialized areas were attributed to the combined contributions of power plants, industry and pollution from traffic. In some of the most detailed work undertaken, Tóth *et al*. (2016) published maps of soil heavy metal contents in the European Union. It was concluded that the nine elements investigated remained below threshold levels in over two-thirds of the European Union region, although one or more elements were of concern in about 28 percent of the region's total surface area. While natural backgrounds could be the reason for the high concentrations in many areas, the effect of anthropogenic activities in historical and recent mining areas was evident, with elevated concentrations of metals, particularly As, Cd, Pb and Hg (Tóth *et al*., 2016).

Chen *et al*. (2022) published a global meta-analysis of 102 research articles on trace metal contamination near copper mines. Copper, Pb, Cd, Zn, Cr, Ni, Mn and As all had elevated concentrations in soils near copper mines. Copper, Cd and As were recognised as the greatest risk to human health, and thus were priorities for control. The highest total carcinogenic risk values were associated with Cu, with lower risk values for Cd, Cr, As and Ni. Non-carcinogenic risks were generally lower but were close to the critical threshold for children, with As being the highest contributor to non-carcinogenic risks. Nickel, Cr, Zn and Mn were elevated at some sites but were considered to have a low potential ecological risk.

Radionuclides occur naturally in the environment, but some have been redistributed globally because of atmospheric nuclear weapon tests or use. More concentrated contamination occurs at sites where radioactive elements such as uranium (U) have been mined, or where nuclear powerplant operations or accidents have occurred. Radioactive decay emits radiation that is damaging to living cells, and as many radionuclides have long half-lives, those will remain a hazard to living organisms for thousands of years. The radionuclides of most concern are strontium-90 (Sr-90) (with a half-life of 29 years), caesium-137 (Cs-137) (with a half-life of 30 years) and plutonium-239 (Pu-239) (with a half-life of 24 000 years) (FAO and UNEP, 2021a, 2021b).

The term “asbestos” includes a wide range of naturally-occurring hydrated mineral silicate fibres that have been mined and used in a wide range of products including building materials, reinforced concrete, fireproofing, thermal insulation and brake linings. Asbestos fibres, when inhaled, can cause malignant tumours. Asbestos use has been banned in many countries, but major consumers of asbestos still exist (FAO and UNEP, 2021a, 2021b).

3.7.3 Organic pollutants

Organic contaminants are C-based molecules with thousands of different compounds (Table 3.6). Some organic compounds of concern may be derived from natural processes such as wildfires or volcanic eruptions. However, most are synthetic compounds produced for specific purposes, including pesticides, and industrial chemicals, derived from the petrochemical industry (FAO and UNEP, 2021a, 2021b). Organic compounds are used in almost every aspect of modern human life from manufactured products to the transport industry and food processing and packaging. Probably the greatest progress since the publication of the 2015 SWSR report has been in our understanding of the extent and consequences of pollution by pesticides. The reports by FAO and UNEP (2021a, 2021b) highlighted the need to assess the toxic effects of mixtures of pesticides in the soil.

Tang and Maggi (2021) modelled the global soil contamination by mixtures of pesticide residues using a mechanistic, spatially explicit and time-resolved model with a resolution of 0.5° × 0.5° (approximately 55 km by 55 km at the equator). Tang and Maggi (2021) estimated that 70 percent of global croplands (around 8.3 million km^2) had multiple pesticide residues in the topsoil. The countries with the highest average pesticide residue were Cyprus, Japan, the Kingdom of the Netherlands, New Zealand, and the United Kingdom. Orchard and grape cropping systems were predicted to have the highest average total pesticide residue in soil (a global average of 2.35 mg/kg) followed by vegetables and fruits. Soils in croplands for maize and soybean had a predicted mean global pesticide residue of about 1 mg/kg, and wheat fields had the lowest at a global average of 0.12 mg/kg. In the global study of Li *et al*. (2023) pesticides were detected (including fungicides, herbicides and insecticides) in 63 percent of sampled natural ecosystem soils, with no differences in the occurrence of pesticides between urban and natural green spaces.

Multiple studies have assessed the ecotoxicological risk that predicted and measured pesticide contamination levels pose for soil organisms. Silva *et al.* (2019) reported that while threshold concentrations for harm were included in the legislation of some countries for now widely banned, highly persistent and obsolete pesticides (such as dichlorodiphenyltrichloroethane [DDT], hexachlorocyclohexanes [HCHs], atrazine, and dieldrin), thresholds for approved, currently used pesticides did not exist. Instead, levels in the soil were only interpreted based on their predicted environmental concentrations (PECs), which were submitted as part of the pesticide registration process.

The pesticide risk assessment employed by Silva *et al*. (2019) relied on the comparison of toxicity exposure ratios (TERs) and trigger values. Toxicity exposure ratios for single residues were calculated by dividing specific ecotoxicologically relevant concentrations by the higher of the initial PEC or the accumulated PEC. Two ecologically relevant concentrations were widely used: the concentration resulting in the mortality of 50 percent of the exposed individuals (the lethal concentration [LC_{50}]) for acute and short-term toxicity, and the highest no observed effect concentration (NOEC) for chronic or reproductive toxicity assessments.

In Europe, the trigger points that indicated an unacceptable risk for indicator organisms were set at a TER lower than 10 for acute exposures and a TER lower than 5 for chronic exposures (Ockleford *et al.* 2017). The in-soil indicator organisms specified in European protocols were the earthworms *Eisenia fetida* and *E. andrei,* the Collembola *Folsomia candida* and *F. fimetaria*, and the mite *Hypoaspis aculeifer*. Effects on the N transformation of microorganisms were assessed by comparing a treated

sample with a control rather than assessing with TERs. Effects above 25 percent after 100 days of exposure were considered to represent an unacceptable risk (Silva *et al*., 2019).

Tang *et al.* (2021) used the PEST-CHEMGRIDS v1.0 database to produce a global map of potential pollution by 92 active pesticide ingredients in soil, surface water, groundwater and atmosphere for 168 countries. The authors calculated a predicted environmental concentration (PEC), based on the recommended application rates for each pesticide, and compared this to the predicted no-effect-concentration (PNEC), which was derived from each active ingredient's ecotoxicity. The risk score for each grid cell was calculated as the maximum risk point across the four environmental compartments.

Overall, 74.8 percent of global agricultural land (or approximately 28.8 million km^2) was at some risk of pesticide pollution, with 64 percent at risk of pesticide pollution by more than one active ingredient, and 31 percent judged to be at high risk (Tang *et al*. 2021). The distribution of the risk scores among the regions (Table 3.7) showed that North America, Europe and Asia had the greatest pesticide pollution risk levels, with Oceania having the lowest. Maps that combined high pollution risk, high water scarcity and high biodiversity identified watersheds at high risk of pesticide pollution, including the Parana watershed in Argentina, the Murray watershed in Australia, the Huang He watershed in China, the Indus watershed in India, and the Orange watershed in South Africa (Tang *et al*. 2021).

Table 3.7. Summary of percentage of agricultural land in different risk score classes for potential pesticide pollution

Risk score	Rating	North America	Latin America and the Caribbean	Africa	Europe and Central Asia	Asia	Oceania
		Agricultural land in each risk score class (%)					
≤0	Negligible	13	15	35	3	24	66
0–1	Low	6	16	20	2	8	13
1–2	Medium	8	7	22	8	5	4
2–3	Medium	36	29	12	25	24	8
3–4	High	30	30	9	55	29	9
>4	High	7	4	3	7	9	<1
Total agricultural land (km^2)		4.0 × 106	6.2 × 106	9.1 × 106	3.8 × 106	12.5 × 106	3.0 × 106

◊ *Source:* **Tang, F.H.M., Lenzen, M., McBratney, A. & Maggi, F.** 2021. Risk of pesticide pollution at the global scale. *Nature Geoscience*, 14(4): 206–210. https://doi.org/10.1038/s41561-021-00712-5

The predicted high levels of mixtures of pesticides, based on the global modelling of Tang *et al.* (2021) is supported by the regional assessments of measured concentrations and their ecological consequences in Europe. For example, Silva *et al*. (2019) assessed pesticide residues in 317 topsoil samples from the ten European Union member states. The analysis showed that 58 percent of the soils had multiple residues present, and 25 percent of the soils had a single residue detected. Only 17 percent of the tested agricultural topsoil had no pesticide residues detected. They suggested that urgent attention was required to address the toxicity of pesticide mixtures in the soil. Similarly, in France, Pelosi *et al.* (2021) investigated residues of currently used pesticides in soils and earthworms from 60 sites, including conventionally managed and organically managed fields and off-field habitats. All the soils sampled contained one or more of the 31 pesticides measured, with 83 percent of soils containing at least five pesticides and 38 percent with ten or more. Ninety percent of the soils contained a mixture of at least one herbicide, fungicide and insecticide and there was a mixture of pesticides present in 92 percent of the earthworms sampled.

Walder *et al.* (2022) examined pesticide residues and the bacterial and fungal microbiota at 60 sites in Switzerland, including 20 conventionally managed wheat fields, 20 conventionally managed with no-till and 20 organically managed wheat fields without synthetic pesticide applications. Of the factors tested, soil, climate and geographic variables primarily determined the soil microbiome, and the effects of agricultural management were relatively small in comparison. Pesticides had a greater impact on the microbiome than any other management factor. Pesticide residues had a stimulating effect on the diversity of fungi but had a consistent negative association with bacterial diversity indices. It was noted that regulatory assessments of pesticides only tested the immediate response of a single microbial indicator – soil N mineralization – and pointed to the need to expand efforts in the assessment process by increasing the number of microbial indicators and assessing the long-term effects of pesticide mixtures.

3.7.4 Emerging pollutants

There is increasing concern regarding compounds that have been developed over the last century and are now used in a wide range of human health, beauty, clothing, packaging and other material products, as well as by agricultural, energy, pharmaceutical and other industries. Many are organic or inorganic contaminants but are included as "emerging contaminants" as they have not previously been monitored in the environment (FAO and UNEP, 2021a, 2021b). The thousands of potential pollutants also include plant and animal hormones, endocrine disrupters, antibiotics, antibiotic-resistant microbes, and microplastics.

Microplastics, and associated chemicals, such as bisphenol (a constituent of plastics) and phthalates (plasticizers), are emerging soil pollutants (FAO and UNEP, 2021b). Common sources include the application of sewage sludge as fertilizer and wastewater as irrigation, plastic mulching of horticultural crops, littering, tyre wear and atmospheric deposition (Büks and Kaupenjohann, 2020).

Liu *et al.* (2023) reported that microplastics were ubiquitous in urban and natural green space surface soils, worldwide, including Antarctica, with an average of 916 items/kg of soil (with a range of 167–3 482 items/kg of soil), mainly consisting of polypropylene and polyester fibres. A global review of 23 studies, sampling a total of 223 sites (Büks and Kaupenjohann, 2020), showed that soil microplastic concentrations in the vicinity of urban areas were generally ten times higher than rural areas, with values on some industrial sites exceeding the common range by two to four orders of magnitude. The median soil microplastic content was 1 167 items/kg of soil and 0.6 mg/kg of soil. Although studies of microplastics have increased since 2015 there is still limited knowledge of the concentrations of microplastics in relation to adverse effects (Büks and Kaupenjohann, 2020).

Antibiotic-resistant genes were described as ubiquitous in both urban and adjacent natural green spaces in the global study of Liu *et al*. (2023), though the proportion of different types varied across locations. There was a greater diversity of antibiotic-resistant genes in urban than adjacent natural areas and lower levels in Antarctica, which supported a link to human antibiotic use. In a global study of 1 088 soil metagenomic samples, Zheng *et al*. (2022) reported that antibiotic-resistant gene numbers were higher ($P < 0.01$) in agricultural soils than in non-agricultural habitats, with higher levels in East Asia, India, eastern United States, and Western Europe (all areas characterised by dense human populations). Levels were also high in soils characterized by abundant C, N and P, such as northern Europe and New Zealand.

3.8 Urban expansion and changes to soil functions

3.8.1 Introduction

Urbanization is defined by the ITPS (2022) as the conversion of agricultural, natural or semi-natural areas to urban uses including sparse settlements, urban fringes, industrial estates and transport infrastructure (discussed in Section 2.2.7).

Substantial growth in urban lands occurred during the period up to 2015: for example, He *et al.* (2019) estimated a total increase of urban land of 126 percent between 1992 and 2016 (Table 3.8). The lowest percentage increases occurred in regions that were already highly urbanized, such as Europe and Central Asia and the highest in regions such as sub-Saharan Africa that had higher rural populations in 1992. Between 2015 and 2020 Africa (with a 1.3 percent annual increase in urban population), and Asia (with a 1.1 percent annual increase in urban population), were projected to have the highest rates of urbanization (UNDESA, Population Division, 2019).

Table 3.8. Urban land expansion between 1992 and 2016

Region/country	Urban land in 1992 (km²)	Urban land in 2016 (km²)	Increase between 1992 and 2016 (%)
Asia	65 629	209 815	220
China	15 865	88 047	455
India	8 012	22 450	180
Viet Nam	190	4 689	2 368
Indonesia	1 685	6 076	261
Malaysia	1 189	5 026	323
North America	78 856	148 535	88
United States	70 704	137 093	94
Latin America and the Caribbean	36 191	99 376	175
Brazil	15 181	43 998	190
Argentina	5 528	16 905	206
Mexico	5 321	13 249	149
Europe and Central Asia	81 478	121 339	49
Türkiye	1 622	9 066	459
Italy	7 688	13 141	71
France	7 706	13 018	69
Spain	2 971	8 216	177
Russian Federation	21 000	24 639	17
Sub-Saharan Africa	7 700	27 597	258
South Africa	2 957	8 126	175
Southwest Pacific	4 798	14 481	202
Australia	4 412	12 762	189
Near East and North Africa*			
Islamic Republic of Iran	2 905	8 721	200
Saudi Arabia	1 646	6 657	304
World	274 652	621 143	126

*Data could not be calculated for the Near East and North Africa (NENA).

Source: **He, C., Liu, Z., Gou, S., Zhang, Q., Zhang, J. & Xu, L.** 2019. Detecting global urban expansion over the last three decades using a fully convolutional network. *Environmental Research Letters*, 14(3): 034008. https://doi.org/10.1088/1748-9326/aaf936

A key concern with urbanization is the loss of food production due to the conversion of agricultural land to urban uses. Gardi, Florczyk and Scalenghe (2021) attempted a broad estimation of the intrinsic soil fertility associated with land converted to urban uses, using a soil productivity index. The overall impact on agricultural production capability of the 145 000 km^2 land converted to urban use between 2000 to 2014 was estimated to be 63 million tonnes of cereals. It was acknowledged that the soil type measures used gave a crude estimate of final food security but did give an indication of the scale of the problem.

The projected increases in global and urban populations are examined in Chapter 4. The projected increases in urban population will increase the area of urban land and require the conversion of non-urban land to urban uses. Chen *et al.* (2020) modelled the likely sources of land to be converted under the five main Shared Socioeconomic Pathways used for global change impact forecasting (Table 3.9). In every scenario, cropland was the greatest potential source of new urban land.

3.8.2 Effects of urbanization on soil functions and ecosystem services

Van Vliet, Eitelberg and Verburg (2017) estimated the impact of urbanization on food production capacity from 2000 to 2040 (Table 3.9). It was estimated that urban land would increase from 213 million ha in 2000 (or 2.1 percent of the terrestrial surface) to 621 million ha in 2040 (or 4.7 percent). The increase in urban use occurred preferentially in land that was both suitable and available for crop production. The predicted increase between 2000 and 2040 could cause the displacement of almost 65 million tonnes of crop production. In turn, this loss could lead to the expansion of agriculture onto 35 million ha of new cropland, with all the attendant impact on soil functions discussed elsewhere in this report. The greatest displacement of crop production by urbanization was predicted to occur in China (13.9 million tonnes) followed by sub-Saharan Africa (12.8 million tonnes), Central and South America (8.0 million tonnes) and India (7.4 million tonnes).

Table 3.9. Change in urban area in major world regions and displacement of food production due to urbanization

Region	Urban land*			Urban land in locations available for crop production			Displacement of food production due to urbanization
	2000 (million ha)	2040 (million ha)	Change (%)	2000 (million ha)	2040 (million ha)	Change (%)	(%)
Central and South America	37.5	100.6	63.1	22.2	51.8	29.6	2.9
North America	42.8	111.7	68.8	20.1	32.5	12.4	2.9
Sub-Saharan Africa	11.3	68.4	57.1	6.9	42.7	35.8	3.6
Near East and North Africa	15.9	29.1	13.2	5.9	8.2	2.5	7.4
Europe	59.4	92.0	32.5	43.9	61.2	17.2	4.8
Russia and Central Asia	23.8	47.3	23.5	11.6	18.6	7.0	0.5
India	16.6	45.9	29.3	13.7	34.0	20.2	3.3
China	40.2	85.9	45.7	24.7	40.1	15.4	12.5
Asia Pacific	20.2	30.7	10.5	15.1	22.0	7.0	2.0
Oceania	3.1	9.7	6.6	1.6	4.1	2.5	1.3
World	270.9	621.3	350.4	165.5	315.3	149.8	3.7

*** Urban land refers to peri-urban and village systems and urban systems combined.**

Source: Adapted from **Van Vliet, J., Eitelberg, D.A. & Verburg, P.H.** 2017. A global analysis of land take in cropland areas and production displacement from urbanization. *Global Environmental Change*, 43: 107–115. https://doi.org/10.1016/j.gloenvcha.2017.02.001

Conversion to urban land also impacts other soil functions and ecological goods and services beyond food production. The direct impacts differ depending on the soil modifications that occur during conversion. Morel, Chenu and Lorenz (2015) used a broad definition of urban areas that included industrial, traffic, mining, and military areas as well as urban uses. The soils of urban areas were classified into four groups: pseudo-natural soils, vegetated engineered soils, dumping site soils, and sealed soils.

Pseudo-natural sites such as urban forests and urban gardens, and vegetated engineered soils (where topsoil is spread over a disturbed subsoil) continue to offer a considerable range of ecological services, including high levels of food and non-food biomass, water storage and runoff and flood protection, and the protection of biodiversity, as well as cultural services such as recreation and tourism. Dumping site soils can be reclaimed to offer a limited range of services, whereas sealed soils are very limited in the services they can offer.

Soil sealing is defined by ITPS (2022) as the permanent covering of the soil surface with impervious materials such as concrete or asphalt, tar seal as well as buildings and other structures that could not be easily removed. Sealed surfaces supported roadways and structures as well as protecting below ground infrastructure such as cables, pipes and tunnels. The greatest impact of soil sealing was on the provisioning services of food and biomass production, the regulation of hydrological flows, the potential for C sequestration and loss of biodiversity (ITPS, 2022).

Gong *et al.* (2020) used a 34-year record of LANDSAT data to estimate the growth in artificial impervious area (equivalent to soil sealing) and reported that the total area of artificial impervious surfaces in 2018 was 79.8 million ha, which was one–and-a-half-times greater than in 1990. Increases were highest in Asia and North America followed by Europe, South America, and Africa. At the national level, China and the United States had the greatest growth in impervious areas.

Reviews by O'Riordan *et al.* (2021), Morel, Chenu and Lorenz (2015), Decoville and Feltgen (2023) and Minixhofer and Stangl (2021) identified the impacts of soil sealing on urban hydrology as its most significant effect on ecosystem services, and found that modifications in water flow within the soil could lead to the destabilization of surface soil material and increases in the incidence of fatality-producing shallow landslides.

The establishment of photovoltaic (PV) panel systems (or solar farms) on agricultural land has continued since 2015. The installation of PV systems requires large areas of land and leads to competition between agriculture and renewable energy (Asa'a *et al.*, 2024). The impact on soil functions has been little studied and can be minimized through agrivoltaics systems, which are installations that allow continued crop growth and harvest (Asa'a *et al.*, 2024).

The projected increases in urbanization will inexorably impact provision of ecosystem services by soils. Measures to mitigate the impact of this expansion are discussed in Section 5.3.6 of this report.



3.9 Physical degradation

3.9.1 Introduction

The physical matrix of the soil consists of solid particles and pore space and it is within the pore space that roots grow, gases are exchanged and water moves through the soil. The natural configuration of pore space is determined by soil properties such as the soil texture (i.e. the relative proportion of sand, silt, clay, and gravel), soil structure, organic matter and the organisms living in the soil and growing within it. Activities such as the use of heavy machinery, livestock trampling, or ploughing can alter the pore network profoundly and lead to changes in the functions delivered by the soil.

3.9.2 Soil compaction, hardsetting and workability

Soil compaction is described as the loss of pore space due to pressure exerted at the soil surface. The effects of compaction on soils have been well established (Yang *et al.*, 2022b; Panagos *et al.*, 2024). Compared to uncompacted soils, compacted soils are denser, have a lower number of macropores and a lower ability to transmit air and water, which in turn lowers the water infiltration rate of water into the surface and the hydraulic conductivity (i.e. the ability to transmit water within the soil profile). These changes in soil physical properties increase the risks of waterlogging on level surfaces and of runoff and water erosion on sloping surfaces. Compacted soils also impede root elongation and development, which reduces soil nutrient uptake and crop development and ultimately causes yield loss. The altered soil aeration and drainage affect nutrient transformations and can increase GHG emissions. Decreased aeration and increased wetness may also predispose compacted soils to infection of root rot diseases. In forest soils, compaction caused by forest machinery ultimately hinders root and tree growth and can lower overall site productivity (Labelle *et al.*, 2022; Nazari *et al.*, 2021).

Concerns about soil compaction have increased due to the rise in the size and weight of farm machinery in many developed countries. For example, Schjønning *et al.* (2015) present data for nine self-propelled combine harvesters for the period from 1958 to 2009 that show that the weight of fully loaded machines increased by a factor of 6 and the mean ground pressures exerted by the machines increased by 43 percent over the same time.

Soil crusting is a second type of physical issue that occurs in soils. Soil crusts form at the soil surface and range from a few mm thick to multilayered, thicker crusts. Soil crusts form at the soil surface upon drying of soil seals caused by raindrop impact. Raindrop-induced soil sealing reduces infiltration and inhibit aeration (gas exchange), while soil crusts inhibit germination and seedling emergence and limit development and functioning of roots (Laker and Nortjé, 2019). Hardsetting is a similar phenomenon and occurs where most (if not all) of the A-horizon soil sets to a hard, structureless mass during drying (Daniells, 2012). The layer that is set hard is thereafter difficult or impossible to cultivate until the profile is rewetted. Both pose agronomic limitations due to a restricted time when they can be tilled, the difficulty in producing a good seedbed, the constraints posed to seedling emergence and constraints to root growth.

Soils susceptible to compaction, crusting and hardsetting have low structural stability typically due to low organic matter content and a high content of silt and fine sand (Yang *et al.*, 2022b; Daniels, 2012; Laker and Nortjé, 2019). Crusting and hardsetting are also associated with soils with a higher-than-normal Na content, which causes soil aggregates to disperse and (upon drying) crusts and hardset layers to form.

While soil compaction is a specific type of degradation process, the structural stability of the soil

contributes to a more complex soil quality called workability. Workability refers to the condition of the soil when tillage operations can be conducted without causing structural damage (Obour *et al.*, 2017). Soil is considered workable when tillage operations produce suitable seedbeds for crop establishment without smearing or compaction. Workability depends on a combination of tillage systems, farm machinery and soil factors such as soil water content, bulk density, texture, clay and SOM.

Workability is commonly cited by farmers as a key element in their evaluation of soil productivity and soil health in many settings (e.g. Barrera-Bassols and Zinck, 2003; Yageta *et al.*, 2019; Castellanos-Navarrete *et al.*, 2015; Jiménez *et al.*, 2024; Klein *et al.*, 2024).

In practice, soil conditions for seedbed preparation are mostly based on qualitative field assessment by farmers, who often crumble soil to see how it breaks (Obour *et al.*, 2017). This test is formalized in soil descriptions in the field as soil consistence, which is the resistance of soil aggregates to an applied force that is usually applied by the fingers of the soil assessor (FAO, 2006).

Soil consistence and workability are directly related to soil moisture content, the effects of previous farm operations on the soil surface and the machinery used for tillage and seeding. Soil moisture conditions are highly dynamic through time and space and no global or regional datasets on machinery size (including weight and working width) are available (Sonderegger and Pfister, 2021). This makes modelling workability at any scale beyond a single field very difficult. Bouma (2014) and Bouma *et al.* (2022) have stated that complex attributes such as workability can only be estimated if real-time monitoring of soil moisture conditions or dynamic modelling of soil moisture is available.

The inability to model complex attributes such as workability is further complicated by the challenges in field assessment of soil compaction itself. Soil compaction is itself highly variable in time and space and there is no commonly measured soil property to assess it. Panagos *et al.* (2024) used a proxy measure of compaction (known as the soil packing density) to assess compaction as part of the intensive LUCAS sampling programme in Europe. Ideally, this would be tested in a broader range of environments to assess its general applicability. Compaction is also readily altered by tillage or other surface disruptions, and these rapid alterations also complicate its measurement and modelling.

Overgrazing can substantially increase soil compaction, primarily through the repeated trampling of livestock, which compresses soil particles and reduces pore space in the upper soil layers. Numerous studies have shown that intensive grazing typically leads to higher soil bulk density, lower porosity, and reduced infiltration rates compared with ungrazed or moderately grazed systems (Byrnes *et al.*, 2018; Lai and Kumar, 2020). These structural changes can impair water movement, restrict root growth, and decrease soil aeration, ultimately affecting plant productivity and ecosystem functioning. Global syntheses and meta-analyses indicate that the magnitude of soil compaction depends on grazing intensity, soil texture, soil moisture conditions, and management practices, with the strongest effects generally occurring under continuous heavy grazing and on moist or fine-textured soils (Byrnes *et al.,* 2018). Consequently, appropriate grazing management – such as reducing stocking rates or implementing rotational grazing – can mitigate compaction and help maintain soil structure and soil health in grazed ecosystems (Lai and Kumar, 2020).

Soil compaction is also a significant concern in forestry. Soil compaction is one of the most pronounced consequences of mechanized logging operations, caused by imposed vertical and horizontal stresses on the soil (Nazari *et al.*, 2021). Skid trails and rutting from machinery used in logging operations can greatly disturb physical, chemical and biological properties of soils and severely alter the forest ecosystem function. Assessments of trafficability prior to forest harvest can reduce the impact of logging and surface amendments such as mulches and brush mats can reduce the impact of logging (Labelle *et al.*, 2022).

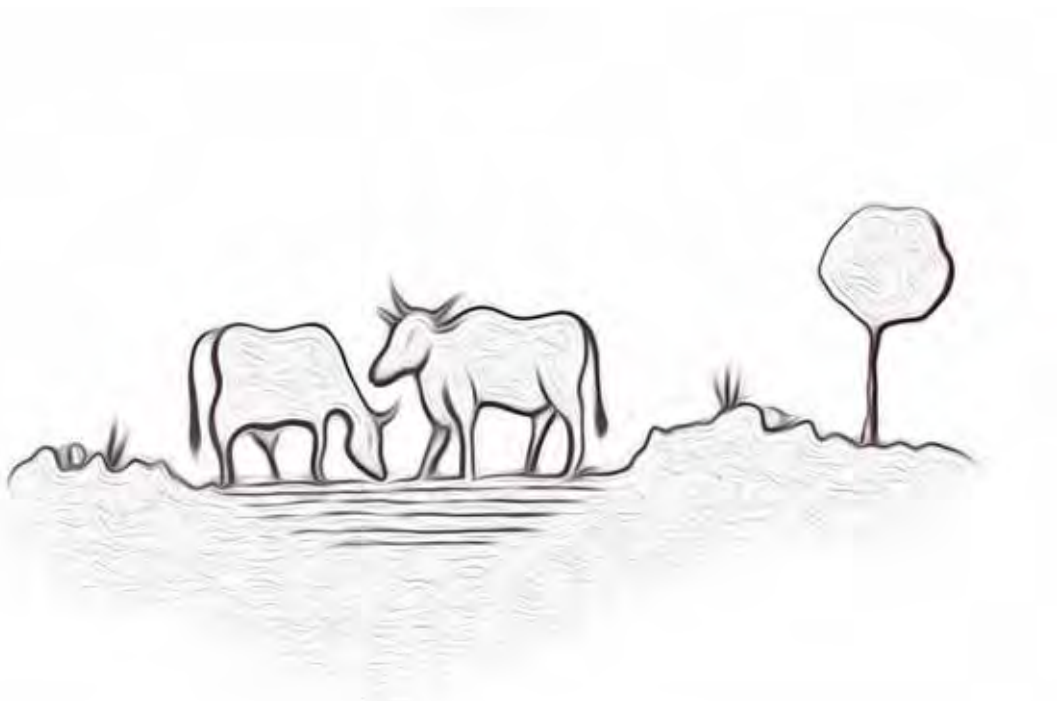
The impacts of compaction on the properties on forest soils can be severe. Nazari *et al.* (2021) completed a meta-analysis of 67 studies on compaction in forest soils with a total of 748 pairs

of observations. Across all studies, compaction significantly increased soil bulk density (with an average effect size of 8.9 percent) and decreased soil total porosity by 10.1 percent. The soil saturated hydraulic conductivity (a measure of its ability to transmit water) was reduced on average by 40 percent across all studies, showing that the density and porosity changes substantially reduced the soil's ability to transmit water through the profile.

There are no global or even regional estimates for the costs of soil compaction. On a country scale, the costs of soil compaction in England and Wales (in the United Kingdom) were estimated by Graves *et al.* (2015), who found that approximately 40 percent of the total agricultural land in the two countries was at risk of compaction. They estimated that compaction reduced crop yield by 4 to 5 percent, relative to uncompacted land. The reduced infiltration of water also led to a 3 to 10 percent increase in runoff from agricultural fields and hence an increase in offsite flooding.

Results from field experiments by Schjønning *et al.* (2015) on the impact of compaction on crop yields showed overall decreases on the order of 5 to 10 percent over a decade, although in some cases only very minor effects were seen. It was suggested that field experiments might have underestimated the effect of compaction-induced poor drainage, which could have reduced the number of workable days in the field, which, in turn, could have affected the conditions for establishing the crop. Poor drainage might have also caused problems for crop harvests in periods with high precipitation, as the possibility of the total loss of a year's crop would be much more serious to the farmer than the average effect of compaction.

Recently there has been considerable research focus on the effects of compaction on N_2O release from soils. Hernandez-Ramirez, Ruser and Kim (2021) completed a meta-analysis of compaction effects based on 22 studies with 108 available comparisons. It was reported that compaction had increased N_2O emissions in 82 percent of the cases, with the strongest effects in forest soils followed by pasture soils. The effect was smaller for vegetable and annual crops but was still significant. The authors noted that a subset of studies did not show an effect, or showed a decrease in emissions due to compaction and suggested that the mechanisms by which compaction causes changes to N_2O emissions were complex and only partially understood.



3.10 Summary

3.10.1 Introduction

As indicated in Chapter 1, the main focus of the 2026 *Status of the world's soil resources* report is the assessment of the current status of the main threats to soil functions and the ability of SSM to address these threats. The threats chosen for inclusion are soil erosion, nutrient mismanagement, the loss of SOC, salt accumulation, soil pollution, the loss of soil biodiversity, soil physical deterioration, and urbanization and soil sealing. The chapter focuses on global-scale studies on the extent and severity of threats to soil functions.

3.10.2 Summary for soil erosion

The results of modelling studies suggest that water erosion rates are unacceptably high (i.e. greater than 6 t/ha annually) in a substantial percentage of global cropland. The global modelling study of Borrelli *et al*. (2017) found that 6.1 percent of the land area (or about 7.5 million km^2 in total) had water erosion rates greater than 10 t/ha annually. The global modelling study by Carr *et al*. (2020) estimated that average water erosion rates were 11 t/ha annually for 58 percent of maize-producing land and 6 t/ha annually for 46 percent of wheat-producing land. There is limited agreement across the two studies about the specific areas experiencing unacceptably high erosion.

The effects of climate change on water erosion are difficult to predict as they depend largely on how human societies respond to climate change, as presented in the five SSPs. Both Borrelli *et al.* (2020) and Eekhout and De Vente (2022) predicted increases in water erosion overall but with considerable uncertainty about the magnitude of the increase.

The estimated annual costs of water erosion due to reduction in crop yield are relatively small but the cumulative impact increases through time. The offsite costs of water erosion through sedimentation in reservoirs and surface waterways, degraded water quality and GHG emissions from eroded soil and nutrients is likely to be substantially greater than the onsite costs, but these offsite costs are challenging to estimate. Ideally, the threshold levels of erosion used to identify unsustainably high rates in the future will incorporate information on offsite impacts.

The summarized research falls short in providing an integrated analysis that synthesizes climatic data, land management practices and socioeconomic impacts. To address this gap, future research should prioritize the development of interdisciplinary models capable of predicting soil erosion patterns under diverse scenarios. These models should incorporate advanced technologies such as remote sensing for monitoring purposes and inform policies that promote sustainable land management practices.

Wind erosion is a serious problem in arid and semi-arid regions. While there are wide differences in the estimates of total wind erosion and rates on a per hectare basis across the various modelling studies of global wind erosion, there is more agreement about the locations with high dust emissions. Most dust emissions due to wind erosion are from natural sources, principally in North Africa, Saudi Arabia, the Taklamakan Desert of East Asia, southwest United States and northern Mexico, the deserts of Chile and Peru in South America and the Namib Desert in southeast Africa. The greatest sources of anthropogenic emissions are in the Sahel in sub-Saharan Africa, the highlands of Saudi Arabia, northern China and Mongolia, the high plains of the United States and the Sonoran Desert in Mexico, river valleys in Patagonia, southeast Africa, and areas in Australia.

Overall, tillage erosion has become well-established as a significant erosion process (FAO, 2019b). Regional and global modelling of tillage erosion has lagged behind those for water and wind erosion,

in part because tillage erosion modelling requires detailed information on slope characteristics and farm equipment that is not widely available. The studies that do exist suggest that tillage erosion may be a threat comparable to water erosion.

3.10.3 Summary for soil organic carbon loss

Soils are the largest land-based store of C. In the IPCC's Sixth Assessment report (Canadell *et al.*, 2021) permafrost soils were estimated to store approximately 1 200 billion tonnes of C and mineral and wetland soils outside of the permafrost region stored 1 700 billion tonnes, with terrestrial vegetation storing a further 450 billion tonnes of C.

Mineral soils

Digital soil mapping has been used to produce detailed global and regional maps of SOC distribution since 2015. These maps draw upon an ever-growing global database of soil properties that serves as one source of information for the mapping procedures. Digital soil mapping has been used to develop estimates of the loss of SOC over the past centuries, with an estimate of 133 billion tonnes of C for the upper 2 m of soil (Sanderman, Hengl and Fiske, 2017). Hotspots of SOC loss are widespread through most regions and are associated with areas undergoing land-use conversion from forest and grassland to cropping and areas of higher soil erosion rates by water and wind. The effect of climate change until 2100 is projected to cause a small increase in the rate of SOC loss. Research published since the release of the IPCC Sixth Assessment report (Canadell *et al.*, 2021) (Hashimoto, Ito and Nishina, 2023; Ren *et al.*, 2024; Nissan *et al.*, 2023) has strengthened the finding that warming climates will lead to higher rates of soil respiration and losses of C from the soil to atmosphere.

The lack of a comprehensive assessment of the regional or global economic cost of SOC loss from mineral soils is a major impediment to the development of effective policymaking to limit SOC losses.

Wetland soils

Organic and mineral wetlands soils store large amounts of C relative to the area of land they occupy. Global peatlands are estimated to store about 19 percent of global SOC stores (i.e. about 540 billion tonnes [Jackson *et al.*, 2017] out of a total of 2 900 billion tonnes [Canadell *et al.*, 2021]) although they only occupy about 3 to 4 percent of the land mass. Mineral wetlands (i.e. those without an organic layer or with only a thin organic layer) are widespread in many landscapes, but no separate estimate of their overall C storage exists. Both types of wetlands are sources of the potent GHG, CH_4. Dean *et al.* (2018) estimated that natural wetlands were the source of 30 percent of total CH_4 emissions.

Peatlands and mineral wetlands in northern latitudes are primarily at risk due to climate change. The higher temperatures can both increase the rate of decomposition and increase the incidence of fire. Under the current warming rates, C loss due to fire in Arctic boreal peat have been estimated to increase from 143 million tonnes of C in 2015 to 256 million tonnes of C by 2050 and 544 million tonnes of C annually by 2100 (Lin, Liu and Huang, 2021). Peatlands in the tropical zone are juxtaposed with densely populated urban areas and are subject to multiple land-use pressures. Fire-related losses are very high in these regions, along with land clearing for crops and timber or pulp production.

Coastal mangrove forests are also under multiple threats, including coastal erosion, conversion to agriculture and aquaculture, extreme climate events, mangrove clearing and expansion of human settlements. Adame *et al.* (2021) estimated total cumulative emissions from mangrove loss from 2010 to 2100 of 653 million tonnes of C.

Permafrost

Permafrost is a major store of SOC, with approximately 1 200 billion tonnes out of a total of 2 900 billion tonnes being stored in permafrost (Canadell *et al.*, 2021). Soil organic carbon storage in permafrost soils is at risk due to the effects of global warming. A secondary threat arising from permafrost thaw is damage to human infrastructure (the loss of ground ice through melt destabilizes the near-surface layer of the soil and reduces the structural support for infrastructure).

Direct measurements of temperatures in permafrost have documented the rise in temperature, and projections for the future have estimated substantial decreases in the area of permafrost by 2100. The net ecosystem balance of C in the permafrost zone is, however, complex to estimate. The many uncertainties and potentially offsetting tendencies (e.g. enhanced plant growth) caused the IPCC to express considerable caution in their summary about SOC loss from permafrost in the Climate Change 2021 report (Canadell *et al.*, 2021). It was concluded that:

> *thawing terrestrial permafrost will lead to C release under a warmer world (high confidence). However, there is low confidence on the timing, magnitude and linearity of the permafrost climate feedback owing to the wide range of published estimates and the incomplete knowledge and representation in models of drivers and relationships. It is projected that CO_2 released from permafrost will be 18 (3.1–41) billion tonnes of C $°C^{-1}$ by 2100, with the relative contribution of CO_2 vs CH_4 remaining poorly constrained* (Canadell *et al.*, 2021, p. 728).

3.10.4 Soil biodiversity

Soil is the most biodiverse system on earth, hosting approximately 59 percent of all the planet's biodiversity (Anthony, Bender and Van de Heijen, 2023).

Habitat modification or loss is the greatest threat to soil organisms (Bach *et al.*, 2020). The largest driver of habitat loss is the conversion of forests and grasslands to agriculture, and many studies have shown that agricultural fields support smaller and less diverse soil communities than grasslands and forested areas (FAO *et al.*, 2020). The meta-analysis by Qu *et al.* (2024) found that the conversion of forest to cropland, grassland or plantation generally led to ecosystems that were dominated by bacteria (rather than fungi) and where the fungal communities were more homogenous than in native soils. The changes in fungal diversity associated with deforestation may also have especially important effects on soil functions such as organic matter decomposition and nutrient availability in low N ecosystems (Dong *et al.*, 2024).

Within agricultural landscapes, the intensity of production can further reduce soil biodiversity, with the dependence of intensive agriculture on irrigation, heavy machinery, tillage and the increasing use of agrochemicals such as fertilizer and pesticides all causing significant habitat modification (Geisen, Wall and Van der Putten, 2019).

The summary of threats to biodiversity highlighted the strong control of soil pH on soil biodiversity. Generally, soil biodiversity and the contributions of soil-dwelling organisms to soil functions are at an optimum in soils within a neutral pH range of approximately 5.5 to 7.0. Drivers such as deforestation and application of fertilizers that shift pH cause major shifts in the soil microbiome.

3.10.5 Nutrient mismanagement

Nutrient budgets are an essential tool to ensure the sustainable use of nutrients. Ideally, inputs of nutrients should be roughly in balance with outputs of nutrients in crops and animal products. If outputs are greater than inputs, mining of soil nutrient reserves occurs and plant yields will drop

below the maximum levels possible in a given location. If inputs are substantially greater than outputs, the excess nutrients can cause harm to air and water quality.

Global estimates of N use efficiency are in the range of 45 to 50 percent (Kaltenegger and Winiwarter, 2020; Lassaletta *et al.*, 2014; Bouwman *et al.,* 2017), indicating that approximately only half of added N is exported in agricultural products.

Nutrient mining is a major cause of soil degradation as it reduces the soil's capacity to supply nutrients, which, in turn contributes to a yield gap: the difference between the actual yield achieved and the potential yield that could be achieved under optimal conditions. Optimum plant nutrition requires that all nutrients be available in the required forms and amounts. The absence of a required nutrient can severely limit plant growth and can cause a soil to be unresponsive when, for example, high amounts of N or P are added. The problem of non-responsive soils is common in sub-Saharan Africa (Sileshi *et al.*, 2022).

Acidification is an increase in the acidity in the soil. Increasing soil acidity can reduce the availability of important plant nutrients and has a strong effect on the global distribution of soil bacteria, fungi and larger soil animals such as earthworms. The main anthropogenic-induced causes of acidification are from the deposition of acidifying gases and particles from the atmosphere and the addition of acidifying fertilizers in cropping systems or the cultivation of legumes. In some areas, the draining of potentially acid sulphate soils can also lead to localized acidification.

Nutrient excess causes major offsite problems. Two forms of gaseous emissions of N (N_2O and NH_3) are of greatest concern. Nitrous oxide is a potent GHG (Jia *et al.* 2019). Agriculture is responsible for approximately two-thirds of the global N_2O emissions into the atmosphere, driven primarily by increases in manure production and synthetic N fertilizer use from the mid-twentieth century onward (Jia *et al.,* 2019). Emissions from croplands have been responsible for over 80 percent of recent increases in atmospheric N_2O concentration (Tian *et al.*, 2019).

Ammonia emissions from soils are linked to environmental and human health concerns. Bauer, Tsigaridis and Miller (2016) cited studies showing that in the past 70 years, global NH_3 emissions more than doubled (from 23 million tonnes annually to 60 million tonnes annually). Increases in emissions between 2008 and 2018 are largely linked with increases in synthetic fertilizers and biomass burning related to agricultural practices (Van Damme *et al.*, 2021).

In addition to the harmful effects of N_2O and NH_3, atmospheric deposition of acidifying N (mainly in the forms of NOx and NH_3) is a global driver of soil acidification. This acidification occurs when urbanization, industrialization, mining, construction or dredging release acidifying substances into the air or water.

Leaching is the dominant cause of rising nitrate concentrations in groundwater and contributes to ecosystem eutrophication and water quality degradation. The emission factor used by IPCC (2006) assumes that nitrate loss by leaching accounts for 30 percent of applied N fertilizer. Wang *et al.* 2019 reported that leaching increased along with the increasing N application rates. Nitrate leaching with increasing N inputs also rose rapidly when the N input exceeded the optimum rate for plant uptake.

Phosphorus in soil originates from soil weathering, atmospheric deposition and inputs of P fertilizers (both inorganic and organic [animal manures and human waste]). Chen and Graedel (2016) reported that in 1961, 56 percent of agricultural inputs were from organic sources (principally manure) and 29 percent from inorganic fertilizers. In the 2017–2021 period, the average P inputs across countries was 48 percent from synthetic fertilizer and 52 percent from manure (FAOSTAT, 2024).

Most of the added P that is not taken up by the crop is retained by the soil (legacy P) or stored in sediments when the eroded soil is deposited on alluvial flood plains, or on lake and ocean floors. Phosphorus adsorbed by soil can persist for decades (Zhang *et al.,* 2020). The release of P from the

P legacy pools may mask the effects of management actions that have been taken to reduce P inputs to aquatic systems.

The greatest concern with excess P applications is the risk of water pollution. Globally, the largest portion of P is in particulate form (eroded soil) that is transported with runoff waters from agricultural fields. In regions where fertilization rates are high, a larger percentage of P is transported in dissolved form.

3.10.6 Salt accumulation

Soils that contain salts occur naturally in semi-arid and arid areas, as well as on coastal margins. However, anthropogenic activities, particularly associated with land clearance and irrigation in drier regions, can lead to salt accumulation in soils which may inhibit plant growth.

FAO (2024) reported that the global area of salt-affected soils amounts to 1 381 million ha, or 10.7 percent of the total land area, with a further 1 038 million ha potentially at risk of salinization and this estimate is close to that from other sources. There is broad agreement across authors (e.g. Wicke *et al*., 2011; Hassani, Azapagic and Shokri, 2020; FAO, 2021) on the areas with the highest proportion of salt-affected soils. There are greater differences in the estimates of cropland affected by salinity. FAO (2024) estimates based on the GSASmap (FAO, 2021) and the European Space Agency (ESA) Climate Change Initiative (CCI) land cover map (ESA, 2017) indicated that around 10 percent of irrigated cropland, 10 percent of rainfed cropland and 8 percent of agricultural land worldwide was affected by salinity or sodicity.

Modelling studies (Hassani, Azapagic and Shokri, 2020; Ivushkin *et al.*, 2019) estimated changes in the area of salt-affected soils in the 1980–2018 period. Only South America had a statistically significant increasing trend in the total area of saline soils. Asia and North America had statistically significant decreasing trends. For sodicity, Asia, South America and North America showed statistically significant increasing trends:

Salt accumulation associated with irrigation is a major driver of salinization. Older estimates by FAO (2011) were that 34 million ha (or 11 percent of the total irrigated area) were affected by some level of salinity and that a further 60 to 80 million ha were affected to some extent by waterlogging and related salt accumulation.

A second global source of anthropogenic-induced salinization is the inundation of coastal wetlands by seawater, a process driven by sea level rise and storm surges associated with anthropogenic-induced global climate change (Chen and Mueller, 2018). The third major source of anthropogenic-induced salinization occurs due to clearance of deep-rooted, perennial vegetation such as grasses, trees and shrubs from soils overlying shallow saline aquifers (Herbert *et al.*, 2015). The resulting decrease in evapotranspiration allows more water to reach the groundwater table, which then rises until the saline groundwater is in contact with the roots of plants and the upper soil profile.

Modelling by Hassani, Azpagic and Shokri (2021) predicted that up to the year 2100, potential increases in salinization could occur in dryland areas of South America, southern and western Australia, Mexico, the southwest of the United States and South Africa. A decrease in salinity was predicted for the drylands of the northwest of the United States, the Horn of Africa, Eastern Europe, Turkmenistan and west Kazakhstan.

FAO (2024) estimated yield losses due to salinity on a land area of 644 million ha and found that the total relative yield loss was 0.5 percent, ranging from 0.03 percent for soybean to 5.05 percent for bean yields. Relative yield losses for other major crops were 0.28 percent for wheat, 0.32 percent for maize, 0.69 percent for rice and 0.21 percent for cotton.

3.10.7 Soil pollution

Soil pollution is defined as the presence in the soil of a chemical or substance out of place or present at a higher-than-normal concentration that has adverse effects on any non-targeted organism. Soil pollutants can be broadly subdivided into organic and inorganic chemicals and emerging contaminants.

A major advance in our understanding of soil pollution since 2015 was the 2021 publication of a global assessment by FAO and UNEP (2021a, 2021b). Most of the polluted sites of greatest concern are associated with point sources and are discussed in greater detail in the regional assessments in this report.

Inorganic pollutants include trace metals, asbestos and radioactive elements. Khan *et al.* (2021) reviewed global soil pollution by trace elements and reported that the soils in and surrounding urban areas in older, industrialized regions generally had high levels of trace element pollution, which they attributed to power plants, industry and pollution from traffic. Mining has left a legacy of contamination in many areas (Chen *et al*., 2022) including radionuclide contamination associated with uranium mining.

Most organic contaminants are synthetic compounds produced for specific purposes, including pesticides and industrial chemicals, derived from the petrochemical industry (FAO and UNEP, 2021a, 2021b). Studies on pesticide contamination since 2015 have established that contamination by pesticide residues are widespread in cropland (Tang and Maggi (2021)) and in natural green spaces (Li *et al*., 2023).

Many organic or inorganic contaminants are included as "emerging contaminants" as they have not previously been monitored in the environment (FAO and UNEP, 2021a, 2021b). The thousands of potential pollutants also include plant and animal hormones, endocrine disrupters, antibiotics, antibiotic-resistant microbes, and microplastics. Microplastics have been shown to be ubiquitous in urban and natural green space surface soils worldwide, including Antarctica (Liu *et al.,* 2023) and generally ten times higher in soils of urban areas than rural areas (Büks and Kaupenjohann, 2020). Antibiotic-resistant genes were described also as ubiquitous in both urban and urban-adjacent natural green spaces in the global study of Liu *et al*. (2023). The review by FAO and UNEP (2021a, 2021b) emphasizes the urgent need to study the impact of these emerging contaminants on soil functions, especially the organisms that live in the soil.

3.10.8 Urban expansion and soil sealing

Substantial growth in urban lands occurred during the period up to 2015. For example, He *et al.* (2019) estimated a total increase of urban land of 126 percent between 1992 and 2016 (see Table 3.9). The lowest percentage increases occurred in regions that were already highly urbanized, such as Europe and Central Asia and the highest in regions such as sub-Saharan Africa that had higher rural populations in 1992. Africa (with a 1.3 percent increase in urban population), and Asia (with a 1.1 percent increase in urban population), continued to have the highest rates of urbanization between 2015 and 2020 (UN-DESA, 2019).

This increase in urbanization is projected to continue into the future. Van Vliet, Eitelberg and Verburg (2017) estimated its impact on food production capacity in the period from 2000 to 2040. They estimated that urban land would increase from 213 million ha in 2000 (or 2.06 percent of the Earth's surface), to 621 million ha in 2040 (or 4.72 percent of the Earth's surface). The predicted increase between 2000 and 2040 could cause the displacement of almost 65 million tonnes of crop production.

Soil sealing due to urbanization has the greatest harmful impact on soil functions. Gong *et al.* (2020) used a 34-year record of LANDSAT data to estimate the growth in artificial impervious area (equivalent to soil sealing). They estimated that the total area of artificial impervious surfaces in 2018 was 79.8 million ha, which was one-and-a half-times greater than in 1990. Increases were highest in Asia and North America followed by Europe, South America, and Africa. The projected increases in urbanization will inexorably impact the provision of ecosystem services by soils.

3.10.9 Soil compaction

Compared to uncompacted soils, compacted soils are denser, have a lower number of macropores and a lower ability to transmit air and water, which in turn lowers the water infiltration rate of water into the surface and the hydraulic conductivity (i.e. the ability to transmit water within the soil profile). Compacted soils also impede root elongation and development, which reduces soil nutrient uptake and crop development and ultimately causes yield loss. In forest soils, compaction caused by forest machinery ultimately hinders root and tree growth and can lower overall site productivity.

Soil crusts form at the soil surface upon drying of soil seals caused by raindrop impact. Crusts range from a few mm thick to multilayered, thicker crusts. Raindrop-induced soil sealing reduces infiltration and inhibit aeration (gas exchange), while soil crusts inhibit germination and seedling emergence and limit development and functioning of roots (Laker and Nortjé, 2019).

While soil compaction is a specific type of degradation process, the structural stability of the soil contributes to a more complex soil quality called workability. Workability refers to the condition of the soil when tillage operations can be conducted without causing structural damage (Obour *et al.*, 2017). Workability is commonly cited by farmers as a key element in their evaluation of soil productivity and soil health in many settings.

There are no global or even regional estimates for the costs of soil compaction or changes to soil workability due to the absence of country-specific machinery data and data on dynamic soil moisture conditions.



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Chapter 4 | Direct and indirect drivers of threats to soil functions

4.1 Overview of drivers

Drivers refer to the underlying causes or pressures – such as population growth, unsustainable agricultural practices, industrial activities and climate change – that exacerbate threats to soil functions. In this chapter, the focus is on the most important drivers that affect the threats to soil functions and the regional differences in the operation of these drivers. National-level statistics for the drivers are presented in each of the regional assessment chapters (chapters 8–14).

In this report, the terms "direct drivers" and "indirect drivers" replace the terms "drivers" and "pressures" as used in the Driver-Pressure-State-Impact-Response (DPSIR) framework. This change is consistent with the 2018 IPBES land degradation report (Montanarella, Scholes, and Brainich, 2018) and the 2019 IPCC report on climate change and land (Olsson *et al.*, 2019), both of which did not use the drivers/pressures terminology but instead adopted the direct/indirect drivers format.

Direct drivers of soil change are changes to management, land use or environmental conditions that affect soils directly. Direct drivers are further divided into natural and anthropogenic direct drivers:

- Natural direct drivers: These are direct drivers that are not the result of human activities and are beyond human control (such as natural climate and weather patterns, natural fire cycles, volcanic eruptions, earthquakes and tsunamis).
- Anthropogenic direct drivers: These are direct drivers that are the result of human decisions and actions (such as land clearance, intensification of agriculture, tillage and residue management). The most relevant anthropogenic direct drivers of processes of soil change are shown in Table 4.1 and are the main focus of this chapter.

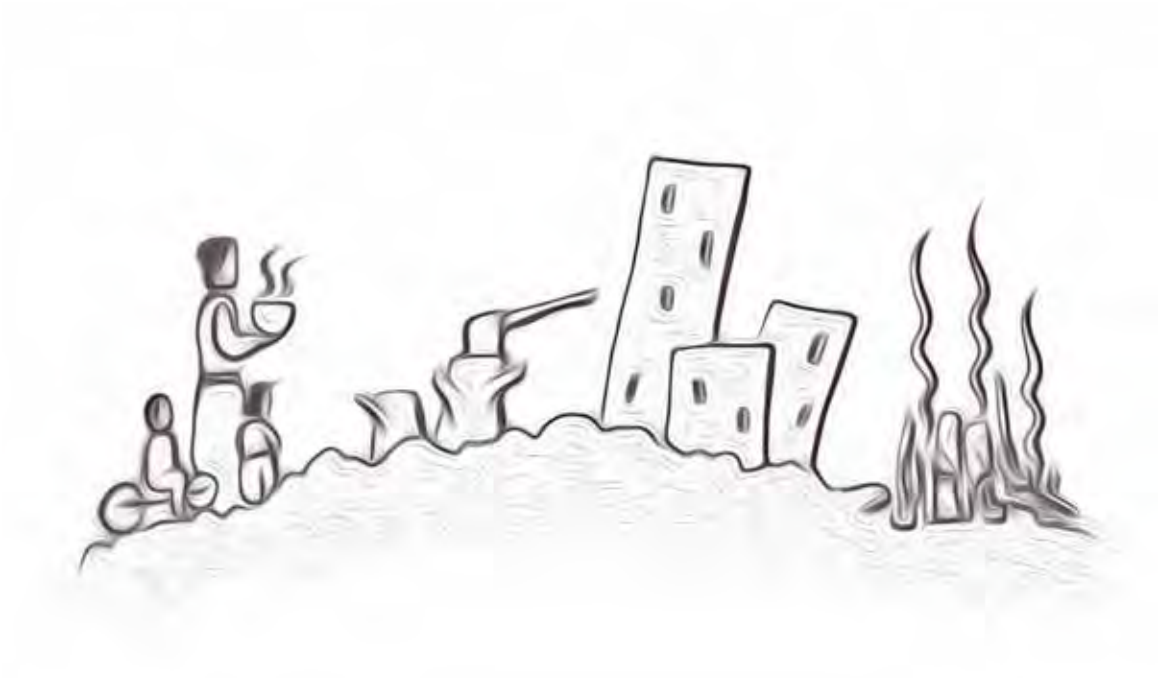


Table 4.1. Anthropogenic direct and indirect drivers of relevance to threats to soil functions

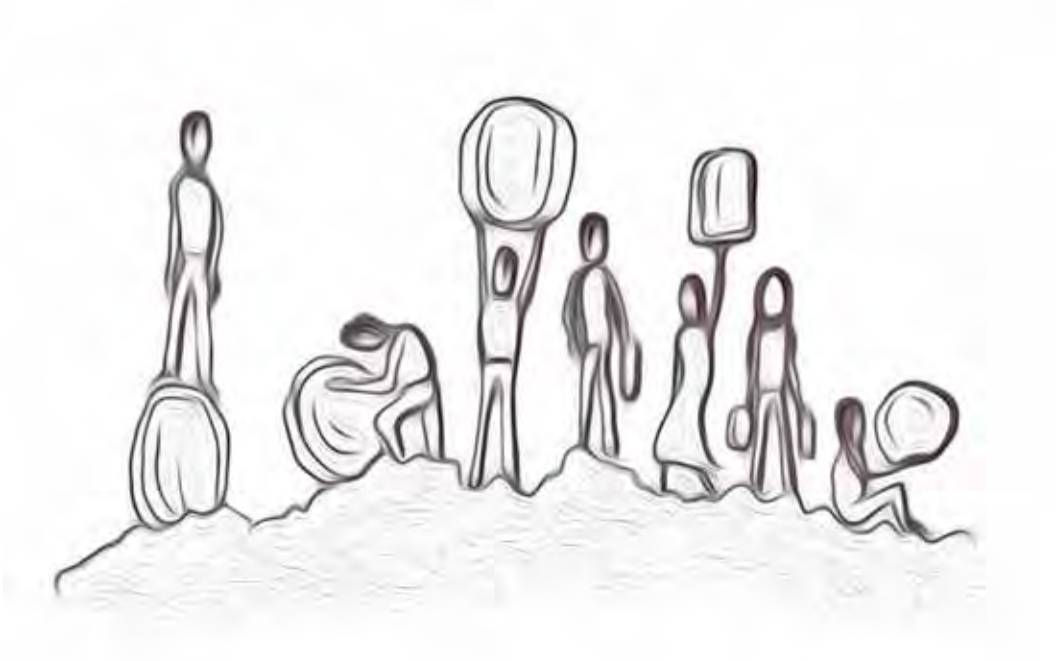
Anthropogenic direct driver	Example subcategories of direct driver	Specific threats to soil functions linked to the driver
Croplands and agroforestry management	Change in extent of croplands and agroforestry systems (including drainage of wetlands, soil management, harvesting and fallow cycles, agricultural inputs, and irrigation)	Soil erosion, change in carbon stocks, soil compaction, change in soil nutrient content, change in runoff and infiltration regimes of water, nutrients and agrochemicals, soil and water salinization, sedimentation, water contamination, atmospheric pollution and deposition, and acidification
Grazing land management	Change in extent of grazing lands and stocking rates	Soil erosion, soil compaction, change in carbon stocks, change in soil nutrient content, change in runoff and infiltration regimes of water, and nutrients and agrochemicals
Forest and tree plantation management	Change in extent of managed and planted forests, harvesting intensity and silvicultural techniques	Soil erosion, change in carbon stocks, soil compaction, change in soil nutrient content, change in runoff and infiltration regimes of water, nutrients and agrochemicals, sedimentation, and water contamination
Extractive industry development	Mine type, extraction and refining techniques, pollutant discharge and spoil disposal, and reclamation	Soil pollution, soil compaction, water contamination, and altered runoff regimes
Urbanization and infrastructure and industrial development	Urban growth, land clearance, roads and railways, and other infrastructure development	Soil pollution, soil compaction, water contamination, altered runoff regimes, and soil sealing
Conflicts	Soil disruption by munitions, mining of land and release of contaminants	Soil pollution, soil compaction, altered runoff regimes, and destruction of conservation structures and irrigation
Anthropogenic indirect driver	**Example subcategories of indirect driver**	**Management and land-use decisions linked to indirect driver**
Demographic	Population growth rate, and migration and population mobility (including to urban centres)	Land conversion, increased use of agrochemicals, increases in irrigation area, and increases in urbanization and soil sealing
Economic	Demand and consumption, poverty, commercialization and trade, urbanization, and industrialization	Land conversion, increased use of agrochemicals, increases in irrigation area, and increases in urbanization and soil sealing
Science, knowledge and technology	Education, Indigenous Peoples' knowledge systems, taboos, access to technology, innovation, and communication and outreach	Adoption of sustainable soil management (SSM) practices and adoption of new technology
Institutions and governance	Public policy (regulatory and incentive based), certification, international agreements and conventions, competencies of formal institutions, and informal institutions (social capital)	Recognition of role of soil in climate change mitigation, development of policies to encourage adoption of SSM, and agreements to limit the spread of soil pollution
Cultural	Worldviews, values, religion, consumer behaviour, and diet	Recognition of Indigenous Peoples' knowledge systems, concerns about agrochemical use, and recognition of the role of soil in climate change mitigation

◊ *Source:* **Montanarella, L., Scholes, R. & Brainich, A.** 2018. *The IPBES assessment report on land degradation and restoration.* Bonn, Germany, Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES).

Indirect drivers operate by altering the level or rate of change of one or more direct drivers. They are the underlying causes of environmental change and usually operate externally to the site in question (such as population change, social change, access rights to land, economic and agricultural policies, and international agreements). Indirect drivers of soil change are broader societal trends and measures that lead to changes in direct drivers. Population change is the principal indirect driver examined in this chapter. The other indirect drivers are examined in the context of facilitating adoption of SSM in Chapter 6 of this report.

Climate change is both a direct and indirect driver of soil change. For example, changes in the temperature regime of a soil with permafrost can cause thawing and subsidence of the soil surface (i.e. a direct driver of soil change). Changes to the temperature or precipitation regimes at the regional scale can lead to shifts in the extent of croplands (i.e. an indirect driver of soil change). The impact of climate change is examined throughout this report and is not included in this chapter.

An important addition to the 2025 report is the greater recognition of the importance of conflict as a driver of soil change, with conflict acting as both a direct and an indirect driver of change. Direct impacts include disruption of the soil through explosion of munitions, or compaction of the surface through the off-road movement of military vehicles. Indirect impacts can occur when disruptions to fertilizer supply affect nutrient balances in cropping systems, potentially leading to nutrient mining from the soil. These effects are examined in Section 4.4 of this chapter.



4.2 Indirect drivers

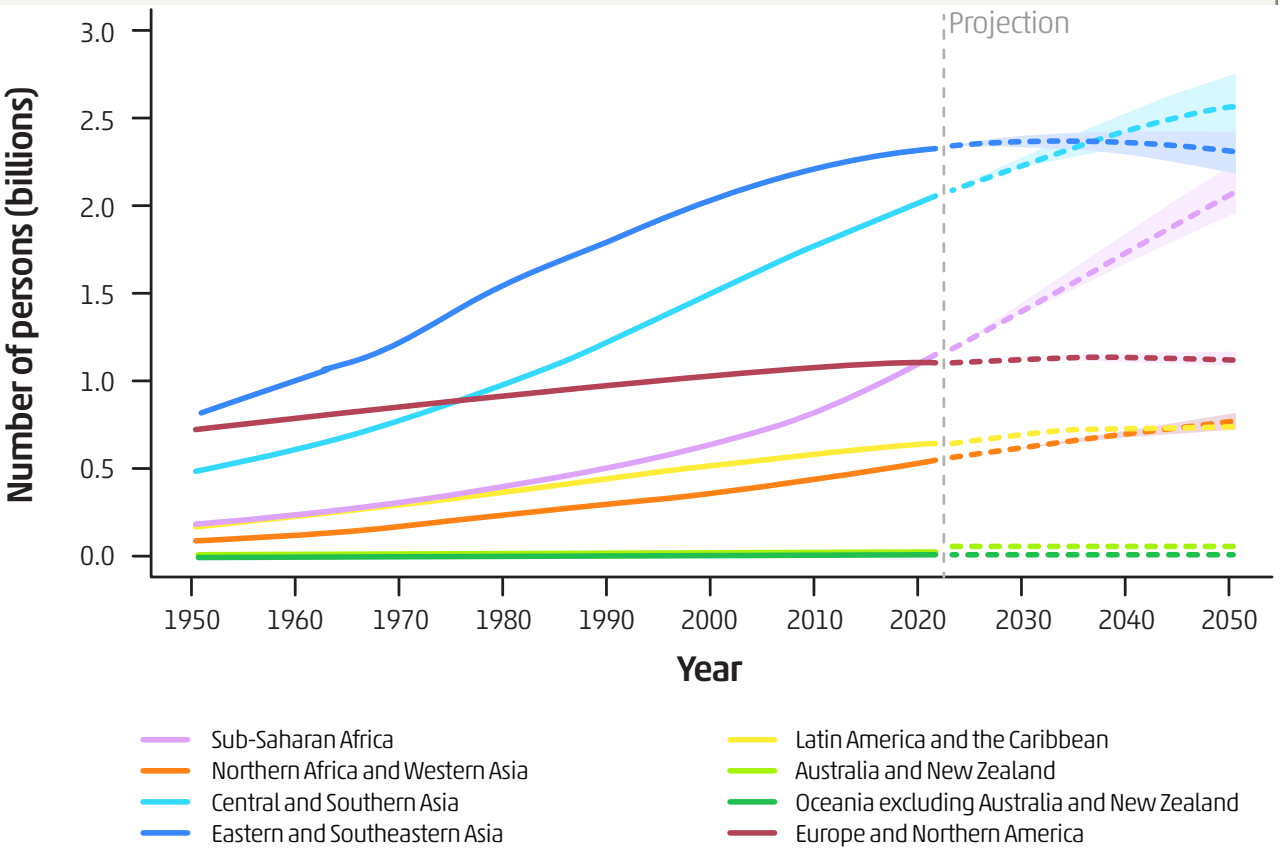
4.2.1 Population change

Population change is the major indirect driver of changes in the threats to soil functions. Growing populations require more food, wood products, non-renewable sources such as petroleum and minerals, and land for housing These requirements in turn lead to changes in the distribution of land uses and to the management practices used within specific land uses.

While long-term projections of population growth are uncertain, the medium-term projections (until 2050) are relatively certain insofar as the size and age structure of the population have been largely set by demographic processes that have already taken place (UN DESA, 2022).

The global population continues to increase, although at a lower rate than for most of the twentieth century (UN DESA, 2023, 2024) (Figure 4.1). The global population was 5.7 billion people in 1995, 7.3 billion in 2015 and 8.1 billion in 2024. It is projected to rise to 9.8 billion by 2054 and reach a peak of 10.2 billion in 2084 (UN DESA, 2024).

Figure 4.1. Population for 1950–2022 and projections with 95 percent confidence intervals for 2022–2050 by region



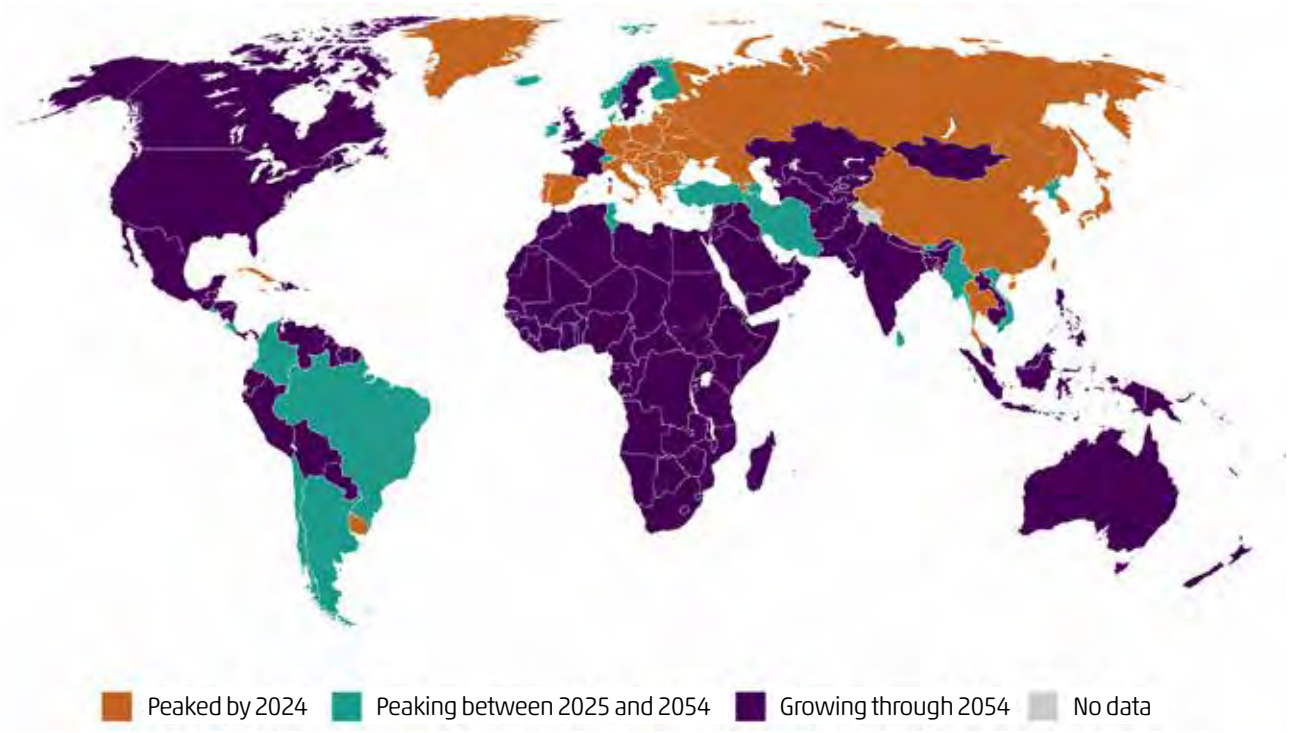
◊ *Source:* **UN DESA**. 2023. World Population Prospects 2022: Summary of Results. New York, USA. https://www.un.org/development/desa/pd/sites/www.un.org.development.desa.pd/files/wpp2022_summary_of_results.pdf

The pattern of population growth across regions is uneven. Population growth reflects the balance of births and deaths as well as migration between countries and regions. Population growth since 2015 has been low or stable in the Southwest Pacific (Oceania, Australia and New Zealand), Europe and North America, Latin America and the Caribbean, and the Near East and North Africa (NENA) (Western Asia in Figure 4.2) (UN DESA, 2023). The rate of population increase has been higher in

sub-Saharan Africa, Central Asia, and South, East and Southeast Asia. However, the projections post-2022 are very different, with the populations of both sub-Saharan Africa and Central and Southern Asia being predicted to continue to increase through to 2050, whereas in Eastern and southeastern Asia, populations are expected to stabilize and then begin to decline before 2050. The decreases in Eastern Asia are largely driven by China, whose population is predicted to decrease from 1 420 billion in 2024 to 1 221 billion in 2054 (UN DESA, 2024). However, the potential environmental benefits (including soil) of a decline in population may be largely offset by increased individual wealth and, therefore, increased consumption per person.

Subsequent projections of the timing of observed or projected peak population by country (UN DESA, 2024) show different patterns within the regions (Figure 4.2). The population of every country in sub-Saharan Africa is projected to keep growing through to 2054. Growth also continues in India and Pakistan in Southern Asia and in most countries of Central Asia. Continued growth but at lower rates are also projected for North America, the Southwest Pacific and some countries in South America. The population of most countries in Europe and Eastern Asia had already peaked by 2024.

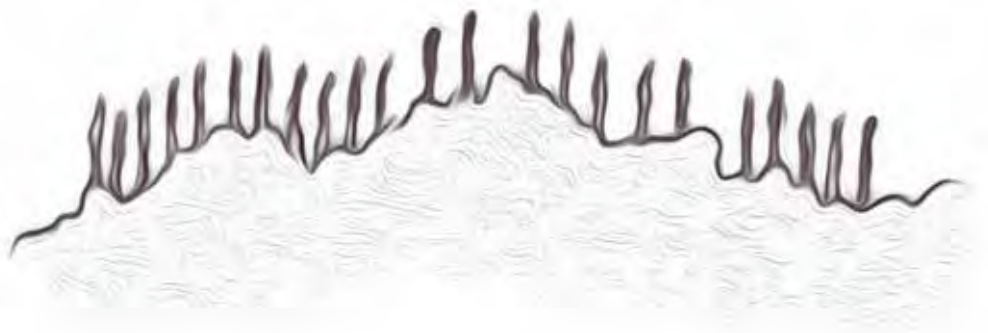
Figure 4.2. Countries grouped by timing of observed or projected peak population



Refer to the disclaimer on page ii for the names and boundaries used in this map. Dotted line represents approximately the Line of Control in Jammu and Kashmir agreed upon by India and Pakistan. The final status of Jammu and Kashmir has not yet been agreed upon by the parties.

Final boundary between the Republic of Sudan and the Republic of South Sudan has not yet been determined. Final status of the Abyei area is not yet determined.

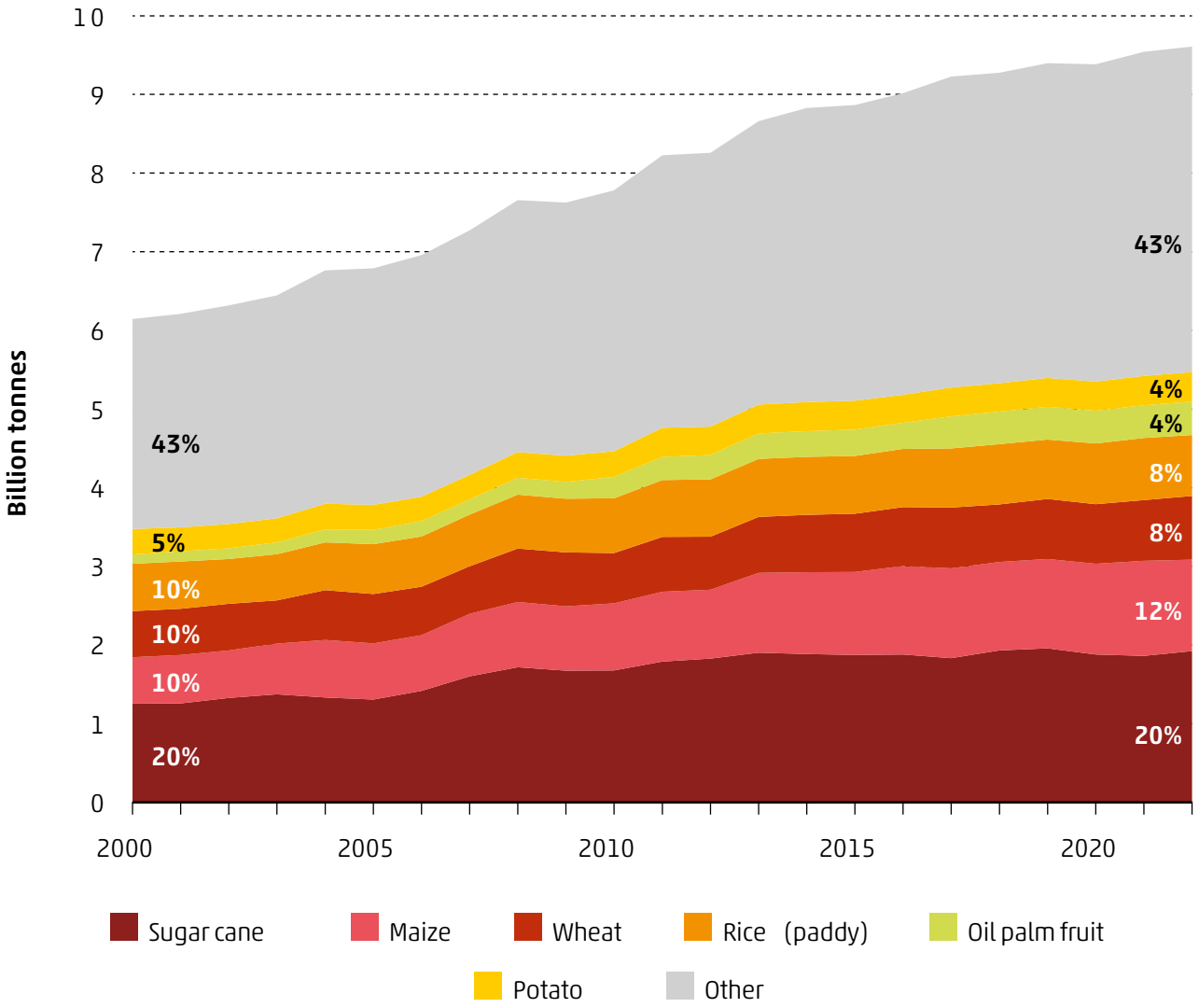
◊ *Source:* **UN DESA**. 2024. *World Population Prospects 2024: Summary of Results*. New York, USA. https://desapublications.un.org/publications/world-population-prospects-2024-summary-results



4.2.2 Demand for food and wood

The addition of 0.8 billion people between 2015 and 2024 has substantially increased the demand for food, wood and other commodities. The increased demand for food is evident in the increased world production of primary crops (Figure 4.3). Although the rate of increase slowed during the 2015–2022 period compared with the 2000–2015 period, total crop production continued to increase, reaching a total of 9.6 billion tonnes in 2022 (FAO, 2024a).

Figure 4.3. World production of primary crops in the main commodity groups



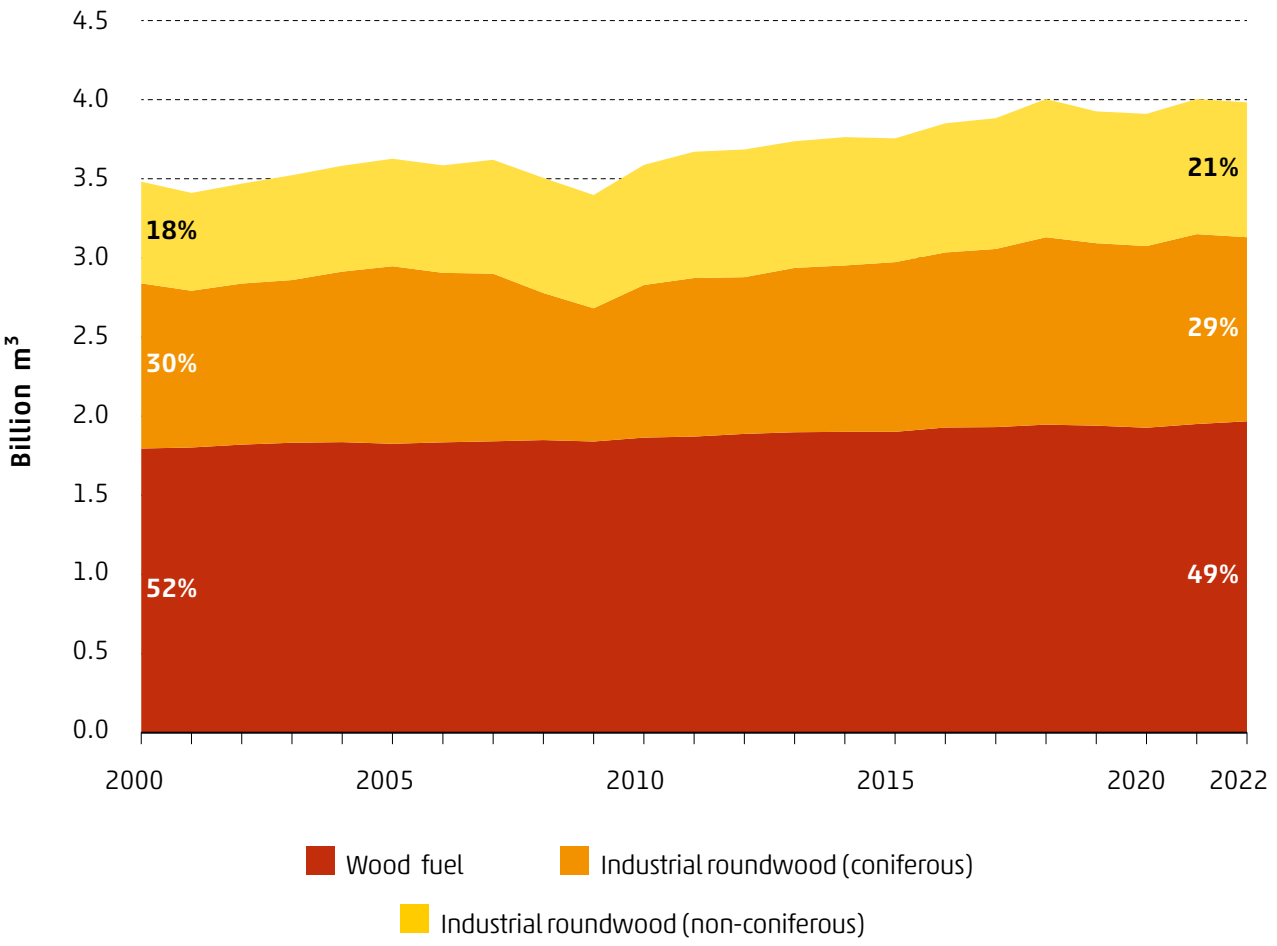
Note: Percentages indicate the shares in the total and may not tally due to rounding.

Source: **FAO.** 2024. *World Food and Agriculture – Statistical Yearbook 2024.* Rome. https://doi.org/10.4060/cd2971en

The greatest increase in any of the main commodities has been in vegetable oils, which rose from 92 million tonnes in 2000 to 215 million tonnes in 2021 (FAO, 2024a), with palm oil registering the largest increase in this time. In 2021, palm oil production has been the most heavily concentrated of any commodity, with 62 percent of global palm oil being produced in Indonesia and 22 percent in Malaysia (FAO, 2024a).

Similar trends to those observed in food production have occurred in the harvesting of forest products for fuel and industrial wood products, showing increases from 2015 to 2022 (Figure 4.4). The majority of wood products are used for fuel, primarily in the Global South (FAO, 2024a). In all, 1.97 billion m^3 of roundwood was harvested for fuel in 2022, with 0.73 billion m^3 in Africa and 0.69 billion m^3 in Asia (FAO, 2024a).

Figure 4.4. World production of roundwood by type



Note: Percentages indicate the shares in the total and may not tally due to rounding.

Source: **FAO.** 2024. *World Food and Agriculture – Statistical Yearbook 2024.* Rome. https://doi.org/10.4060/cd2971en

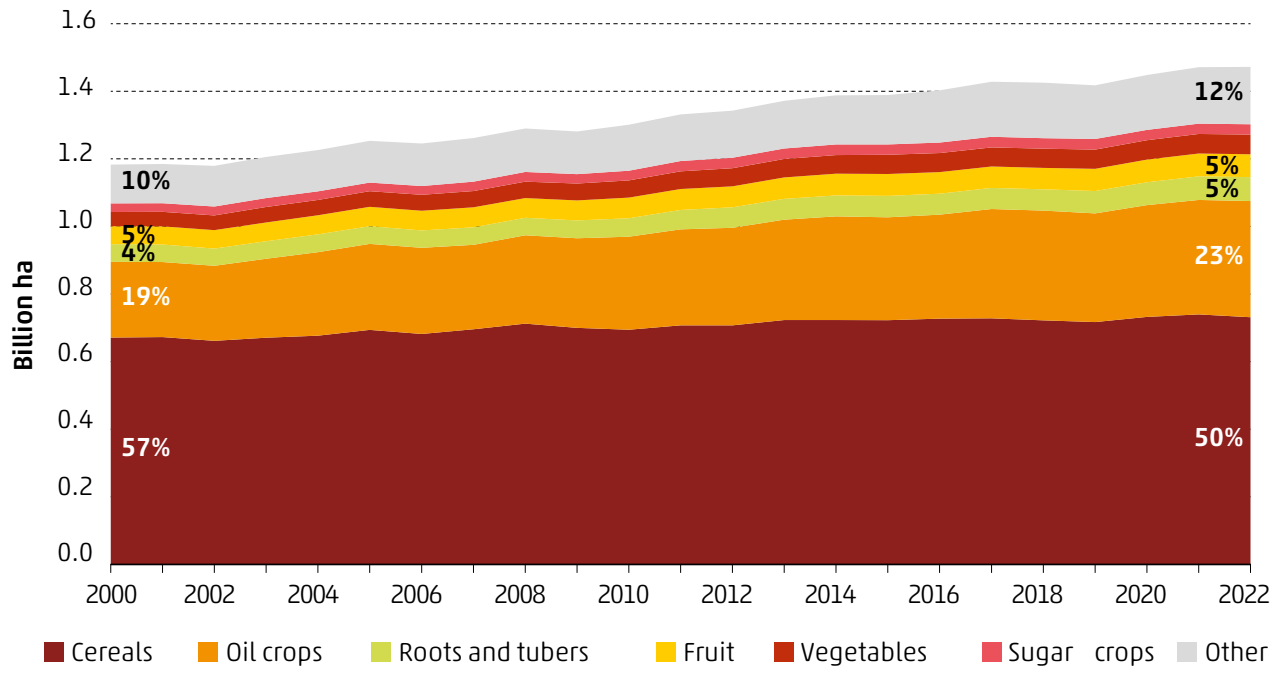


4.3 Direct drivers

4.3.1 Land-use change

The increased demand for food and related increases in demand for fuel and fibre has led to a continuous increase in the global harvested area of main crops (Figure 4.5). The total harvested area increased by 24 percent between 2000 and 2022 to 1.47 billion ha, with an increase of 80 million ha between 2015 and 2022 (FAO, 2025). Cereal crops account for more than half the harvested area, but vegetable oils have substantially increased their share, with an absolute increase of 120 million ha since 2000.

Figure 4.5. Harvested area of world primary crops by commodity group



Note: Percentages indicate the shares in the total and may not tally due to rounding.

Source: **FAO.** 2024. *World Food and Agriculture – Statistical Yearbook 2024.* Rome. https://doi.org/10.4060/cd2971en

The increase in harvested area and changing dietary preferences cause more land to be brought into agricultural production. Generally, the conversion of forestland to cropland will increase threats to soil functions:

- Forestland has substantially lower rates of soil erosion in all its forms than cropland (Borrelli *et al.*, 2017);
- Soil organic carbon (SOC) loss after conversion of forestland to cropland is the highest of any land-use change (Beillouin *et al.*, 2023);
- Deforestation causes shifts in the community composition of microorganisms (Qu *et al.*, 2024) and the disruption of natural nutrient cycles (Veldkamp *et al.*, 2020);
- Machinery used to clear the land and till the soil can cause physical deterioration and increase runoff (Nazari *et al.*, 2021)
- Alterations to the hydrological cycle may lead to increased salinization (Herbert *et al.*, 2015).

- The forced displacement and eviction of Indigenous Peoples from their ancestral lands breaks the continuity of their food generation schemes and territorial management practices and fallow cycles. When geographic space is restricted, these cycles are shortened, directly disrupting natural nutrient replenishment, reducing fertility, and accelerating severe soil degradation across key biomes and ecosystems hold by Indigenous Peoples (FAO and Alliance of Bioversity International and CIAT, 2021; Nelson *et al.*, 2024).

FAO (2024d) calculated the change in forestland over time as a net rate of change in forest area, which is the difference between forest expansion and deforestation, estimating the net change at −4.7 million ha per year in the 2010–2020 period. This was significantly lower than in the two previous decades (−7.8 million ha per year in the 1990–2000 period and −5.2 million ha per year in the 2000–2010 period).

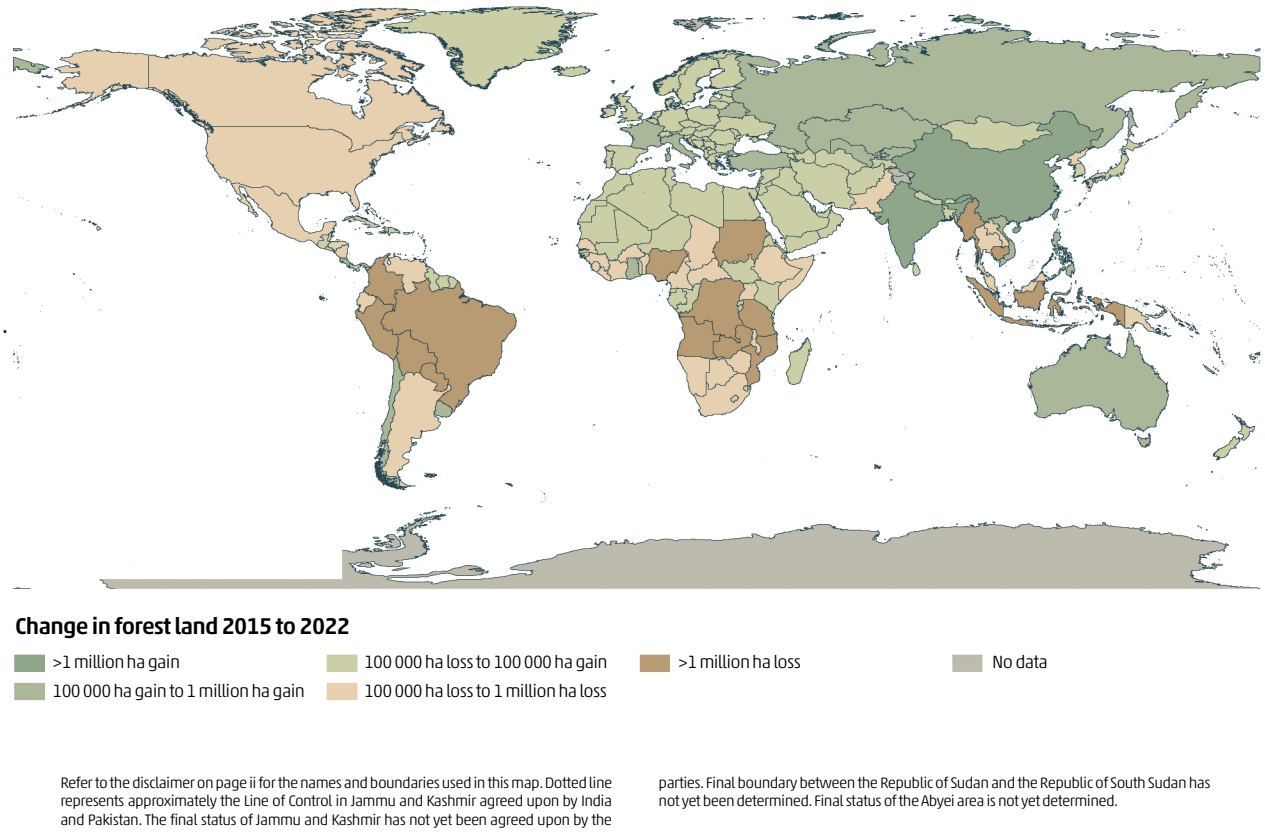
On a regional basis, the loss of forestland between 2015 and 2022 is greatest in sub-Saharan Africa, NENA and Latin America and the Caribbean (Table 4.2 and Figure 4.6). The loss in NENA is almost entirely due to deforestation in the Sudan, where the forest area decreased by 1 195 000 ha between 2015 and 2022 (FAO, 2024b). Of the other countries that experienced deforestation greater than 1 million ha between 2015 and 2022, five are in Latin America and the Caribbean, six are in sub-Saharan Africa and three are in Asia (Figure 4.6). FAO (2024d) indicated that deforestation had slowed in some countries since 2021, with decreases in the rate of deforestation of 8.4 percent in Indonesia over the 2021–2022 period and by 50 percent in Brazil's Legal Amazon in 2023.

Table 4.2. Regional statistics for the area of cropland, permanent pastures and meadows and forestland in 2015 and 2022 and the percentage change from 2015 to 2022

Region	Year	Cropland	Permanent pastures and meadows	Forestland
Asia	2015	476 603 000	570 997 000	563 110 000
	2022	477 066 000	570 139 000	570 675 000
	Change (2015–2022) (%)	0.09	−0.15	1.34
Europe and Central Asia	2015	354 983 000	446 096 000	1 054 663 000
	2022	354 406 000	447 087 000	1 058 206 000
	Change (2015–2022) (%)	−0.16	0.22	0.33
Latin America and the Caribbean	2015	173 938 000	491 357 000	953 824 000
	2022	176 067 000	467 276 000	935 291 000
	Change (2015–2022) (%)	1.22	−4.90	−1.94
Near East and North Africa	2015	82 300 000	405 326 000	42 700 000
	2022	84 591 000	405 196 000	41 607 000
	Change (2015–2022) (%)	1.92	−0.03	−2.56
North America	2015	197 806 000	277 194 000	657 211 000
	2022	193 257 000	276 233 000	656 649 000
	Change (2015–2022) (%)	−2.29	−0.35	−0.09
Sub-Saharan Africa	2015	237 644 000	699 767 000	628 007 000
	2022	254 687 000	700 113 000	601 284 000
	Change (2015–2022) (%)	7.12	0.05	−4.25
Southwest Pacific	2015	33 671 000	327 529 000	184 435 000
	2022	34 049 000	341 644 000	185 227 000
	Change (2015–2022) (%)	1.12	4.30	0.43

Source: **FAO**. 2024. *FAOSTAT: Land Use*. [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

Figure 4.6. Country map of change in forestland area (ha) between 2015 and 2022

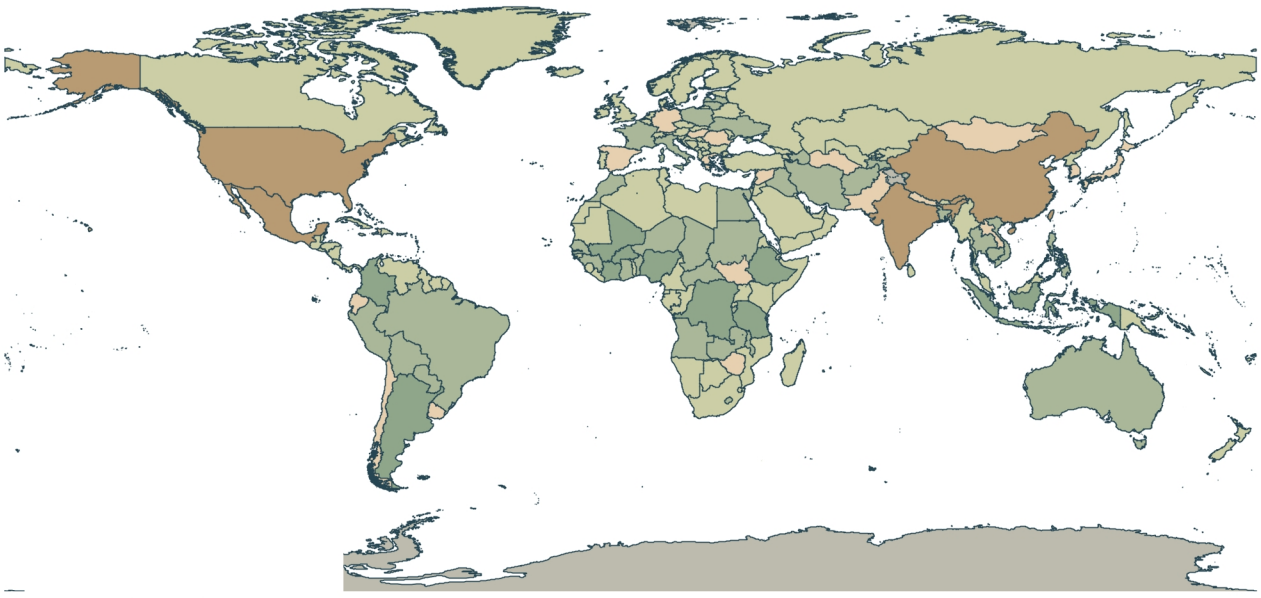


Source: **FAO.** 2024. *FAOSTAT: Land Use.* [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

It cannot be assumed that the loss of forestland is directly linked with the conversion to cropland. Potapov *et al*. (2021) estimated changes in cropland (equivalent to FAO's arable land definition) and the transitions between different uses using remote sensing data. They reported that 49 percent of new cropland land between 2003 and 2019 replaced natural woody and herbaceous vegetation. The highest conversions from natural vegetation (excluding conversions to irrigation) were in Africa (79 percent of all gross arable land gain area), Southeast Asia (61 percent) and South America (39 percent). Fifty-one percent of cropland expansion was due to pasture conversion and recultivation of abandoned arable land. Nearly all cropland expansion in Australia, Europe, New Zealand, and northern Asia was reported to be within pastures and long fallows (i.e. with no crop cultivation for more than four years). Seventy-five percent of cropland expansion in North America and 61 percent in South America was through the conversion of pastures and multiyear fallows.

The net changes in cropland and permanent pastures and meadows in the 2017–2022 period (FAO, 2024b) (Table 4.2) are consistent with the earlier estimates by Potapov *et al.* (2021). The expansion of cropland has been greatest in sub-Saharan Africa (Figure 4.7), which has also experienced the greatest loss of forestland. Of the ten countries that gained over 1 million ha of cropland between 2015 and 2021, seven are in sub-Saharan Africa, two are in Latin America and the Caribbean and one is in Southeast Asia. Decreases in cropland area are greatest in China, India, Mexico, and the United States (Figure 4.7).

Figure 4.7. Country map of change in cropland area (ha) between 2015 and 2022



Refer to the disclaimer on page ii for the names and boundaries used in this map. Dotted line represents approximately the Line of Control in Jammu and Kashmir agreed upon by India and Pakistan. The final status of Jammu and Kashmir has not yet been agreed upon by the parties. Final boundary between the Republic of Sudan and the Republic of South Sudan has not yet been determined. Final status of the Abyei area is not yet determined.

Source: **FAO.** 2024. *FAOSTAT: Land Use.* [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

Changes in permanent pastures and meadows in the 2017–2022 period are more concentrated than the changes in cropland (Figure 4.8). Australia (with a gain of 15 147 000 ha) and Turkmenistan (with a gain of 6 109 000 ha) account for 86 percent of all reported gains (FAO, 2024b). Brazil has the highest loss of permanent pastures and meadows (12 952 000 ha), with substantial losses also in Argentina, Chile and Colombia, and in Latin America and the Caribbean (FAO, 2024b).

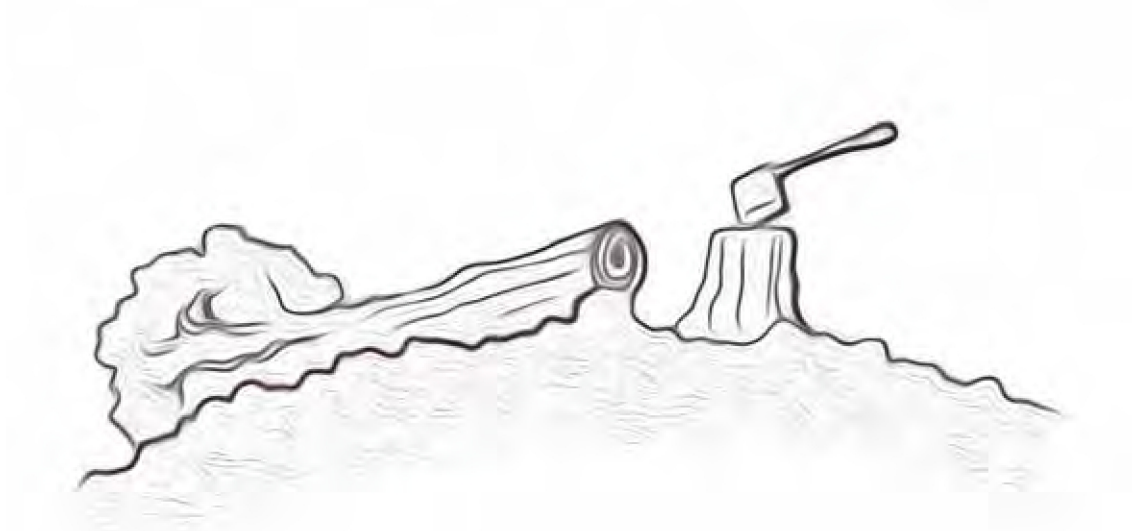
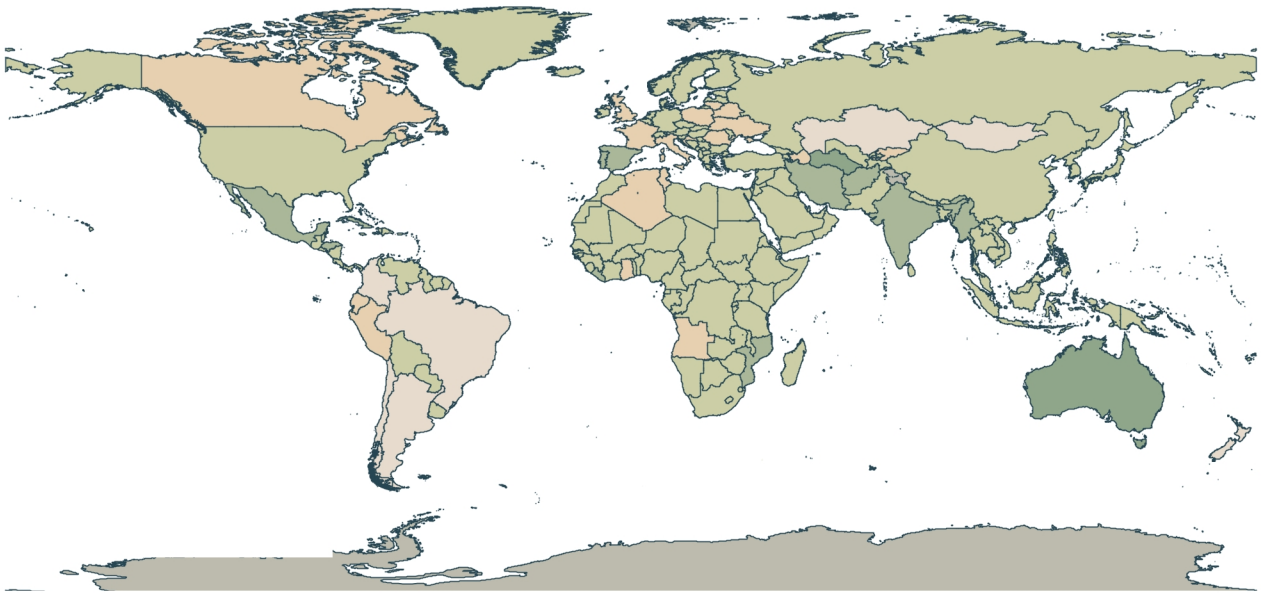


Figure 4.8. Country map of change in permanent meadows and pastures (ha) between 2015 and 2022



◊ *Source:* **FAO (Food and Agriculture Organization of the United Nations)**. 2024. *FAOSTAT: Land Use*. [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

Reforestation can reverse many of the negative effects of deforestation but substantial improvement in soil properties can take decades to occur (Veldkamp *et al.*, 2020). Reforestation has been by far the greatest in China, with 13 443 000 ha of net forestland gain between 2015 and 2022 (Figure 4.6). India has the second-highest area of net forestland gain (1 865 000 ha). The majority of countries with net forestland gain in this period are in Europe and Central Asia (Figure 4.6).

The conversion of wetlands to other land uses is a concern due to high levels of SOC storage in wetlands compared to other terrestrial landscapes and the important role of wetlands in surface water storage and recharge of groundwater (FAO and ITPS, 2021).

Conversions to cropland have been one of the main drivers of the loss of wetlands through time. Fluet-Chouinard *et al.* (2023) estimated that the global area of natural wetlands declined by 3.4 million km^2 since 1700. This estimate corresponds to a loss of 21 percent of the 15.8 million km^2 of wetlands estimated from the same time period. Wetland drainage for upland croplands was the most common cause of loss of natural wetlands (2 million km^2 or 61.7 percent of total loss), followed by conversion to flooded rice (18.2 percent), urban areas (8 percent), forestry (4.7 percent), non-rice wetland cultivation (4.3 percent), pasture (2 percent) and peat extraction (0.9 percent).

The highest annual global rates of wetland area loss (0.46 million km^2 or 2.2 percent) occurred in the 1950s when government programmes subsidized drainage for agriculture and forestry across North America, Europe and China (Fluet-Chouinard *et al.*, 2023). Between 2010 and 2020 there was a notable loss of wetlands in Bangladesh, Brunei, Indonesia, Malaysia, and Thailand with smaller areas of loss in several countries in sub-Saharan Africa and in the northeast of the United States.

Extraction of peat from peatlands has relevance for the management of SOC, as peatlands are major

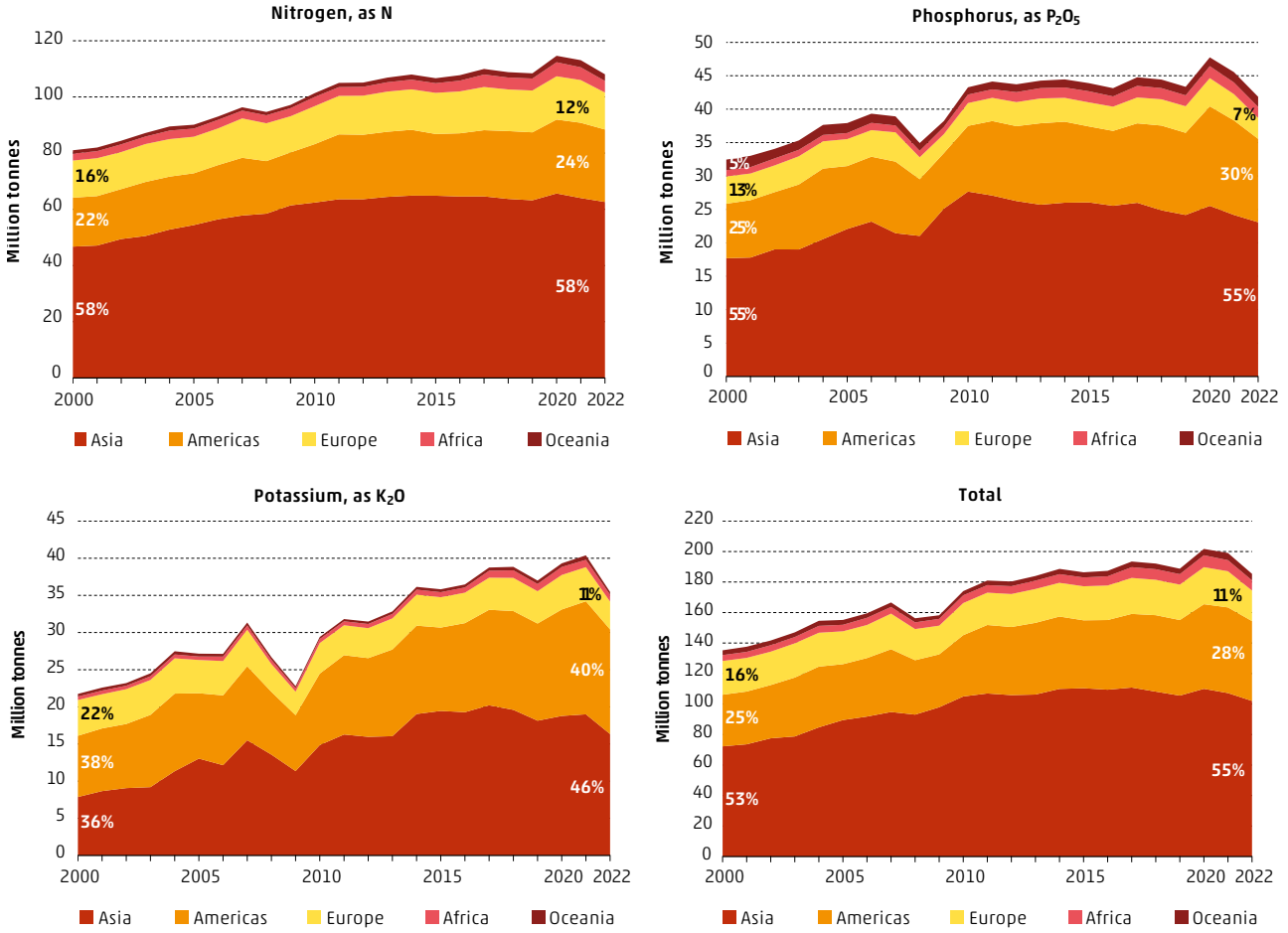
reservoirs of carbon. Fluet-Chouinard *et al.* (2022) estimated that approximately 0.51 million km^2 of peatlands had been lost since 1700 through peat extraction. Räsänen *et al.* (2023) cited statistics that, outside of tropical areas, around 10 percent of the loss of peatlands in their natural state was due to peat extraction, compared to losses due to agriculture (50 percent), forestry (30 percent) and urbanization (5 percent).

Excavated peat is used both an energy source and in horticulture (USGS, 2023). The top four countries for peat extraction in 2022 were Finland (4 900 tonnes), Germany (2 350 tonnes), Sweden (2 100 tonnes) and Latvia (1 800 tonnes). Belarus, Canada, Poland, and the Russian Federation all excavated between 900 and 1 800 tonnes of peat in 2022. Policy interventions in several countries have led to decreased peat extraction. For example, Ireland has gone from being the second-highest extractor of peat up to 2017 to having no peat extraction in 2023 (Räsänen *et al.*, 2023; USGS, 2023).

4.3.2 Changes in agrochemical use

The increasing demand for food and other commodities has also led to an increase in inorganic fertilizer use. The use of inorganic fertilizers increased from 2015 to 2020 but all three of the main types of inorganic fertilizers have decreased between 2020 and 2022 (Figure 4.9). FAO (2024b) attributed the decrease to higher natural gas prices and conflict-related supply chain disruptions.

Figure 4.9. Inorganic fertilizer use by region



Note: Percentages indicate the shares in the total and may not tally due to rounding.

Source: **FAO**. 2024. *World Food and Agriculture* – Statistical Yearbook 2024. Rome. https://doi.org/10.4060/cd2971en

The most effective method to assess nutrient use is to calculate a comprehensive nutrient balance

(see Section 3.5), which allows areas of nutrient mining and nutrient excess to be identified. For both N and P, inorganic fertilizer is just one source of inputs. For N, the other inputs to be considered are atmospheric deposition, biological N fixation and N inputs from manure. For P, manure is the other input. Excess P inputs through time can accumulate in the soil and this residual soil P is available for future crops (Zou, Zhang and Davidson, 2022). For P, the soil itself can be a long-term source or, in soils with high P retention, a sink. For both N and P, the nutrients exported from the field in harvested products are the output.

The regional summaries of N budgets (Table 4.3) show relatively high positive N budgets (i.e. higher N inputs than outputs) for five regions but low positive budgets in sub-Saharan Africa and negative budgets in the Southwest Pacific region. In Chapter 3, regions or countries with annual N balances greater than 80 kg/ha per year and NUEs less than 50 are classified as having excessive N inputs. As a region, Europe and Eurasia have average N levels indicative of N excess but both Asia and Latin America and the Caribbean have over ten countries in the N excess group (Figure 4.10). Nitrogen mining is defined in Chapter 3 as a negative N balance, with seven countries in sub-Saharan Africa and ten Pacific Island countries in the Southwest Pacific showing considerable evidence of nutrient mining.

Table 4.3. Average global nitrogen use statistics for 2017 to 2021

Region and number of countries	Average nitrogen (N) budget (2017–2021) (kg/ha)	Average inorganic fertilizer use (2017–2022) (kg/ha)	Average NUE (2017–2022) (kg/ha)	Number of countries with N excess	Number of countries with N mining
Asia (25 countries)[1]	90.7	72.9	45.6	10	0
Europe and Central Asia (51 countries)[2]	85.5	80.7	45.0	18	1
Latin American and the Caribbean (34 countries)	67.5	54.8	49.1	11	0
Near East and North Africa (18 countries)	78.7	66.9	48.9	4	1
North America (2 countries)	35.6	61.4	66.8	0	0
Sub-Saharan Africa (48 countries)	17.9	7.7	68.4	1	7
Southwest Pacific (16 countries)	–22.8	16.1	181.9	1	10

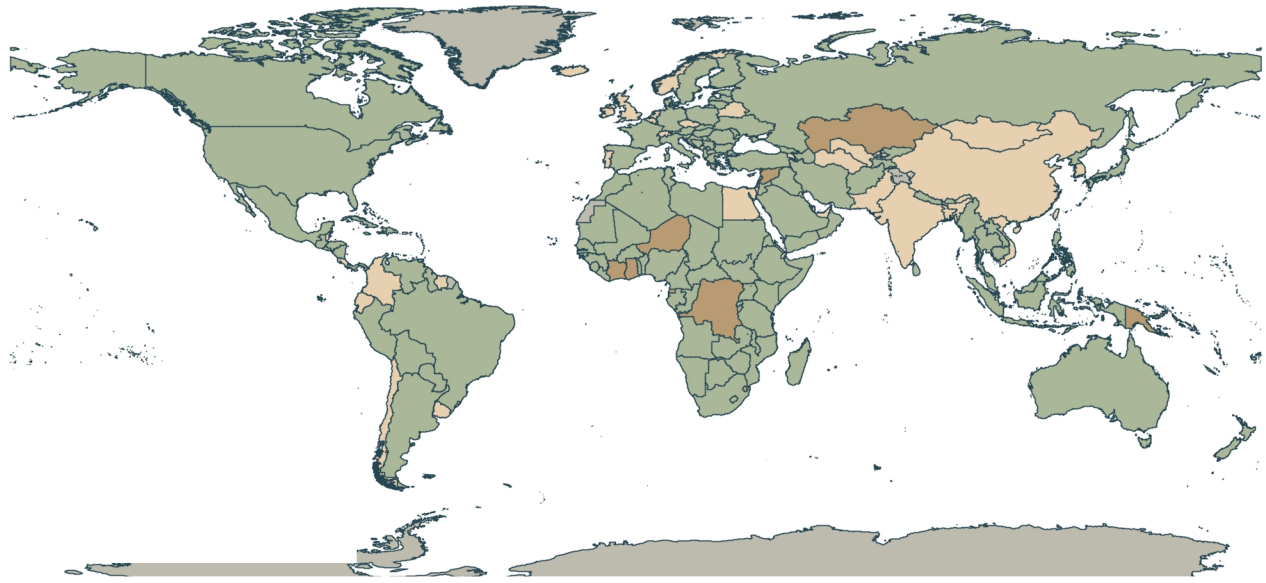
Note: NUE = nitrogen use efficiency.

[1] N budgets and NUE for Asia excludes Singapore (N budget = 1 413 kg/ha).

[2] N budgets and NUE for Europe and Central Asia excludes the Faroe Islands (N budget = 16 703 kg/ha).

Source: **FAO**. 2024. *FAOSTAT: Fertilizers by Nutrient*. [Accessed on 9 January 2024]. https://www.fao.org/faostat/en/#data/RFN. Licence: CC-BY-4.0.

Figure 4.10. Country map of average nitrogen nutrient budget for the 2017 to 2021 period



◊ *Source:* **FAO.** 2024. *FAOSTAT: Fertilizers by Nutrient.* [Accessed on 9 January 2024]. https://www.fao.org/faostat/en/#data/RFN. Licence: CC-BY-4.0.

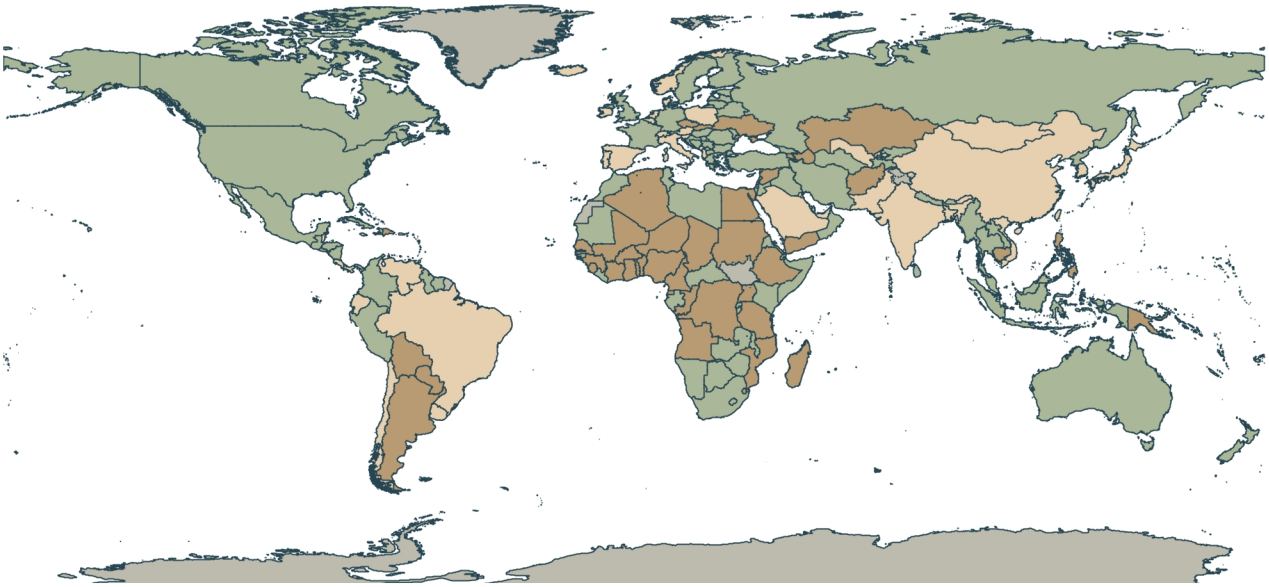
For P budgets, as outlined in Chapter 3, two criteria are used to identify nutrient excess: the average annual P budgets in excess of 6.9 kg of P/ha, and long-term accumulation of P reserves (1961 to 2019) in the soil (Zou, Zhang and Davidson, 2022). The authors defined P mining as a negative P budget in the 2017–2021 period. Countries with P excess are common in Asia, Europe and Central Asia, and Latin America and the Caribbean (Table 4.4 and Figure 4.11) while the majority of countries in sub-Saharan Africa have experienced nutrient mining.

Table 4.4. Average global phosphorus use statistics for 2017 to 2021

Region and number of countries	Average P budget (2017–2021) (kg/ha)	Number of countries with P excess	Number of countries with P mining
Asia (23 countries)	30.0	10	3
Europe and Central Asia (50 countries)	9.6	19	6
Latin American and the Caribbean (34 countries)	10.5	15	4
Near East and North Africa (18 countries)	10.7	5	5
North America (2 countries)	0.7	0	0
Sub-Saharan Africa (47 countries)	0.3	1	25
Southwest Pacific (15 countries)	4.7	4	4

◊ *Source:* **FAO**. 2024. *FAOSTAT: Fertilizers by Nutrient.* [Accessed on 9 January 2024]. https://www.fao.org/faostat/en/#data/RFN. Licence: CC-BY-4.0.

Figure 4.11. Country map of average phosphorus nutrient budget for the 2017 to 2021 period



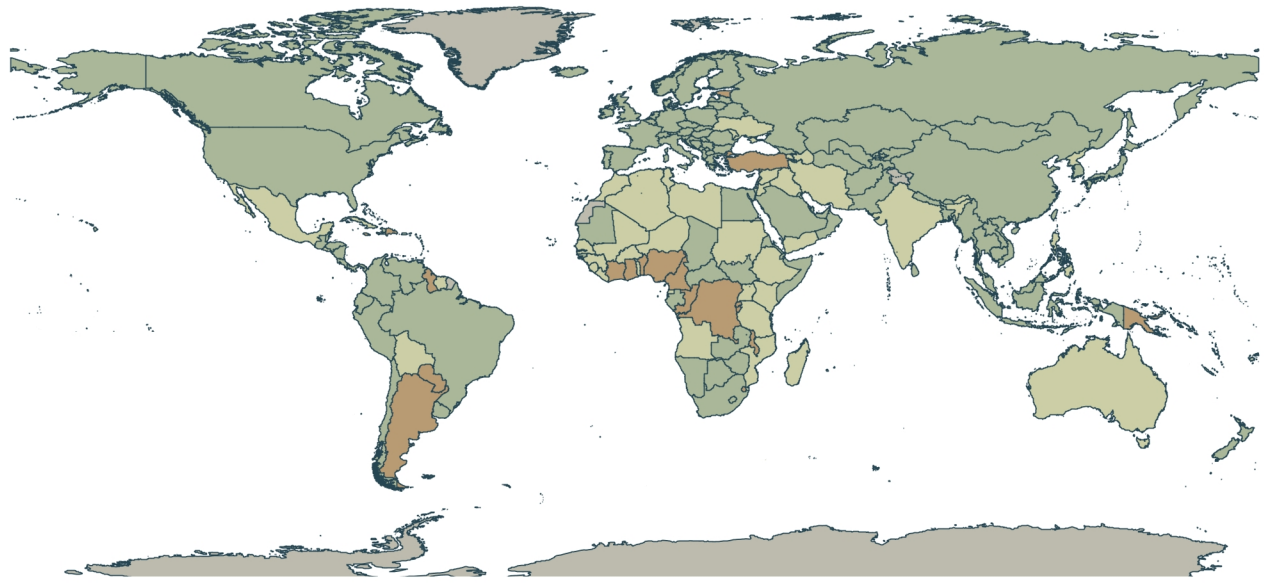
◊ *Sources:* **FAO**. 2024. *FAOSTAT: Fertilizers by Nutrient*. [Accessed on 9 January 2024]. https://www.fao.org/faostat/en/#data/RFN. Licence: CC-BY-4.0.

◊ **Zou, T., Zhang, X. & Davidson, E.A.** 2022. Global trends of cropland phosphorus use and sustainability challenges. *Nature*, 611(7934): 81–87. https://doi.org/10.1038/s41586-022-05220-z

Annual budgets for potassium (K), the third nutrient required in large amounts, show few countries with negative balances indicative of nutrient mining (Figure 4.12). In addition, some counties have large K reservoirs in the soil that can meet crop requirements without K fertilizers. Excess K inputs are not linked with environmental issues.



Figure 4.12. Country map of average potassium nutrient budget for the 2017 to 2021 period



Source: **FAO**. 2024. *FAOSTAT: Fertilizers by Nutrient*. [Accessed on 9 January 2024]. https://www.fao.org/faostat/en/#data/RFN. Licence: CC-BY-4.0.

The available global maps of nutrient use omit small island nations, being too small to be represented. However, the Small Island Developing States (SIDS) are still a group of 39 states and include 18 Associate Members of UN regional commissions that face unique socioeconomic and environmental vulnerabilities (UN-OHRLLS, 2025).

The statistics on nutrient use across the 39 SIDS that have nutrient data available (FAO, 2024c) illustrate the range in challenges faced by those nations (Table 4.5). Across the 39 states, only three (Seychelles, Dominica and Mauritius) show inputs of N, P and K that are acceptable (i.e. not showing either mining or excess). Thirteen of the SIDS have levels of either N, P or both nutrients that are indicative of nutrient excess (Table 4.5). Singapore is an extreme case with very high values for excess N, P and K. The high levels are driven by inputs from manure, with nutrients from manure equalling 95 percent of N inputs and 97 percent of P inputs (FAO, 2024b). Excess P levels in Antigua and Barbuda, Barbados, the Cook Islands and Trinidad and Tobago are also associated with inputs from manure greater than 80 percent of total P inputs (FAO, 2024b).

Nutrient mining is also common in SIDS, especially in the island countries in the Southwest Pacific region (FAO, 2024c) (Table 4.5). Seventeen SIDS have indicators of mining of N including 13 of 15 SIDS in the Southwest Pacific region. Nine SIDS have indicators of mining of P and 17 of mining of K. Of the SIDS in the Southwest Pacific region, five (Kiribati, the Federated States of Micronesia, Papua New Guinea, Solomon Islands, and Vanuatu) have indicators on mining of N, P and K. For SIDS in the other regions, only Comoros in the sub-Saharan Africa region shows mining of all three nutrients.

Table 4.5. Small Island Developing States with average values (2017–2021) for nitrogen, phosphorus and potassium indicative of nutrient excess or mining

Status of nutrient balance	Small Island Developing States (SIDS)
Excess N and P	Bahamas, Barbados, Belize, New Caledonia, Puerto Rico, Singapore, and Trinidad and Tobago
Excess N	Suriname
Excess P	Antigua and Barbuda, Cook Islands, Fiji, Jamaica, Saint Lucia, and Tuvalu
Mining N, P and K	Comoros, Kiribati, Federated States of Micronesia, Papua New Guinea, Solomon Islands, and Vanuatu
Mining P and K	Dominican Republic
Mining N and K	Sao Tome and Principe, Samoa, and Niue
Mining N	Antigua and Barbuda, Cape Verde, Haiti, Nauru, Saint Kitts and Nevis, Timor-Leste, and Tonga
Mining P	Guinea-Bissau
Mining K	Cuba, Fiji, Grenada, Guyana, Jamaica, Saint Vincent and the Grenadines, and Suriname
No mining or excess NPK	Dominica, Mauritius and Seychelles

◊ *Source:* **FAO**. 2024. *FAOSTAT: Fertilizers by Nutrient*. [Accessed on 9 January 2024]. https://www.fao.org/faostat/en/#data/RFN. Licence: CC-BY-4.0.

The draft outcome document for the Fourth International Conference on SIDS (UN, 2024) stated that to achieve food security, the SIDS and the international community needed to promote sustainable agricultural practices and deploy agricultural technologies that were resilient to the adverse impacts of climate change. Any measures taken to achieve food security needed to also help with climate adaptation, mitigate emissions and promote the sustainable use of biodiversity. This report includes sections on the particular challenges faced by islands of the Caribbean (Section 10.3), sub-Saharan Africa (Section 15.5) and the Pacific Island countries (Section 13.5).

A driver of changes in nutrient use is the adoption of SSM practices to make more efficient use of added nutrients. These are examined in detail in Section 5.3 of this report. A major emphasis in many fertilizer-intensive countries of the Northern Hemisphere has been the adoption of a set of nutrient best management practices that are referred to as the 4Rs: the right source, right rate, right timing and right placement (Johnston and Bruulsema, 2014; Davidson *et al.,* 2016). The selection of the right source and rate is central to achieving an optimal nutrient balance, while the right timing and placement are essential to ensure that uptake of the applied nutrients is as high as possible.

Of the 4Rs, global data is only available for some of the nutrient sources. FAOSTAT (e.g. FAO, 2024b) provides national-level data on N, P and K inputs from manure, as well as N inputs from atmospheric deposition and BNF, which are used in the calculation of nutrient budgets presented previously. Research studies have also used crop models or remote sensing to spatially distribute these national-level statistics within countries (e.g. Kaltenegger and Winiwarter, 2020).

There are no widely available national-level statistics on the other three parts of the 4Rs approach. For example, the use of enhanced efficiency fertilizers (EEFs) is presented in Section 5.3 as an approach that can substantially reduce the harmful impacts of mismanagement of N fertilizers on the environment but there is no globally relevant information on their use.

4.3.3 Changes in agricultural and forestland management

Adoption of SSM practices reduces the threats to soil functions (FAO, 2017) and these reductions are discussed in detail in Chapter 5 of this report. An accurate assessment of the adoption of SSM would allow a quantitative appraisal of the reduction in the threats, but unfortunately only partial information is available on adoption at the global (or even regional) scale.

The regional assessments of the threats to soil functions for Latin America and the Caribbean, North America and the Southwest Pacific presented in this report highlight the importance of adoption of no-till and reduced tillage for reducing soil erosion and stabilizing SOC storage. Estimates of changes to tillage systems are mainly available for SSM techniques, although comprehensive national reporting is not available (Erb *et al.*, 2017).

Kassam, Friedrich, and Derpsch (2022) presented estimates for adoption of conservation agriculture, which included no-till and reduced tillage adoption as a key component. It was estimated that in 2018/2019, approximately 205.4 million ha were under conservation agriculture, corresponding to about 14.7 percent of the total global cropland (Table 4.6) (an increase of 98.9 million ha or 92.9 percent from 2008/2009). It was also estimated that the rate of adoption had increased through time, with the rate of increase being about 5 million ha per year during the period from 1990 to 2008/2009, but increasing to about 10 million ha per year from 2008/2009 to 2018/2019.

Table 4.6. Global spread of conservation agriculture cropland area (ha) in different regions for 2008/2009 and 2018/2019, and corresponding percentage change

Region	Conservation agriculture cropland area (2008/2009) (ha)	Conservation agriculture cropland area (2018/2019) (ha)	Change in conservation agriculture cropland area (2008/2009 to 2018/2019) (%)	Percentage of cropland area in conservation agriculture (2018/2019) (%)
South and Central America	49 564 000	82 996 000	67.5	68.7
North America	40 004 000	65 937 000	64.8	33.6
Australia and New Zealand	12 162 000	23 293 000	91.5	74.0
Russian Federation and Ukraine	100 000	6 900 000	6 800	4.5
Europe	1 560 000	5 602 000	259	5.2
Asia	2 630 000	17 529 000	567	3.6
Africa	485 000	3 143 000	548	1.1
Near East and North Africa[1]		370 000		
Total	106 505 000	205 400 000	92.9	14.7

Note: the regions do not correspond to GSP regions.

Source: **Kassam, A., Friedrich, T. & Derpsch, R.** 2022. Successful Experiences and Lessons from Conservation Agriculture Worldwide. *Agronomy*, 12(4): 769. https://doi.org/10.3390/agronomy12040769

Yield benefits of no-till and reduced tillage are greatest under rainfed management in dry climates where no-till matched yields (for maize and legumes) or exceeded yields (for wheat, oilseeds and cotton) from conventional tillage (Pittelkow *et al.*, 2015) (see Section 5.2.5). The large area of no-till and reduced tillage in drier areas of Australia, Canada, Argentina and the United States (Porwollik *et al.*, 2019; Kassam, Friedrich and Derpsch, 2022) shows the importance of yield benefit to the adoption of this practice (Table 4.7).

Table 4.7. Top 15 countries for conservation agriculture

Country	Conservation agriculture area (2008/2009) (ha)	Conservation agriculture area (2018/2019) (ha)	Change in conservation agriculture area (2008/2009 to 2018/2019) (%)
United States	26 500 000	44 049 000	66.2
Brazil	25 502 000	43 000 000	68.6
Argentina	19 719 000	32 907 000	66.9
Australia	12 000 000	22 927 000	91.1
Canada	13 481 000	21 739 000	61.3
China	1 330 000	9 000 000	576.7
Russia Federation	ND	6 000 000	–
India	ND	3 500 000	–
Paraguay	2 400 000	3 158 000	31.6
Kazakhstan	1 300 000	3 000 000	130.8
Bolivia (Plurinational State of)	706 000	1 858 000	163.2
South Africa	368 000	1 607 000	336.7
Uruguay	655 000	1 278 000	95.1
Spain	650 000	1 000 000	53.8
Ukraine	100 000	900 000	800.0

Note: ND = no data.

Source: **Kassam, A., Friedrich, T. & Derpsch, R.** 2022. Successful Experiences and Lessons from Conservation Agriculture Worldwide. Agronomy, 12(4): 769. https://doi.org/10.3390/agronomy12040769

Cover crops offer benefits similar to no-till and reduced tillage in reducing threats to soil functions. They complement the tillage practices as the yield benefits from cover crops are greater in non-water-limited locations (Blanco-Canqui *et al.*, 2022; Garba, Bell and Williams, 2022). Unfortunately, data on the area of adoption of cover crops is lacking (Feng *et al.* 2024). An estimated area of cover crops ranged from 4 percent of cropland in China and the United States to 9 percent in the European Union (Feng *et al.* 2024). Progress has been made on the use of satellite-based remote sensing to estimate cover crops (e.g. Do Nascimento Bendini *et al.*, 2024), but no national or regional estimates are available at this time.

The adoption of management systems that incorporate one or more of the specific SSM practices can also drive changes in the threats to soil functions, such as conservation agriculture which has been discussed previously. Organic agriculture, which involves crop and animal production systems without the use of synthetic fertilizer and pesticides, is also widely recognized. As of 2020, seventy-six countries had fully implemented standards for organic production, with 74.9 million ha reportedly under organic agriculture, representing 1.6 percent of total global agricultural land (Willer *et al.*, 2022). Australia had the largest area (35.7 million ha), followed by Argentina (4.5 million ha) and Uruguay (2.7 million ha). Regionally, the Southwest Pacific had the highest area (attributable to Australia's figures) followed by Europe (17.1 million ha), Latin America (9.9 million ha), Asia (6.1 million ha), North America (3.7 million ha) and Africa (2.1 million ha). Over two-thirds (51 million ha) of organic agricultural land were grassland or grazing areas and 13.1 million ha was in arable land use. Permanent crops such as olive (*Olea europaea*), nut, coffee (*Coffea* sp.), grape (*Vitis* sp.) and cocoa (*Theobroma cacao*) accounted for a further 5.2 million ha (Willer *et al.*, 2022).

Although only 7 percent of the total organic area is made up from permanent crops, a high percentage of organic producers grow permanent crops. There were at least 3.4 million organic producers in 2020 (Willer *et al.*, 2022), with 56 percent of organic producers in Asia, followed by Africa (24 percent). India has the highest number of organic producers (1.6 million) followed by Ethiopia (220 000) and the United Republic of Tanzania (149 000).

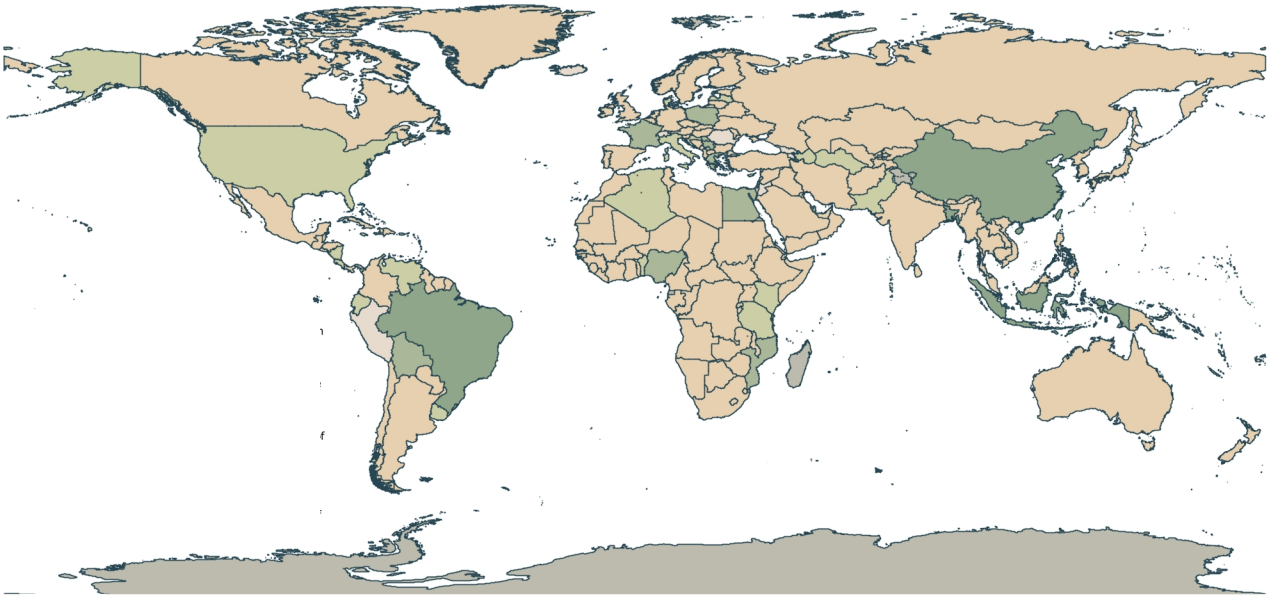
Agroforestry is an umbrella term for a wide variety of practices that integrate trees and shrubs in the agricultural landscape with the shared goal of diversifying and sustaining production for increased socioeconomic and environmental benefits (Muchane *et al.*, 2020). Agroforestry is reported in the *Global Forest Resources Assessment* (FAO, 2020) in the category of "Other land with forest cover" along with orchards and palms (all with a tree canopy cover with more than 10 percent of trees able to reach a height of 5 m at maturity). In 2020, FAO (2020) reported a combined area of 45.4 million ha of agroforestry over seventy-one countries and territories, mostly in Asia (31.2 million ha) and Africa (12.8 million ha). There was an estimated 1.28 million ha of agroforestry in North and Central America.

More recently, regenerative agriculture has been promoted as a potentially viable approach to crop and animal production with a primary objective of improving SOM content. There is currently no universally agreed definition of regenerative agriculture and no regional or global estimates of its adoption through time. The practices associated with regenerative agriculture are discussed more fully in Chapter 5 of this report.

Irrigation is an important way of increasing yields in water-limited areas. If irrigation is improperly implemented, it can lead to increases in salt accumulation (as discussed in Section 3.10.6). Globally, about 22.5 percent of cropland is equipped for irrigation (FAO, 2024a), with approximately 70 percent of the area equipped for irrigation in Asia. The land area equipped for irrigation increased from 338 million ha in 2015 to 354 million ha in 2025 (FAO, 2024b). The largest increases in area occurred in Asia (especially in China and Southeast Asia) and in Brazil (Figure 4.13).



Figure 4.13. Country map of change in land equipped for irrigation (ha) between 2015 and 2022



Change in land area equipped with irrigation from 2015 to 2022

>1 million ha gain
100 000 ha gain to 1 million ha gain
100 000 ha loss to 100 000 ha gain
100 000 ha loss to 1 million ha loss
>1 million ha loss
No data

Refer to the disclaimer on page ii for the names and boundaries used in this map. Dotted line represents approximately the Line of Control in Jammu and Kashmir agreed upon by India and Pakistan. The final status of Jammu and Kashmir has not yet been agreed upon by the parties. Final boundary between the Republic of Sudan and the Republic of South Sudan has not yet been determined. Final status of the Abyei area is not yet determined.

Source: **FAO**. 2024. *FAOSTAT: Land Use*. [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

Despite the importance of forest harvest management for reducing threats to soil functions, there are no global or regional estimates of the extent of specific silvicultural or harvest techniques. Harvesting information is important because different forest management techniques impact on soil functions such as SOC change and GHG emissions from soils (reviewed in Section 5.3.1 and Section 5.3.2).

4.3.4 Urban expansion, infrastructure development and extractive industries

Urban expansion is a major driver of land cover and soil change. Urbanization was defined by ITPS (2022) as the conversion of agricultural, natural or semi-natural areas to urban uses including sparse settlements, urban fringes, industrial estates and transport infrastructure. Much of the urbanized land also undergoes soil sealing, which ITPS (2022) defined as the permanent covering of the soil surface with impervious materials such as concrete, asphalt or tar seal, as well as buildings and other structures that cannot be easily removed. The conversion of land covers such as forest or croplands to urban uses also leads to the loss of ecosystem services such as food, fibre or biofuel production, water infiltration and groundwater recharge and, in some cases, higher rates of threats to soil functions such as soil erosion and loss of SOC.

Globally, more people live in urban areas than in rural areas (UN DESA, 2019). In 2018, 4.2 billion people resided in urban settlements compared to 3.4 billion in rural areas (UN, 2019). The 2018 levels of a regional population living in urban areas were highest in Northern America (82.2 percent) and Latin America and the Caribbean (80.7 percent), followed by Europe (74.5 percent), Oceania (68.2), Asia (49.9 percent) and Africa (42.5 percent). It was predicted that by 2050, the levels of regional population living in urban areas would total nearly 90 percent in Northern America and Latin

America and the Caribbean, nearly 85 percent in Europe, 59 percent in Africa, 66 percent in Asia and about 70 percent in Oceania (UN DESA, 2019).

There has been rapid urban expansion in the period since 1990 (He *et al.* 2019; Zhou *et al.* 2018; Liu *et al.* 2018; UN DESA, 2019). For example, He *et al.* (2019) estimated that between 1992 and 2016, global urban land had increased from 27.5 million ha to 62.1 million ha, with those values corresponding to an urban expansion area of 34.6 million ha and an annual growth rate of 3.5 percent (Table 4.7). The rate of urban expansion differs greatly between countries and regions. Regions such as North America and Europe that experienced high rates of urbanization prior to the 1990s, had lower rates of growth by 2016, while countries in Asia, Africa, and South America (Table 4.8 and Figure 4.14)) with low urbanization rates prior to 1992 had relatively rapid growth in urban extent up to 2016 (He *et al.* 2019; Zhou *et al.* 2018).

Table 4.8. Urban land expansion between 1992 and 2016

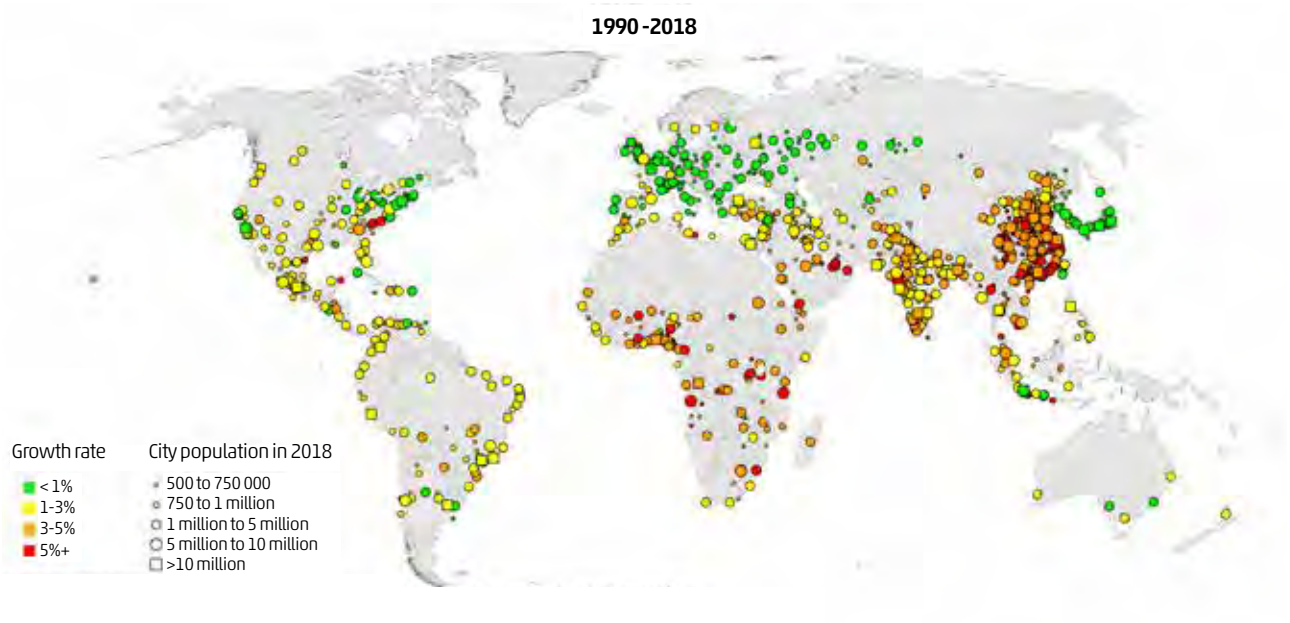
Region/country	Urban land area (1992) (km^2)	Urban land area (2016) (km^2)	Urban land area increase (1992–2016) (%)[2]
Asia	65 629	209 815	220
China	15 865	88 047	455
India	8 012	22 450	180
Viet Nam	190	4 689	2368
Indonesia	1 685	6 076	261
Malaysia	1 189	5 026	323
North America	78 856	148 535	88
United States	70 704	137 093	94
Latin America and the Caribbean	36 191	99 376	175
Brazil	15 181	43 998	190
Argentina	5 528	16 905	206
Mexico	5 321	13 249	149
Europe and Central Asia	81 478	121 339	49
Türkiye	1 622	9 066	459
Italy	7 688	13 141	71
France	7 706	13 018	69
Spain	2 971	8 216	177
Russian Federation	21 000	24 639	17
Sub-Saharan Africa	7 700	27 597	258
South Africa	2 957	8 126	175
Southwest Pacific	4 798	14 481	202
Australia	4 412	12 762	189
Near East and North Africa[1]			
Iran (Islamic Republic of)	2 905	8 721	200
Saudi Arabia	1 646	6 657	304
World	274 652	621 143	126

[1]Data could not be calculated for the Near East and North Africa.

[2]The values for percentage increases have been recalculated and corrected here using the original land areas cited in He et al. (2019).

◊ *Source*: **He, C., Liu, Z., Gou, S., Zhang, Q., Zhang, J. & Xu, L.** 2019. Detecting global urban expansion over the last three decades using a fully convolutional network. *Environmental Research Letters*, 14(3): 034008. https://doi.org/10.1088/1748-9326/aaf936

Figure 4.14. Urban growth rate between 1990 to 2018 for urban areas with populations with 500 000 inhabitants or more



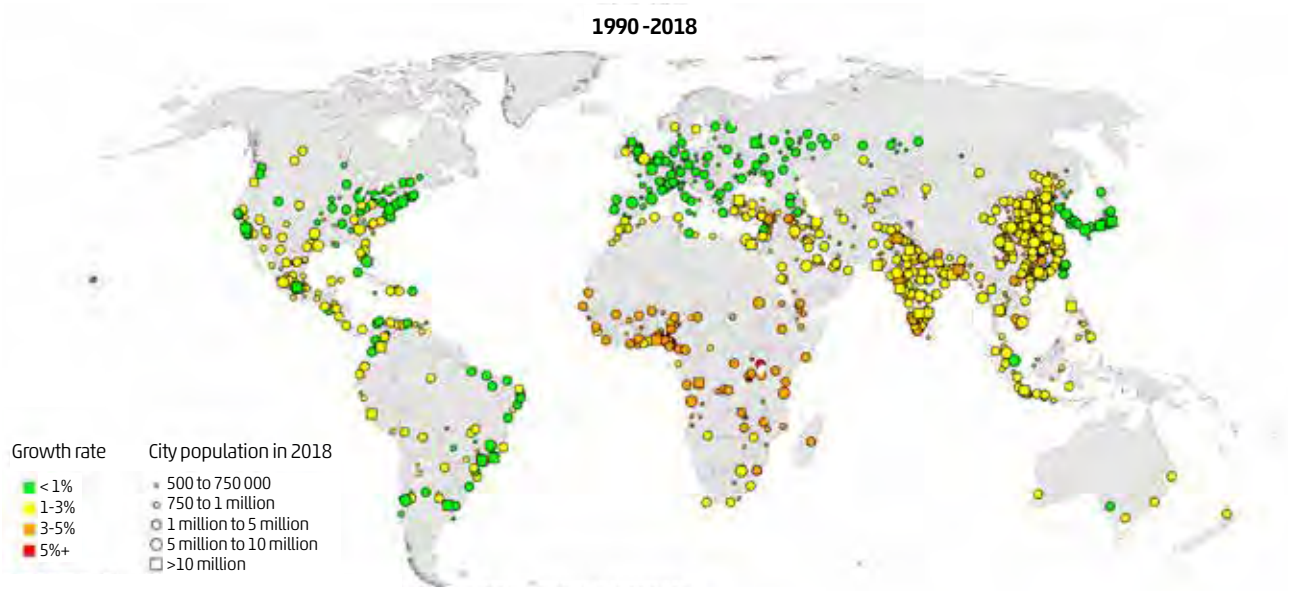
Refer to the disclaimer on page ii for the names and boundaries used in this map. Dotted line represents approximately the Line of Control in Jammu and Kashmir agreed upon by India and Pakistan. The final status of Jammu and Kashmir has not yet been agreed upon by the parties. Final boundary between the Republic of Sudan and the Republic of South Sudan has not yet been determined. Final status of the Abyei area is not yet determined.

◊ *Source:* **UN DESA Population Division**. 2018. World Urbanisation Prospects 2018: Maps. In: *UN DESA Population Division*. New York, USA. [Cited 19 June 2025]. https://population.un.org/wup/maps?tab=Growth%20Rates%20of%20Urban%20Agglomerations&year=1990-2018

Projections of the rate of urbanization into the near future (i.e. to 2030) are important for evaluating the likely impact of urbanization on soil threats. The rate of urbanization is projected to slow in many areas in the period up to 2030. The global map of predicted urbanization between 2018 to 2030 (Figure 4.14) generally shows lower rates of urbanization than in the 1990–2018 period (Figure 4.15). The highest rate of urban growth is associated with urban areas in sub-Saharan Africa, the Near East and a few cities in India and China.



Figure 4.15. Predicted urban growth rate between 2018 and 2030 for urban areas with populations with 500 000 inhabitants or more



Refer to the disclaimer on page ii for the names and boundaries used in this map. Dotted line represents approximately the Line of Control in Jammu and Kashmir agreed upon by India and Pakistan. The final status of Jammu and Kashmir has not yet been agreed upon by the parties. Final boundary between the Republic of Sudan and the Republic of South Sudan has not yet been determined. Final status of the Abyei area is not yet determined.

◊ *Source:* **UN DESA Population Division**. 2018. World Urbanisation Prospects 2018: Maps. In: UN *DESA Population Division*. New York, USA. [Cited 19 June 2025]. https://population.un.org/wup/maps?tab=Growth%20Rates%20of%20Urban%20Agglomerations&year=2018-2030

The study by Van Vliet, Eitelberg and Verburg (2017) examined the effects of urban land growth on cropland (termed "land take" in the study). They reported that urban growth had disproportionately caused the loss of cropland over other land covers, an observation supported by the modelling done by Chen *et al.* (2020) for the 2015–2100 period. Van Vliet, Eitelberg and Verburg (2017) distinguished between land that was suitable for crop production (defined as all land that could be used for rainfed farming based on the biophysical properties of a location) and available land (all land that is suitable for crop production, minus land that is legally protected or covered by natural vegetation) (Table 4.9).



Table 4.9. Change in urban area in major world regions and displacement of food production due to urbanization

Region	Urban land[1]			Urban land in locations available for crop production			Displacement of food production due to urbanization
	2000	2040	Change	2000	2040	Change	
	(million ha)		(%)	(million ha)		(%)	(%)
Central and South America	37 5	100.6	63.1	22.2	51.8	29.6	2.9
North America	42.8	111.7	68.8	20.1	32.5	12.4	2.9
Sub-Saharan Africa	11.3	68.4	57.1	6.9	42.7	35.8	3.6
Near East and North Africa	15.9	29.1	13.2	5.9	8.2	2.5	7.4
Europe	59.4	92.0	32.5	43.9	61.2	17.2	4.8
Russia and Central Asia	23.8	47.3	23.5	11.6	18.6	7.0	0.5
India	16.6	45.9	29.3	13.7	34.0	20.2	3.3
China	40.2	85.9	45.7	24.7	40.1	15.4	12.5
Asia Pacific	20.2	30.7	10.5	15.1	22.0	7.0	2.0
Oceania	3.1	9.7	6.6	1.6	4.1	2.5	1.3
World	270.9	621.3	350.4	165.5	315.3	149.8	3.7

[1]Urban land refers to peri-urban and village systems and urban systems combined.

Source: **Van Vliet, J., Eitelberg, D.A. & Verburg, P.H.** 2017. A global analysis of land take in cropland areas and production displacement from urbanization. *Global Environmental Change*, 43: 107–115. https://doi.org/10.1016/j.gloenvcha.2017.02.001

The land lost to food production due to urbanization (up to the year 2000), was 166 million ha, or 4.71 percent of all land that was available for crop cultivation. It was projected to increase to 8.84 percent of all land that is available for crop cultivation globally by 2024, with the highest regional values in China (16.4 percent) and Europe (18.0 percent). The cropland converted to urban uses decreased food production on average by 3.7 percent in 2020 compared to 2000. The greatest displacement of food production was projected to be in China (12.5 percent of production displaced between 2000 and 2040) and NENA (7.4 percent of production displaced between 2000 and 2040). Globally, the displacement of crop production due to urbanization was predicted to be 3.7 percent by 2040 (Van Vliet, Eitelberg and Verburg, 2017).

Chen *et al.* (2020) modelled future urban land expansion (2015 to 2100) under the five shared SSPs:

1. Sustainable development (SSP1).
2. Middle-of-the-road development (SSP2).
3. Regional rivalry (SSP3).
4. Inequality (SSP4).
5. Fossil-fuelled development (SSP5).

The trend in urban expansion differed across the five SSPs, with SSP5 (fossil-fuelled development) having the highest rate of urban area expansion, with continued increases through 2100. While the other SSPs showed growth to about 2060 to 2070 followed by levelling off or declines, the projections depend on population growth projects. For example, China showed a probable decrease in urban

land area after 2050 due to projected population declines. The authors also estimated which land covers would be converted to urban lands under the five SSPs (Table 4.10). In each case, the majority of land cover loss was from cropland, followed by forestland. The cropland losses equalled about 1 percent of global cropland area and direct losses of global crop production were projected to exceed 1 percent and could exceed 4 percent for specific crop types.

Table 4.10. Projections of the proportion of urban land expansion between 2015 and 2100 taken from different land covers

Land cover	Proportion of urban land expansion (2015–2100) (%)				
	SSP1	SSP2	SSP3	SSP4	SSP5
Forestland	26.98	26.36	21.42	22.89	29.29
Grassland	12.01	10.86	5.71	7.49	14.64
Wetland	2.10	2.46	2.77	3.51	2.23
Bare land	3.76	4.68	7.28	5.86	2.98
Cropland	55.14	55.63	62.82	60.25	50.84
Other	0.1	0.01	0.00	0.00	0.02

Note: SSP1 = sustainable development, SSP2 = middle-of-the-road development, SSP3 = regional rivalry, SSP4 = inequality (SSP4), and SSP5 = fossil-fuelled development.

◊ *Source:* **Chen, G., Li, X., Liu, X., Chen, Y., Liang, X., Leng, J., Xu, X.** *et al.* 2020. Global projections of future urban land expansion under shared socioeconomic pathways. *Nature Communications*, 11(1): 537. https://doi.org/10.1038/s41467-020-14386-x

Extractive industries take raw materials from the Earth and process them. Raw materials include fossil fuels (coal, oil and gas), minerals (such as bauxite, phosphate, potash, Cu, Au, diamonds, and rare earth minerals) and aggregates (such as sand, gravel and clay) (UNECE, 2021), along with the extraction of peat for sale as a soil amendment or a biofuel (discussed at the end of Section 4.3.1). Extractive industries have two main impacts on soils and soil functions: soil excavation or burial during the extraction and processing of materials, and the pollution of soils adjacent to the sites of the raw materials extraction.

The mining and fossil fuel sectors play a strong economic role in 63 countries (UNEP and IRP, 2019). The extent of soil disruption from mining and fossil fuel extraction was summarized in the *Global Assessment of Soil Pollution* (FAO and UNEP, 2021a, 2021b). Summaries of soil pollution and mine-related disturbances are included in the regional assessment chapters of this report.

Urbanization and extractive industries also interact with the other drivers of threats to SSM to change the rate and scale of impacts on soil functions. For example, in their discussion of the factors that contributed to the formation of gullies and ravines by water erosion, Kuhn *et al.* (2023) included vegetation removal, overgrazing, deforestation, forest burning, infrastructure construction (such as roads), arable land obstacles that promoted the accumulation of surface runoff, hydrological changes in rural and urban areas, trails used by animals or humans, retention basin overflows, and mining activities or other interventions as activities that concentrated surface flow and led to gully and ravine formation. This emphasizes that few of the drivers examined occurred in isolation and that interactions between them could multiply their harmful impact on the soil.



4.4 Conflict as a driver of soil change

The period since 2015 has seen the number of violent conflicts reach the highest number since the Second World War (UN, 2023). These conflicts have a terrible human impact, and also affect the broader environment including soils and are considered in this report as one of the drivers of the threats to soil functions.

Conflicts have an immediate impact on the production of biomass for provision of food, feed, fibre and energy by agriculture and forestry. When conflict erupts in productive regions, it can limit access to arable land and reduce soil productivity due to physical degradation and chemical contamination, posing risks to food security and water quality locally and globally (Behnassi and Haiba, 2022; Lin *et al.*, 2023).

Even after hostilities end, the long-term negative impacts of conflict on soils can continue to threaten soil functions. Military activities, such as the movement of heavy vehicles, digging and munitions explosions, directly damage soils by compacting and disrupting soil layers, degrading soil structure and leading to erosion, rutting and mud formation, thereby decreasing available arable land and overall soil productivity (Broomandi *et al.*, 2020; Certini, Scalenghe and Woods, 2013). The effects of conflict on soils can be long-lasting.

The presence of landmines and unexploded ordnance, collectively referred to as explosive remnants of war (ERW), further restricts soil use by making the land unsafe for farmers and foresters to access and by destroying machinery and infrastructure (Neyter, Zorya and Muliar, 2024). Restoring soil access in conflict settings requires extensive demining and clearance of ERW. This clearance process is slow and high risk, and can inadvertently cause additional soil damage through compaction, erosion, biodiversity loss and further chemical contamination (Berhe, 2007).

The United Nations Mine Action Service (UNMAS) has active programmes to remove and destroy mines and other unexploded ordnance in 20 programmes around the world. In 2023, 419 km^2 of land were confirmed safe through the UNMAS programmes. However, large land areas still require urgent attention. For example, UNMAS (2023) cited 1 290 km^2 of land contaminated by mines in Afghanistan alone.

Beyond physical damage, soil pollution due to armed conflicts poses an additional threat. Explosives, munitions, military waste and damaged industrial facilities can release contaminants such as heavy metals, hydrocarbons, explosive residues and POPs (Solokha *et al.*, 2024; Broomandi *et al.*, 2020; Certini, Scalenghe and Woods, 2013; Rodríguez-Seijo *et al.*, 2024). Studies on military-caused soil pollution have increased substantially, rising from approximately 150 per year in 2015 to almost 300 in 2022 (Stadler, Temesi and Lakner, 2022). For example, results based on soil sampling in a zone of heavy shelling in Ukraine found higher concentrations of the trace metals, including Pb, Cd, Co and Cu, compared to adjacent unshelled areas (Solohka *et al.*, 2023). These contaminants can leach into soil and groundwater, rendering them potentially unsafe for agricultural use. A review by Rodríguez-Seijo *et al.* (2024) found that measures of microbial biodiversity (such as biomass and microbial decomposition), or soil enzymatic activities (such as dehydrogenase activity), are adversely impacted over time by high levels of toxic elements produced by conflicts.

A final impact of conflict on soils is the building of military-related infrastructure (e.g. barracks, camps and firing ranges) and structures for manoeuvres and drills (Stadler, Temesi and Lakner, 2022). These cause land-use conversions such as deforestation and are similar in impact to urbanization and soil sealing.

Another impact of conflicts on soils stems from disruptions to the supply of agricultural inputs, including fertilizers and pesticides. In Syria, these disruptions, combined with the deteriorating economic situation, led to a decrease in the average total application of N, P and K from 40.9 kg/ha in 2011 to 11.1 kg/ha in 2013 and 4.2 kg/ha in 2022 (FAO, 2024c). In 2022, the Syrian Arab Republic had

negative nutrient balances for N, P and K and experienced nutrient mining of all three nutrients (See Section 11.3.4 for further information as well as Figure 11.8, Figure 11.9 and Figure 11.10 in the same chapter).

Since the first months of the armed conflict in the Gaza Strip in Palestine, three UN agencies (FAO, the United Nations Institute for Training and Research [UNITAR] and the United Nations Satellite Centre [UNOSAT]) in cooperation with the Ministry of Agriculture have issued periodic reports on the damage to agricultural sector in Gaza. According to FAO-UNOSAT geospatial assessment, in the Gaza strip, as of 28 September 2025, 86.9 percent (13 080 ha out of 15 053 ha) of cropland has been damaged. North Gaza Governorate has the largest proportion of damaged cropland (94.1 percent and 2 946 ha). With regard to the crop categories, orchards and other trees showed the largest damage proportion, with 7 871 ha (60.2 percent) to total cropland damage (FAO, UNITAR and UNOSAT, 2025).

The damage on the Palestinian agricultural sector, especially soil, and the extent of its pollution and deterioration as a result of the conflict cannot be accurately estimated until the conflict ends and technical teams are able to conduct field surveys. Thus, it is very important to emphasize that internal and external technical and financial support should be ready to estimate the extent of soil pollution and to restore land and soils for agriculture.

Ukraine's soils were affected by major degradation processes such as erosion, organic C losses, over-compaction, salinity, acidification, and pollution (Baliuk, Kucher and Maksymenko, 2021). Information on the state of these threats can be found in the regional assessment chapter for Europe and Central Asia (Chapter 9) of this report.

Scientists from the National Scientific Center "Institute for Soil Science and Agrochemistry Research named after O.N. Sokolovsky" (NSC ISSAR) created maps of the impact of armed conflicts on the soil cover of Ukraine from May 2022 and March 2023 (published in Bailuk *et al*., 2024). Based on the March 2023 map, Bailuk *et al*. (2024) estimated that 9.2 million ha of black soils in the Ukraine had been degraded as a result of the conflict.

Seventeen types of land degradation were identified (Baliuk, Kucher and Plisko, 2024), grouped under five types of degradation related to armed conflicts (Table 4.11):

1. mechanical degradation (soil profile damage from explosions, fortifications and trenches);
2. physical degradation (compaction from heavy military equipment);
3. physical and chemical degradation (acidification, alkalization and salinization from soil mixing caused by explosions);
4. chemical contamination (localized toxic pollutants from destroyed equipment and weapons); and
5. biological degradation (increased soil toxicity, reduced biological activity and microorganism loss).

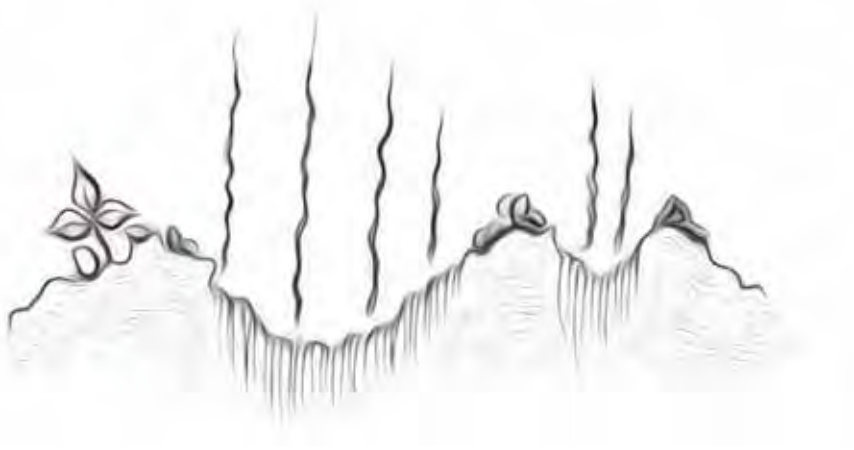


Table 4.11. Types of land and soil degradation related to armed conflict in Ukraine

Types of degradation	Specific types of degradation
Mechanical	Activation of erosion processes and landslides
	Mechanical disturbances of the morphological structure of the soil profile
	Mixing of genetic horizons
	Formation of funnels, ditches and pits
Physical	Deterioration of physical properties of soils
	Persistent changes in particle size distribution
	Persistent changes in aggregate composition (destructuring)
	Compaction
Chemical	Deterioration of humus state of soils
	Nutrient depletion of the soil
	Salinization
	Pollution and contamination
Physicochemical	Acidification and alkalization
Biological	Reduction (narrowing) of biodiversity
	Reduction of soil biological activity
	Deterioration of the state of sanitation
	Soil toxicity

In addition, several other drivers are expected to have long-term negative impacts on soil conditions even after conflicts end. These include: contamination of soil surfaces and profiles by military debris, equipment remnants, personnel, ammunition, and fuel residues; extensive mining of agricultural land and forest shelterbelts; destruction of irrigation systems and shelterbelts; flooding of large areas from breached dams and reservoirs; and groundwater pollution (Figure 4.16).

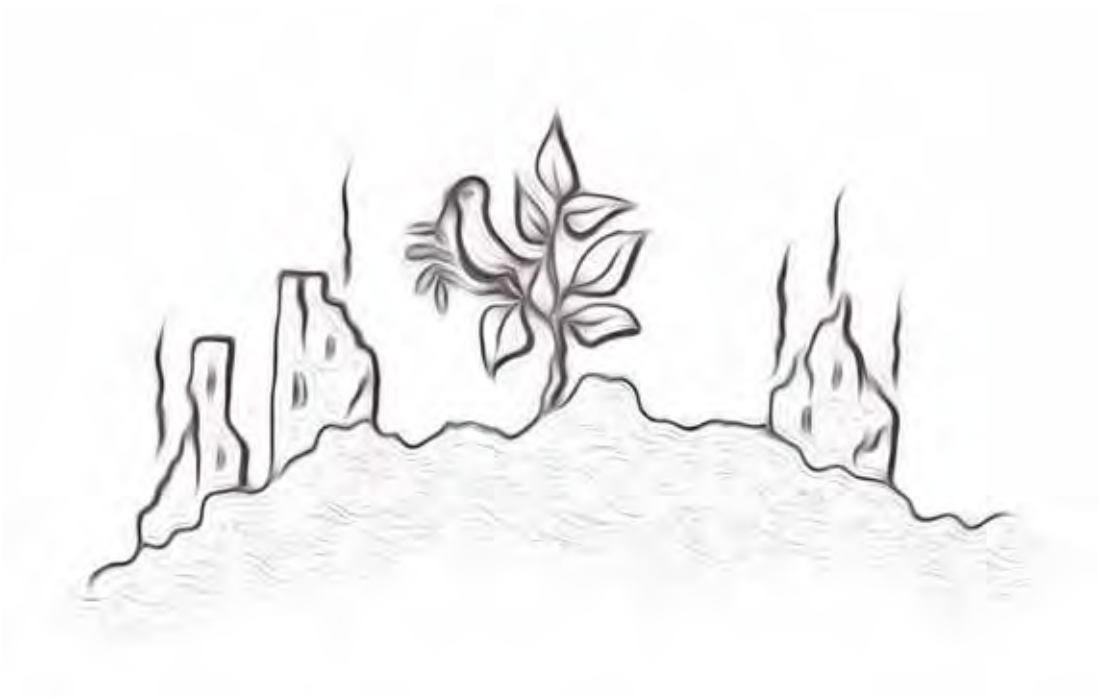


Figure 4.16. Soil disruption caused by conflict in Ukraine



1. © Hanna Honcharova, 2. © Yurii Afanasiev, 3. © Yevhenii Panasenko, 4. © Yurii Zalavskyi, 5. © Vyacheslav Kikonenko and Alina Revtje-Uvarova, 6. © Maxym Solokha, and 7. © Yurii Afanasiev

Notes: 1, 2, 3, 6 and 7 show soil disturbance due to explosions, while 4 and 5 show signage for the presence of mines and mines collected from the agricultural field.

Using satellite data, available sources and selective field research, the spatial distribution of degradation types was analysed and maps of their impacts were created at national, regional, district and local levels. In the Kharkiv region, approximately 40 percent of its predominantly black soil territory was affected, encompassing 1.3 million ha, with 0.9 million ha of agricultural land (Bailuk *et al.*, 2024). In the Kharkiv district, initial estimates revealed localized chemical contamination and shell craters over 6 816 ha, with 1 373 ha of burnt fields, and 700 ha with destroyed military equipment and artillery positions (Bailuk *et al.*, 2024). The NSC ISSAR's national project also surveyed soils in the Chkalovska territorial community (*hromada*), mapping affected areas, assessing black soil degradation and developing fertility restoration measures (Bailuk *et al.*, 2024).

The FAO assessments of conflict impacts on agricultural land in Ukraine highlight substantial biophysical damage and productivity losses. In Kharkiv Oblast, armed conflict has caused widespread soil disturbance and contamination, with over 420 000 craters, affecting 4 213 ha of land directly and placing an additional 28 268 ha at risk of contamination, alongside significant volumes of displaced soil and localized compaction (Table 4.12). At the same time, broader agricultural impacts are evident in Mykolaiv Oblast and across Ukraine, where conflict has driven a decline in cultivated land (–17 percent from 2021 levels), large-scale cropland abandonment, and sharp reductions in crop yields and production, particularly for key staples such as wheat, barley, maize, and sunflower. Together, these findings demonstrate that the armed conflict has not only degraded soil quality and land resources but also significantly undermined agricultural productivity and food production capacity, reinforcing the need for targeted land restoration, demining, and recovery interventions.

Table 4.12. Damage and losses to land and soil due to armed conflict in Ukraine

Indicator	Damage and losses (value)
Losses of the land fund / agricultural capacity	
Cropland abandonment in conflict-affected areas (in a whole country)	~2.34–2.40 million ha (~7% of national cropland)
Reduction in cultivated land (Mykolaiv oblast, 2021–2024)	~0.31 million ha decrease (–17% from baseline)
Yield reductions (Mykolaiv oblast, 2021–2024)	Yields declined after 2021 and remained below baseline for wheat and barley, sunflower, and maize, while sunflower fell to around 0.9 tonnes/ha by 2024, and maize declined from 7.7 to 2.4 tonnes/ha.
Damage to land resources and soils (physical and contamination impacts)	
Agricultural land directly disturbed by craters and explosions (Kharkiv oblast)	4 213 ha (0.7% of cultivated land)
Potentially contaminated agricultural land (Kharkiv oblast)	28 268 ha (4.4% of cultivated land)
Total number of craters identified (in the studied area covering 37 percent of Kharkiv oblast)	420 615 craters
Average crater density	66 craters/km^2
Estimated volume of displaced soil	376 310 m^3

◊ *Source:* **Mohammed, I., Senel, G., Ostapenko, S., Lucas, T.A., Amaya Ortiz, L., Ostapenko, P., Bonchkovskyi, O., Shvaiko, V. & Henry, M.** 2026. *A remote-sensing assessment for monitoring agricultural land in Mykolaiv Oblast, Ukraine – A satellite-based change analysis of cropland and productivity, 2021–2024.* Rome, FAO. https://doi.org/10.4060/cd9075en

◊ **Senel, G., Dadhich, G., Ostapenko, P., Bonchkovskyi, O., Shvaiko, V., Ostapenko, S., Lucas, T.A., Amaya Ortiz, L. & Henry, M.** 2025. *Mapping impacted soils of war-affected farmers in Kharkiv Oblast – Impact assessment of agricultural land 2021–2023.* Rome, FAO. https://doi.org/10.4060/cd6127en

Planning land and soil restoration in areas affected by conflicts after conflict ends is urgent and would require national and international technical and financial support. This global task can only be solved with systematic support of the international community in the following areas:

1. implementing pilot projects for soil rehabilitation;
2. developing infrastructure for surveying disturbed soils;
3. establishing a modern system for remote soil surveying and monitoring; and
4. preparing and enacting a global support programme for large-scale restoration of soils degraded by conflicts.

4.5 Summary

Population growth and demand for commodities

The global population continues to increase, although at a lower rate than for most of the twentieth century (UN DESA, 2023, 2024) (Figure 4.1). The global population was 5.7 billion people in 1995, rising to 7.3 billion in 2015 and 8.1 billion in 2024. It is projected to rise further to 9.8 billion by 2054 and reach a peak of 10.2 billion in 2084 (UN DESA, 2024).

The addition of 0.8 billion people between 2015 and 2024 has substantially increased the demand for food, wood and other commodities (UN DESA, 2024). Similar trends to food production occur for harvest of forest products for fuel and industrial wood products, which had small increases from 2015 to 2022 (Figure 4.4). The majority of wood products are used for fuel, primarily in the Global South (FAO, 2024b).

Land-use change

The increased demand for food and related increases in demand for fuel and fibre have led to a continuous increase in the global harvested area of main crops (FAO, 2024b) (Figure 4.5). The total harvested area has increased by 24 percent between 2000 and 2022 to 1.47 billion ha, with an increase of 80 million ha between 2015 and 2022 (FAO, 2025).

The increase in harvested area and changing dietary preferences causes more land to be brought into agricultural production. The conversion of forestland to cropland has the greatest impact through higher erosion rates and losses of SOC, shifts in the community composition of microorganisms and disruption of natural nutrient cycles. FAO (2024a) estimated the net change in forestland at −4.7 million ha per year in the 2010–2020 period. On a regional basis, the loss of forestland between 2015 and 2022 has been greatest in sub-Saharan Africa, NENA and Latin America and the Caribbean (FAO, 2024c). Reforestation has been by far the greatest in China, with approximately 13.4 million ha of net forestland gain between 2015 and 2022. India has had the second-highest area of net forestland gain, with 1.9 million ha (FAO, 2024c).

The expansion of cropland has been greatest in sub-Saharan Africa, which has also experienced the greatest loss of forestland (FAO, 2024c). Of the ten countries that gained over 1 000 000 ha of cropland between 2015 and 2021, seven are in sub-Saharan Africa, two are in Latin America and the Caribbean and one is in Southeast Asia. Decreases in cropland area are greatest in China, India, Mexico, and the United States (FAO, 2024c).

Conversions to cropland have been a driver of the loss of SOC from wetlands (Fluet-Chouinard *et al.,* 2023). Wetland drainage for upland croplands was the most common cause of loss of natural wetlands since 1700, followed by conversion to flooded rice, urban areas, forestry, non-rice wetland cultivation, pasture and peat extraction (Fluet-Chouinard *et al.,* 2023).

The increasing demand for food and other commodities has also led to an increase in inorganic fertilizer use. The use of inorganic fertilizers increased from 2015 to 2020, but all three of the main types of inorganic fertilizers have decreased between 2020 and 2022 (Figure 4.9). FAO (2024b) attributed the decrease to higher natural gas prices and conflict-related supply chain disruptions.

The regional summaries of N budgets (Table 4.3) show relatively high positive N budgets (i.e. higher N inputs than outputs) for five regions but low positive budgets in sub-Saharan Africa and negative budgets in the Southwest Pacific region.

Countries with P excess are common in Asia, Europe and Central Asia and Latin America and the Caribbean (Table 4.4 and Figure 4.11) while the majority of countries in sub-Saharan Africa have experienced nutrient mining of P.

Changes in agricultural and forestland management

Adoption of SSM practices reduces the threats to soil functions (FAO and ITPS, 2017). An accurate assessment of the adoption of SSM would allow a quantitative appraisal of the reduction in the threats but unfortunately at best, only partial information is available on adoption at the global (or even regional) scale.

Kassam, Friedrich, and Derpsch (2022) presented estimates for the adoption of conservation agriculture, which included no-till and reduced tillage adoption as a key component. They estimated that in 2018/2019, approximately 205 million ha were under conservation agriculture, corresponding to about 14.7 percent of the total global cropland.

Cover crops offer benefits comparable to no-till and reduced tillage in reducing threats to soil functions, and they complement those tillage practices. Yield benefits from cover crops are also greater in non-water-limited locations (Blanco-Canqui *et al.*, 2022; Garba, Bell and Williams, 2022). Unfortunately, data on the area of adoption of cover crops is lacking (Feng *et al.* 2024).

Agroforestry is an umbrella term for a wide variety of practices that integrate trees and shrubs in the agricultural landscape with the shared goal of diversifying and sustaining production for increased socioeconomic and environmental benefits (Muchane *et al.*, 2020). In 2020, 45.4 million ha was reported to be under agroforestry across 71 countries and territories, mostly in Asia (31.2 million ha) and Africa (12.8 million ha) (FAO, 2020).

More recently, regenerative agriculture has been promoted as a potentially viable approach to crop and animal production with a primary objective to improve SOM content. There is currently no universally agreed definition of regenerative agriculture and no regional or global estimates of its adoption through time.

Urban expansion

Projections of the rate of urbanization into the near future (i.e. to 2030) are important for evaluating the likely impact of urbanization on soil threats. The rate of urbanization is projected to slow in many areas in the period up to 2030, with the global map of predicted urbanization between 2018 and 2030 (Figure 4.14) (UN DESA, 2019) showing generally lower rates of urbanization than in the 1990–2018 period. The highest rate of urban growth is associated with urban areas in sub-Saharan Africa, the Near East and a few cities in India and China.

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Conflict as a driver of soil change

The period since 2015 has seen the number of armed conflicts reach the highest number since the Second World War (UN, 2023). These conflicts not only have a human impact, but also affect the broader environment, including soils, and are considered in this report as one of the drivers of the threats to soil functions.

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Beyond physical damage, soil pollution due to conflicts poses an additional threat. Explosives, munitions, military waste and damaged industrial facilities can release contaminants such as heavy metals, hydrocarbons, explosive residues and persistent organic pollutants (Solokha *et al.*, 2024; Broomandi *et al.*, 2020; Certini *et al.*, 2013; Rodríguez-Seijo *et al.*, 2024).

Another impact of conflicts on soils stems from disruptions to the supply and availability of agricultural inputs, such as fertilizers and pesticides.

The conflicts in the Gaza Strip in Palestine and the Ukraine highlight the devastation wrought by armed conflicts on the human use of soils. In the Gaza Strip, FAO, UNITAR and UNOSAT (2025) reported that as of 28 September 2025, 86.9 percent of cropland had been damaged by the conflict. Several other drivers are expected to have long-term negative impacts on soil conditions even after conflicts end. These include: contamination of soil surfaces and profiles by military debris, equipment remnants, personnel, ammunition, and fuel residues; extensive mining of agricultural land and forest shelterbelts; destruction of irrigation systems and shelterbelts; flooding of large areas from breached dams and reservoirs; and groundwater pollution.



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Chapter 5 | Sustainable soil management in support of the Sustainable Development Goals

5.1 Introduction

This chapter synthesizes research since 2015 on how SSM practices can mitigate threats to soil functions and support the SDGs. These practices are also central to the rehabilitation of soil degraded in the past by human activities.

The links between soil threats to soil functions and the SDGs were examined in Chapter 2 of this report. Chapter 3 summarized the global studies on the threats published since 2015 and Chapter 4 examined the drivers that are the underlying causes or pressures that exacerbate threats to soil functions.

Soil management is sustainable if the services provided by soil in support of the SDGs are maintained or enhanced without significantly impairing the soil functions that support ecosystem services and biodiversity (FAO, 2015). In almost every case, the SSM practices examined in the subsequent sections of this chapter were in place prior to publication of the 2015 SWSR report (FAO and ITPS, 2015). What has changed since the publication of that report is the growth in publication of reviews and meta-analyses, and this chapter draws heavily on these results.

A meta-analysis is a set of statistical methods which are used to determine a general result across sometimes hundreds of individual studies (Fohrafellner *et al.*, 2023). Meta-analyses are very useful in a global survey such as that being presented in this report, but they are not without limitations. Fohrafellner *et al.* (2023) examined 31 meta-analyses on SOC published up to 2021 and found that many were flawed on statistical grounds. Overall, it was found that the quality of meta-analyses had improved through the 15-year period examined but the authors cautioned against the uncritical use of many such analyses in policymaking. A second caveat is that the general results of meta-analyses are drawn from a high level of variation across the primary studies used. The applicability of the general result to any specific location has to be assessed in a subsequent step, drawing upon the literature available for soil, environmental and management conditions that occur at the specific site. This need for site-specific information is discussed in Section 6.3 of this report.

This chapter begins with an examination of SSM practices that can be used to address multiple threats to soil functions, such as addition of organic amendments such as manure, the adoption of cover crops, the management of crop residues, the addition of biochar, improved nutrient management, and the adoption of no-till or reduced tillage. The SSM practices are described, and the socioeconomic implications of their use are examined. The key element of the economic evaluation is the impact of the practice on crop yield. Section 5.2 concludes with management systems that involve multiple SSM practices including agroforestry, organic agriculture, conservation agriculture, and regenerative agriculture.

The specific effects of soil management practices on individual soil threats are detailed in subsequent sections of this chapter.

5.2 Cross-cutting sustainable soil management practices

5.2.1 Manure additions

The addition of manure to agricultural soils is an age-old practice, and research since 2015 has added new insights into the many roles that it plays in soil. Here, manure additions are examined primarily for their contribution to crop yield changes.

The contribution of manure to soil nutrient status is substantial. Bouwman *et al.* (2017) estimated that the contribution of manure to total global nutrient inputs ranged between 20 and 25 percent for N and 25 to 30 percent for P. Manure additions are of particular importance in cropping systems where additions of synthetic fertilizer are limited by socioeconomic factors and in organic production systems throughout the world.

Several recent studies have examined the hypothesis that manure additions will increase crop yield beyond the direct nutrient benefit from N and P (Celestina *et al.,* 2019; Hijbeek *et al*., 2017). However, these studies have not found significant yield benefits beyond those associated with N and P content. The nutrients contributed by manure can, however, reduce the quantity of synthetic fertilizers that need to be added to meet crop demands.

Four main constraints to substantially increasing the amount of manure added to soils have all been discussed in the literature. First, manure is already widely added in soils and is already a major contributor to global nutrient inputs (Bouwman *et al*., 2017). Powers *et al.* (2018) state that virtually all manure is currently being applied to soil at some location but often in a suboptimal way.

Second, a large volume of manure must be added to achieve levels of N and P inputs comparable to synthetic fertilizers, while the transport of manure is only financially viable over relatively short distances. This requires proper planning to ensure that the manure produced in intensive livestock operations can be safely distributed in adjacent areas. In sub-Saharan Africa, manures are often low in required P and K, as the biomass that manure is formed from is low in nutrients (Falconnier *et al*., 2023). In many regions, the increased specialization in farming through time has led to large areas where mixed crop–livestock operations are absent and hence there is no economically viable source of manure available (Poulton *et al*., 2018; Powers *et al*., 2018). Poulton *et al.* (2018) stated that specialization was favoured by practical and economic factors and that a return to more mixed farming would probably require significant changes in policies and financial incentives. Moreover, the skills and infrastructure required to work with livestock have been lost through time in specialized crop production areas.

Third, the benefits of manure additions are largely confined to ecological services such as C sequestration and N losses to air and water rather than to crop yield (beyond the direct nutrient content of manure). Given the ease with which nutrients can be added through synthetic fertilizers, the additional handling and application steps required for manures can be unattractive to farmers.

Finally, there is a societal trend in some countries towards reducing livestock production and meat and dairy consumption. This will therefore reduce the future production and use of livestock manures in these countries.

5.2.2 Cover crops

There has been increased interest in cover crops since 2015 in part because of their importance in both conservation agriculture and regenerative agriculture approaches (see Section 5.2.6) (Giller *et al.*, 2021; Kassam, Friedrich and Derpsch, 2022). Cover crops can provide protection from erosion and physical deterioration, as well as providing increased levels of SOC and residual N when grown and left on the surface or ploughed back into the soil as a green manure.

In the past, cover crops were primarily used as part of improved fallow systems and as catch crops. Fallowing is when a field is deliberately left out of production for part or all of the growing season and is used in many cropping systems (Garba, Bell and Williams, 2022). Historically in arid and semi-arid areas, fallowing was used to maximize soil water storage, control weeds and mineralize N for subsequent cash crops (Garba, Bell and Williams, 2022; Sileshi *et al.,* 2009). Cover crops are also a component of sequential agroforestry systems (Muchane *et al.,* 2020).

Catch crops are also used in more humid environments where N leaching is an issue. Uptake of soil N by cover crops decreases N loss from the soil and thereby lessens the harmful environmental consequences associated with N loss (Garba, Bell and Williams, 2022). Ideally, the N in the plant tissue of the cover crops is released in a timely manner during the decomposition of the plant material, so it can be utilized by the subsequent cash crop.

Many plants can be used for cover crops, including many grasses, legumes and brassicas, with the cover crops used either singularly or in combination (Jian *et al.,* 2020; Sileshi *et al.*, 2009). Leguminous species are widely used as cover crops due to their ability to fix N from the atmosphere, increasing the availability of N to subsequent crops.

Although the benefits of cover crops are well established, the effect of cover crops on the yield of cash crops grown after the termination of the cover crop is more variable. Cover crops can compete with the subsequent crop for resources like water and N and hence may impede the establishment and growth of the cash crop (Miner *et al.*, 2020).

The global meta-analysis of Jian *et al.* (2020) found that subsequent crop yields (following a cover crop) showed an average increase of 4.3 percent across 1 462 comparisons in 114 studies, with the greatest increase occurring for cash crops following legumes. Jian *et al.* (2020) reported non-significant increases with mixtures of cover crop species versus slight yield declines with grass cover crops, and significant yield decreases for other types of cover crops (such as brassicas and other broadleaved crops).

The yield benefit following leguminous cover crops is consistent with the yield benefits associated with incorporation of legumes into crop rotations. In a global meta-analysis, Zhao *et al.* (2022) reported that the legume pre-crop effect on main crop yield was predominantly positive (73.6 percent positive, 0.7 percent neutral and 25.7 percent negative), with 50 percent of the data showing a yield increase greater than 10 percent. Averaged across all studies, it was found that the inclusion of legumes in the crop rotation significantly increased subsequent crop yield (by 20.4 percent).

Meta-analyses have found that cover crops can reduce yields in water-limited environments (Blanco-Canqui *et al.*, 2022; Garba, Bell and Williams, 2022). Looking at water-limited environments with <500 mm annual precipitation, Blanco-Canqui *et al.* (2022) reported that cover crops did not affect crop yields in the subsequent crop in 56 percent of comparisons, reduced yields in 38 percent and increased yields in 6 percent, which was attributed to reduced water availability under cover crops. The sites that experienced yield increases under cover crops were predominantly those which had normal or above-average precipitation levels.

Garba, Bell and Williams (2022) completed a meta-analysis of the effects of cover crops on yield and N dynamics in drylands (regions characterized by water scarcity and an aridity index of 0.65 or less). The mean change (percentage) in cash crop yield with cover cropping varied significantly

between dryland climate types. Larger change increases in yield were reported in tropical climates (14.5 percent) followed by continental climates (4.4 percent), while there were decreases in dry, arid and semi-arid climates (11.1 percent) and temperate climates (12.4 percent). It was found that following cover cropping at sites with low mean annual rainfall (<300 mm) led to significant mean reductions of 29 percent in cash crop yield, mean reductions of 24 percent in soil water content and mean reductions of 48 percent in soil mineral N. Where annual precipitation exceeded 700 mm, a neutral cash crop yield response to cover cropping was observed. In higher rainfall regimes (>900 mm), cash crop yield increased on average by 14.5 percent following cover cropping.

The fate of post-termination cover crop residue also influenced cover crop effects. Burning residues resulted in a 43 percent decrease in soil mineral N and a 30 percent decrease in cash crop yield. Rolled and incorporated residues did not significantly impact subsequent crop yield (Garba, Bell and Williams, 2022).

In colder environments, the use of either cover crops or residues on the soil surface can lead to higher dissolved reactive P losses in runoff from fields and increased P inputs into surface waterways, compared to conventionally tilled fields (Van Eerd *et al.*, 2023; Liu *et al.*, 2021). This occurs due to the vegetative cover trapping more snow (also potentially increasing runoff from the field), with the multiple freeze and thaw cycles damaging plant tissues and leading to the release of P from the tissues. The effect of cover crops varies across cold regions and the reasons for this variation are unclear (Van Eerd *et al.*, 2023).

The effects of cover crops on ecosystem services are generally well established and the impacts on yields show a clear association with precipitation and aridity levels. Garba, Bell and Williams (2022) and Miner *et al.* (2020) found that early cover crop termination, careful selection of cover crop functional type and management of cover crop residues post-termination minimized the potential negative effects of cover crops on soil water content and subsequent cash crop yield.

5.2.3 Residue retention, residue return and mulching

Crop residues are a more readily available form of organic inputs to soils compared to manure additions or cover crops and can remain in place after harvest removed from the field. The crop residues that remain in the field can either be left on the soil surface (as in most forms of reduced tillage or no-till) or can be completely incorporated into the soil surface through conventional tillage practices (Turmel *et al.*, 2015). The residues that are removed can be either used for other purposes such as fuel or food for livestock or returned to the soil (Turmel *et al.*, 2015).

Mulching is a broader approach that refers to any material other than soil or living vegetation that is added to the soil surface to form a protective cover (Prosdocimi, Tarolli and Cerdà, 2016) (this includes the addition of branches pruned from trees in orchards). Mulches can also include gravel, geotextiles and crushed stones but such materials have limited applications. Prosdocimi, Tarolli and Cerdà (2016) showed that the various vegetative mulches were effective at reducing erosion and runoff generation but yield effects were not assessed.

Residues are burned, used for animal bedding, sold or used to generate heat through combustion (Lehtinen *et al.*, 2014). Crop residues are burned in the field to facilitate subsequent seeding operations when the seeding equipment designed to seed through heavy residue is not available. Equipment designed to chop and spread residue during harvesting can assist with this. The burning of residues has been restricted or banned in countries such as China (Wang *et al.*, 2023) due to its negative impact on air quality. In many developing countries, the residue from cereal crops has other uses: as a source of feed for animals (through either the introduction of grazing on the harvested field, or by residue removal before using as a farm feed source) (Castellanos-Navarrete *et al.*, 2015), as a commodity to be sold for feeding other people's animals (Pisarek, 2024), or as a biofuel

for the farmer's household (Turmel *et al.,* 2015). At the start of the twenty-first century, there was controversy about the use of residues in ethanol production in developed countries. Soil scientists are generally opposed to crop residue use as biofuels due to the removal of nutrient in the biomass from the field (e.g. Blanco-Canqui and Lal, 2009).

It has proven difficult to determine a clear impact of residue removal on subsequent crop yields. The impacts are highly variable and dependent on the specific management regime and cropping system used, soil- and site-specific factors, and climate during the growing season (Blanco-Canqui and Lal, 2009; MacLaren *et al.*, 2022).

There is evidence that residue retention in combination with no-till or reduced tillage can (on average) increase yields (see Section 5.2.5). A meta-analysis on no-till, residue retention and yield showed that residue retention alone had a significant beneficial effect (median yield increases of 8.2 percent) and no-till combined with residue retention led to median yield increase of 6 percent (Xiao *et al.,* 2021). No-till without residue retention led to a median decline of 1.9 percent in crop yields.

5.2.4 Biochar

Biochar is produced by heating organic matter in oxygen-limited environments. The feedstocks for biochar creation include forestry and sawmill residues, crop residues, and manure and biosolids. Application of biochar to soils is of greatest interest as a means of CO_2 removal from the atmosphere, but studies have also shown that biochar additions to soils can lower N_2O emissions, reduce N fertilizer requirements due to reduced N losses from soils, reduce GHG and NH_3 emissions from manure and compost, increase yield in some soils, increase the water-holding capacity of soils and adsorb organic pollutants and heavy metals (Nabuurs *et al.*, 2020). This section focuses on yield impacts and the economics of biochar application.

In their meta-analysis of 56 studies, Ye *et al.* (2020) reported that under biochar applications, there was an overall mean yield increase of 9.9 percent compared with fertilizer control treatments, while the mean increase was 29.8 percent compared to unfertilized control treatments. It was estimated that the contribution of biochar to the yield increase (beyond the addition of fertilizer) was about 15 percent. The yield benefit was greatest in soils with low SOM contents (i.e. less than 20 g/kg), pH levels below 6.5 and sand-dominated soil textures.

Baveye (2023) argued that economic considerations needed to be included in the ongoing discussion around biochar. The cost of biochar can be high in some regions, with Kochanek *et al.* (2022) citing costs of biochar in high-income countries ranging from USD 325 to USD 550 per tonne to USD 800 per tonne in Australia, and up to USD 5 000 per tonne in the United Kingdom and the United States. They cited payback periods (i.e. where the cumulative yield increases surpass the cost of application) ranging from 15 to 125 years, using the commonly suggested rates of 20 to 100 t/ha. Bach, Wilske and Breuer (2016) reviewed eight studies of the economic return to biochar application from Australia, China, the United States and northwest Europe, with no study showing any resulting profit for the farmer. The authors cautioned, however, that the short duration of most of the studies limited the generality of the result. Bach, Wilske and Breuer (2016) suggested that the profitability of biochar in cropping systems could be readily enhanced if C sequestration benefits were monetized. They noted that a call for C credits for biochar use in agriculture had been almost omnipresent within the biochar research community.

5.2.5 No-till and reduced tillage

No-till (or zero tillage) is a management practice where a crop is planted directly into a seedbed that has not been tilled since the harvesting of the previous crop (FAO and ITPS, 2021). The no-till operation consists of a one-pass planting and fertilizer operation in which the soil and the surface residues are minimally disturbed, and minimal to no disturbance occurs in subsequent weed control. No-till systems eliminate all mechanical seedbed preparation before seeding except for the opening of a narrow strip or small hole in the ground for seed placement to ensure adequate seed to soil contact. Weed control is generally achieved with intensive application of herbicides, or in some cases, with cover crops and crop rotation. Reduced tillage incorporates many elements of no-till, but is associated with higher levels of surface disturbance and crop residue incorporation. Most meta-analyses have focused on no-till rather than reduced tillage and the term no-till will be used throughout the following section.

Adoption of no-till has been shown to lead to decreases in crop yield in humid sites but no change or small increases in semi-arid and arid sites. The widely cited meta-analysis by Pittelkow *et al.* (2015a) drew upon 678 studies with 6 005 paired observations, representing 50 crops and 63 countries. Across all crop categories and locations, no-till was found to have reduced yields by 5.1 percent but there were significant differences among crop categories and between climate zones.

Land aridity and the water management employed (i.e. rainfed vs irrigated) had a clear association with no-till effects on yield. No-till performed best under rainfed management in arid and semi-arid climates, where no-till matched conventional tillage yields for maize and legumes or exceeded yields for wheat, oilseeds and cotton. The other three combinations of aridity and water management (irrigated and dry, rainfed and humid, and irrigated and humid) showed consistent yield decreases for each major crop category except for oilseeds and cotton. Using the same aridity classification used by Pittelkow *et al.* (2015a), Cui *et al.* (2022) also found differences between humid and dry regions, with no significant difference from conventional tillage in dry regions and a 9 percent decrease overall in humid regions.

Across the crop categories, Pittelkow *et al.* (2015a) found that for cereals, average crop decreases were lowest for wheat (2.6 percent decrease) and higher for rice (7.5 percent decrease) and maize (7.6 percent decrease). No-till yields matched conventional tillage yields for oilseed, cotton and leguminous crops. The highest yield decreases were for root crops (21.4 percent decrease) and horticultural and other miscellaneous crops (20.4 percent decrease). Yield decreases were generally highest in the first one to two years after establishment of no-till and lessened subsequently.

In their meta-analysis on no-till, residue retention and yield, Xiao *et al.* (2021) found that the adoption of no-till alone led to a significant median decline in crop yield of 1.9 percent, while residue retention alone had a significant beneficial effect, with median yield increases of 8.2 percent. No-till combined with residue retention led to a median yield increase of 6 percent. These results suggest that the simultaneous implementation of no-till and residue management is key to preventing yield decreases under no-till management.

Cui *et al.* (2022) and Pittelkow *et al.* (2015b) showed that inclusion of residue retention and crop rotation in no-till systems significantly increased yields compared to no-till without these practices. Overall, Pittelkow *et al.* (2015b) found that the largest yield decreases occurred when no-till was implemented alone (9.9 percent decrease). When both residue retention and crop rotation were added to no-till, overall crop losses were only 2.5 percent. For rainfed agriculture in dry areas under no-till in combination with the other two practices, yields were significantly higher (7.3 percent increase), while when no-till was applied alone, yields were significantly lower (11.9 percent decrease). Similarly, Cui *et al.* (2022) found that including crop rotation and residue retention or return with no-till improved wheat and maize yield reductions from a decrease of 5 percent to a decrease of 2.4 percent.

The yield penalty associated with adoption of no-till in tropical and subtropical regions can be lessened with proper N fertilization. Lundy *et al.* (2015) found that no-till yield declines were most consistently observed at low rates of N fertilization during the first two years of no-till adoption. Applications of N fertilizer at annual rates of up to 85 ± 12 kg N/ha significantly reduced the yield penalty for no-till adoption in these regions. In addition, it was reported that when crop rotation was not practiced, or when residues were removed from the field, no-till yield declines were magnified by low rates of N fertilization. The authors observed that N fertilizer additions of at least 85 ± 12 kg N/ha were necessary to offset yield declines due to no-till and that this rate was greater than rates typically used in sub-Saharan Africa.

Adoption of no-till can also be a challenge in organic agriculture due to the prohibition of herbicides for organic certification (discussed in Section 5.2.6). In humid sites, cover crops are often used for weed control in organic production but their high water use limits their use in semi-arid systems (Lehnhoff *et al.*, 2017), meaning that tillage is the major tool for weed control. After a review of alternative approaches for weed control in organic production systems, Lehnhoff *et al.* (2017) concluded that achieving the goal of eliminating tillage in water-limited environments was elusive.

While the economic benefits of no-till include reduced operating costs for labour, fuel and tillage machinery, when compared to conventional tillage (Wade, Claassen and Pailler, 2022), the early adoption of no-till in countries like Argentina (Peiretti and Dumanski**,** 2014) and Canada (Awada, Lindwall and Sonntag, 2014) initially occurred due to groups of innovative farmers, supported by the agrochemical industry, recognizing that conventional tillage was a major cause of soil erosion and soil organic matter loss. Subsequent changes such as decreases in the price of weed-control herbicides and increases in the price of fuel caused more widespread adoption, but the beneficial effect of no-till on erosion control and increased water retention remains a major driver of adoption, while environmental and human health concerns due to the increased use of herbicides seem mostly neglected. In Australia, other agronomic considerations – such as early season weed management options and the ability to seed in a timelier manner – were important in the decision to adopt (D'Emden, Llewellyn and Burton, 2008). Hence, a broad suite of socioeconomic, agronomic and environmental considerations may come into play when considering the adoption of no-till and this may be a challenge where one or more of these enabling conditions are absent.

In Argentina, no-till is the dominant sowing system, used by over 90 percent of farmers, for several reasons (such as soil erosion control and fuel saving) (Peiretti and Dumanski, 2014). Between 2008/2009 and 2018/2019, Kassam, Friedrich and Derpsch (2022) estimated that the area under conservation agriculture in Argentina (including no-till and reduced tillage) increased from 19.7 million ha to 32.9 million ha.

Progress continues to be made on alternative tillage systems to no-till, such as strip and ridge tillage that achieve many of the same protective benefits for soil and leave most of the soil undisturbed. Strip tillage is where row crops are grown in a narrow (5 to 15 cm wide) tilled strip with the area between the rows left undisturbed (Achankeng and Cornelis, 2023). Ridge tillage involves preparation of raised and usually permanent seedbeds that are seeded with minimal soil disturbance. In their meta-analysis of European sites, Achankeng and Cornelis (2023) found that (compared to conventional tillage) no-till reduced yields by 5.1 percent on average, whereas strip tillage and ridge tillage increased yields by 5 percent. There were also differences across textures, with strip tillage performing best on sandy loams, ridge tillage performing best on loams and no-till on sandy clay loams. This type of analysis of the conditions under which practices perform the best is critical information for adoption but is lacking in many meta-analyses.

5.2.6 Improved nutrient management

Nutrient mismanagement occurs through nutrient application in excess of plant demand and through insufficient applications of nutrients which leads to depletion of inherent soil nutrients (nutrient mining) and thus reduced plant yield and lower nutritional value of the plant products. Sustainable use of N is a concern across many disciplines and since 2015, major initiatives such as the UNEP Working Group on Nitrogen (UNEP, 2024) have been initiated.

The studies discussed previously in Section 3.5 focused on the nutrient balance concept, which states that there should be a balance between nutrient inputs to the soil and nutrient outputs in plant products through time. The analyses of many field studies and simulation modelling provide compelling evidence that there is a strong relationship between nutrient balance and losses of nutrients to the environment, and that achievement of a balance though time is essential for the sustainable management of nutrients (McLellan *et al.*, 2018; Davidson *et al.,* 2016). Nutrient balances are also used as key indicators in many national and regional agri-environmental assessments such as those of the European Environmental Agency (EEA) (2023) and OECD (2019).

The set of nutrient best management practices are referred to in some countries as the 4Rs: right source, right rate, right timing and right placement (Johnson and Bruulsema, 2014; Davidson *et al.,* 2016). The selection of the right source and rate is central to achieving an optimal nutrient balance, while the right timing and placement are essential to ensure that uptake of the applied nutrients is as high as possible. Adoption of the 4Rs is effective for reducing both nutrient excess and nutrient mining and reducing the harmful impacts of nutrient excess on the environment.

Optimizing sources of nutrients and nutrient usage

The main sources of N inputs into soil are synthetic fertilizers, manure and biological N fixation, with the relative contribution of each source of N differing across regions (Table 5.1). The effects of various practices on crop yield and mitigation of N pollution have been assessed by a number of meta-analyses since 2015, with the recent meta-analysis by Gu *et al.* (2023) providing a comprehensive range of practices (Table 5.2) and assessed effects (Table 5.3). The nutrient benefits of manures were examined previously, in Section 5.2.1.

Table 5.1. Average percentage of total nitrogen (N) inputs from fertilizer, manure and biological N fixation in 2015 for the seven regions and N fertilizer additions (kg/ha)

Region	N from fertilizer (%)	N from manure (%)	Biological N fixation (%)	Average N fertilizer use (2015) (kg/ha)
Asia	37.6	25.4	19.5	81.6
Europe and Central Asia	52.3	27.2	11.0	90.5
Latin America and the Caribbean	35.6	33.9	20.9	62.1
Near East and North Africa	44.5	18.1	14.5	95.6
North America	60.3	8.4	25.5	70.7
Sub-Saharan Africa	13.8	23.6	46.7	12.2
Southwest Pacific	24.1	41.8	25.7	41.5

Note: Averages are calculated across national values in each region. Other sources of nitrogen (N) (not shown) are irrigation inputs and atmospheric deposition.

◊ *Sources:* **Gu, B., Zhang, X., Lam, S.K., Yu, Y., Van Grinsven, H.J.M., Zhang, S., Wang, X.** *et al.* 2023. Cost-effective mitigation of nitrogen pollution from global croplands. *Nature*, 613(7942): 77–84. https://doi.org/10.1038/s41586-022-05481-8

◊ **FAO**. 2024: *FAOSTAT: Fertilizers by Nutrient*. [Accessed on 9 January 2024]. https://www.fao.org/faostat/en/#data/RFN. Licence:CC-BY-4.0.

Table 5.2. Practices to mitigate nitrogen pollution from global croplands

Enhanced efficiency Fertilizers	Slow or controlled release fertilizers and nitrification and urease inhibitors
Organic amendments	Organic amendments include sphagnum manure, peat, wood chips, grass clippings, straw, compost (manure-based and plant-based), biosolids, sawdust, and wood ash
Legumes	Inclusion of legumes in cropping system
Rate	Optimizing N fertilizer rate based on soil N supply and crop needs (part of the 4Rs)
Type	Shift from urea to ammonium sulphate/nitrate accompanied by lime ($CaCO_3$) (part of the 4Rs)
Time	Splitting fertilizer application into smaller applications throughout the growing season that coincide with the times that the crops most need fertilizer (part of the 4Rs)
Placement	Deep placement compared to surface application (part of the 4Rs)
New cultivar	Adoption of new crop varieties
Irrigation	Inclusion of drip irrigation or optimization of irrigation application
Tillage	No-till

◊ *Source:* **Gu, B., Zhang, X., Lam, S.K., Yu, Y., Van Grinsven, H.J.M., Zhang, S., Wang, X.** *et al.* 2023. Cost-effective mitigation of nitrogen pollution from global croplands. *Nature*, 613(7942): 77–84. https://doi.org/10.1038/s41586-022-05481-8

Table 5.3. Effect of management practices on percentage change of yield, nitrogen use efficiency, ammonia volatilization, nitrous oxide emission, nitrogen oxides emission, nitrogen leaching and nitrogen runoff

Practice	Yield1	NUE[1]	NH_3 volatilization	N_2O emission	NO_x emission	N leaching	N runoff
	Percentage increase (+) or decrease (−)						
Enhanced efficiency fertilizers	+24	+21	−69 (12)	−35 (98)	−46 (32)	−29 (7)	−52 (14)
Organic amendments	n.s.	+38	−38 (18)	n.s.	−19 (20)	−22 (19)	−27 (17)
Rate[2]	n.s.	+36	−42 (46)	−58 (94)	−26 (26)	−39 (120)	−45 (42)
Type	n.s.	+86	−66 (26)	n.s.	−37 (5)	n.s.	No data
Timing	+27	n.s.	n.s.	−38 (14)	−74 (15)	46 (17)	No data
Placement	+18	+59	−72 (14)	−33 (3)	No data	39 (6)	No data
Legume inclusion	+19	+60	−37 (2)	n.s.	No data	−37 (17)	−34 (15)
New cultivar	+6	+22	n.s.	−7 (4)	No data	n.s.	n.s.
No-till	n.s.	No data	111 (11)	n.s.	n.s.	−20 (17)	n.s.
Drip or optimal irrigation	n.s.	+8	−36 (10)	−45 (14)	−93 (34)	−69 (28)	No data

Notes: Values shown are the overall mean across studies. n.s. = not significant.

[1] The number of studies in the meta-analysis of studies are in parentheses after the percentage amounts but are not presented for percentage change in yield or nitrogen use efficiency.

[2] Rate compares optimum rate to the highest fertilizer input rate used in the study.

◊ *Source:* **Gu, B., Zhang, X., Lam, S.K., Yu, Y., Van Grinsven, H.J.M., Zhang, S., Wang, X.** *et al.* 2023. Cost-effective mitigation of nitrogen pollution from global croplands. *Nature*, 613(7942): 77–84. https://doi.org/10.1038/s41586-022-05481-8

Several studies have shown that nutrient uptake from synthetic fertilizers can readily displace the nutrients available from other sources. For example, MacLaren *et al.* (2022) found that when crop demand for N was already met through fertilizer, only a small additive benefit of the other practices was observed. Zhao *et al.* 2022 reported that legumes significantly increased crop yields for the following crop (by a median value of 20.4 percent), but that the yield advantage was partially or completely lost when N fertilizer was added to the following crop (the N in the fertilizer being substituted for the N from the legume residues).

Enhanced efficiency fertilizers hold the promise of better synchronicity between plant N demands and N fertilizer supply (Lam *et al.*, 2022; Li *et al.*, 2018). Gu *et al.* (2023) estimated that EEFs increased NUE by 21 percent across the studies included in their meta-analysis. Efficiency enhancing products in fertilizers include:

- urease inhibitors, which can decrease loss of N as NH_3;
- nitrification inhibitors, which can reduce loss of N_2O, leaching of nitrate and denitrification; and
- controlled-release fertilizers, where a surface coating regulates the release of N through time.

Urease and nitrification inhibitors can also be combined into a double inhibitor. Commercial forms of these EEFs are widely available: Siman *et al.* (2023) listed five commercially available formulations of slow-release N fertilizer, 20 controlled-release N fertilizers, 5 nitrification inhibitors and 3 urease inhibitors.

Gu *et al.* (2023) (Table 5.6) estimated that EEFs increased yield on average by 24 percent compared to conventional fertilization. Li *et al.* (2018) also found a yield benefit to EEFs but found that the increases in yield did not cover the costs of EEFs relative to conventional fertilizer for maize and rice crops, with only rice showing a sufficient yield increase to cover the increased cost of EEFs. Enhanced efficiency fertilizers cost more than regular synthetic fertilizers, cited by Li *et al.* (2018) to be 1.3 to 1.6 times the cost of conventional fertilizers, and cited by Pan *et al* (2023) to be 1.5 to 5 times the cost of conventional fertilizers. Li *et al.* (2018) and Lam *et al.* (2022) both noted that while the economic benefit to society of reducing N losses would make the use of EEFs economical, this broad advantage was unlikely to cause widespread adoption by farmers. Both sets of authors emphasized that incentives to adopt EEFs could create an economic shift and then lead to much greater adoption.

Li *et al.* (2018) also noted that EEFs must be added to the soil using correct rate, placement and timing if their benefit was to be realized:

> *EEFs are not a panacea for addressing the <reactive nitrogen> challenge without the elimination of fertilizer mismanagement and the implementation of knowledge-based management practices* (Li *et al.*, 2018, p. e519).

Zou, Zhang and Davidson (2022) listed four alternative sources for P inputs beyond fertilizers and manure:

1. phosphorus recovery from wastewater treatment plants (e.g. struvite);
2. the reuse of wastewater and urine;
3. phosphorus recovery from food production and consumption systems; and
4. the liberation and use of the legacy/residual P in soils.

Techniques to access the legacy soil P include double-cropping, crop rotation, liming to reduce acidity (and hence increase P availability) and no-till. The first three sources are viable where agricultural production is located close to densely populated areas. Powers *et al.* (2019) mapped the distribution of manure-rich cultivated areas and populous cultivated areas and showed that substantial areas had the proximity preconditions for high levels of P reuse in cropping systems.

Biofertilizers are promoted as a sustainable alternative to synthetic fertilizers and as source of nutrients in organic production systems. Biofertilizers are defined as formulations comprised of living microbial cells with a single strain or multiple strains that promote plant growth by increasing

nutrient availability and acquisition (Mitter *et al.*, 2021). They differ from other inoculants used as biopesticides or phytostimulators (Mącik, Gryta and Frąc, 2020). In 2019, biofertilizers accounted for approximately 5 percent of the total fertilizer market and more than 150 products based on microbial strains were registered for agricultural purposes (Mącik, Gryta and Frąc, 2020). Nitrogen-fixing biofertilizers have the greatest market share, drawing upon both symbiotic N-fixing bacteria from the family *Rhizobiaceae* and free-living non-photosynthetic N-fixing bacteria mainly belonging to *Azotobacter* and *Azospirillum* genera. Other biofertilizers include those that increases the availability of P for plants, plant growth-promoting biofertilizers, K biofertilizers, and Zn and S-solubilizing biofertilizers (Mącik, Gryta and Frąc, 2020).

The development of new biofertilizers can be difficult. Mitter *et al.* (2021) listed four types of challenges to product development, including soil and environmental issues, plant- and inoculant-related challenges and various practical issues. This latter group included costs for production at a commercial scale, farmer accessibility, regulation and intellectual property issues. Despite this, the authors believed that the very major recent advances in microbial science would lead to the development of new biofertilizers in the future.

Optimizing rate of nutrient additions

The traditional approach to developing nutrient recommendations for crops involves soil sampling before seeding or during plant development, assessing the level of plant-available nutrients and then adding sufficient nutrients to ensure that the total level of nutrient is sufficient to meet crop yield targets (Olfs *et al.*, 2005; Grzebisz, Niewiadomska and Przygocka-Cyna, 2022). The most popular soil nutrient tests measure mineral (inorganic) forms of N (nitrate and ammonium) and P (orthophosphate).

Gu *et al.* (2023) compared the effect of fertilization at the right rate (determined from appropriate soil or plant analysis) versus the highest rate of N fertilizer used in each study. The effect on yield was non-significant, but significant reductions were observed across studies in the range of 20 to 80 percent for NH_3 volatilization, N_2O emissions, N leaching and N runoff (Table 5.3). The scale of reduction increased as the difference between the highest rate and the right rate increased. Overall, the decreases in N loss led to an overall mean increase of 34 percent in NUE when the right rate was applied rather than the highest rate used in the study, but the highest rate used in research studies may have been unrealistically high compared to usual farm practices.

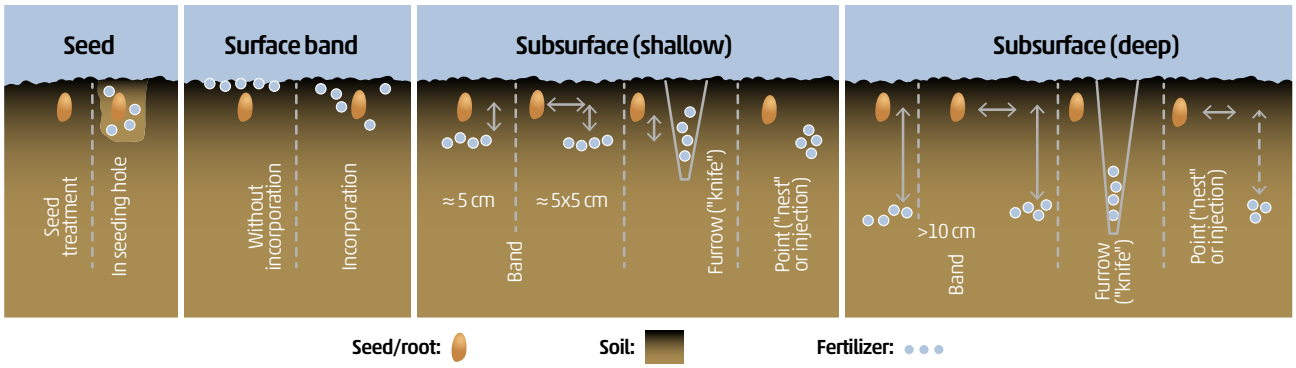
The inclusion of different sources of nutrients complicates the setting of the right rate, which is essential if the nutrient inputs are to be balanced with plant outputs. Inclusion of nutrients from organic amendments and legume residues is more complex, as the nutrients must be converted from an organic to a mineral form (mineralization) for plant uptake. The degree and rate of mineralization depend on the weather during the growing season and on soil characteristics, which adds uncertainty to selection of the right rate of nutrient inputs. The contributions of the legacy pool to crop growth in any given year is also dependent on soil characteristics. Additionally, the procedures for sampling soils and assessing the nutrient content can be subject to error (Schut and Giller, 2020).

Although the legacy P pool has been known for years, recent research has highlighted the contributions of the soil N pool to N uptake by crops (Yan *et al.*, 2020; Grandy *et al.*, 2022). In a synthesis of field and pot studies, Yan *et al.* (2020) reported that most (58 percent) of the N in crops came from sources other than the fertilizer added in that crop year. Grandy *et al.* (2022) suggested that as soils became healthier, they were better able to supply N from multiple sources and that soil test recommendations based on snapshots of mineral N supply were unable to capture the contributions from diverse sources.

Optimizing placement and timing of nutrient additions

Studies on the placement of fertilizers typically contrast fertilizer application by broadcast spreading of fertilizers (with or without subsequent incorporation), with concentrated subsurface placement of fertilizers in bands or furrows adjacent to seeds (Figure 5.1). The subsurface placements can occur at varying depths (typically 5 to 10 cm, 10 to 15 cm or greater than 15 cm in experimental designs).

Figure 5.1. Fertilizer placement techniques



◊ *Source*: Adapted from **Nkebiwe, P.M., Weinmann, M., Bar-Tal, A. & Müller, T.** 2016. Fertilizer placement to improve crop nutrient acquisition and yield: A review and meta-analysis. *Field Crops Research*, 196: 389–401. https://doi.org/10.1016/j.fcr.2016.07.018

Nkebiwe *et al.* (2016) reviewed 40 field studies published between 1982 to 2015 and reported that subsurface placement overall led to a 3.7 percent mean increase in yield and a 11.9 percent higher nutrient content in the above ground parts of plants compared to broadcast applications. The yield increases differed for different fertilizer types and combinations: yield increases were highest for urea and soluble phosphate together (27.3 percent increase) followed by ammonium and phosphate (14.7 percent increase), urea alone (11.6 percent increase), ammonium alone (3.8 percent increase) and phosphate alone (no increase). There was also no significant yield increase for subsurface solid manures. The authors recommended the subsurface placement of combinations of ammonium and soluble P or urea and soluble P as an effective way to improve yield.

Chen, Xiao and Chen (2023) reviewed the effect of depth of placement in 96 studies. They found higher yield increases overall (10.7 percent) due to subsurface placement than Nkebiwe *et al.* (2016). Yield increases were similar for placement depths of 5 to 10 cm (11.4 percent increase) and >10 cm (9.1 percent increase). Gu *et al.* (2023) found mean yield increases of 19 percent for the placement studies they examined (Table 5.6).

Deeper placement of fertilizers, as implemented in the studies summarized by Nkebiwe *et al.* (2016) and Chen, Xiao and Chen (2023), required a high degree of mechanization and high energy inputs (Chen, Xiao and Chen, 2023) and therefore the technique was not suitable when farmers had not access to agricultural machinery. For farming systems with hand seeding and fertilization, microdosing (applying a small rate of fertilizer within the planting pocket or in its close vicinity) was a viable alternative approach to placement (Aune, Coulibaly and Giller, 2017). Placement of manure and seeds together in planting basins was also a placement technique suitable for smallholder farming (Kuyah *et al.*, 2021).

The right timing of nutrient additions allows the availability of the nutrient to be better matched with critical growth stages of the crop and contributes to efficient nutrient uptake (Davidson *et al.*, 2016). Timing is strongly tied to the specific fertilizer being used and the results tend to be site- and fertilizer-specific. When new fertilizer options are introduced, such as controlled release fertilizers, it is necessary to run parallel studies on correct timing. For example, the introduction of controlled-release urea fertilizer in China has stimulated research on the one-time application of fertilizer rather than application at multiple periods during the growth season (Cui *et al.*, 2022).

Soil texture, probable rainfall or irrigation and fertilizer properties also need to be considered when determining amounts and timing of fertilizer applications.

Social factors in selection of fertilization programmes

The implementation of the 4Rs programme requires accurate, site-specific information on rates, timing, sources and placement but adoption also requires that farmers apply the fertilizers according to the recommendations. Research from regions with high fertilizer application rates such as North America (Davidson *et al.* 2016) and South Asia (Begho, Eory and Glenk**,** 2022) has shown that even with knowledge-based recommendations available, other considerations may determine the actual application practices. The primary reason for this is that farmers often apply more fertilizer than is recommended as an "insurance policy" to obtain the highest yield possible, given uncertainties about weather and crop prices (Davidson *et al.* 2016). This insurance value is most prevalent in regions with subsidized fertilizers, such as much of South and East Asia (Begho, Eory and Glenk**,** 2022). In these regions, risk-averse farmers tend to overapply fertilizer if they perceive that it is less costly than growing a crop in soil lacking N, which otherwise would prevent the crop from attaining full potential (Begho, Eory and Glenk**,** 2022). A final but important social factor is the cost of new machinery to implement the 4Rs programme and the logistics around the timing of inputs.

5.2.7 Organic agriculture

Organic agriculture involves crop and animal production systems without the use of synthetic fertilizer and pesticides. Although the specific regulations governing organic accreditation differ across countries, a similar range of prohibited and required activities occur in most regulatory systems (Meemken and Qaim, 2018). In organic production, the use of synthetic fertilizer, synthetic chemical pesticides, genetically modified organisms and sewage sludge are prohibited. The required activities may include use of organic seeds and locally adapted varieties, specific measures to improve soil fertility (such as crop rotation, organic fertilizer and erosion control) and pest and weed control achieved only through mechanical, biological or thermic measures. As of 2020, 76 countries have implemented standards for organic agricultural production (Willer *et al.*, 2022). Information on the adoption by country and region can be found in Section 4.2.3 of this report.

Studies summarized by Meemken and Qaim (2018) found that the adoption of organic production reduced farm energy use and emissions of N_2O and GHGs, and more generally, lowered the leaching loss of nitrate, and increased many measures of microbial biodiversity.

It is well established in the literature that yields of crops are lower in organic agriculture than in conventional agriculture. In their meta-analysis, De La Cruz *et al.* (2023) reported that across 786 comparisons, overall yield was reduced by 18.4 percent compared to conventional agriculture. Average yield reductions were greatest for root and tuber crops (29 percent), followed by leguminous crops (25 percent), cereals (18 percent), oilseed and oleaginous crops (16 percent) and vegetables and melons (*Cucumis melo*) (15 percent). Only fruits and nuts did not show a statistically significant difference between the two production systems. Similar results were reported in three earlier meta-analyses summarized by Meemken and Qaim (2018). As well as lower yields, organic agriculture is associated with higher year-to-year variations in yield than conventional systems. Knapp and Van Der Heijden (2018) found that organic agriculture per unit yield had significantly lower temporal stability compared to conventional agriculture.

Despite the lower yields associated with organic production, organic agriculture can be significantly more profitable than conventional systems due to the premiums paid for organic crops. According to Crowder and Reganold (2015), the benefit–cost ratios (8 to 7 percent reduction) and net present values (27 to 23 percent reduction) of organic agriculture without organic premiums were

significantly lower than conventional agriculture. When actual premiums were applied, organic agriculture was significantly more profitable (22 to 35 percent) and had higher benefit–cost ratios (20 to 24 percent) than conventional agriculture. The authors found that organic premiums were between 29 and 32 percent, but that break-even premiums necessary for organic profits to match conventional profits were only between 5 and 7 percent, even with organic yields being 10 to 18 percent lower.

However, wide variations in profitability occur within regions. For example, in five case studies of groups of farms in sub-Saharan Africa, Schader *et al.* (2021) found higher total gross margins (292 percent) for one case study, but a much lower or a slightly negative percent difference for the other four, due in part to a limited or incomplete adoption of organic practices in the other sites. This was attributed to limited knowledge or capacity in managing the organic production system, a lack of suitable organic biomass and other organic inputs for soil fertility management and plant protection inputs, and the lack of stable markets that generated organic price premiums.

5.2.8 Agroforestry

Agroforestry is an umbrella term for a wide variety of practices that integrate trees and shrubs in the agricultural landscape with the shared goal of diversifying and sustaining production for increased socioeconomic and environmental benefits (Muchane *et al.*, 2020). Agroforestry is reported in the *Global Forest Resources Assessment 2020* (FAO, 2020) in the category of "other land with forest cover" (having a tree canopy cover of more than 10 percent of trees able to reach a height of 5 m at maturity), along with orchards and palm trees. Seventy-one countries and territories reported an area of 45.4 million ha of agroforestry in 2020, mostly in Asia (31.2 million ha) and Africa (12.8 million ha) (FAO, 2020), with an estimated 1.28 million ha of agroforestry in North and Central America.

Given the wide range of practices, it is difficult to assess the benefits of agroforestry across multiple locations. Implementing agroforestry also often involves other practices discussed previously in this section, such as the implementation of agroforestry in sub-Saharan Africa which often involves the use of cover crops and mulches. In these cases, it is difficult to separate the effect of trees or shrubs from that of the cover crops (Kuyah *et al.,* 2021; Muchane *et al.,* 2020). A variant of agroforestry of particular relevance for wind erosion control is the use of shelterbelts, which are rows of shrubs or trees grown perpendicular to the dominant wind direction (Vypritskiy and Yuferev, 2023; FAO and ITPS, 2021c).

Studies on the economic and environmental benefits of agroforestry are also difficult due to the range of practices involved. Meta-analyses summarized by Kuyah *et al.* (2021) and Muchane *et al.* (2020) showed that agroforestry in the humid and subhumid tropics could improve crop yields, reduce pest problems, increase soil fertility, and increase infiltration and thereby reduce runoff and water erosion and increase soil moisture. However, Sileshi *et al.* (2009) reported that in approximately half of the summarized studies, yield increases from the inclusion of woody legumes and herbaceous green manure legumes in fallows were insufficient to offset the yield losses incurred during the fallow period when the legumes were grown.

5.2.9 Regenerative agriculture

Regenerative agriculture has received wide media coverage in the past decade, including two popular films: Kiss the Ground (Tickell and Tickell, directors, 2020) and Common Ground (Tickell and Tickell, directors, 2023). The increasing popularity of the approach is closely linked to concerns about declining soil health, specifically soil biological health. Unlike organic and conservation agriculture, there is no widely agreed-upon definition of regenerative agriculture, although six main principles (Table 5.4) have often been identified (Jaworski *et al.*, 2023; Rehberger *et al.*, 2023). Giller *et al.* (2021)

identified organic regenerative agriculture as a distinct variant of regenerative agriculture that prohibited the use of synthetic fertilizer and pesticides (as is the case in organic agriculture). The integration of livestock involves several techniques identified as regenerative, such as rotational grazing, adaptive multipaddock grazing or holistic planned grazing.

Table 5.4. Practices commonly associated with the main principles of regenerative agriculture

Principle	Practices
Minimize soil disturbance	No-till or reduced tillage, conservation agriculture, controlled traffic, and deep-rooting crops for compaction alleviation.
Maximize crop diversity	Diverse crop rotations, multispecies cover crops and agroforestry.
Keep soil covered	Cover crops, residue retention or return, agroforestry, and grazing management.
Maintain living roots year-round	Cover crops, leys (arable land temporarily used for forage crops or grazing), winter crops, ratoon crops, late season summer crops, and agroforestry.
Reduce or eliminate synthetic fertilizers	Animal manures, compost, green manure and cover crops, inclusion of nitrogen (N)-fixing crops in rotation, inoculation of soils and manures, and enhanced efficiency N sources.
Reduce or eliminate pesticides	Diverse crop rotations, multispecies cover crops, biological pest control, and agroforestry.

◊ *Sources*: Adapted from **Giller, K.E., Hijbeek, R., Andersson, J.A. & Sumberg, J.** 2021. Regenerative Agriculture: An agronomic perspective. *Outlook on Agriculture*, 50(1): 13–25. https://doi.org/10.1177/0030727021998063

◊ **Rehberger, E., West, P.C., Spillane, C. & McKeown, P.C.** 2023. What Climate and Environmental Benefits of Regenerative Agriculture Practices? An Evidence Review. *Environmental Research Communications*, 5(5): 052001. https://doi.org/10.1088/2515-7620/acd6dc

There are many practices that can be adopted under the regenerative agriculture approach (Table 5.4). Many such practices have individually been shown to lead to improved soil biological attributes and increased soil organic matter (Giller *et al.*, 2021; Jordon *et al.*, 2022). The bulk of the scientific literature on regenerative agriculture has examined studies on the individual practices listed in Table 5.4, but advocates of the approach argue that practice-specific studies cannot capture the benefits created when multiple practices are implemented simultaneously (Sands *et al.*, 2023).

The lack of consideration of the holistic benefits of regenerative agriculture complicates any assessment of the effect of its adoption on yields (Rehberger *et al.*, 2023). Sands *et al.* (2023) stated that a focus on productivity above all else was misplaced, and that instead, the value of the regenerative approach lay in the improved relationship between farmers and the land. Sands *et al.* (2023) proposed a definition of regenerative agriculture that placed such values first as:

> *A way of farming comprised of entangled values and practices and founded in Indigenous principles*[1] *of loving-caring for the Earth. This approach to farming values 1) reciprocity, 2) respect, 3) collective (human and non-human) wellbeing, 4) knowledge cocreation, and 5) (re)localization, and it is often practiced through some combination of 1) minimizing soil disturbance, 2) maintaining vegetative soil cover, 3) maximizing diversity, 4) integrating livestock, and 5) minimizing synthetic agrichemicals.* (Sands *et al.*, 2023, p. 33).

[1] The mention refers to Indigenous Peoples principles.

5.2.10 Conservation agriculture

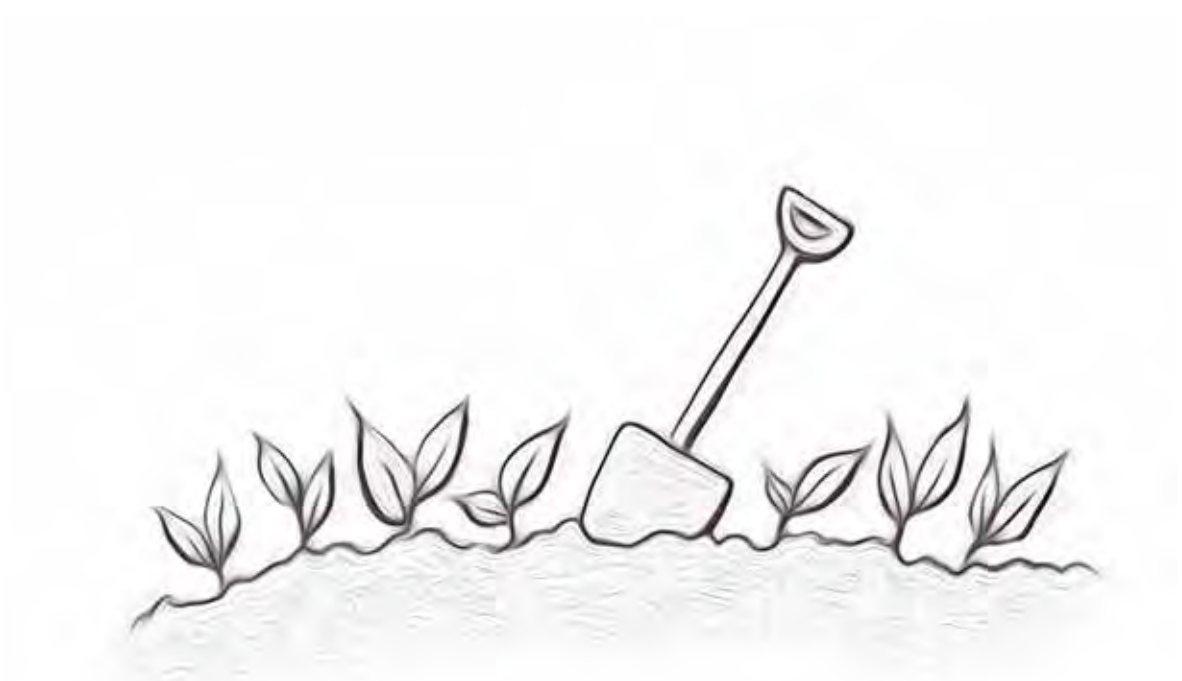
According to Kassam, Friedrich and Derpsch (2022), conservation agriculture had three key principles:

1. continuous minimum tillage or no-till;
2. the maintenance of a permanent mulch cover (through crop residue retention or cover crops); and
3. a diversification of species in the cropping system (ideally with both annual and perennial crops, using a balanced mix of legumes and non-legumes and cover crops).

The authors compiled global data estimating that in 2018/2019, the total cropland area in conservation agriculture was 205.4 million ha, corresponding to 14.7 percent of global cropland. It was estimated that the adoption of conservation agriculture had been greatest in Australia and New Zealand (with 74 percent of total cropland), South and Central America (68.7 percent of total cropland) and North America (33.6 percent of total cropland). The country-specific totals are reported in the regional assessment chapters of this report.

There is strong support for the finding that yield benefits and erosion control attributed to no-till can only occur in combination with residue retention (Xiao *et al.*, 2021; Mhazo, Chivenge and Chaplot, 2016). Giller *et al.* (2015) stated that no-till adoption in the absence of an effective soil mulch cover can be disastrous and can lead to accelerated soil erosion.

Critics of conservation agriculture (such as Giller *et al.*, 2015) have argued that the benefits (including yield increases) claimed by some advocates have been overstated and that the area estimates and yield comparisons are often based only on no-till or reduced tillage adoption rather than on adoption of all three foundational practices.



5.3 Sustainable soil management to improve food security and support sustainable food systems (SDG 2 and SDG 11)

5.3.1 Introduction

The specific target of SDG 2: Zero Hunger that has greatest relevance for soil science is Target 2.4: By 2030, ensure sustainable food production systems and implement resilient agricultural practices that increase productivity and production, that help maintain ecosystems, that strengthen capacity for adaptation to climate change, extreme weather, drought, flooding and other disasters and that progressively improve land and soil quality (UN, 2022).

The specific indicator, Indicator 2.4.1, concerns the proportion of agricultural area under productive and sustainable agriculture. The specific SSM practices examined in this section (i.e. practices that reduce soil erosion, nutrient mining, salt accumulation and acidification) are all known to reduce soil threats that reduce soil and crop productivity.

Sustainable Development Goal 11: Sustainable Cities and Communities, calls for inclusive, safe, resilient, and sustainable cities and covers the spatial aspect of urbanization by its indicator of land consumption (SDG Indicator 11.3.1: Ratio of land consumption rate to population growth rate). The term “land consumption” used in the UN indicator is used to discuss the conversion of natural and agricultural land to urban uses.

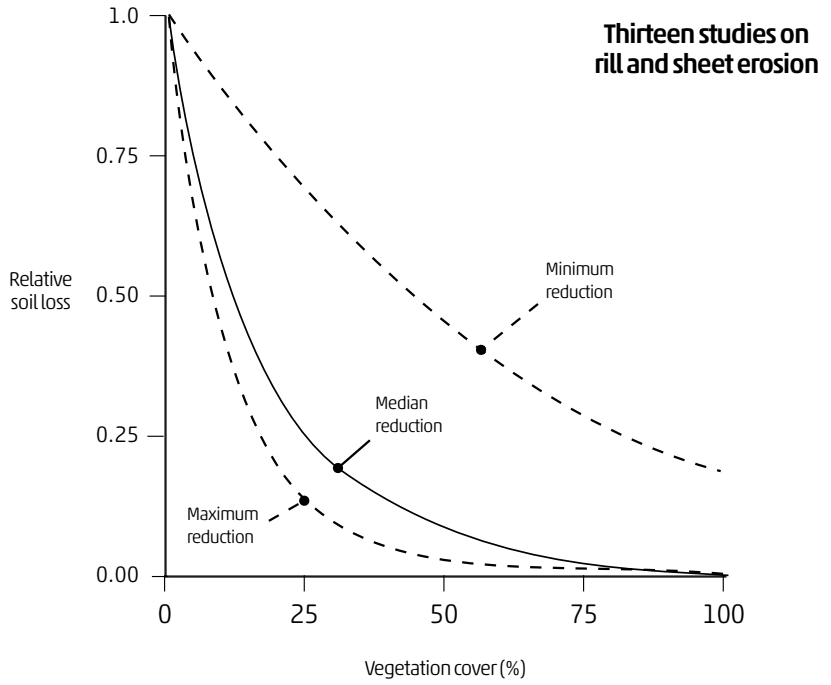
5.3.2 Reducing soil erosion caused by water, tillage and crop harvest

Measures to conserve soil through a reduction in soil erosion have a long history (Baveye *et al.,* 2011) and there are ample research literature and technical support available, such as the World Overview of Conservation Approaches and Technologies (WOCAT) (WOCAT, n.d. [a]). Specific conservation techniques and case studies are also presented in the six volumes of the Recarbonizing Global Soils project (FAO and ITPS, 2021a, 2021b, 2021c, 2021d, 2021e, 2021f).

The major innovations since 2015 do not involve new techniques to reduce erosion but instead employ meta-analyses of the efficacy of different erosion control measures, focusing on sheet and rill erosion by water.

New reports support the well-established finding that vegetative cover of either living plants or crop residues is the key control for sheet and rill erosion in agricultural soils (FAO, 2019) (Figure 5.2). The above ground vegetation intercepts raindrops and lessens their impact on soil detachment, reduces surface sealing and compaction, and hence lessens runoff, while below ground, plant roots increase soil microporosity and hence the infiltration rate of water, thereby reducing runoff, while also increasing the resistance of the soil to detachment of soil particles by flowing runoff. Plants (and the soil organic matter formed from them) also contribute to the formation of stable aggregates that resist the detachment of soil particles.

Figure 5.2. Graphical summary of 13 studies on the relationship between relative soil loss by interrill and rill erosion and vegetation cover



◊ *Source:* **FAO.** 2019. *Soil erosion: the greatest challenge for sustainable soil management.* Rome. https://openknowledge.fao.org/server/api/core/bitstreams/a268e967-a373-4a56-adac-ae1d7ab3567f/content

The various approaches to soil erosion reduction have been classified by WOCAT into four major groups: agronomic, vegetative, structural and overall management regime change (WOCAT, n.d. [b]) (Table 5.5). Agronomic measures can typically be adopted within the existing cropping system and have the least impact on current land use. Vegetation and structural measures require more substantial investments of labour and funds by land users and often require land to be devoted to the measures themselves, thereby reducing the area available for cropping. Changes in the overall management regime have the greatest impact on land users and their communities (Section 6.3).

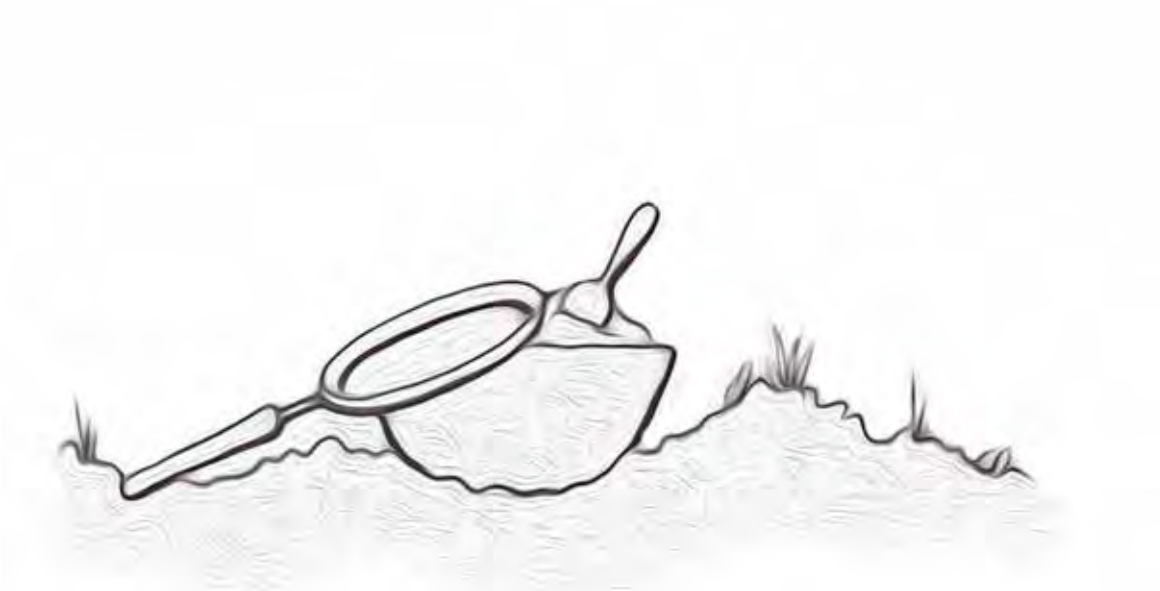


Table 5.5. Summary of types and examples of soil conservation measures

Type of measure	Characteristics	Examples
Agronomic	Usually associated with annual crops. Repeated routinely each season or in rotational sequence. Of short duration and not permanent. Do not lead to changes in slope profile. Normally independent of slope.	Cover cropping, intercropping, intercropping, relay cropping, and crop rotations. Use of compost/manure, mulching, trash lines, and green manure. Zero tillage (no-till), minimum tillage and contour tillage. Breaking compacted subsoil and deep ripping.
Vegetative	Use of perennial grasses, shrubs or trees. Of long duration. Often lead to a change in slope profile. Often aligned along the contour or against the prevailing wind direction. Often spaced according to slope.	Agroforestry, windbreaks, hedgerows, and live fences. Grass strips along the contour, and riparian buffers along riverbanks.
Structural	Of long duration and permanent. Involve major earth movements and/or construction with various materials. Often intended to control runoff, erosion and wind velocity. Often lead to a change in slope profile. Often aligned along the contour or against the prevailing wind direction. Often spaced according to slope.	Bench terraces (slope of terrace bed <6%) or forward-sloping terraces (slope of terrace bed >6%). Earth bunds, stone bunds and semicircular bunds. Check dams. Diversion/drainage ditches. Retention/infiltration ditches and microcatchments. Sand dune stabilization. Rotational grazing using fences
Management regime	Involve a fundamental change in land use. Usually involve no agronomic and structural measures. Often result in improved vegetation cover. Often reduce the intensity of use.	Land-use change (e.g. from cropland to grazing land, forest to agroforestry, afforestation, and bare land to crop). Exclosure by closing areas to grazing and agriculture. Change from herding to fencing, adjusting stocking rates, rotational grazing, and grazing to forage cutting (for stall feeding). Exclusion of natural waterways. Area closure/resting and protection.

◊ *Source*: Adapted from **WOCAT.** n.d. WOCAT. Classifications and definitions. In: *WOCAT*. Bern, Switzerland. [Cited 09 June 2022]. https://www.wocat.net/en/global-slm-database/slm-practices-technologies-and-approaches/classifications-technologies

Agronomic measures are the simplest to implement and several meta-analyses (Table 5.6) have documented substantial (50 to 90 percent) reductions in water erosion (Du *et al*., 2022; Jia *et al*., 2020; Mhazo, Chivenge and Chaplot, 2016; Xiong, Sun and Chen, 2018; Xiao *et al*., 2021a). Residue retention, cover crops and mulching all contribute to the covering of the soil surface and the reduction of water erosion, with cover crops providing the greatest protection (averaging about an 80 percent reduction in soil erosion) (Du *et al*., 2022; Liu *et al*., 2021). Contour tillage has a more restricted literature support base but was reported to reduce water erosion by approximately 50 percent in a meta-analysis of studies in China (Jia *et al*., 2020) (Table 5.6).

Erosion reductions for reduced- and no-till agriculture were more variable, ranging from substantial reductions to non-significant statistical differences compared to conventional tillage (Table 5.6). In

part, the variation is due to the differences in handling of residues between different implementations of no-till and reduced tillage. Xiao *et al.* (2021) reported that the reduction of sediment under no-till (55 to 17 percent reduction with a median of 26 percent) was lower than that under residue retention (98 to 7.4 percent reduction, with a median of 55.5 percent). The reduction in sediment under no-till plus residue retention was intermediate between the two.

Table 5.6. Compiled values for changes in water erosion from five meta-analyses grouped by agronomic, vegetative, structural and management regime

Practice	Change in water erosion (%)	Author
Agronomic		
No-till	42% reduction	Du *et al.* (2022)
Reduced tillage	75% reduction	
No-till	57% reduction	Xiong, Sun and Chen (2018)
Reduced tillage	n.s.	Xiong, Sun and Chen (2018)
No-till versus conventional Temperate maritime region	66% reduction	Mhazo, Chivenge and Chaplot (2016)
No-till versus conventional Temperate continental	69% reduction	Mhazo, Chivenge and Chaplot (2016)
No-till versus conventional Subtropical climate	n.s.	Mhazo, Chivenge and Chaplot (2016)
No-till versus conventional Tropical climate	n.s.	Mhazo, Chivenge and Chaplot (2016)
No-till without residue retention	26% reduction	Xiao *et al.* (2021)
Residue retention	55.5% reduction	Xiao *et al.* (2021)
Residue retention	64% reduction	Du *et al.* (2022)
Cover crop	92% reduction	Du *et al.* (2022)
Cover crop	72.7% reduction	Liu *et al.* (2021)
Mulching	54% reduction	Xiong, Sun and Chen (2018)
Mulching	51.2% reduction	Liu *et al.* (2021)
Deep tillage	72% reduction	Xiong, Sun and Chen (2018)
Contour tillage	49% reduction	Jian *et al.* 2020
Vegetative		
Agroforestry	86% reduction	Du *et al.* (2022)
Strip cropping	59% reduction	Xiong, Sun and Chen (2018)
Hedgerow	79% reduction	Xiong, Sun and Chen (2018)
Buffer strips	89% reduction	Xiong, Sun and Chen (2018)
Structural		
Terraces	88% reduction	Xiong, Sun and Chen (2018)
Contour bunds	85% reduction	Xiong, Sun and Chen (2018)
Management regime		
Cultivated conversion to grass cover	97% reduction	Xiong, Sun and Chen (2018)
Afforestation	97% reduction	Xiong, Sun and Chen (2018)
Steep cropland to non-crop	88% reduction	Xiong, Sun and Chen (2018)
Bare land to cultivated land	70% reduction	Xiong, Sun and Chen (2018)
Exclosure	62% reduction	Xiong, Sun and Chen (2018)

Note: n.s. = not significant.

Sources: See References.

Soil losses were lower under no-till compared to conventional tillage in sites in temperate maritime zones (80 percent reduction) and temperate continental climate zones (58 percent reduction) but that there were no significant differences between no-till and control plots in subtropical and tropical regions (Mhazo, Chivenge and Chaplot, 2016). The lack of effect of no-till on soil loss was supported by data on sediment concentration at the research sites in tropical climates, where the median difference between no-till and conventional tillage was close to zero. The results of Mhazo, Chivenge and Chaplot (2016) on regional differences may be confounded by the differences in residue management between temperate regions and subtropical and tropical regions, as residue removal for other purposes is more common in subtropical and tropical regions.

Vegetative measures also supported reductions in water erosion across the studies (Table 5.6). Agroforestry typically involves a number of specific erosion-reducing measures, such as cover crops and mulching, as well as the direct effect of the trees themselves. Measures implemented across the slope contour such as strip cropping and buffer strips were generally effective at reducing water erosion. For example, in sub-Saharan Africa, soil loss was 50 percent lower with grass strips than without (Wolka, Mulder and Biazin, 2018).

Structural measures such as terraces and contour bunds are also generally effective at reducing water erosion (Table 5.6). There are a range of vegetative measures available with varying levels of initial labour and upkeep. For example, in sub-Saharan Africa, trench-ridge systems such as *fanya juu* and soil bunds (described in Mekonnen *et al.*, 2017) have been implemented. Stone bunds were shown by Wolka, Mulder and Biazin (2018) to reduce soil loss by more than 60 percent on average.

The most common structural measure to fight water erosion on agricultural land is terracing. Terraces increase the surface area for cultivation on slopes and increase water retention while reducing soil erosion by water (Arnáez *et al.*, 2015; Wei *et al.*, 2016). The mean efficacy of terracing in controlling erosion was 11.5 times higher than the control plots without terracing, and terraces reduced erosion in 141 out of 154 available cases from the literature (Wei *et al.,* 2016).

Despite their proven efficacy in reducing erosion, there has been widespread abandonment of terraces in many developed countries, beginning in the early twentieth century. The main drivers of abandonment are the absence of labour (and a rural population generally), low agricultural prices and high maintenance costs, and the limited accessibility of some terraces (Wei *et al.*, 2016). The collapse of terraces after their abandonment is common, with significant collapse of stone walls and mass movement of soil in abandoned terrace systems. With heavy rainfall, shallow landslides can transition into debris flows that impact whole hillsides (Arnáez *et al.*, 2015). Poorly designed terraces are associated with fatality-inducing shallow landslides in urban areas (Ozturk *et al.*, 2022).

Changes in the overall management regime can lead to the greatest reductions in water erosion (Table 5.6). In many cases, measures are best implemented as part of an overall landscape approach to soil and water conservation (Shames, Heiner and Scherr, 2017). Landscape approaches target specific areas of land units (e.g. catchments or river basins that are at particular risk) for the implementation of measures such as the establishment of forest or grass cover, while implementing agronomic and vegetative measures on other, less vulnerable sites. The specific implemented measures differ across sites and hence landscape approaches cannot be readily synthesized using tools such as meta-analysis, but they are integral to many successful implementation programmes.

Reduced tillage is also a primary control on rates of tillage erosion. In general, tillage erosion rates increase linearly with tillage depth (as depth increases, more soil is subject to movement and downslope displacement) (Van Oost *et al.*, 2007). Generally, mouldboard ploughs (with a mean tillage depth of approximately 25 cm) cause more tillage erosion than chisel ploughs (with a mean tillage depth of approximately 15 cm), while secondary tillage operations such as the use of cultivators, harrows or discs are shallower and cause correspondingly lower tillage erosion rates. Reduced tillage and no-till may therefore greatly reduce rates of tillage erosion in tandem with reductions in the rate of sheet and rill erosion by water. However, that the speed and depth of

non-inversion tillage is important, as pointed out by Öttl *et al.* (2022), who showed that the use of chisel ploughs at high speed and at tillage depths of 25 cm increased tillage displacement relative to mouldboard ploughs.

Two general approaches are used to reduce erosion by gullying (Bartley *et al.*, 2020; Frankl *et al.*, 2021). The first approach involves reducing runoff in the catchment before it reaches the gully head (i.e. the point where incision begins). This can be accomplished by increasing the vegetation cover (as discussed previously), creating ponds or other water-retention measures to intercept runoff, or by terracing in the catchment above the gully head. The latter approach is to focus on management of the gully itself, typically through the installations of check dams throughout the gully. Check dams reduce the velocity of flow and cause sedimentation behind the dam, lowering the overall slope of the gully in the medium term. As reported by Bartley *et al.* (2020), hillslope flow diversion, runoff trapping, revegetation and check dams are often combined in China, reducing sediment yields from gullies by approximately 60 to 75 percent.

Erosion through crop harvest can be reduced if farmers remove adhering soil in the field rather than transporting it offsite (Panagos, Borrelli and Poesen, 2019). This also has the advantage of reducing subsequent processing charges for the produce and, hence, is a benefit for both farmers and subsequent processers. Panagos, Borrelli and Poesen (2019) cite examples of innovations in sugarbeet harvest that have reduced the soil transported by sugar beets by at least 50 percent.

5.3.3 Reducing nutrient mining

Nutrient mining occurs when the cumulative losses of nutrients from a site (through crop harvest, erosion and runoff losses, leaching and gaseous emissions) exceed the nutrient inputs from all sources. In some countries, surpluses of P have built up in the soil due to excess supply, and mining of these legacy P sources is a good practice (Zou, Zhang and Davidson, 2022). However, this section will focus instead on countries where no such surplus exists and nutrient mining is significantly limiting crop yields.

The countries that have experienced nutrient mining in the 2017–2021 period have been discussed in Section 4.3.2. The nutrient mining of N, P and K has been most prevalent in sub-Saharan Africa, with 33 of 48 countries experiencing mining of one or more of the elements (FAO, 2024a) (Figure 4.10, Figure 4.11 and Figure 4.12). This is also common in the Pacific Island countries, with 11 countries showing evidence of nutrient mining (FAO, 2024a) (Table 4.5).

The chronic lack of nutrients in many countries of sub-Saharan Africa leads to crop yields well below potential yields and crops that are less nutritious (i.e. lower in protein and basic nutrients). The need to increase synthetic fertilizer use is widely, though not universally, accepted. Vanlauwe *et al.* (2010) developed the concept of integrated soil fertility management (ISFM) to increase crop productivity in a profitable and environmentally friendly way. They defined ISFM as:

> *A set of soil fertility management practices that necessarily include the use of fertilizer, organic inputs, and improved germplasm combined with the knowledge on how to adapt these practices to local conditions, aiming at maximizing agronomic use efficiency of the applied nutrients and improving crop productivity* (Vanlauwe *et al.*, 2010, p. 19).

The authors additionally emphasized the need to address secondary limitations to increased yield, such as micronutrient deficiencies.

Organic inputs are already a major source of nutrients for farming systems in sub-Saharan Africa. As discussed in Section 5.2, estimates by Gu *et al.* (2023) based on FAOSTAT data show average N inputs in sub-Saharan Africa of 47 percent from biological N fixation, 24 percent from manure and

14 percent from synthetic N fertilizer. The average estimated N fertilizer use in 2015 from all sources was 12.2 kg/ha.

The fertility contributions of organic materials are also limited by other uses for organic residues and the low nutritional value of the residues. For example, in the study of farms in Kenya by Castellanos-Navarrete *et al.* (2015), the majority of crop residues were fed to cattle at the expense of soil fertility. The animal manures had low concentrations of N and P (reflecting the low N and P contents of animal feeds), and poor storage of manure also led to substantial losses of N (up to 62 percent) by NH_3 volatilization and leaching losses of up to 25 percent of P. The combined inputs of nutrients from both manure and crop residues were too small to sustain crop production and inputs of mineral fertilizers were required.

The addition of legumes to crop rotations has been widely promoted as a means of increasing N availability through biological N fixation. In their meta-analysis of the yield benefits of legumes, Zhao *et al.* (2022) found that the benefits on subsequent crop yields were substantial, with the introduction of legumes significantly increasing the main crop yield by 43 percent on average.

In some areas of Africa, P deficiencies limit the efficacy of biological N fixation as a N source. Plants dependent on symbiotic N fixation from the atmosphere have a high P requirement for the formation of the nodules where N fixation occurs (Graham and Vance, 2003) and N fixation will be limited by low P availability.

A number of factors act to limit the feasibility of substantial increases in synthetic fertilizers in sub-Saharan Africa. The cost of synthetic fertilizer is higher there than in other regions (Davidson *et al*. 2016). Within the African continent, there are substantial differences between countries (Bonilla-Cedrez *et al*., 2021). The ratio of N fertilizer to maize price (using fertilizer and grain price data from 2000 to 2018) created very unfavourable conditions for fertilizer use in Mali, Mozambique, Sierra Leone, Togo, and Uganda, and very good conditions for use in Ghana, eastern South Africa and western Kenya (Bonilla-Cedrez *et al*., 2021). Using a separate analysis based on modelling site-specific internal rate of returns on fertilizer use in maize, McCullough, Quinn and Simons (2022) found that fertilizer use was unlikely to be profitable in Eritrea, Ethiopia, Malawi, Mozambique, the Sudan and Zimbabwe. Clearly, uncertainty exists about the economic returns from fertilizer use in the region.

Of particular concern in African soils is the relatively high percentage of sites that are either non-responsive or have very low response to added nutrients (Sileshi *et al*., 2022). A summary of studies in Africa by these authors showed that the percentage of non-responsive soils was both widespread and severe in some countries, with between 5 and 25 percent of soils being non-responsive and a similar percentage possibly showing low yield responses.

Microdosing of synthetic and organic fertilizer can partially address the constraints surrounding fertilizer inputs. Microdosing administers small amounts of the nutrient source at the right time and close to the seedlings and is widely considered an entry point to sustainable intensification of agricultural systems in dry areas (Aune, Coulibaly and Giller, 2017; Kuyah *et al*., 2021). Kuyah *et al*. (2021) found that microdosing led to increases in crop yield in all 22 studies they considered and to increases in soil quality in two studies, no effect in one study and a decrease in soil quality in two studies. They cited a meta-analysis that found that microdosing increased yields by 47 percent compared to studies with no nutrient inputs. The practice originated in the Sahel before spreading to Southern Africa (Kuyah *et al*., 2021).

However, a major constraint is that manual microdosing is labour intensive and time consuming. It may also lead to nutrient mining of P and K, although this mining can be reduced if organic manures are added in addition to synthetic fertilizers (Kuyah *et al*., 2021).

5.3.4 Reducing acidification

Acid soils are defined by Von Uexküll and Mutert (1995) as having a pH less than 5.5 when measured in water. These soils are widespread, with an estimated 30 percent of the ice-free land area having acid soils (Section 3.5.5). The great majority are naturally acidic soils but the acidification of soil by human activities is increasing. In the 2015 SWSR report (FAO and ITPS, 2015), soil acidification was identified as the number one soil threat in the Southwest Pacific (predominantly in Australia), with approximately 70 percent (11 million ha) of southwestern Australia being moderately to strongly affected by acidity (Petersen, 2015).

The main causes of human-induced acidification are the addition of fertilizers with ammonium or sulphate, the loss of base cations through crop harvesting or accelerated leaching, N_2 fixation by legumes and acid deposition from the atmosphere (Fageria and Nascente, 2014; Agegnehu *et al.*, 2021; Kunhikrishnan *et al.*, 2016) (see Section 3.5.3).

Liming (i.e. the addition of Ca and Mg-rich materials in various forms) is a time-tested approach to correct acidity issues and obtain optimum crop yields. The direct effects of liming include decreased acidity (i.e. an increase in soil pH), increased Ca, and (if dolomitic limestone is used) increased Mg (Fageria and Nascente, 2014). As the pH increases, several indirect effects occur, including increased availability of P and molybdenum (Mo), increases in symbiotic N_2 fixation by legumes, N uptake efficiency and improvements in root growth of annual crops, which increases absorption of water and nutrients (Fageria and Nascente, 2014; Agegnehu *et al.*, 2021).

Two recent meta-analyses documented the major yield increases that occurred in acid soils when $CaCO_3$ was added. Li *et al.* (2019) reported that across all crops, liming significantly increased crop yields, with the largest effect size (66.5 percent) under dolomite, while the smallest effect size (13.2 percent) occurred under burnt $CaCO_3$. Significant yield increases occurred for maize, wheat, barley (*Hordeum vulgare*), rapeseed (*Brassica napus*), beans, forage legumes and grasses, significant but lower increases observed for groundnut, rice and soybean, and no significant differences occurred for sorghum (*Sorghum bicolor*), tubers and tobacco (*Nicotiana tabacum*). Enesi *et al.* (2023) reported generally similar crop responses, but oat (*Avena sativa*) did not show a significant response.

The differences among crop response to acidity have been well known for decades (Fageria and Nascente, 2014; Agegnehu *et al.*, 2021), with many economically significant plant species being regarded as being tolerant to acid soil conditions. Agegnehu *et al.* (2021) showed that crops such as upland rice, cassava (*Manihot esculenta*), mango (*Mangifera indica*), citrus (*Citrus*), pineapple (*Ananas comosus*), sugar cane (*Saccharum officinarum),* cowpea (Vigna unguiculata), tubers and plantain (Musa) were suitable for acid soils. Many acid-tolerant species have their origin in tropical and subtropical regions, suggesting that adaptation to soil constraints is a part of the evolutionary process (Fageria and Nascente, 2014; Agegnehu *et al.*, 2021). The planting of acid-tolerant crops (especially soybean) was a factor in the increase in agricultural production in the Cerrado region of Brazil (Fageria and Nascente, 2014).

The profitability of $CaCO_3$ additions has also been well established in the literature, although the strongest analyses are at national or subnational scales. The costs of field applications of $CaCO_3$ are determined especially by the transportation cost as well as the source. Large amounts of $CaCO_3$ must be added to significantly offset soil acidity. For example, at Rothamsted and Woburn (two sites studied in the United Kingdom), Holland and Behrendt (2021) cited applications rates of 9 to 15 t/ha of $CaCO_3$ (as crushed limestone) for the low-rate treatment and 45.5 to 52.5 t/ha for the high-rate treatment over the course of the treatment.

The two sites were studied over a 35-year period (Holland and Behrendt, 2021). The difference in long-term economic performance between the highest returning treatment, the high $CaCO_3$

treatment, and the control (no $CaCO_3$) was around GBP 436 per ha annually at Rothamsted and GBP 208 per ha annually at Woburn. The major factor determining profitability was the price of the crop, as the range of total liming costs had little effect. For the acid soils of southwestern Australia, Petersen (2015) showed that the loss of production due to subsoil acidity averaged AUD 141 per ha annually, totalling AUD 1.6 billion annually nationally. Liming at appropriate rates increased the annual profit by AUD 63 per ha, with AUD 11 returned for every dollar invested. For Ethiopian soils, Warner *et al.* (2023) estimated that the impact of moving pH from 5.5 to 6.5 through a $CaCO_3$ remediation strategy could increase yields by 22 percent for wheat and 19 percent for barley. The yield increases were substantially higher than the benefit achieved through a fertilizer enhancement project.

Warner *et al.* (2023) highlighted impediments in some areas to the adoption of a liming strategy, including the large application requirements (and hence high transport costs), a lack of farmer awareness and weak or non-existent $CaCO_3$ supply chains. The effects of $CaCO_3$ additions often take time to become evident to farmers and this limits adoption, even where research has clearly demonstrated a benefit and where decision support systems for calculating appropriate rates are in place.

5.3.5 Reducing salt accumulation

Numerous practices to reduce accumulation of salts in soils have been developed. The *Global Status of Salt-affected Soils* report (FAO, 2024b) included mitigation strategies that were aimed at the reduction of salinity or sodicity levels in the root zone and adaptation strategies that were aimed at coping with existing salinity levels (Table 5.7).

Table 5.7. Summary of mitigation and adaptation methods for salt-affected soils

	Physical
Mulches	Mulching of the surface reduces the evaporation of water from the soil, preventing the upward movement of salts, and inhibits the accumulation of salts on the soil surface.
Interlayers from loose materials	Adding a layer of any material (such as sand, pebbles or straw) in the subsoil creates an effective barrier to capillary movement and prevents the upward movement of salts from the subsoil and groundwater.
Leaching	Washing out material from the soil, both in solution and suspension reduces the salt content of soil by adding water to salt-affected land and washes salt from the soil into the groundwater. Leaching is effective on soils with good internal drainage due to a favourable structure or coarse texture.
Drainage	Drainage is the natural or artificial removal of surplus surface water and groundwater and dissolved salts from any area to enhance crop growth. In an artificial system, surplus ground or surface water is removed by means of natural or artificial subsurface or surface conduits.
Surface scraping	The removal of the topsoil by scraping it off and removing any spots of efflorescence on the surface is a temporary cure rather than a solution, and salinity may develop again if other methods are also not followed simultaneously.
Compost and plant residue incorporation	Organic matter amendment can effectively reduce salinity and sodicity obstacles by improving soil structure, water-holding capacity and fertility level.
Biochar	Biochar is a specific organic soil amendment that can improve the physical, chemical and biological properties of salt-affected soils.
Low rank coal	Low-rank coal is high in lignin and low in cellulose and hemicellulose. It is typically derived from sources such as peat, lignite, or brown coal. It has the potential to enhance soil composition, improve the soil's ability to retain water, and increase the availability of nutrients.
Deep tillage	Ameliorative deep tillage methods for sodic soils include deep ploughing, when the topsoil and subsoil are reversed, deep mixing when the topsoil and soil are well mixed and subsoiling when the field is partially tilled and the soil is generally not reversed.

	Physical
Land shaping	Land shaping practices such as farm pond, deep furrow and high ridge, and paddy-cum-fish cultivation have been developed in India for the restoration and productivity enhancement of degraded (saline) coastal land.
Land levelling	Land levelling causes the water obtained from precipitation and irrigation to infiltrate evenly, enhancing the effect of leaching salts, and eliminating the local accumulation of salts.
	Chemical
Gypsum and other calcium-containing amendments	Chemical amelioration for sodic soils is used by applying chemical amendments that help to reduce soil sodicity, improve soil structure and promote the growth of plant roots.
	Biological
Adjustment of planting time and place	Many crops are not able to tolerate salinity in their germination and early seedling period, but can tolerate salinity later, during the vegetative, reproductive, and grain-filling stages.
Inclusion of salt-tolerant crops	Crops such as barley (*Hordeum vulgare*), rapeseed (*Brassica napus*), millet (*Panicum* sp.), and sugar beet (*Beta vulgaris*) are salt-tolerant.
Agroforestry	Plant species from salt shrubs to large trees are suitable for growing on salt-affected soils. Several tree species can thrive and grow in salty and waterlogged soils, and these species can reclaim saline and waterlogged areas.
Bioremediation and phytoremediation	Bioremediation can remove salts from topsoil in saline soils or improve permeability of sodic soils. Bioremediation is an environmentally- and economically-beneficial measure among biological methods for the conservation and improvement of soil fertility.
	Adaptation
Development of salt-tolerant crops	Breeding of salt-tolerant crops and the domestication of halophytes and other non-conventional plants enable the continued use of salt-affected soils for farming.
Halopriming	Halopriming is a seed priming technology where seeds are soaked in aerated inorganic salt solutions under controlled temperature and moisture conditions. It promotes the seeds emergence rate for healthy seedling vigour.

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

Salinity and sodicity require different management approaches. The salts that cause salinity are generally highly mobile and can be removed from the soil profile by several methods. Sodicity causes the degradation of soil structure and requires both physical disruption to change the structure of the soil and often the addition of chemical amendments to remove the harmful Na ions from the soil profile (Vargas *et al.* 2018; Stavi, Thevs and Priori, 2021; Cuevas *et al.*, 2019).

Flushing of salts from the rooting zone by surface application of water and removal of the flushing water through a drainage system can lower the salinity to the point where saline-tolerant crops can be grown (Vargas *et al.,* 2018). Cuevas *et al.* (2019) found that flushing was effective at reducing salinity by an average of 39 percent, increasing yield by an average of 24 percent across five studies. However, the ultimate disposition of the water used for flushing is a concern and leads to questions about the long-term viability and environmental costs of this method (Stavi, Thevs and Priori, 2021).

Irrigation without the implementation of drainage to remove excess water from the soil profile is a common cause of salinization. Ritzema (2016) reported that only about 22 percent of irrigated lands were drained and that salinity and related waterlogging issues affected about 10 to 16 percent of irrigated land globally. However, controlled drainage is an effective method that can be employed to manage these issues. It is a practice that manages within-soil flow and water table depth by using shallow drainage conduits and control structures in drainage ditches or sumps (Hopmans *et al.*, 2021).

Phytoremediation uses salt-accumulating plants to take up salts from the soil solution and concentrate the salts in their tissues, with the plant material then being removed from the site (Litalien and Zeeb, 2020). However, Cuevas *et al.* (2019) did not find a significant reduction in salinity or increased yield across 17 phytoremediation studies, and the disposal of the salt-laden plant

residues also presented a further issue (Qadir *et al.*, 2014). Litalien and Zeeb (2020) suggested that the removed plant material could be used as fodder or compost or in the production of biofuel.

Remediation of sodicity in soils requires ameliorative deep ploughing to disrupt the unfavourable soil structural conditions in the rooting zone and to mix $CaCO_3$ into the upper soil material (Vargas *et al.* 2018). Deep ploughing is often coupled with addition of gypsum ($CaSO_4 \cdot 2H_2O$) which releases Ca^{2+} that displaces Na^+ from the charged soil particles. In some cases, flushing water needs to be added to remove the displaced Na from the rooting zone. Results from Ukraine indicated that the beneficial effects of a one-time deep ploughing plus dolomite treatment persisted for 40 to 50 years but the effects have been more ephemeral in other areas (Vargas *et al.,* 2018).

Soil tillage and the subsequent loss of soil cover can be a risk in sodic soils with shallow groundwater containing high levels of soluble salts, such as the case of the Inland Pampas of Argentina (Taboada *et al*. 2021). There are rehabilitation technologies available which consider the influence of groundwater as a source of salts and Na^+ on the soil surface, such as the provision or retention of surface vegetation to avoid the arrival of salts to the topsoil. In these kind of sodic soils, other suitable technologies include the application of organic products, biological improvement through planting of adapted species, grazing systems, gypsum application, and direct sowing.

The economic benefits of the various sodicity treatments have not been widely examined. Qadir *et al.* (2014) summarized various case studies but stated that the economic estimates were variable due to the different practices used, differences in site conditions and the general economic context.

5.3.6 Reducing loss of agricultural land to urban expansion

The loss of agricultural land to urban expansion is considered to be one of the most irreversible forms of soil degradation (Gardi, Florczyk and Scalenghe, 2021; Peake and Robb, 2021). There are considerable changes in ecosystem services (especially the provisioning service of food production) associated with conversion of agricultural land to urban uses (see Section 2.2.6 and Section 3.8). Given the projections for increased urban population growth throughout the twenty-first century (see Section 4.2.5) the pressures on agricultural land from urbanization will only increase through time.

Peake and Robb (2021) outlined three categories of measures to control land conversion to urban use:

1. regulations and laws to prohibit or discourage specific actions that lead to conversion (e.g. construction and soil sealing);
2. restrictions on development involving conversion of use; and
3. generic incentives to preserve land or enhance its ecosystem services.

The second category (restrictions on development) are most widely applied and may involve laws based on zoning to protect specific areas (e.g. green belts around cities or nature reserves) or more general laws to protect prime farmland.

Many protective measures are implemented at the local, municipal or subnational levels (FAO and UNEP, 2020). Farmland protection involves two key components: a process to evaluate and prioritize specific areas of land (i.e. prime farmland) and the implementation of the required legal instruments and enforcement structures (Peake and Robb, 2021). In a survey of nearly 200 countries (using FAOLEX, FAO's online database of policies and laws), Peake and Robb (2021) reported that while 24 countries had policies advocating avoidance of land conversion to urban use, and 21 countries had laws prohibiting or strongly restricting loss of all prime farmland, only 9 had specific targets. Naumann *et al.* (2018) examined the European context for policies that restricted land conversion to

urban use and associated soil sealing and noted that, while countries such as Germany had strong objectives for limiting land take, such targets were not necessarily embedded in municipal policies, where land-use decisions were made.

As discussed in Section 2.2.7 and Section 3.8, the greatest loss of soil functions is associated with sealed soil surfaces. Decoville and Feltgren (2023) showed that the proportion of sealed surfaces varied between European cities and that it was possible to identify areas where suboptimal land-use decisions had led to an anomalously high proportion of imperviousness. Features such sealed car parks or disused industrial land could be targeted for partial or complete unsealing (Pannicke-Prochnow and Albrecht, 2024).

The trade-offs between urbanization and its effects on soil functions and ecosystem services are complex:

> *However, it is not sufficient to define a comprehensive strategy for land preservation. Indeed, the fight against land take cannot be isolated from other crucial issues to which our societies are exposed, such as the need for new housing, jobs, and public facilities. Anti-land take strategies must therefore ensure that when land take occurs, the use of the newly converted space is efficient, that it provides real added value for society, and that it is located in areas that are suitable for it. The systematic reduction, wherever possible, of new forms of non-essential soil sealing thus appears to offer much more potential for a sustainable, fair, and livable answer to the "no net land take" objective than a systematic intensification of the use of free spaces within urban areas.* (Decoville and Feltgren, 2023, p. 7).



5.4 Water security and water resources (SDG 6)

5.4.1 Introduction

Three specific targets of SDG 6: Clean Water and Sanitation are of relevance in this section: Target 6.3 (improve water quality, wastewater treatment and safe reuse), Target 6.4 (increase water-use efficiency and ensure fresh water supplies) and Target 6.6 (protect and restore water-related ecosystems). Sustainable soil management practices can reduce erosion-induced sedimentation of waterways and agrochemical pollution of waterways and increase water-use efficiency in agricultural systems, all of which directly contributed to the soil-related targets for SDG 6. Preservation and restoration of wetlands is a further contributor to Target 6.6 and is examined in Section 5.8 of this chapter.

5.4.2 Reducing runoff and sedimentation

The measures to reduce runoff are for the most part similar to those used to reduce water erosion (since runoff is responsible for detachment and transport of soil during storm events). As with water erosion, measures that keep a cover of living or dead plant material on the soil surface can be very effective in reducing runoff (Table 5.8). Of the agronomic measures, residue retention, mulching and especially cover crops lead to substantial decreases in runoff, ranging from 25 to 81 percent. The higher ground cover associated with agroforestry also substantially decreases runoff (58 percent reduction), as well as changes in the management regime that increase ground cover, such as the conversion of cropland to grass, afforestation, exclosures (areas where livestock are excluded), or even conversion of bare land to cultivated land.

Table 5.8. Compiled values for changes in runoff from seven meta-analyses grouped by agronomic, vegetative, structural and management regime

Practice	Change in runoff (%)	Author
Agronomic		
No-till versus reduced tillage	21.9% reduction	Sun *et al.* (2015)
No-till versus mouldboard tillage	27.2% reduction	
No-till with residue retention versus reduced tillage	34% reduction	
No-till with no residue retention versus reduced tillage	n.s.	
No-till	30% reduction	Du *et al.* (2022)
Reduced tillage	64% reduction	
No-till	n.s.	Xiong, Sun and Chen (2018)
Reduced tillage	n.s.	
No-till versus conventional Temperate maritime region	39% reduction	Mhazo, Chivenge and Chaplot (2016)
No-till versus conventional Temperate continental	27% reduction	
No-till versus conventional (subtropical climate)	141% increase	
No-till versus conventional (tropical climate)	25% increase	
No-till without residue retention	10% reduction	Xiao *et al.* (2021)
Residue retention	45.5% reduction	
Residue retention	39% reduction	Du *et al.* (2022)
Cover crop	81% reduction	

Practice	Change in runoff (%)	Author
Cover crop	50% reduction	Liu *et al.* (2021)
Mulching	25% reduction	Xiong, Sun and Chen (2018)
Mulching	39% reduction	Liu *et al.* (2021)
Deep tillage	48% reduction	Xiong, Sun and Chen (2018)
Contour tillage	36% reduction	Jian *et al.* 2020
Vegetative		
Strip cropping	n.s.	Xiong, Sun and Chen (2018)
Hedgerow	n.s.	Xiong, Sun and Chen (2018)
Buffer strips	73% reduction	Xiong, Sun and Chen (2018)
Exclosure	39% reduction	Xiong, Sun and Chen (2018)
Structural		
Terraces	47% reduction	Xiong, Sun and Chen (2018)
Contour bunds	36% reduction	Xiong, Sun and Chen (2018)
Management regime		
Agroforestry	58% reduction	Du *et al.* (2022)
Cultivated conversion to grass cover	74% reduction	Xiong, Sun and Chen (2018)
Afforestation	56% reduction	Xiong, Sun and Chen (2018)
Steep cropland to non-crop	53% reduction	Xiong, Sun and Chen (2018)
Bare land to cultivated land	47% reduction	Xiong, Sun and Chen (2018)
Exclosure	39% reduction	Xiong, Sun and Chen (2018)

Note: n.s. indicates a non-significant effect.

◊ *Sources:* **Du, X., Jian, J., Du, C. & Stewart, R.D.** 2022. Conservation management decreases surface runoff and soil erosion. *International Soil and Water Conservation Research*, 10(2): 188–196. https://doi.org/10.1016/j.iswcr.2021.08.001

◊ **Xiong, M., Sun, R. & Chen, L.** 2018. Effects of soil conservation techniques on water erosion control: A global analysis. *Science of The Total Environment*, 645: 753–760. https://doi.org/10.1016/j.scitotenv.2018.07.124

◊ **Mhazo, N., Chivenge, P. & Chaplot, V.** 2016. Tillage impact on soil erosion by water: Discrepancies due to climate and soil characteristics. *Agriculture, Ecosystems & Environment*, 230: 231–241. https://doi.org/10.1016/j.agee.2016.04.033

◊ **Xiao, L., Kuhn, N.J., Zhao, R. & Cao, L.** 2021. Net effects of conservation agriculture principles on sustainable land use: A synthesis. *Global Change Biology*, 27(24): 6321–6330. https://doi.org/10.1111/gcb.15906

◊ **Jian, J., Lester, B.J., Du, X., Reiter, M.S. & Stewart, R.D.** 2020. A calculator to quantify cover crop effects on soil health and productivity. *Soil and Tillage Research*, 199: 104575. https://doi.org/10.1016/j.still.2020.104575

The effect of no-till or reduced tillage on runoff is complex (Table 5.8). Some studies have shown no significant effect of changes to tillage, but two factors may be responsible for the substantial variation across studies. First, Mhazo, Chivenge and Chaplot (2016) found differences across climate regions, with substantial reduction in runoff in temperate regions under no-till, but substantial increases in runoff in subtropical and tropical climates (Table 5.8). Second, Xiao *et al.* (2021) reported no-till without residue retention had only a small reduction in runoff associated with it, whereas no-till with residue retention or residue retention alone had substantial benefits in runoff reduction. Mhazo, Chivenge and Chaplot (2016) also reported that the presence of a surface mulch was critical in the implementation of no-till, finding that the runoff coefficient was 41 percent lower under no-till than conventional till in plots with mulch cover, whereas there were no significant differences between no-till and conventional tillage on bare soils. While the interaction between different climate zones and residue retention was unclear in their study, tropical and subtropical regions such as Argentina and Brazil have successfully and widely adopted no-till and residue retention practices.

Structural measures such as terraces and contour bunds have also been shown to substantially reduce runoff across many studies (Table 5.8). Other small-scale structural measures for rainwater harvesting are widely used in Africa to reduce runoff and increase water storage (Olsson *et al.*, 2019) (see Section 5.4.2).

Soil sealing in urban areas reduces both infiltration (and hence increases runoff) and surface roughness, which increases the speed of runoff and the risk of flooding (Decoville and Feltgren, 2023). Sound urban planning should aim to reduce the percentage of sealed soil to a minimum and to use pervious materials as much as possible instead of impermeable material in both core urban areas and peri-urban areas surrounding cities (Naumann *et al.*, 2018). Unsealing (i.e. the complete or partial removal of the physical barrier or covering layers that seal the soil) can lead to the restoration of a fuller range of ecological services in the unsealed surface (Pannicke-Prochnow and Albrecht, 2024). City planning that incorporates a range of green infrastructure can also substantially reduce runoff and delay and flatten peak flows (Minixhofer and Stangl, 2021). Green infrastructure includes measures such as parks, forests, gardens, green water stormwater infrastructure, low-impact development, urban drainage systems, bioswales, rain gardens, permeable surfaces, green roofs, and walls.

Sedimentation is the deposition of eroded soil after it has been transported from its point of detachment and can occur within the source field or offsite at the edge of the field or beyond. Accelerated rates of sedimentation into waterbodies have harmful consequences, and therefore, trapping sediment (and the agrochemicals associated with the sediment) before it reaches waterbodies is a major goal of watershed management.

Management practices designed to trap sediments include both vegetative and structural measures (Mekonnen *et al.*, 2015). Vegetative measures that reduce runoff velocity and sediment transport and enhance sediment deposition include ensuring that grass strips, shrub and tree buffers, riparian vegetation and grassed waterways are in place along contours, at the edge of fields or along streams or other waterbodies.

Structural measures can be designed to intercept runoff, reduce sediment transport and trap sediments either from surface runoff or channel flow. Examples of such measures include trench ridge structures such as *fanya juu* and soil bunds (Mekonnen *et al.*, 2017), terraces, check dams, dams, basins, and ponds. Trench-ridge systems and terraces are normally constructed onsite and check dams, dams, basins and ponds are constructed to intercept runoff and sediment offsite. Offsite measures such as check dams are built from stone, sandbags, wood or gabions in the drainage channel, whereas ponds or basins are constructed using dikes or stone barriers.

Experimental studies of individual catchments (e.g. Mekonnen *et al.*, 2017) and critical reviews across multiple studies (e.g. Mekonnen *et al.*, 2015) have shown that appropriate measures typically trap approximately 50 to 75 percent of sediment. The percentage is highest when integrated sediment trapping measures are strategically placed throughout the watershed. Measures such as grass or riparian buffers are also highly effective at trapping and retaining nutrients such as N and P transported in particulate form (Nummer *et al.*, 2018; Gu *et al.*, 2017) as well as pesticides (Chen, Grieneisen and Zhang, 2016), which reduce the loading in surface waters.

5.4.3 Increasing water-use efficiency in agriculture

Two water management issues are covered in this section: moisture scarcity and waterlogging. The management of saline water in irrigation is discussed in Section 5.3.5.

Reducing the overall water footprint of agriculture involves both soil and crop management practices (Table 5.9), with an initial stage being to increase soil water availability. Heitman, Kool and Carvalho (2023) stated that for dryland agriculture, three types of interventions could be effective: maximizing rainfall capture, maximizing soil water storage capacity and suppressing unproductive evaporative water losses.

Table 5.9. Main soil-related technology and practices to reduce the water footprint of crop production

Strategies	Technology and practices
Increasing yield	Soil nutrient management (optimizing crop rotation, residue retention and reuse, erosion control, appropriate tillage, proper application and timing of manure or artificial fertilizer). Precision irrigation (synchronizing water application with crop water demand). Weed and pest control through crop rotation, proper tillage and biological pest control.
Reducing non-beneficial evapotranspiration	Crop scheduling to reduce evaporation during the fallow period. Plant spacing and row orientation. Minimum tillage to reduce soil water evaporation. Improved irrigation techniques (drip and subsurface irrigation). Effective control of weeds to reduce transpiration losses.
Enhancing effective use of rainfall	Water harvesting and supplemental irrigation.

◊ *Source*: **Mekonnen, M.M. & Hoekstra, A.Y.** 2014. Water footprint benchmarks for crop production: A first global assessment. *Ecological Indicators*, 46: 214–223. https://doi.org/10.1016/j.ecolind.2014.06.013

Many authors stress the need to maintain or improve soil structure at the surface and within the soil profile to ensure the highest possible infiltration of precipitation into the soil. The management-driven soil properties that control infiltration into the soil are macroporosity and pore continuity, plant and mulch cover, stability of surface soil aggregates, and the roughness of the soil surface (Bodner, Nakhforoosh and Kaul, 2015).

The techniques discussed in Section 5.4.1 on reducing runoff are all relevant for maximizing rainfall capture. An important additional method for increasing rainfall capture is microcatchment rainwater harvesting, which has been widely adopted in sub-Saharan Africa (Biazin *et al.*, 2012; Mekonnen, 2021; Jägermeyr *et al.*, 2016). The specific methods differ across subregions in sub-Saharan Africa (Table 5.10), but all are designed to capture runoff and allow greater infiltration of the intercepted runoff into the soil. Biazin *et al.* (2012) reported that a number of studies showed improvements by up to 30 percent of soil water content in the rooting zone. Such measures also intercept sediment in the runoff and hence reduce offsite sedimentation problems.

Table 5.10. Descriptions of microcatchment and in-field water harvesting methods used in sub-Saharan Africa

Type of rainwater harvesting systems	Description	Countries or subregions of wider application
	Microcatchments	
Pitting (Zai pits, Ngoro pits, trenches, tassa pits, etc.)	Zai pits: A grid of planting pits is dug in less permeable or rock-hard plots (organic matter may be added to the bottom of the pits). Ngoro pits: A series of regular traditional pits are dug (1.5 m^2 by 0.1–0.5 m deep) with the crops grown on the ridges around the pits. Trenches: Pits are made along the contour, sometimes with a bund downslope.	West Africa (Burkina Faso, Mali, Niger) East Africa (Ethiopia Kenya, Somalia, the United Republic of Tanzania, and Uganda) and Southern Africa (South Africa and Zimbabwe)
Contouring (stone/soil bunds, hedgerows and vegetation barriers)	Stone and soil bunds: A stone or sometimes an earthen bank of 0.50–0.75 cm height is piled on a foundation along the contour in a cultivated hillslope and may be stabilized with grasses or other fodder plant species. Hedgerows: Strips of land are marked out on the contour and left unploughed in order to form permanent, cross-slope barriers of naturally established grasses and herbs or shrubs are planted along the contour.	East Africa (Ethiopia, Kenya and the United Republic of Tanzania), West Africa (Burkina Faso) and South Africa

Type of rainwater harvesting systems	Description	Countries or subregions of wider application
Terracing (fanya juu are semicircular and hillside terraces)	Bunds are built in association with a ditch, either along the contour or on a gentle lateral gradient and are constructed in different forms. In fanya juu terraces, the embankment is in the upslope position.	East Africa (Ethiopia, Kenya and the United Republic of Tanzania)
Microbasins (negarims are usually diamond-, half moon- or eyebrow-shaped)	Different shapes of small basins, surrounded by low earth bunds formed to enable the runoff to infiltrate at the lowest point, where the plants are grown.	East Africa (Ethiopia, Kenya, the United Republic of Tanzania, and Uganda) West Africa (Burkina Faso, Mali and Niger)
	***In situ* rainwater harvesting.**	
Ridging	Basins wider than the traditional furrows are created either by manual hoeing or during tillage using a modified ploughing instrument.	Many parts pf sub-Saharan Africa
Mulching	The use of both crop residues and material from non-cultivated areas, including stones, are aimed at covering the soil.	Western and Eastern Africa
Furrowing and pot-hoeing	Different furrowing techniques are used before and after planting to conserve soil moisture in areas where oxen ploughing and hand-hoeing practices are common. In the Sahel, small shallow holes are dug manually at correct intervals, and the seeds are covered with soil. Two weeks after the emergence of the crop fertilizer is added about 10 cm from the plant	Eastern and Western Africa
Conservation tillage	Conservation tillage encompasses a wide range of tillage techniques ranging from non-inversion ploughing and reduced tillage to ripping and subsoiling.	Many parts of sub-Saharan Africa (Ethiopia, Kenya, South Africa and the United Republic of Tanzania)

◊ *Source:* **Biazin, B., Sterk, G., Temesgen, M., Abdulkedir, A. & Stroosnijder, L.** 2012. Rainwater harvesting and management in rainfed agricultural systems in sub-Saharan Africa – A review. *Physics and Chemistry of the Earth, Parts A/B/C*, 47–48: 139–151. https://doi.org/10.1016/j.pce.2011.08.015

Reducing unproductive evaporative losses from soil is best accomplished by providing a surface cover on croplands throughout the periods when growing crops cannot protect the soil surface. Plants or plant residues are the most practical and inexpensive way to provide surface cover, although cover crops may reduce soil moisture in drier regions to levels that reduce crop growth (see Section 5.2.2). Non-crop mulches also reduce evaporation, but concerns about the use of plastics and the cost of non-crop mulches limit their use in many areas (Bodner, Nakhforoosh and Kaul, 2015).

Waterlogging is an issue of excess water rather than water scarcity and occurs when all the soil pore space is saturated with water. Flooding (i.e. the submergence of plants by floodwaters) and soil waterlogging is projected to increase in future under simulated climate changes. Liu *et al.* (2023) projected that the 3 to 11 percent yield loss caused by waterlogging historically would rise to between 10 and 20 percent by 2080.

The major structural soil management approach to waterlogging is the implementation of soil drainage through surface drainageways, subsurface drainage conduits or a combination of both. Implementation of drainage with irrigation programmes is essential to reduce waterlogging, although Ritzema (2016) reported that only 22 percent of irrigated lands were drained. Raised beds are also useful for crops in areas prone to waterlogging (Kaur *et al.*, 2020).

An important agronomic measure to reduce waterlogging is the use of cover crops, which can decrease waterlogging by improving soil structure, reducing compaction, increasing the water infiltration rates (Kaur *et al.*, 2020) and increasing water uptake. Altering sowing times and adoption of waterlogging-tolerant genotypes can decrease the yield penalty due to waterlogging (Liu *et al.*, 2023).

5.4.4 Reducing agrochemical pollution of water

The widespread pollution of surface waters and ground water by agrochemicals (especially N and P) has harmful effects on both the ecosystem and human health (see Section 3.5.1). Research since 2015 has established that there are several SSM practices that can significantly reduce the risk of water pollution from agrochemicals.

Nitrogen pollution of groundwater and surface waters occurs primarily from the leaching of N from the soil profile into the groundwater and from loss of N in surface runoff. The meta-analysis of Gu *et al.* (2023) showed that significant reductions in N leaching could be achieved through the use of EEFs, the application of organic amendments such as manure and biochar with of the right rate of fertilizer application, the adoption of no-till, the inclusion of legumes in the crop rotation, and deployment of drip or reduced amounts of irrigation water (Table 5.6). Although Gu *et al.* (2023) did not find a significant effect of fertilizer type, an earlier meta-analysis by Wang *et al.* (2019) found lower emissions factors (i.e. the fraction of added N lost) for urea and controlled-release fertilizer relative to ammonium nitrate. Cover crops also significantly reduced N leaching for leguminous, non-leguminous and mixed cover crops (Abdalla *et al.,* 2019).

Most of the measures that reduce N leaching are also associated with reductions in N runoff, with significant reductions associated with EEFs, organic amendments and the right rate of fertilizer inputs (Table 5.6) (Gu *et al.*, 2023). The use of in-field and riparian buffers has offsetting effects, with N leaching increasing on average by 16 percent, while N runoff is decreased by 57 percent (Gu *et al.*, 2023). In this case, the retention of N in the buffers allows a greater period of time for leaching to occur. Cover crops also cause significant decreases in overall runoff from sites of around 50 percent (Jian *et al.*, 2020) (see Section 5.2.1).

Agriculture is a major source of P inputs in waterways. The major pathway for P inputs to reach waterways is from soil erosion by water, where the P is associated with mineral and organic soil particles (Alewell *et al.* (2020). The measures discussed in Section 5.2.1 to reduce soil erosion are critical to reduce P losses from this pathway. As well as the in-field measures discussed in Section 5.2.1, Singh *et al.* (2020) identified measures such as constructed wetlands and the conservation of existing wetlands as important sinks for P that had been lost from agricultural fields.

P can also be lost in a dissolved form though drainage systems associated with artificial drainage of cropland (Sims, Simard and Joern, 1998). Artificial drainage is regionally important in areas such as central Canada and the midwestern United States, where the majority of streamflow in many agricultural watersheds is from tile drains (King *et al.* 2015). Measures that mitigate the harmful effects of drainage waters include controlled drainage through water management, drainage water recycling, saturated buffers and constructed or restored wetlands (Mitchell *et al.*, 2023; Singh *et al.* 2020).



5.5 Human-induced climate change (SDG 13: Climate Action)

5.5.1 Introduction

Actions to address human-induced climate change is the focus of SDG 13: Climate Action. Of particular relevance to soil science is Target 13.2: Integrate climate change measures into national policies, strategies and planning. The specific indicator is Indicator 13.2.2: Total greenhouse gas emissions per year (UN, 2022). The three GHG emissions of relevance to soils are CO_2, CH_4 and N_2O. Emissions of CO_2 are one consequence of loss of SOC from soils and practices that increase SOC levels are an important potential mitigation measure in climate action. Soils are an important source of CH_4 and N_2O emissions and hence practices that reduce emissions of both gases from soils have been an important research focus since 2015. The potential for CO_2 for inorganic carbon sources in soils has also been an emerging issue in the past decade along with practices that can minimize emissions.

5.5.2 Estimates of mitigation potential of soil-based practices

The most authoritative estimates of the mitigation potential of soil-based measures are those of the IPCC in their Sixth Assessment Report (AR6) (Nabuurs *et al.*, 2022) which draw heavily on the country-based estimates on land-based measures developed by Roe *et al.* (2021). The estimates are made from both published, sector-based estimates and results from integrated assessment models. The two documents provide information on both the technical potential (i.e. the total possible with available technology regardless of the cost) and cost-effective measures (i.e. those measures possible with costs of up to USD 100 per tonne of CO_2 equivalents). The use of CO_2 equivalents allows a standard costing across the different GHGs. This section will primarily focus on the potential available at the USD 100 per tonne threshold rather than the technical potential. Generally, the cost-effective sectoral estimate is about 40 percent of available technical potential.

Roe *et al.* (2021) estimated that land-based measures had the potential to mitigate approximately 10 to 15 billion tonnes of CO_{2eq} annually by 2050, corresponding to about 20 to 30 percent of the mitigation needed to achieve the 1.5 °C temperature target agreed to in the *Paris Agreement to the United Nations Framework Convention on Climate Change* (12 December 2015).

Forests and other ecosystems provide the largest share of land-based mitigation. In sectoral estimates, there is a cost-effective potential of 6.6 ± 2.9 billion tonnes of CO_{2eq} annually, or 48 percent of the total land-based potential (Roe *et al.*, 2021).

The specific potentials of the land-based measures relevant to SSM are summarized in Table 5.11. Reductions in deforestation and degradation have the highest cost-effective potential (3.4 billion tonnes of CO_{2eq} annually) followed by afforestation, reforestation and forest ecosystem restoration (1.6 billion tonnes of CO_{2eq} annually) and improvements in forest management (1.1 billion tonnes of CO_{2eq} annually).

Table 5.11. Estimated annual mitigation potential (billion tonnes CO_2-eq annually) over the 2020–2050 period of selected land-use options

Main category of mitigation	Subcategory	Potential at a cost of <USD 100/tonne CO_2-eq	Technical potential[1]
Agriculture (total through carbon sequestration and reductions in CH_4 and N_2O emissions)		4.1	11.2
Agriculture (through overall carbon sequestration)		3.4	9.5
	Soil carbon management in croplands	0.6	1.9
	Soil carbon management in grasslands	0.9	1.0
	Agroforestry	0.8	4.1
	Biochar	1.1	2.6
Agriculture (through reductions in CH_4 and N_2O emissions)		0.6	1.7
	Manure management	0.1	0.3
	Nutrient management	0.2	0.3
	Improving rice cultivation	0.2	0.3
Forest and other ecosystems (total through protection, management and restoration)		7.3	13
Forest and other ecosystems (through protection)		4.0	6.2
	Reducing deforestation and degradation	3.4	4.5
	Loss and degradation of peatlands	0.48	0.86
	Coastal wetlands	0.17	0.8
	Grasslands and savannas	0.04	0.2
Forest and other ecosystems (through restoration)		2.1	5
	Afforestation, reforestation and forest ecosystem restoration	1.6	3.9
	Peatland restoration	0.4	0.79
	Coastal wetland restoration	0.1	0.3
Forests and other ecosystems (through management)		1.2	1.8
	Improving forest management	1.1	1.7
	Fire management (grassland and savanna fires)	0.05	0.1

[1] **Technical potential is the total potentially achievable at any cost.**

Source: **Nabuurs, G.-J., Mrabet, R., Abu Hatab, A., Bustamante, M., Clark, H., Havlík, P., House, J.** *et al.* 2020. Agriculture, Forestry and Other Land Uses (AFOLU). In: P.R. Shukla, J. Skea, R. Slade, R. Fradera, M. Pathak, A. Al Khourdajie, M. Belkacemi *et al.*, eds. *Climate Change 2022 Mitigation of Climate Change. Working Group III contribution to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*, p. 185. Geneva, Switzerland, Intergovernmental Panel on Climate Change (IPCC).

Measures related to agriculture are grouped into C sequestration measures (with cost-effective potentials of 3.4 billion tonnes of CO_{2eq} annually) and measures that reduce CH_4 and N_2O emissions (with smaller cost-effective potentials of 0.6 billion tonnes of CO_{2eq} annually). The specific soil management approaches that can achieve these potentials are examined in detail in the following sections with an emphasis on studies produced since 2020 (due to the information not being available to the authors of the Roe *et al.* [2021] synthesis).

5.5.3 Increasing soil organic carbon content

There has been an extraordinary effort made by the soil science community since 2015 to quantify the changes in SOC levels associated with different land management decisions, due to the possibility of the removal of C from the atmosphere by increasing the store of C in the soil (called C sequestration). Carbon storage in a given land unit can also be increased by transferring C from another location, in which case there is no removal of C from the atmosphere (and hence no C sequestration) (Chenu *et al.*, 2019). Soil C increases can, however, lead to improvements in soil functions such as water and nutrient-holding capacity.

The results from thousands of individual studies on the soil management–soil C relationship can be summarized using meta-analyses (Table 5.12). For SOC, the large number of meta-analyses allow second-order meta-analyses to be performed (Beillouin *et al.*, 2023). The net result is a more authoritative understanding of the management of SOC than was available in 2015.

Table 5.12. Summary of meta-analyses of soil organic carbon changes due to management practices in croplands

Study	Beillouin *et al.* (2022)	Bolinder *et al.* (2020)	Bolinder *et al.* (2020)	Haddaway *et al.* (2017)	Lessman *et al.* (2022)
Metric	Change (%)	Change (%)	Stock change rate (kg SOC/ha annually)	SOC stocks to 30 cm (t/ha)	SOC stock change per year (t/ha annually)
Soil depths	Variable	Variable	Variable	0 to 30 cm	Variable
Biochar	66.6	Not assessed	Not assessed	Not assessed	Not assessed
Organic amendments	29.1	30.3	529	Not assessed	0.6
Organic amendments versus mineral fertilizer	33.8	Not assessed	Not assessed	Not assessed	Not assessed
Cover crops	12.8	10.5	358	Not assessed	Not assessed
No-till or reduced tillage	9.3	Not assessed	Not assessed	4.61	0.18
Crop residue retention	13	9.9	212	Not assessed	0.21
Mineral fertilization	9.4	7.2	302	Not assessed	0.15
Agroforestry	20.0	Not assessed	Not assessed	Not assessed	Not assessed
Perennial crops/orchards	20.3	Not assessed	Not assessed	Not assessed	Not assessed
Perennial energy crop	11.8	Not assessed	Not assessed	Not assessed	Not assessed
Crop rotation	6.5	Not assessed	Not assessed	Not assessed	0.18

[1] No-till vs high intensity tillage.

◊ *Sources:* **Beillouin, D., Cardinael, R., Berre, D., Boyer, A., Corbeels, M., Fallot, A., Feder, F. & Demenois, J.** 2022. A global overview of studies about land management, land-use change, and climate change effects on soil organic carbon. *Global Change Biology*, 28(4): 1690–1702. https://doi.org/10.1111/gcb.15998

◊ **Bolinder, M.A., Crotty, F., Elsen, A., Frac, M., Kismányoky, T., Lipiec, J., Tits, M., Tóth, Z. & Kätterer, T.** 2020. The effect of crop residues, cover crops, manures and nitrogen fertilization on soil organic carbon changes in agroecosystems: a synthesis of reviews. *Mitigation and Adaptation Strategies for Global Change*, 25(6): 929–952. https://doi.org/10.1007/s11027-020-09916-3

◊ **Haddaway, N.R., Hedlund, K., Jackson, L.E., Kätterer, T., Lugato, E., Thomsen, I.K., Jørgensen, H.B. & Isberg, P.-E.** 2017. How does tillage intensity affect soil organic carbon? A systematic review. *Environmental Evidence*, 6(1): 30. https://doi.org/10.1186/s13750-017-0108-9

◊ **Lessmann, M., Ros, G.H., Young, M.D. & De Vries, W.** 2022. Global variation in soil carbon sequestration potential through improved cropland management. *Global Change Biology*, 28(3): 1162–1177. https://doi.org/10.1111/gcb.15954

At a broad level, the research has identified and quantified three approaches to reduce SOC loss or (ideally) increase SOC storage:

1. preserve existing SOC stores by preventing harmful land-use conversions;
2. adopt management practices that preserve or enhance existing SOC contents of managed soils; and
3. reduce losses of SOC through soil erosion.

The management practices that reduce soil erosion are discussed in Section 5.3.1.

Preservation of existing stores of SOC is critical to prevent further emissions of SOC to the atmosphere and to maintain the soil functions provided by the soils in their present state. The losses of SOC associated with conversions of forests, wetlands and grasslands to cropland range from 16 to 26 percent of the original SOC (see Section 3.3).

Conversions of cropland to forest or grassland can lead to substantial increases in SOC storage through time. Beillouin *et al.* (2023) estimated that forestland SOC increased on average by 57 percent and grassland SOC by 26 percent after conversion from cropland. It was also estimated that the conversion of cropland to uses such as perennial crops and orchards or agroforestry led to average increases in SOC of approximately 20 percent, in part because permanent crops led to higher inputs of C deeper in the soil profile (Chenu *et al.,* 2019).

Soil organic carbon content of the soil can be significantly increased by the adoption of SSM practices in croplands and grasslands (Table 5.11 and Table 5.12). The practices can be divided into three broad groups: exogenous (the additions of C-rich material produced elsewhere), changes to in-field soil management and changes to the cropping system.

The greatest increases are associated with the addition of C-rich material such as biochar and manure (Table 5.11). Biochar and some forms of manure management involve a transfer of C from the point of production and processing to the site, where the C is added to the soil. Because of this, a fuller life cycle is required to assess the C sequestration benefits.

In their second-order meta-analysis, Beillouin *et al.* (2023) reported that biochar increased SOC by the largest amount, followed by other organic amendments. A 2024 meta-analysis of the effect of biochar on SOC reported that biochar additions across 101 field experimental sites increased SOC stocks on average by 6.1 tonnes C/ha compared to plots with no biochar additions (Li *et al.,* 2024*)*. The specific effects of biochar on SOC, and more generally on the GHG balance, depends on the feedstock used to produce the biochar (e.g. wood vs crop residues) and the technology used to produce the biochar (Smith, 2016; Majumer *et al.*, 2019).

Like biochar, manure additions also substantially raise SOC levels due in part to the organic carbon (OC) content of the manure. For example, the meta-analysis of Liu *et al.* (2020) used data from 521 observations from 92 studies, which were drawn heavily from China and Europe. The average OC content of the manures ranged from 407 g SOC/kg manure for green manures to approximately 205 g SOC/kg manure for cattle manure. The average increase in SOC after manure additions was 27 percent. The average increase in SOC was least in sites in cold climates (11 percent in sites where the mean annual temperature was <5 °C and between 20 and 25 percent for temperatures of 5 °C and above). The average increase in SOC was greatest for moist climates (26 percent for sites with a mean annual precipitation between 250 and 1 000 mm compared to only 17 percent for drier and wetter sites). The increases in SOC were highest for green and swine manures and lowest for cattle and poultry manure.

Changes to management practices on cropland may lead to increases in SOC. The SOC changes arise from changes in the quantity of organic matter inputs, the depth in the soil profile that the inputs are added to, or the processes that decompose organic matter but do not involve additions of C from offsite.

Cover crops have been shown to increase SOC in several meta-analyses (Table 5.12). The SOC benefits are evident across climate zones, but the results are more variable in water-limited environments. Across 17 sites with precipitation <500 mm, Blanco-Canqui *et al.* (2022) found that the average annual stock change in SOC was 0.43 tonnes SOC/ha, but no accumulation of SOC occurred in 7 of the 17 locations. Both Blanco-Canqui *et al.* (2015) and Jian *et al.* (2020) reported that the greatest increases in SOC due to cover crops occurred in fine-textured (clayey) soils and the lowest increases were in coarse-textured soils. Cover crops also substantially decreased wind and water soil erosion (see Section 5.3.1), reducing SOC losses through the erosion pathway.

The role of tillage in SOC change has been intensively studied in the post-2015 period. Initially, the evidence pointed to a substantial increase in SOC associated with adoption of reduced tillage or no-till in place of conventional tillage with inversion or mixing of the topsoil. However, a series of meta-analyses demonstrated that the SOC increase was largely associated with the upper 20 to 30 cm of soil, and that, on average, small decreases of SOC occurred in the subsoil (Chenu *et al.*, 2019). For example, Haddaway *et al.* (2017) showed that C stock increased under no-till compared to high-intensity tillage by (on average) 4.6 t/ha in the upper soil (0–30 cm) over at least ten years, while no effect was detected in the full soil profile. It was concluded that the higher SOC stocks or concentrations in the upper soil promoted a more productive soil with higher biological activity and provided resilience to extreme weather conditions. Chenu *et al.* (2019) concluded that:

> *Hence when reducing soil tillage, there may be an additional SOC storage in superficial soil layers, but no or little SOC sequestration for the land unit considered if the whole soil profile is considered.* (Chenu *et al.*, 2019, p. 46).

Like cover crops, reduced tillage with residue retention significantly reduces soil erosion from all causes and hence leads to higher SOC retention in fields (see Section 5.3.1).

A practice closely associated with reduced tillage is the handling of crop residues after harvest. At a broad level crop residues can be left in place (or retained), returned to the soil after processing off-field or removed entirely, and used for other purposes such as animal feed, bedding or for the production of energy. In their second-order meta-analysis, Beillouin *et al.* (2023) found an average SOC gain of 12.8 percent for residue retention and was almost identical to the average SOC gain under cover crops (12.8 percent). In their focused meta-analysis of straw return after harvest, Li *et al.* (2023) reported an average global increase of 3.7 tonnes SOC/ha over the duration of the trials, with greatest gains for maize (approximately 5 tonnes C/ha) and rice (approximately 4 tonnes C/ha) and least gains for wheat (approximately 2.5 tonnes C/ha). No significant differences were observed for no-till vs conventional tillage plots, climate zones (considered as cold temperate, warm temperate and tropic), soil texture or soil pH.

Fertilization with synthetic fertilizers also leads to higher SOC storage (Table 5.12). The increases are associated with increases in biomass production associated with the use of mineral fertilizers (see Section 5.3.3). There are GHG emissions associated with production and transport of fertilizers and (in the case of N fertilizers) emissions of GHGs due to the transformation processes with the soil, and hence (like biochar and manure) a full life cycle analysis must be completed to assess their full GHG balance.

The response to management of SOC in grasslands is generally similar to the response in cropland (Table 5.13). For example, Beillouin *et al.* (2023) found on average that SOC in grasslands increased by 33.6 percent due to organic amendments, 32 percent due to biochar, 25.7 percent due to agroforestry and 3.6 percent due to mineral fertilization. Grazing of grasslands led (on average) to losses of 16 percent of SOC but with a wide range of responses (from losses of 31.4 to increases of 11.4 percent).

Table 5.13. Summary of average percentage change of soil organic carbon due to management practices in grasslands and forestland

Management practice	Grassland	Forest
Mineral fertilization	3.6% increase	5.5% increase
Organic amendments	33.6% increase	
Agroforestry	25.7% increase	
Species mixture	6.6% increase	9% increase
Liming	5.7% increase	
Litter removal	3.5% loss	14.5% loss
Grazing	15.5% loss	
Intensive versus less intensive grazing	9.9 %loss	
Conifer versus hardwood stands		7.1% loss
Forest harvesting		8.1% loss
Prescribed fire		20.9% loss

◊ *Source:* **Beillouin, D., Cardinael, R., Berre, D., Boyer, A., Corbeels, M., Fallot, A., Feder, F. & Demenois, J.** 2022. A global overview of studies about land management, land-use change, and climate change effects on soil organic carbon. *Global Change Biology*, 28(4): 1690–1702. https://doi.org/10.1111/gcb.15998

In forestland, a number of practices have been shown to lead to substantial decreases in SOC (Table 5.13). Beillouin *et al.* (2023) reported a mean loss of 8.1 percent due to forest harvest and 14.5 percent due to removal of the forest organic litter layer. A focused meta-analysis by James *et al.* (2021) showed losses between 15.4 percent for whole-tree harvest alone and 24.9 percent for whole-tree harvest plus removal of limbs and the forest litter layer. Although fewer studies were available, the effects of prescribed fire were also severe, with losses of 20.9 percent on average (Beillouin *et al.,* 2023). Increases in forestland SOC storage were associated with mineral N addition and the selection of N-fixing species (Mayer *et al.*, 2020).

5.5.4 Reducing nitrous oxide and methane emissions from soil

Gaseous emissions of N_2O from soils are a major contributor to increasing atmospheric GHG concentrations (also see Section 2.2.3 and Section 3.5.1). Nitrous oxide is released during microbial transformation of N inputs to soils from all sources, with synthetic fertilizer providing the largest sources of N. The IPCC methods for calculating a national inventory of emissions use an emission factor of 0.016 kg N_2O-N per kg of N in synthetic fertilizer in wet climates, and 0.005 kg N_2O-N per kg of N in dry climates (Hergoualc'h *et al.*, 2019). Organic amendments, animal manures, N in crop residues and N from soil organic matter decomposition are assigned an emission factor of 0.006 kg N_2O-N per kg of N in wet climates and 0.005 kg N_2O-N per kg of N in the materials in dry climates.

Given this relationship, any measures that improve NUE (see Section 5.3.2) are likely to lead to reductions in N_2O emissions. Using the N balance is a superior management approach compared to considering N inputs alone, as once crop uptake demand has been satisfied, any surplus N is susceptible to gaseous loss pathways (Eagle *et al.,* 2020). Using N balance to guide input decisions can minimize N_2O emissions while still achieving high crop yields (Maaz *et al.*, 2021).

The use of EEFs has been shown to substantially lower emissions (Akiyama, Yan and Yagi, 2009; Gurung *et al.*, 2021; Pan *et al.*, 2023; Gu *et al.*, 2023). Young, Ros and de Vries (2021) found mean reductions of 36 percent for N_2O emission after application of EEFs, while Gu *et al.* (2023) found mean reductions of 35 percent (Table 5.3). Of the EEFs, nitrification inhibitors caused the greatest reduction (48 to 38 percent) followed by controlled release and urease inhibitors (38 percent reduction to 6 percent increase). A 2023 meta-analysis by Pan *et al.* (2023) focusing on EEFs in potato production also found that nitrification inhibitors were the most effective at reducing N_2O emissions (with a mean reduction of 49.4 percent).

Biochar causes N_2O emission reductions in a similar range to EEFs. The meta-analysis of Kaur *et al.* (2023) of other meta-analyses found mean reductions in N_2O emissions of 38.8 percent. The impacts of biochar on emissions generally increased with an experimental duration of one and a half years before reducing. A meta-analysis by Zhang *et al.* (2023) reported mean reductions of N_2O emissions due to biochar additions of 16.2 percent.

Deep placement of synthetic fertilizer has also been shown to substantially decrease N_2O release from soils. A global meta-analysis by Wu *et al.* (2024) reported that deep placement of fertilizer reduced N_2O emissions on average by 16.2 percent, with the most suitable fertilization depth at 15 to 25 cm. They also found that deep placement increased grain yield by 13.5 percent on average. The meta-analysis also showed that deep placement significantly decreased emissions from paddy fields but that there was no significant effect on emissions from drylands. This was contrary to the 2021 meta-analysis of Chen, Liao and Liao (2021), who reported a significant increase in emissions from drylands. Wu *et al.* (2024) attributed the difference to the variability of precipitation in drylands.

Reductions in tillage have been shown in several meta-analyses to have minimal overall impacts on N_2O and CH_4 emissions (Van Kessel *et al*., 2013; Ruis *et al*. 2022; Li *et al*., 2023). Van Kessel *et al*. (2013) showed that significant decreases in N_2O only occurred in sites in dry climates after ten years or more of no-till management. Li *et al.* (2024) did find that reduced tillage was one of the most effective mitigation options for N_2O from specialty crops, along with drip irrigation and fertigation.

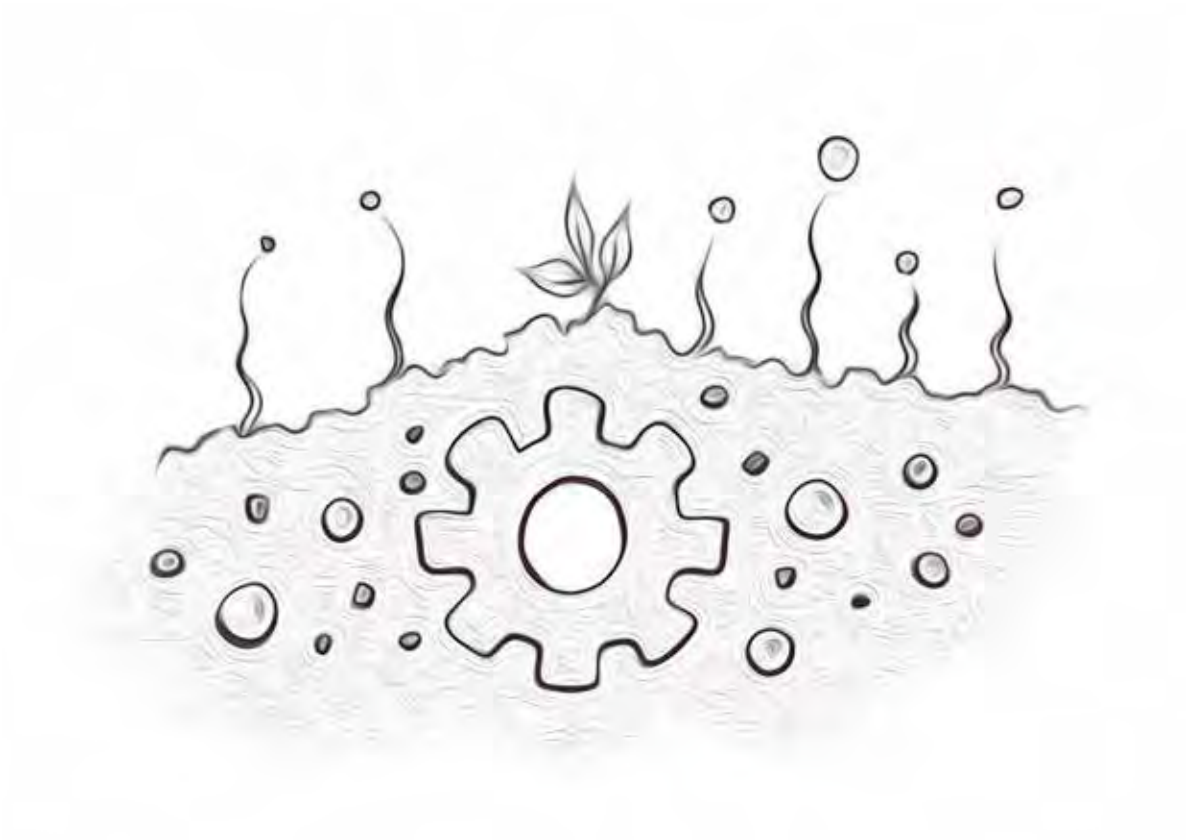
The effects of additions of organic material such as manure, other organic fertilizers or straw on N_2O emissions must be considered in the broader context of emissions of CH_4 and CO_2 so as to determine the full global warming potential (GWP). The meta-analysis of He *et al.* (2023) found that while on average, the replacement of synthetic fertilizers by organic fertilizers significantly reduced N_2O emissions, they significantly increased emissions of CH_4 and CO_2 and thereby increased the GWP. The increase in GWP was greatest for paddy fields, whereas high latitude areas with little precipitation were more suitable for promoting the application of organic fertilizers to replace chemical fertilizers. Similarly, Xia *et al.* (2017) found that the use of animal manure improved the GWP in upland fields but had a negative impact in paddy fields. Straw return to paddy fields likewise caused a minor decrease in N_2O but substantial increases in CH_4 and CO_2 (He *et al*., 2024).

Minimizing GHGs from paddy fields has been a particular focus of research given their high rates of emissions, as around 30 percent of global agricultural CH_4 emissions are from paddy fields, along with 11 percent of global N_2O emissions (Zhang *et al*., 2024). Recent meta-analyses by Zhang *et al.* (2024) for the United States and a global database and Zhou *et al.* (2024) for Monsoon Asia showed that changes in water regimes from continuous flooding to intermittent flooding (also called multiple drainage or non-continuous irrigation) caused the greatest reduction in emissions. Zhou *et al.* (2024) found a mean reduction in CH_4 emissions of 27.7 percent with multiple drainage. Zhang *et al.* (2024) considered CH_4 and N_2O together, and reported that non-continuous irrigation reduced net GHG emissions by approximately 10 percent and non-CO_2 GHG emissions by about 39 percent. The adoption of no-till or reduced tillage, the optimization of the ratio of synthetic N fertilizer and manure, and returning straw to the fields post-harvest also significantly reduced CH_4 and total GHG emissions from paddy fields.

5.5.5 Reducing carbon dioxide emissions from soil inorganic sources

Losses of inorganic C from soils to the atmosphere occur due to deposition of acidic compounds from the atmosphere and from inputs of synthetic N fertilizers (see Section 3.3.5). Zamanian, Zhou and Kuzyakov (2021) suggested that avoidance of over-fertilization and including organic fertilizers as sources of nutrients could reduce CO_2 emissions.

Liming is widely used to neutralize acidity in soils and involves the addition of neutralizing compounds such as crushed $CaCO_3$ or dolomite to soils (see Section 5.3.4). The transformation of $CaCO_3$ is known to release CO_2 (the IPCC emission factor for $CaCO_3$ additions is 13 kg C per tonne for dolomite and 12 kg C per tonne for limestone) (IPCC, 2006). Zhang *et al.* (2022) reported that while liming increased CO_2 emissions by 41 percent, it also decreased soil N_2O emissions by 38 percent and CH_4 emissions by 20 percent. Overall, however, the change in the net GWP was not significantly different from zero.



5.6 Soil pollution and human and ecosystem health (SDG 3: Good Health and Well-being)

5.6.1 Introduction

Soil pollution is explicitly mentioned in Target 3.9: By 2030, substantially reduce the number of deaths and illnesses from hazardous chemicals and air, water and soil pollution, and contamination (UN, 2022). The *Global Assessment of Soil Pollution* (FAO and UNEP, 2021a, 2021b) also stated that soil pollution hits the most vulnerable hardest, especially children and women (SDG 5: Gender Equality). The supply of safe drinking water is also threatened by the leaching of contaminants into groundwater and runoff (SDG 6: Clean Water and Sanitation). Soil pollution also contributes to land degradation and the loss of terrestrial and aquatic biodiversity (SDG 15: Life on Land and SDG 14: Life below Water, respectively), as well as decreased security and resilience of cities (SDG 11: Sustainable Cities and Communities). Human health is also harmed by NH_3 emissions from soils. Ammonia emissions are associated with presence of fine-particulate matter (particulate matter with an aerodynamic diameter smaller than 2.5 µm) in the atmosphere, which has been linked to many health impacts, including acute lower respiratory infections.

5.6.2 Reducing soil pollution and remediating polluted sites

Reducing soil pollution requires action at several levels. Internationally there are legal instruments designed to eliminate or reduce use of harmful compounds in the environment including soils. The *Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and their Disposal* (22 March 1989), the *Rotterdam Convention on Prior Informed Consent for Hazardous Chemicals and Pesticides in International Trade* (10 September 1998), the *Stockholm Convention on Persistent Organic Pollutants* (22 May 2001), and the *Minamata Convention on Mercury* (10 October 2013) are all legal instruments targeted at the control of specific harmful compounds, and are discussed further in FAO and UNEP (2021).

Voluntary guidelines have also been adopted by FAO Member States to tackle soil pollution, which can serve as a basis for action for countries that lack adequate national legislation on soil pollution. The VGSSM (FAO, 2017), the *International Code of Conduct for the Sustainable Use and Management of Fertilizers* (FAO, 2019) and the *International Code of Conduct on Pesticide Management* (FAO and WHO, 2014) all contain provisions specific to reducing soil pollution.

A key way forward identified in the *Global Assessment of Soil Pollution* (FAO and UNEP, 2021) is the need to establish and strengthen monitoring systems for measuring the extent and severity of soil pollution. Monitoring should be included in conventional soil surveys and national reporting mechanisms should be established with soil pollution targets and indicators related to achievement of the SDGs. As an example, the European Union has undertaken an assessment of 118 active ingredients of pesticides and their derivatives at over 3 400 sites as part of the ongoing LUCAS assessment system (Orgiazzi *et al.*, 2022).

As documented in the regional assessments (Chapter 8 to Chapter 14 of this report) there are many sites where soil pollution has taken place in the past and where pollution at new sites continues due to accidents and lack of enforcement of existing controls. Contaminated sites need to be remediated to prevent harm to humans and the broader environment.

The suite of remediation techniques is well-established (Table 5.14) (FAO and UNEP, 2021; Kuppusamy

et al., 2016a, 2016b; Morillo and Villaverde, 2017; Song *et al.*, 2022; Rajendran *et al.*, 2022; Sarker *et al.*, 2024). A major focus in these reviews has been on novel approaches for remediation using nature-based solutions such as phytoremediation, microbial remediation and immobilization techniques using biochar.

Table 5.14. Technologies for remediating polluted soils

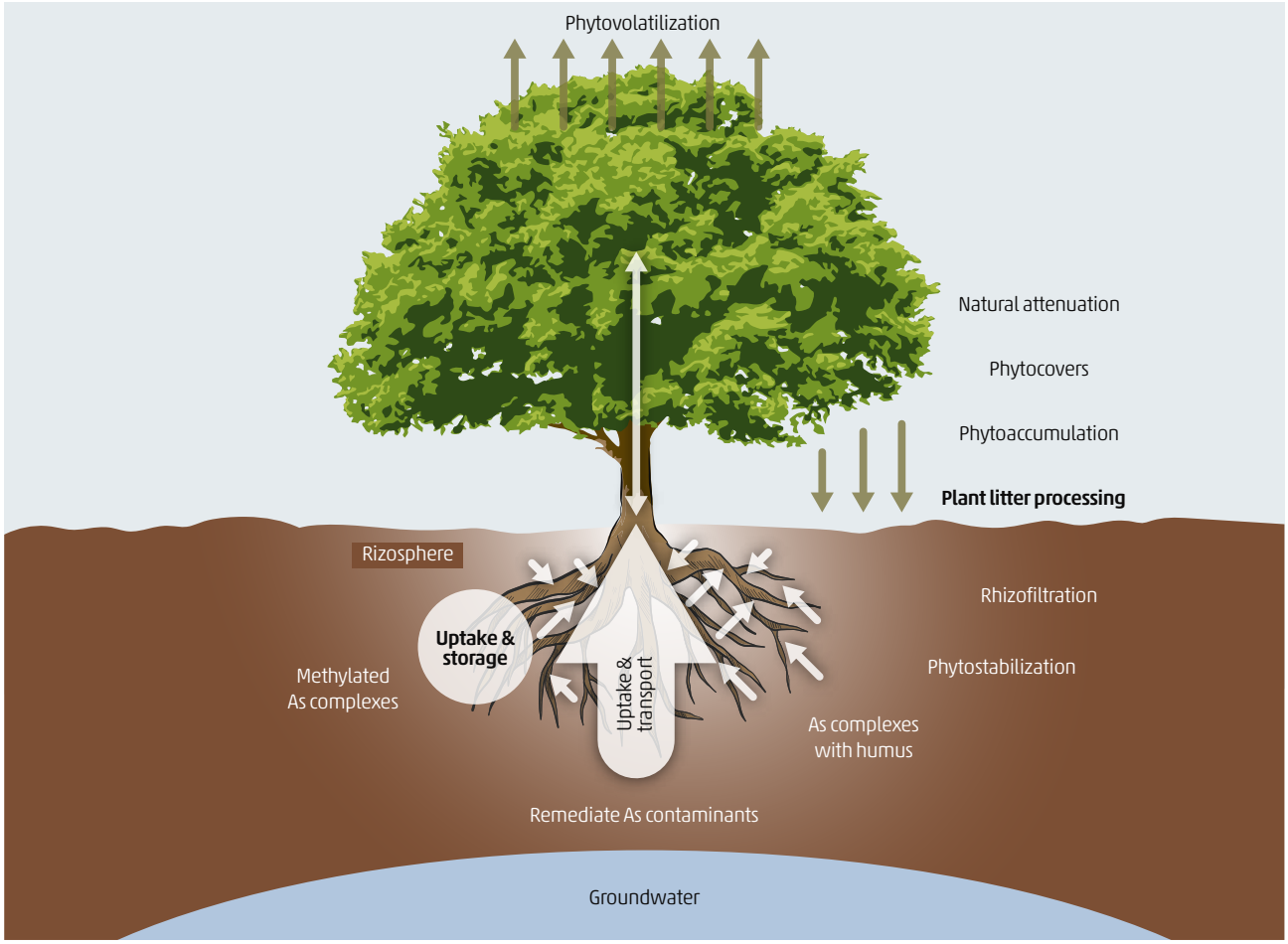
***In situ* biological treatments**	
Biomediation	This is an *in situ* biological treatment that uses soil microorganisms for remediation, and is primarily aimed at degrading organic contaminants, including petroleum hydrocarbons, solvents and pesticides, and transforming types of trace elements to reduce their availability.
Bioventing and biosparging	Bioventing and biosparging are similar techniques that stimulate the biodegradation of organic compounds by the direct injection of air into the soil. Bioventing is undertaken in the soil vadose zone (the unsaturated strata above the water table), and biosparging in the saturated zone below the water table.
Phytoremediation	Phytoremediation describes in situ technologies that use plants for treating polluted soils. It plays two principal roles in remediating polluted soils: stabilizing the contaminants so that they are less mobile and less available to cause harm (phytostabilization) and removing them by facilitating their degradation or transferring them to other media (phytoextraction). Phytoremediation has been used extensively for remediating soils that have been polluted with trace elements but can also be effective in facilitating the removal and biodegradation of organic contaminants.
Vermiremediation	This describes a technique where earthworm species have been found to remove trace elements, pesticides and lipophilic organic contaminants, including polycyclic aromatic hydrocarbons (PAH), from the soil. Earthworms can be tolerant to many chemical contaminants including trace elements and organic contaminants in soil and can bioaccumulate them in their tissues.
***Ex situ* biological treatments**	
Biopiling	This is a bioremediation technology, extensively used for the ex situ remediation of polluted soils by a wide range of petrochemical and other organic chemical contaminants. The technique involves excavating soil which is then homogenized and amended to enhance the conditions for biodegradation by microorganisms.
Bioleaching	This is an extractive technology that uses a solution inoculated with microorganisms to leach trace elements from polluted soils. It is also a process that is used in the mining industry as an alternative to cyanide to extract metals from low-grade ores or mine wastes.
Composting	Composting is an aerobic ex situ technology, similar to biopiling, which involves mixing the excavated polluted soil with a non-hazardous bulking agent and organic amendments (e.g. straw or green waste) to provide the optimum levels of air and moisture to microorganisms.
Landfarming	This is an ex situ bioremediation technique, where excavated polluted soil is transported to the landfarming site and spread in a thin layer onto biologically active land or an impermeable surface. The polluted soil is ploughed into the soil surface, sometimes with the addition of organic amendments (e.g. manure or sewage sludge). This process stimulates biodegradation by the inherent microorganisms, aerobically stimulated by the ploughing, which may be repeated, and by adding nutrients for growth promotion of microorganisms.
Bioreactors	These are open or closed installations that agitate the soil while adding nutrients to stimulate microorganisms to degrade the contaminants.
In situ physical/chemical treatment technologies	
In situ physical treatments	
Electrokinetic separation	This technique uses electrical fields and electrochemical processes to enhance the migration of polar inorganic and organic molecules out of soil. It is useful to increase the extraction rates for contaminants in low permeability soils.
Soil vapour extraction and air sparging	These are similar processes to their biological equivalents, bioventing and biosparging, but rely solely on the air (which may be heated) passing through the polluted soil to volatilize the volatile and semi-volatile contaminants.

In situ chemical treatments	
Solidification and stabilization	Contaminants are physically bound or enclosed within a stabilized mass (solidification), or chemical reactions are induced between the stabilising agent and contaminants to reduce their mobility (stabilization).
Oxidation	Solutions of chemical oxidising agents are injected directly into the polluted soil or placed as a permeable barrier in the path of a contaminant plume. These oxidising agents are able to cause the rapid and complete chemical degradation of many organic contaminants and partial degradation of others, which aids subsequent bioremediation.
Reduction	This describes a range of reactions including the dechlorination of organochlorine compounds and the conversion of trace elements to less toxic and/or mobile forms.
Nanotechnologies	
	The application of reactive nanomaterials for transformation and detoxification of contaminants.
In situ thermal treatments	
	Electrical resistance heating, steam injection and extraction, conductive heating, radio frequency heating, and vitrification are in situ thermal treatments. The main objective of these technologies except for vitrification is to increase the removal efficiency of organic contaminants by heating the soil.
Ex situ physical treatments	
Separation	Size separation is generally undertaken as an initial step in washing processes, to remove the coarse sand and gravel fractions from the fine clay and silt particles, concentrating the contaminants into a smaller volume of soil that can be further treated or disposed of using other ex situ methods.
Soil washing	These are soil washing systems that can be used to address to soils polluted with a wide variety of trace elements, radionuclides, and organic contaminants.
Solidification or stabilization	This is where the contaminants are held in ex situ solidification or stabilization, being either physically bound or enclosed within a stabilized mass (solidification), or where chemical reactions are induced between the stabilizing agent and contaminants to reduce their mobility (stabilization). Ex situ processes, however, typically require the disposal of the resultant materials at a licenced disposal facility.
Ex situ chemical treatments	
Chemical extraction	This is a means of separating hazardous contaminants from soils, thereby reducing the volume of the hazardous waste that must be treated. The technology is similar to soil washing but uses an extracting chemical rather than water and wash-improving additives, such as surfactants.
Chemical reduction and oxidation	The ex situ chemical reduction and oxidation processes are identical to those discussed under the in situ processes previously described.
Dehalogenation	After processing, the contaminated soil is heated in a reactor. This technique has been used to treat soils polluted with intractable chlorinated organic compounds including hexachlorobenzene, polychlorinated biphenyls and polychlorinated dibenzo dioxins, and furans.
Ex situ thermal treatments	
Thermal desorption	As with the in situ thermal desorption, this process uses heat to drive off volatile contaminants, such as petroleum hydrocarbons, chlorinated organics and trace elements such as mercury (Hg).
Incineration	High temperature incineration is a way to treat soils containing extremely high levels of pollution from the most persistent contaminants, including high concentrations of persistent organic pollutants (POPs).
Sequestration	
Engineered landfill	Where there is an urgency to eliminate the polluted soil or where insufficient funds are available for other treatment techniques, excavation and offsite disposal in an engineered landfill is an option. It is also an option for soils with contaminants that cannot easily be treated, such as radioactive isotopes.

◊ *Source:* **FAO & UNEP.** 2021. *Global Assessment of Soil Pollution* – Summary for policymakers. Rome, FAO. https://doi.org/10.4060/cb4827en

Phytoremediation is the most widely deployed of these nature-based techniques (Alshehri *et al.*, 2023). Phytoremediation uses plants to extract, stabilize, degrade or volatize contaminants (Figure 5.3). In the review by Alshehri *et al.* (2023) phytoextraction was used in 65 percent of the studies reviewed and was often targeted at heavy metals and hydrocarbons. Phytoremediation is well suited to slightly or moderately polluted agricultural sites but at more heavily polluted brownfield sites, mobilizing agents may need to be added to the soil to improve the effectiveness of phytoremediation (Hou *et al.*, 2023).

Figure 5.3. The phytoremediation functions of a plant source for remediating arsenic-contaminated soil



Note: As = arsenic.

Source: **FAO & UNEP**. 2021. *Global Assessment of Soil Pollution* – Summary for policymakers. Rome, FAO. https://doi.org/10.4060/cb4827en

Microbial bioremediation uses bacteria and fungi to remove or transform contaminants. Microbial bioremediation has been effective for treating trace metals and pesticides as well as for emerging pollutants such as perfluorooctanoic acid (PFOA) and perfluorooctanesulfonic acid (PFOS), and antibiotics (Rajendran *et al.*, 2022; Hou *et al.*, 2023). Microbial remediation may involve the native microflora already present in the soil, but to be most effective often requires the addition of nutrients such as N, P or micronutrients (a process called biostimulation), or inoculation with enriched consortia of single pollutant-degrading strains (called bioaugmentation) (Morillo and Villaverde, 2017). Although microbial bioremediation can be carried out at the contaminated site, in some cases it is more desirable to transport the contaminated soil and mix it with topsoil elsewhere (land farming), or create piles where O can be pumped into the mounds to enhance degradation (called biopiles).

Phytoremediation and microbial bioremediation are less expensive than alternative techniques, with Morillo and Valverde (2017) quoting costs of USD 10 to 50 per tonne for phytoremediation versus USD 2 200 to 2 400 per tonne for excavation of soil and final disposal in landfills. Phyto- and bioremediation also have greater public acceptance than the traditional methods such as landfilling (Hou *et al.*, 2023; Sarker *et al.*, 2024). The bioremediation methods are, however, only effective at relatively shallow depths of contamination and may require considerable time to substantially reduce contamination. Hou *et al.* (2023) stated that the relative slowness of the bioremediation approaches had limited their adoption in China where the remediation market was development driven and required faster methods.

While the emphasis in the research literature on nature-based solutions is encouraging, many pollutants such as radionuclides or the persistent organic pollutants controlled by the Stockholm Convention still require traditional excavation, containment and incineration approaches or other physical and chemical remediation techniques. Continuing identification and risk assessment at contaminated sites globally is essential before the appropriate remediation technique can be selected and a successful remediation programme implemented (FAO and UNEP, 2021).



5.6.3 Reducing ammonia emissions from soils

Ammonia emissions from soils have agronomic, environmental and human health impacts (see Section 3.5.4). Pan *et al.* (2016) estimated that globally on average, 18 percent of N added as synthetic fertilizer was lost as NH_3.

Fortunately, there are a variety of measures that can significantly reduce NH_3 emissions from soils (Table 5.15). Highest reductions are associated with EEFs, specifically those with urease inhibitors, double (i.e. both urease and nitrification) inhibitor and controlled-release fertilizers (Ti *et al.*, 2019; Lam *et al.*, 2022; Pan *et al.*, 2016).

Table 5.15. Changes in ammonia release to atmosphere (volatilization) associated with management practices in four meta-analyses

	Gu *et al.* (2023) (%)	Ti *et al.* (2019) (%)	Lam *et al.* (2022) (%)	Pan *et al.* (2016) (%)
Enhanced efficiency fertilizers (EEFs)	69% decrease			
Urease inhibitors		53.1% decrease	50 to 74% decrease	53.7% decrease
Nitrification inhibitors		42.6% increase	13 to 52% increase	38% increase
Double inhibitors			36 to 75% decrease	
Controlled-release fertilizers		56.8% decrease	27 to 83% decrease	68% decrease
Non-urea-based fertilizers/right type	66% decrease			74.5% decrease
Ammonium nitrate		88.3% decrease		87.9% decrease
Ammonium sulphate		82.9% decrease		78.8% decrease
Right rate	42% decrease			
Right/deep placement	72% decrease	48% decrease		54.7% decrease
Right time	n.s.			n.s.
Legumes included	37% decrease			
No-till	111% increase			
Irrigation	36% decrease	38.8% decrease		34.5% decrease
Amendments including biochar	38% increase			
Deep manure placement		98.7% decrease		
Residue retention		n.s.		25.5% increase
Replacement of nitrogen (N) from synthetic fertilizer by manure		67% decrease		

Note: n.s. indicates a non-significant effect.

◊ *Sources:* **Gu, B., Zhang, X., Lam, S.K., Yu, Y., Van Grinsven, H.J.M., Zhang, S., Wang, X.** *et al.* 2023. Cost-effective mitigation of nitrogen pollution from global croplands. *Nature*, 613(7942): 77–84. https://doi.org/10.1038/s41586-022-05481-8

◊ **Lam, S.K., Wille, U., Hu, H.-W., Caruso, F., Mumford, K., Liang, X., Pan, B.** *et al.* 2022. Next-generation enhanced-efficiency fertilizers for sustained food security. *Nature Food*, 3(8): 575–580. https://doi.org/10.1038/s43016-022-00542-7

◊ **Pan, B., Lam, S.K., Mosier, A., Luo, Y. & Chen, D.** 2016. Ammonia volatilization from synthetic fertilizers and its mitigation strategies: A global synthesis. *Agriculture, Ecosystems & Environment*, 232: 283–289. https://doi.org/10.1016/j.agee.2016.08.019

◊ **Ti, C., Xia, L., Chang, S.X. & Yan, X.** 2019. Potential for mitigating global agricultural ammonia emission: A meta-analysis. *Environmental Pollution*, 245: 141–148. https://doi.org/10.1016/j.envpol.2018.10.124

Of the 4Rs (right rate, source, timing and placement), source and placement have the largest roles to play in reducing NH_3 emissions. Highest losses are associated with urea fertilizers and significant reductions occur if alternative N sources such as ammonium nitrate, ammonium sulphate or manure are used (Table 5.15). Urea is, however, the most widely used synthetic N fertilizer due to the high N content and ease of application (Adalibieke *et al.*, 2023).

Deep placement of N fertilizer or manure substantially reduces NH_3 emissions (Table 5.15) and also accounts for the substantial increases in emissions associated with no-till (compared to conventional

tillage) and residue retention. Both practices are associated with minimal or no soil disturbance and hence the shallow placement (or surface broadcasting) of fertilizers.

The combined effect of improved N fertilizer practices can be significant. Xu *et al.* (2024) calculated global emission factors for NH_3 emissions for rice, wheat and maize crops. Emission factors are defined as the difference between NH_3 emissions under N fertilization and those without N fertilization, divided by the total amount of N fertilizer applied. The global mean emission factors were highest (7.6 to 12.7 percent) for urea that was surface broadcast once in the season under no-till. The lowest emission factors (3.4 to 4.7 percent) were for the deep placement of EEFs under conventional tillage. Hence, substantial reductions in NH_3 emissions can be achieved with deep placement.

5.7 Halt the loss of soil biodiversity and reduce wind erosion (SDG 15: Life on Land)

5.7.1 Introduction

Many elements of SDG 15: Life on Land (UN, 2022) have relevance for SSM. Indeed, Target 15.1: Ensure the conservation, restoration and sustainable use of terrestrial and inland freshwater ecosystems, would pertain to almost all the SSM practices discussed previously. Three specific elements in the targets of SDG 15 are addressed in this section:

1. Target 15.5 specifies a halt to the loss of biodiversity. The preservation of biodiversity is also included in Indicator 15.1.2 and Target 15.4.
2. Soil erosion by wind is an important component of desertification and combating desertification is the focus of Target 15.3.
3. Wetland soils are at the interface of the terrestrial and inland freshwater ecosystems specified in Target 15.1. The SSM practices relevant to this target are also examined in this section.

5.7.2 Halt the loss of soil biodiversity

A key message of the 2020 *State of knowledge of soil biodiversity* report (FAO *et al.*, 2020) was that:

> *The essential contributions of soil organisms are threatened by soil-degrading practices. Policies that minimize soil degradation and protect soil biodiversity should be a component of biodiversity protection at all levels* (FAO *et al.*, 2020, p. 18).

The major recommendations from the FAO *et al.* (2020) report were:

- Sustainable soil management practices should be adopted by farmers and land users to prevent and minimize soil biodiversity loss.
- Soil biodiversity needs to be reflected in national reports and national biodiversity strategies and action plans (NBSAPs).
- Strengthen education and capacity-building in the adoption of molecular tools to contribute to human, plant and soil health.

- Soil remediation and ecosystem restoration plans need to include soil health and soil.
- Biodiversity considerations.
- There is a need to promote a necessary shift toward including biological indicators of soil health alongside physical and chemical indicators.
- There is a need to standardize sampling and analysis protocols worldwide to enable collation of large, comparable datasets.
- Increase intersectoral and interinstitutional collaboration and avoid duplication or fragmentation, due to soil polices often being under the responsibility of different ministries.
- Policies and urban planning need to integrate soil biodiversity into SSM and ecosystem restoration plans to guarantee healthy soils for all and reduce urban threats to soil biodiversity.

Section 3.4 in this report on the threats to soil biodiversity highlights the strong influence that soil pH has over soil biodiversity and underscores the importance of managing soil pH to ensure it remains within the optimum range important for the maintenance or enhancement of soil biodiversity (generally, soil biodiversity and the contributions of soil-dwelling organisms to soil functions are at optimal in soils with a pH range of approximately 5.5 to 7.0). As documented in Section 3.4, deforestation causes (on average) a decrease in soil pH that reduced diversity and hence, avoiding deforestation is an important measure to maintain biodiversity.

The addition of $CaCO_3$ or similar soil amendments is the most common way to manage acidic soil pH levels (see Section 5.3.4). Holland *et al.* (2018) cited multiple studies that documented generally beneficial impacts of liming on the abundance of bacteria (especially rhizobia), fungi, nematodes and earthworms and a decrease in the abundance of pathogens. Both N and C in microbial biomass were also observed to increase with liming, although increased C mineralization and hence losses of C were also observed. There were a limited number of studies of liming effects on microarthropods but there is evidence that the effects of liming may be complex. For example, in their study on liming effects on mesofauna in a Loblolly pine (*Pinus taeda* L.) plantation in Brazil, de Oliveira *et al.* (2024) found that liming increased oribatid mites but decreased Collembola numbers in the forest floor litter layer (where most organisms were found).

A meta-analysis of liming effects by Zhang *et al.* (2022) on GHG emissions found that liming increased both microbial and fungal biomass, with a much stronger effect on bacterial biomass. Qiu *et al.* (2024) reported that soil pH exerted a critical, non-linear effect on N_2O emissions through its effect on the soil denitrifying bacteria, with enzymes produced by microorganisms that favour N_2O emissions being highest at a pH of 6.0 and decreasing at higher and lower pH levels. Consequently, shifts to more acidic soil conditions would lead to decreased N_2O emissions from acidified soils. It was concluded that the overlap in high N_2O emissions factors and a pH optimal for most crops (i.e. pH 5.5 to 6.5) presented a daunting challenge for reducing N_2O emissions through manipulating soil pH.

Inclusion of organic fertilizers such as manure has a positive effect on soil biodiversity. The effects of synthetic versus organic N fertilizers were assessed in the meta-analyses of Dang *et al*. (2022), Morugán-Coronado *et al*. (2022) and in an influential long-term experiment by Hartmann *et al.* (2015). In their examination of long-term organic and conventional farming, Hartmann *et al.* (2015) found that the application of manure altered the composition of the microbial community, increasing richness, while decreasing evenness (or homogenization) of the soil microbiota and thus creating positive effects for soil functions. Dang *et al.* (2022) summarized 105 studies on microbial biodiversity, finding that although additions of straw and of synthetic N fertilizer reduced diversity, manure

additions increased diversity. Soil pH had the largest effect on biodiversity, and it was suggested that both synthetic N fertilizer and straw probably decreased soil pH, leading to lower soil bacterial diversity. Morugán-Coronado *et al.* (2022) found that while fertilization (i.e. both organic and mineral fertilization) had no particular effect on bacterial or fungal biomass, there was a significant positive effect of organic fertilizer alone on bacterial and fungal biomass. These studies support the finding that organic fertilizers can increase microbial biodiversity compared to synthetic N fertilizer.

A similar effect of synthetic versus organic fertilizers on larger soil fauna was observed by de Graaf *et al*. (2019). It was found that while the diversity of soil fauna (nematodes, microarthropods and earthworms) was significantly and negatively impacted by synthetic N fertilizers, organic fertilizers had no significant effect of on soil fauna (although with large variation in results across the study).

Adoption of reduced tillage or no-till has been shown in many studies to increase soil biodiversity and biomass of organisms. In their meta-analysis, Li *et al.* (2020) showed that adoption of no-till led to increases in bacterial diversity (measured using the Shannon index) but resulted in no statistically significant changes in soil fungal diversity. Moreover, the positive effect of no-till on bacterial diversity only occurred where residue retention was included. Both Morugán-Coronado *et al.* (2022) and Chen *et al.* (2020) found that conservation tillage (no-till and reduced tillage) significantly increased both fungal and bacterial biomass compared to conventional tillage. For earthworms, Briones and Schmidt (2017) performed a meta-analysis showing that no-till alone increased earthworm abundance by a mean of 137 percent and earthworm biomass by a mean of 196 percent and that conservation agriculture (no-till with residue retention or cover crops) increased abundance and biomass by 127 percent and 101 percent, respectively. On average, microbial biomass C and N was also increased by approximately 25 percent with the adoption of cover crops alone, but the effects on soil biodiversity were small or statistically non-significant (Kim *et al.*, 2020).

Finally, Gunstone *et al.* (2021) showed in their summary of studies that 70 percent of pesticide–organism interactions showed negative results, which were evident in both field and laboratory studies, across all the types of pesticides they studied and in a wide variety of soil organisms. It can therefore be concluded that reducing the use of pesticides is important for maintaining soil biodiversity.

5.7.3 Reducing loss of wetlands and organic soils

The need for maintenance or enhancement of the functions provided by wetland soils has become increasingly evident in the past decade, especially as the C storage function of wetlands has become better documented. The conservation of wetlands is first and foremost a land-use decision rather than an alteration of existing management practices, as is the case with many of the SSM practices discussed previously.

The retention or restoration of wetlands should be considered in the land-use decision-making processes. This is most likely to happen if wetland protection is included in policies that govern land-use decisions in a country or subnational jurisdiction. Since conversion to cropland is the major cause of wetland loss, a key element is the incorporation of wetland issues into national policies or measures on agriculture. The 2018 *Global Wetland Outlook* (Ramsar Convention, 2018) reported that of the 170 parties (as of 2018) to the Ramsar Convention, 48 percent had incorporated wetland issues into their national policies on agriculture, 35 percent had partially incorporated them, 4 percent planned on incorporating them and 5 percent had not incorporated them (8 percent did not submit information). By 2018, 73 parties reported that a national Wetland Policy was in place and a further 18 had an instrument under development.

It was estimated that from 1975 to 2015, both inland, and marine and coastal wetlands had declined by approximately 35 percent. While the extent of anthropogenic wetlands (such as those from rice

paddies and reservoirs) had almost doubled during that time to approximately 12 percent of all wetlands, it did not compensate for the loss of natural wetlands. The Ramsar Convention estimated that the average rate of wetland loss since 1907 had been 0.78 percent per year.

The purpose of the Ramsar Convention is to promote wetland conservation and wise use at the global level. The *Global Wetland Outlook* (Ramsar Convention, 2018) urged action at many levels to address wetland loss and degradation, and in particular:

- enhance the network of Ramsar Sites and other wetland protected areas;
- integrate wetlands into planning and the implementation of the post-2015 development agenda;
- strengthen legal and policy arrangements to protect all wetlands;
- implement Ramsar guidance to achieve wise use;
- apply economic and financial incentives for wetland conservation for communities and businesses;
- integrate diverse perspectives into wetland management; and
- improve national wetland inventories and track wetland extent.

5.7.4 Reducing wind erosion

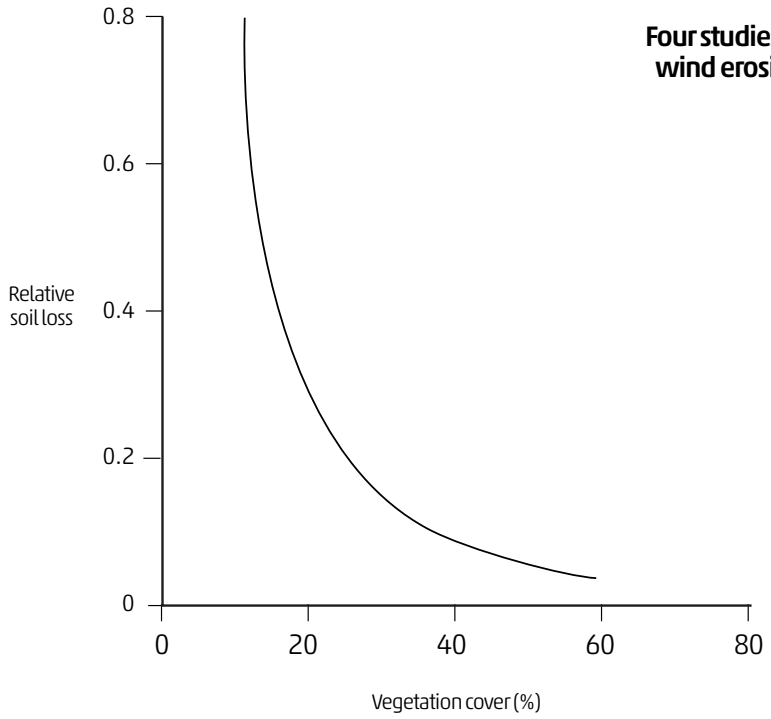
Desertification is land degradation in arid, semi-arid and dry subhumid areas, collectively known as drylands, resulting from many factors, including human activities and climatic variations (UNCCD), 1994). Since 2015 there have been several UN-based summaries of desertification, including the IPCC report on climate change and land (see Olsson *et al.*, 2019; Mirzabaev *et al.*, 2019), the *World Atlas of Desertification* (WAD) (see Cherlet *et al.*, 2018). and the two editions of the *Global Land Outlook* (UNCCD, 2017, 2022).

As emphasized by UNCCD in both editions of the *Global Land Outlook*, the response to desertification must be multifaceted. One element is sustainable cropping, including nutrient conservation, the use of manure, compost, and mulches, fertilizer microdosing, integrated pest management strategies, the selection of suitable and sustainable crop mixes, and soil conservation techniques such as no-till and agroforestry. Other elements include rangeland management, water security, policy incentives, research and capacity building, and investment (UNCCD, 2017).

Wind and water erosion are commonly cited as one of the major soil-related threats in drylands (water erosion is discussed in Section 5.3.1). The management techniques to limit wind erosion are well established (FAO, 2019). Primary among these is the maintenance of a vegetative cover of either growing plants or plant residues on the soil surface. The relationship between relative soil loss is exponential, with wind erosion reductions of 80 to 90 percent above vegetative covers of 40 percent or more (Figure 5.4).



Figure 5.4. Graphical summary of four studies on the relationship between relative soil loss by wind erosion and percent vegetation cover by wheat



◊ *Source*: **FAO.** 2019. *Soil erosion: the greatest challenge for sustainable soil management*. Rome. https://openknowledge.fao.org/server/api/core/bitstreams/a268e967-a373-4a56-adac-ae1d7ab3567f/content

The meta-analysis by Xiao *et al.* (2021) on wind erosion control measures in China showed that (on average), residue retention caused the greatest reduction in wind erosion (about 60 percent), while no-till caused a reduction of approximately 44 percent. The authors suggested that no-till agriculture reduced wind erosion by maintaining a crust on the soil surface, while residue protected the soil surface from the direct wind shear. Cover crops can reduce wind erosion through both vegetative protection of the soil surface and the creation of large soil aggregates that resist detachment by wind, but the water loss associated with cover crops limits any adoption of cover crops in drylands (Blanco-Canqui *et al.*, 2021).

Shelterbelts are the major structural measure to reduce wind erosion (FAO and ITPS, 2021c). Shelterbelts or windbreaks are lines of trees or shrubs planted perpendicular to the prevailing wind direction (or in a criss-cross pattern for multiple wind directions) to form shelter for crops and pastures. Shelterbelts have been shown to reduce wind erosion for a horizontal distance between 12 and 17 times the height of the belt (FAO and ITPS, 2021c). Unfortunately, shelterbelts are being removed by some farmers in Canada, the United States and Europe due in part to the adoption of larger farm machinery and the farmer's perception that the area lost to shelterbelts reduces their income.

5.7.5 Reducing shallow landslides

Shallow landslides are commonly triggered by rainstorms and earthquakes. The rainfall-triggered landslides commonly occur on steep slopes and can be caused by heavy rainfall of either short duration with high rainfall intensity, or long duration with lower intensity (Phillips *et al.*, 2021). The initiation of landslides is a complex interaction of precipitation, physical factors and human-controlled factors such as land use and vegetation.

The major nature-based approach to reduce landslides is vegetation management (Dewi and Wijayanti, 2021). The major beneficial effects of vegetation for landslide protection are due to increases in the strength of the soil due to roots and their binding of soil layers, and to the reduction in the pressure exerted by water in soil pores. Larger plants such as trees can have a negative effect, as they can destabilize slopes by treefall due to the effect of strong winds (Kalsnes and Capobianco, 2022).

Many rainfall-induced landslides are triggered by historical and contemporary forest clearance and disturbance (Phillips *et al.*, 2021). Forest removal is associated with a "window of vulnerability" occurring approximately 3 to 20 years after forest removal, due to a reduction in soil strength caused by the decay of the remaining roots prior to the establishment of any new forest cover. Phillips *et al.* (2021) stated that landslide rates increased during this window by approximately two- to tenfold compared to undisturbed forests.

Shallow landslides in urban areas are the major source of fatalities associated with landslides and are often linked to inappropriate surface modifications (Ozturk *et al.*, 2022). Measures such as the re-establishment of tree cover and eliminating poor drainage or leaking pipes can reduce the incidence of these catastrophic events.

The reestablishment of forest in vulnerable areas is the major management measure used to reduce landslide risk in countries like New Zealand. In many cases, this has involved blanket afforestation of reforestation of entire catchments, in part because the knowledge required to target specific portions of catchments is missing.



5.8 Summary

The linkages between soils and the SDGs have been examined in Chapter 2 of this report. In it, the contribution of soil science is deemed to be greatest for SDG 2: Zero Hunger, SDG 3: Good Health and Well-being, SDG 6: Clean Water and Sanitation, SDG 13: Climate Action, and SDG 15: Life on Land. In this final section, the contributions to each of the SDGs of the management practices discussed previously are briefly summarized.

An overarching measure to reduce the harmful impact of soil threats is to reduce the conversion of natural lands, especially forests, to other uses. This has been emphasized throughout this report and is a major factor in the regional assessment of the state and trend of soil threats.



SDG 2: Zero Hunger

Sustainable Development Goal Target 2.4 has the greatest relevance for soil science:

> *By 2030, ensure sustainable food production systems and implement resilient agricultural practices that increase productivity and production, that help maintain ecosystems, that strengthen capacity for adaptation to climate change, extreme weather, drought, flooding and other disasters and that progressively improve land and soil quality* (UN, 2022).

Reducing water erosion, gullying (see Section 5.3.2) and shallow landsliding (see Section 5.7.4):

- Maintaining a vegetation cover of either living plants or crop residues on the soil surface substantially reduces the rate of erosion. This can be achieved by no-till with residue retention on the soil surface, planting cover crops or other mulches.
- Structural measures such as terraces and contour bunds of soil or stone are also effective in reducing erosion.
- The benefits in erosion reduction and in yields are greatest for no-till with residue retention in semi-arid and arid climates, and cover crops in more humid environments.
- Shallow landslides can be minimized through the maintenance of a cover of deep-rooted trees or shrub vegetation.
- Ideally, any of these measures should be implemented as part of an overall landscape approach that targets especially vulnerable parts of the landscape.

Improving nutrient management and reducing nutrient mining (see Section 5.2.6 and Section 5.3.3):

- The foundation of sustainable nutrient management is a proper nutrient balance, where there is balance between nutrient inputs (including micronutrients) to the soil and nutrient outputs in plant products through time.
- The 4Rs approach (right source, right rate, right timing and right placement) (Johnson and Bruulsema, 2014; Davidson *et al.,* 2016) has been adopted in many countries to achieve sustainable nutrient balances.
- There are major yield benefits to the use of EEFs as a source but the cost of the EEFs outweighs the yield benefits across studies.
- Selection of the right rate requires information on the nutrient available in the soil

through soil testing and the nutrient requirements of the crop to be grown. Optimum placement and timing are dictated by site-specific conditions.

- Nutrient mining is most prevalent in sub-Saharan Africa and the Pacific Island countries. Nutrient mining can be identified through assessment of the nutrient balance- but the determining factors are normally socioeconomic and are considered in Chapter 6 of this report.

Reducing acidification (see Section 5.3.4) and salt accumulation (see Section 5.3.5):

- Acidification of soils can reduce their fertility and hence overall productivity. Liming is a well-established practice to raise the pH and improve soil fertility and plant yields.
- Human-induced salinity due to improper irrigation can be addressed through the provision of adequate drainage to remove salts from the rooting zones, but this requires the safe disposal of the saline drainage water.

Reducing loss of agricultural land to urban expansion (see Section 5.3.6):

- Decisions about land-use conversion should always include consideration of the societal consequences of loss of agricultural land and how the extent of sealing of the land surface can be minimized.



SDG 3: Good Health and Well-being

Of greatest relevance to soil science is SDG Target 3.9: By 2030, substantially reduce the number of deaths and illnesses from hazardous chemicals and air, water and soil pollution and contamination (UN, 2022).

Reducing soil pollution and remediating polluted sites (see Section 5.6.2):

- International agreements in place need to be enforced and extended to limit and eliminate the use of particularly harmful contaminants.
- Extending the range of phytoremediation and microbial bioremediation will assist with the remediation of legacy polluted sites.
- New research on emerging contaminants in the soil should inform new policy development.

Reducing ammonia emissions from soils (Section 5.6.4):

- Harmful emission of NH_3 to the atmosphere can be reduced by use of specific EEFs, use of non-urea sources of synthetic fertilizers and deep placement of N fertilizers.



SDG 6: Clean Water and Sanitation

Of greatest relevance to soil science is SDG Target 6.4: Substantially increase water-use efficiency and address water scarcity:

- The maintenance of good soil structure can be achieved by minimizing physical deterioration of soil assists in both minimizing runoff and reducing waterlogging. The

accurate assessment of workability for mechanical operations can reduce physical deterioration in both agricultural and forestry operations.

Increasing water-use efficiency in agriculture (see Section 5.4.3):

- In addition to the measures used to reduce soil erosion, both runoff and flooding can be reduced by measures to retain water on the soil surface include grass strips, shrub and tree buffers, riparian vegetation and small basins and dams placed in areas of water collection across slopes.
- Many of the measures used to reduce runoff can also be used to increase water retention in the landscape in water-limited areas. Mulch covers can also reduce evaporation for the soil surface.

Reducing agrochemical pollution of water (see Section 5.4.4):

- Significant reductions in harmful N leaching to groundwater can be achieved through use of EEFs, the application of organic amendments such as manure and biochar and of the right rate of fertilizer application, the adoption of no-till, the inclusion of legumes in the crop rotation, and the deployment of drip or reduced amounts of irrigation water.
- Most of the measures that reduce N leaching are also associated with reductions in N runoff, with significant reductions associated with EEFs, organic amendments and the right rate of fertilizer inputs.
- The measures discussed in Section 5.2.1 to reduce soil erosion are critical to reduce P losses from surface erosion of P-associated mineral particles.



SDG 13: Climate Action

The target of particular relevance to soil science is SDG Target 13.2: Integrate climate change measures into national policies, strategies and planning. The specific indicator is Indicator 13.2.2: Total greenhouse gas emissions per year (UN, 2022).

- The IPPC (Nabuurs *et al*., 2022) estimated that both reductions in deforestation and degradation and C sequestration measures in agriculture had similar cost-effective potentials for reducing GHGs of about 3.4 billion tonnes of CO_{2eq} annually. Measures that reduce CH_4 and N_2O emissions have smaller cost-effective potentials of 0.6 billion tonnes of CO_{2eq} annually (Section 5.5.1).

Increasing soil organic carbon content (see Section 5.5.3):

- Preservation of existing stores of SOC is critical to prevent further emissions of SOC to the atmosphere and to maintain the soil functions provided by the soils in their present state.
- Soil organic carbon content of the soil can be significantly increased by the adoption of SSM practices in croplands and grasslands.
- The greatest increases are associated with the addition of C-rich material such as biochar and manure. Both may require processing and transport and hence a full lifecycle analysis is required.
- Cover crops can significantly increase SOC, but results are more variable in water-limited environments.

- Adoption of no-till or reduced tillage generally increases SOC stores of the topsoil but may lead to decrease in stores in subsurface layers. Often, no change is observed over the whole soil profile.
- Retention of plant residues on the soil surface significantly increases SOC stores.
- Cover crops, the adoption of no-till or reduced tillage and the retention of plant residues all significantly reduce soil erosion and the loss of SOC through erosion.
- Forest harvest practices that remove limbs and the first surface organic layer increase the loss of SOC from forest stands.

Reducing nitrous oxide and methane emissions from soil (see Section 5.5.4):

- Any measures that improve NUE (see Section 5.3.2) are likely to lead to reductions in N_2O emissions.
- The use of EEFs and biochar can substantially lower N_2O emissions.
- Deep placement of synthetic fertilizer has also been shown to substantially decrease N_2O release from soils.
- Reductions in tillage have been shown in several meta-analyses to have minimal overall impacts on N_2O and CH_4 emissions.
- The replacement of synthetic N fertilizers by organic fertilizers such as manure, on average, can significantly reduce N_2O emissions, although significantly increase emissions of CH_4 and CO_2 and thereby increase the overall GWP of this practice.
- Adoption of intermittent flooding (also called multiple drainage or non-continuous irrigation) in paddy rice production can significantly reduce CH_4 emissions.
- Adoption of no-till and reduced tillage, optimization of the ratio of synthetic N fertilizer and manure and returning straw to the fields post-harvest also significantly reduce CH_4 and total GHG emissions from paddy fields.



SDG 15: Life on Land

Sustainable Development Goal 15: Protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss.

Halting the loss of soil biodiversity (see Section 5.7.2):

- Managing soil pH so that it does not depart from the optimum range of 5.5 to 7.0 helps maintain soil biodiversity and ensures that the contributions of soil-dwelling organisms to soil functions are optimized.
- Inclusion of organic fertilizers such as manure has a positive effect on soil biodiversity.
- The adoption of reduced tillage or no-till has been shown across many studies to increase soil biodiversity and biomass of organisms, especially larger soil fauna.
- Reduced use of pesticides is important for maintaining soil biodiversity.

Reducing loss of wetlands and organic soils (see Section 5.7.3):

- The retention or restoration of wetlands should be considered in land-use

decision-making processes. This is most likely to happen if wetland protection is included in policies that govern land-use decisions in a country or subnational jurisdiction.

- The conversion of wetlands to cropland is the major cause of wetland loss and wetland protection and should be incorporated into national policies or measures on agriculture.
- Broader adoption and full implementation of the Ramsar Convention would better protect wetlands.

Reducing wind erosion and desertification (see Section 5.7.4):

- The UNCCD has proposed agriculture practices such as nutrient conservation, the use of manure, compost and mulches, fertilizer microdosing, integrated pest management strategies, the selection of suitable and sustainable crop mixes, and soil conservation techniques such as no-till and agroforestry to combat desertification.
- The maintenance of a vegetative cover of either growing plants or plant residues on the soil surface is the key measure in combating wind erosion.
- Shelterbelts are an effective method to reduce wind velocity and hence reduce wind erosion.



Conclusion

Rietra, Heinen and Oenema (2022) reviewed the results of existing meta-analyses on crop husbandry and soil management practices and concluded that:

> *While global assessment studies often paint rather pessimistic views on the state of food production, agriculture, and the environment, it is clear that the 174 studies reviewed here present a picture of optimism and hope. Indeed, there is large body of scientific/empirical evidence that some specific practices are more effective than others, i.e., have positive effect sizes relative to conventional practices), and that these positive effects may contribute to the sustainability of crop and food production* (Rietra, Heinen and Oenema, 2022, p. 19).

They found that the benefits of the management practices were relatively large for environmental effects such as nutrient leaching and GHG emissions, but were smaller for crop yield.

The results summarized in this chapter would support the optimistic opinion of Rietra, Heinen and Oenema (2022) that there is considerable scope for increasing cropland productivity and minimizing environmental impacts with the coordinated implementation of currently available management practices.



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Chapter 6 | Facilitating the adoption of sustainable soil management

6.1 Introduction

This chapter examines the policy frameworks, institutional support, economic incentives and capacity-building efforts necessary to encourage the widespread adoption of SSM. As documented in Chapter 5, the adoption of SSM practices can substantially reduce the threats to soil functions, thereby maintaining and enhancing the contributions that soils make to human societies and the SDGs. However, although considerable progress has been made in the adoption of some SSM practices, the seven regional assessments in this report document many cases of continuing high rates of soil threats that negatively affect soil functions. The goal of this chapter is to report the challenges to adoption that have been explored in the literature, and the approaches that can remove the barriers, drawn from successful examples of SSM adoption.

Literature on the adoption of SSM (and sustainable agricultural management more generally) is dominated by economic considerations. The impact of the adoption of specific techniques on crop yield is the most studied area (e.g. Pittelkow *et al.*, 2015; Ye *et al.*, 2020). However, a broader economic analysis considers the yield changes in the context of other costs such as labour, inputs, crop prices and machinery requirements that determine overall profitability of the farming operation (e.g. Che *et al.*, 2023; Begho, Eory and Glenk, 2022). A more holistic socioeconomic approach attempts to understand the decisions made by farmers about adoption within the context of Indigenous Peoples' knowledge systems – which involve their ancestral, traditional and innovative knowledge – together with other traditional knowledge held by the community and the worldview held by members of that community (Kelly *et al.*, 2020). Although the importance of a spiritual worldview is perhaps most evident for many Indigenous Peoples, it can also influence the adoption of SSM techniques for environmental benefits that are not linked to tangible financial rewards. The non-economic considerations are often linked to an ethos of care or stewardship held by members of the community (Ataria *et al.*, 2023).



6.2 Traditional and Indigenous Peoples' knowledge

A major point of agreement across the literature on the adoption of SSM is that initiatives that do not respect traditional and Indigenous Peoples' knowledge often fail. In this context, respect means soliciting information from Indigenous Peoples and other population groups, and evaluating the information gathered using a broader lens than just the western scientific perspective. Innovative ways of creating and co-creating knowledge can be explored, where Indigenous Peoples' knowledge is treated with the same level of respect as academic knowledge – including the representation of Indigenous Peoples in the development of initiatives that may affect their lands, territories and resources – ensuring continuing dialogue during implementation to address issues as they arise.

The need to involve Indigenous Peoples' knowledge systems and other traditional knowledge is recognized across the various perspectives on facilitating adoption. For example, the focused review on economic incentives by Piñeiro *et al.* (2020) stated that a key policy recommendation was to:

> *Know your farmers. Policymakers must be familiar with the farmers, and tailor the incentive programmes for them by incorporating the range of personal, political, institutional and biophysical factors into the design of the programme (Piñeiro et al., 2020, p. 817).*

A second recommendation was:

> *Behavioural preferences matter. Given that people tend to follow the behaviour of others, farmers' preferences should be taken into account when designing incentives, acknowledging that they vary depending on the target population (Piñeiro et al., 2020, p. 817).*

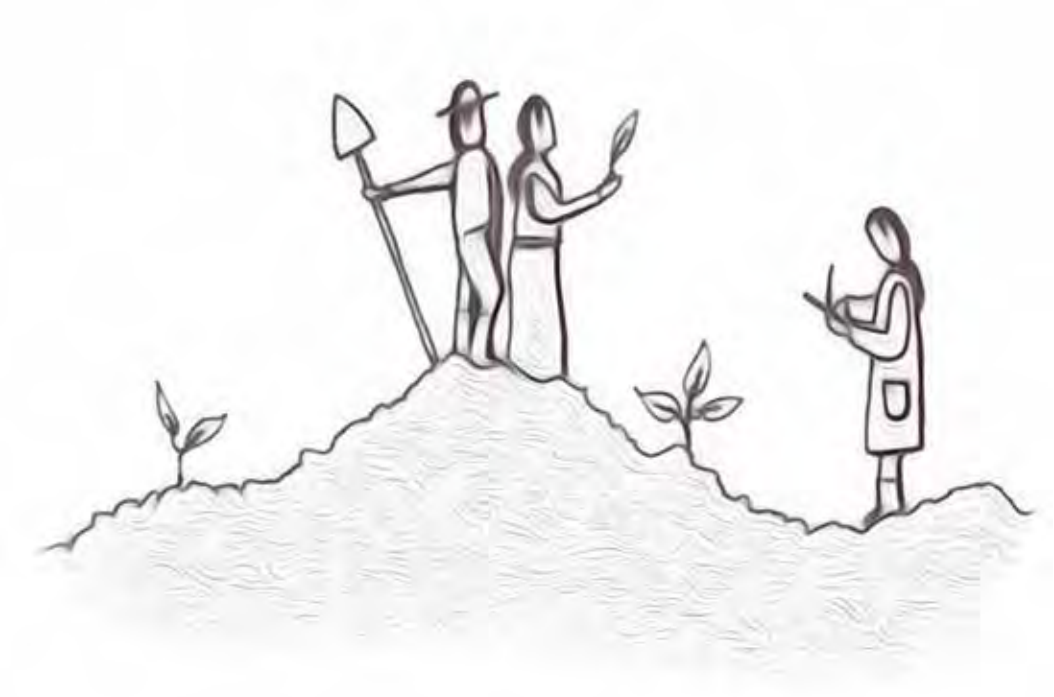
Participation by Indigenous Peoples (and of other population groups) is also one of the key principles of the FAO framing of governance:

> *Individuals should participate in the planning, design, monitoring and evaluation of decisions that affect their rights. The principle of participation means that people should be able to determine their own well-being. The effectiveness of stakeholder participation is contingent upon a consideration of socioeconomic and institutional contexts, and a selection of appropriate tools to foster participation (for example public hearings, stakeholder bodies, website forums, etc.) (FAO and UNEP, 2020, p. 33).*

In addition, and grounded in the provisions of UNDRIP (UN, 2007), Indigenous Peoples hold the collective right to participate in decision-making processes that may affect their lands, territories, resources and ways of life (Article 26 and Article 32). The Declaration affirms the right of Indigenous Peoples to self-determination (Article 3), including the right to freely determine their political status and pursue their economic, social, and cultural development. Article 4 further recognizes their right to autonomy and self-government in matters relating to their internal and local affairs, while Article 18 and Article 19 guarantee Indigenous Peoples the right to participate in decision-making through their own representative institutions and require their free, prior and informed consent (FPIC) before adopting measures that may affect them. Participation is essential for securing the legitimacy of governing systems, defined by Juerges and Hansjürgens (2018) as the acceptance and agreement of the governed individuals to the rules associated with implementation, even if these rules might work against their own interests in the short term. Participation by all relevant actors (especially groups in situations of vulnerability) is a crucial component for the success of sustainable soil governance.

The need to respect the knowledge of Indigenous Peoples and other groups may also not always be fully aligned with national-level soil governance systems of the type discussed in Section 6.1. For example, Kelly *et al.* (2020), in their examination of governance issues around soil erosion action in East Africa, found that national-level policies often diverged from community-level governance practices that had developed over generations. These community-level practices dealt with issues such as damage done to terraces by cattle movement or young goats eating planted seedlings that were not anticipated by the national-level policies. This reinforces an earlier point that such policies should not dictate specific practices to be generally adopted but instead focus on broader principles.

In summary, the common theme across many practitioners in this area is that the adoption of new approaches to management will not be successful unless planners are cognizant of all the influences on local decision-making. For example, Hermans *et al.* (2021) developed an integrated soil health assessment system by working closely with farmers in Malawi. The assessment was developed by discussing the impact of the proposed management change on soil health, with farmers identifying farmer soil health indicators and comparing these with literature, taking soil measurements of the indicators with the help of farmers at the on-farm trials, and discussing the soil measurements results and their connection to farmer observations. It also allowed the agricultural researchers to understand how a soil health assessment was placed within the farmer's overall decision-making process, which included other factors such as labour requirements, resource availability (such as fertilizers and pesticides) as well as social acceptability and dynamics. The failure of researchers to understand the importance of soil information within the overall decision-making process used by farmers has been widely cited as a common failing in development projects (e.g. Giller *et al.*, 2009; Fairhead and Scoones, 2005).



6.3 Soil information

The need to provide useful and accessible soil information, technical advice and facilitation to support land managers in their adoption of SSM has been widely identified in the literature (Ingram *et al.*, 2022; Lobry de Bruyn and Prager, 2024; Smith *et al.*, 2024). Despite this clear need, many problems remain with the delivery of appropriate and useful information. Lobry de Bruyn, Jenkins and Samson-Liebig (2017) stated that:

> *As scientists or policymakers, we have largely assumed what type of soil information landholders or the public are interested in, rather than asking them, as well as the avenues they pursue to seek information, and the forms of delivery they find most accessible (Lobry de Bruyn, Jenkins and Samson-Liebig, 2017, p. 431).*

Soil information in the context of adoption of SSM comprises facts, interpretations, culturally appropriate information and projections that reduce the uncertainty faced by decision-makers (Ingram and Mills, 2019). To be useful, information is often presented to land managers as a recommendation of a particular course of action or as a range of alternatives, and therefore the development of recommendations is a key role for skilled local advisors who can take account of local conditions.

A key requirement for all levels of soil information is that it must be accessible to potential users. Lobry de Bruyn and Prager (2024) used four criteria of accessibility in their study of soil information in the United Kingdom and Australia:

1. How readily discoverable is it?
2. Is the language appropriate for the audience?
3. Is the information presented culturally appropriate?
4. Is it available under an open-source licence?
5. Is there interpretive material to outline the management implications of the information?

These accessibility issues can be especially acute for legacy data that were gathered for one objective but are now being deployed for a second, newer objective, such as soil health assessments.

The recent rise of public interest in soil health has also expanded the categories of information that must be provided to potential adopters of SSM approaches (Lobry de Bruyn and Andrews, 2016; Hermans *et al.*, 2021). Previously, the main categories of soil information were fertility recommendations and practices to reduce specific soil threats (such as erosion or salinization) as appropriate to the landscape setting of the soil manager. Although definitions of soil health differ in their details, the importance of life in the soil is common to all. However, creating accessible information on soil biodiversity and its links to soil functions and to appropriate management practices has been challenging (Lehmann *et al.*, 2020). The broader link between soil health and ecosystem functions also requires an understanding of the major environmental impacts of inappropriate soil management and this knowledge may not be present in many of the providers of advisory services (Veenstra *et al.*, 2024; Ingram and Mills, 2019; Lobry de Bruyn and Andrews, 2016).

The difficulties with communicating information about soil biodiversity are similar to those encountered more generally in biodiversity. Globally produced modelling data on biodiversity have often been shown to conflict with local or national scale information, leading to concerns about the usefulness of global data overall (Buschke *et al.*, 2023). The need for useful data requires action at several levels, ranging from basic science addressing the credibility and validation of data through

to user-focused requirements of capacity and awareness (Table 6.1). Many of the issues evident in biodiversity studies are more acute for microscopic soil organisms. Indicators such as microbial biomass, biodiversity and microbial activity which commonly appear in assessments of soil health (Lehmann *et al.*, 2020) also have issues with availability and usability.

Table 6.1. Seven preconditions of useful biodiversity and soil health information to improve the uptake of soil information in policies and programmes

Preconditions	Supporting actions
Awareness: Users should know the data exist.	**Outreach**: Improve marketing, advertising, publicizing and communication.
Capacity: Users should have the technical ability to use the data.	**Training**: Produce tutorials, user manuals, demonstrations or present training workshops.
Infrastructure: Users should have sufficient internet access and computing power.	**Data optimization and investment**: Package data more efficiently (for example, for mobile access) and enhance or share information and communications technology infrastructure.
Usability: Information should be in an easy-to-use and accessible format and language.	**Design and translation**: Design data platforms to be more user-friendly and intuitive and produce jargon-free explanations in multiple languages.
Legitimacy: Data should be considered authoritative.	**Advocacy and diplomacy**: Advocate and negotiate so that data are accepted by national and international authorities.
Saliency: Data should be fit for purpose.	**Engagement and exchange**: Interact with end users to better understand their needs.
Credibility: Data should be scientifically robust and representative.	**Validation**: Ensure the scientific validity of data through independent verification and peer review.

◊ *Source:* Adapted from **Buschke, F.T., Capitani, C., Sow, E.H., Khaemba, Y., Kaplin, B.A., Skowno, A., Chiawo, D.** *et al.*, 2023. Make global biodiversity information useful to national decision-makers. *Nature Ecology & Evolution*, 7(12): 1953–1956. https://doi.org/10.1038/s41559-023-02226-2

The problem with the legitimacy of global-scale modelling information also occurs with other soil threats. For example, the global erosion models reviewed in Section 3.2 are often challenged in the literature. Often, when erosion models have been compared to field-based erosion monitoring schemes, there is poor agreement between the field-based measurements and the soil erosion models (Batista *et al.,* 2019). In many cases, field data has been used during model development for the calibration of specific elements of the model and hence the data available for validation is further diminished (Jarrah *et al.*, 2020). The review of soil erosion modelling by Borrelli *et al.* (2021) concluded that many modelling studies lacked validation attempts and that new regional and national assessment efforts such as the country-based global soil erosion assessment proposed by the UN (FAO, 2019) needed to include well-designed field-based data collection to evaluate the results of the models.

A second scale-related issue with soil information occurs at the other end of the scale range, with the need for site-specific information required for the successful implementation of many SSM techniques. A commonly cited reason for incomplete or failed programmes for SSM adoption is the mismatch between generalized recommendations and information appropriate for national or regional policies, and the specific soil, socioeconomic and cultural context of local adoption (Lobry de Bruyn and Prager, 2024). Typically, maps for land management need to be at a 1:5 000 scale (i.e. 1 cm on the map equals 5 000 cm or 50 m of the ground surface), while many existing soil resource maps are at scales of approximately 1:100 000 (i.e. 1 cm on map equals 1 km) or coarser (Lobry de Bruyn and Prager, 2024).

The knowledge about soil properties required for the successful adoption of new management practices includes basic soil properties such as pH, soil organic matter content and soil texture, as well as an understanding that some soil properties can be modified by past management and hence differ across fields, even within relatively homogenous landscape units (Sileshi *et al.*, 2022). Sileshi *et al.* (2022) analysed the reasons why some African soils failed to show a response (in terms

of yield gain and agronomic efficiency) to added N and P fertilizers, and found that the probability of non-response was as high as 26 percent on some soil types. The risk of non-response limits the adoption of yield-enhancing synthetic fertilizers and highlights the need for site-specific, tailored management recommendations for nutrient inputs if farmer reluctance to adopt SSM practices is to be overcome.

Importantly, the site-specific data required for the successful implementation of SSM cannot be downscaled from the global maps of soil properties that have been useful for regional and global assessments of the spatial pattern of soil properties (see Chapter 3). Poggio *et al.* (2021) stressed that even sophisticated products such as SoilGrids250m were not meant to be used at a detailed scale and suggested that national data providers often had access to more detailed point data sets and information on related properties than were available to global modelling groups. Information on the most management-sensitive properties (such as nutrient levels, soil organic matter and many measures of soil life), require field-specific sampling for reliable assessments.



6.4. Education and public awareness

The production of soil information is only the first stage in its role of facilitating the adoption of SSM. The information must reach the end users in a format that can be understood and with clear steps to be acted upon. Moreover, Mason, Cornu and Chenu (2023) showed that the end users or stakeholders had a wide range of interest in the use of the information, identifying six classes of stakeholders (farmers, public authorities, advisors, NGOs, researchers, Indigenous Peoples and agricultural schools). The different stakeholders accessed information from many sources: farmers, for example, preferring farmer-to-farmer groups, whereas researchers had a clear preference for the scientific literature (Mason, Cornu and Chenu, 2023). The main barriers that limited access to the relevant information were the lack of connection between stakeholders and the absence of structures that enabled two-way sharing of knowledge. The authors argued for the role of a scientific mediator (or scientifically literate farm advisor) to bridge the gap between researchers and the various stakeholders, as well as to provide land managers with a better understanding of research results.

Farmer-focused research in many countries has promoted the role of farmer-to-farmer meetings to inform possible users and overcome the social and cultural barriers to the adoption of SSM. For instance, farmer-to-farmer meetings were a critical stage in the adoption of no-till in the Canadian Prairies (Awada, Lindwall and Sonntag, 2014) and in the similarly widespread adoption of no-till in Argentina (Peiretti and Dumanski, 2014). Surveys in Europe (Vanino *et al.*, 2023; Mason, Cornu and Chenu, 2023; Ingram and Mills, 2019) and Australia and the United States of America (Lobry de Bruyn and Andrews, 2016) all emphasized the importance of farmer-to-farmer information sharing.

A European study on the skills required for the transition to sustainable agriculture found a clear need for more information on soil health and life in the soil on the part of stakeholders (Veenstra *et al.*, 2024). The skill most sought by stakeholders was knowledge on the ecological functioning of soils, emphasizing the importance of soil biodiversity, plant–soil interactions and the role of soil in biogeochemical cycling. The second highest was on understanding how to use cropping systems and plant cover to protect soils, related to expertise on agricultural systems being linked to SSM. However, traditional agronomic skills were still highly valued, emphasizing the need for a more holistic appreciation of the soil (Veenstra *et al.*, 2024).

The role of advisory services is central to the communication of soil information to users. Ingram and Mills (2019) defined advisory services as sets of organizations that supported and facilitated land managers to solve problems and to obtain information, skills and technologies. Studies of the changes in advisory services through time in Europe (Ingram and Mills, 2019), Australia, and the United States (Lobry de Bruyn and Andrews, 2016; Lobry de Bruyn, Jenkins, and Samson-Liebig, 2017; Lobry de Bruyn, 2019) all emphasized the changing structure of advisory services through time that had seen the decline in government funding of extension in many countries and the rise of the private sector. For much of the twentieth century, information generated by researchers was published in peer-reviewed journals and communicated to potential adopters by government extension workers in a linear information transmission model. Despite the rise of the private sector in advisory services (comprising of a diverse mix of public, private [supply chain and consultants], NGOs and farmer-based organizations [such as farmer unions, associations and cooperatives]), consistent communication of soil advice can be difficult when organizations have differing objectives, priorities and delivery approaches (Ingram and Mills, 2019).

At the same time that the range of providers has expanded, the sources of information have become vastly diversified, due to the rise of the digital information age and the associated changes in the relationship between science and society (Bouma, 2019). This has led to a greater amount of information but with less checks and balances than were previously used in the pre-internet era (Lobry de Bruyn, Jenkins and Samson-Liebig, 2017).

Soil science education at colleges and universities in North America appears to be on the rebound after several studies from the 2000s had shown decreases in enrollment in soil science courses. Studies of universities in the United States (Brevik *et al.*, 2018) and Canada (Diochon *et al.*, 2016) have shown stable or increasing enrollments across many departments. Importantly, the content is being taught across a much broader range of departments than was the case in the twentieth century: for example, in Canada, the majority of soil courses are taught by Geography and Environmental Science departments (Diochon *et al.*, 2016).



6.5. Economic benefits

Piñeiro *et al.* (2020) stated that "linking programmes to economic benefits (productivity or profitability) is essential for farmers to adopt sustainable agriculture practices in the short term" (Piñeiro *et al.*, 2020, p. 810). A review by Giger *et al.* (2018) on the benefits and costs of adoption of sustainable land management (SLM) technologies documented in the WOCAT's Global Sustainable Land Management Database (WOCAT, ELD and UNCCD, 2022; WOCAT, n.d.) came to a similar conclusion. In their survey of adopters, the most frequently mentioned motivations for adopting SLM technology were production (24 percent), increased profitability (20 percent), well-being and livelihood improvement (20 percent) and reduced workload (5 percent). These are all directly or indirectly related to economic considerations, and together, they represent 69 percent of the listed motivations of adopters.

The initial measure of productivity used in most studies is the change in crop yield associated with the adoption of a new SSM technique, compared to non-adoption. Profitability requires an assessment of the direct costs and benefits for the implementation of a new technique, ideally, an assessment of costs and benefits associated with changes to farm operations and in some cases, commodity pricing. These costs and benefits have been examined in Section 5.2 and examples can be briefly summarized for the major cross-cutting SSM techniques as follows:

- **Manure**. No additional yield benefit was shown beyond that associated with the N and P in the manure. The yield benefit was similar to the benefit associated with equivalent amounts of N and P in synthetic fertilizers (Celestina *et al.,* 2019; Hijbeek *et al*., 2017).

- **Cover crops**. Subsequent cash crop yields following cover crops showed an average increase of 4.3 percent (Jian *et al*., 2020). In water-limited environments (i.e. locations with less than 500 mm annual precipitation), cover crops reduced yields in the subsequent crop by 38 percent, did not affect crop yields in 56 percent of comparisons and increased yields in 6 percent (Blanco-Canqui *et al*., 2022). In studies with low mean annual rainfall (<300 mm), cover cropping led to a significant reduction in cash crop yield of 29 percent (Garba, Bell and Williams, 2022). Where annual precipitation exceeded 700 mm, a neutral cash crop yield response to cover cropping was observed. In higher rainfall regimes (>900 mm), cover cropping increased cash crop yield by 14.5 percent (Garba, Bell and Williams, 2022).

- **Residue retention/return**. Residue retention alone had a significant beneficial effect (with median yield increases of 8.2 percent) (Xiao *et al*., 2021). In a study based on data from 30 long-term experiments (MacLaren *et al*., 2022), no significant effect of retaining rather than removing crop residues on crop yields was observed.

- **Enhanced efficiency fertilizers**. Enhanced efficiency fertilizers increased yield on average by 24 percent compared to conventional fertilization (Gu *et al*., 2023). Li *et al.* (2018) also found a similar yield benefit to EEFs but the increases in yield did not cover the costs of EEFs relative to conventional fertilizer for maize and rice crops: only rice showed a sufficient yield increase to cover the increased cost of EEFs.

- **Biochar.** Biochar applications on their own, or in combination with synthetic fertilizer and organic amendments, caused an overall mean yield increase of 9.9 percent compared with fertilizer control treatments (Ye *et al*., 2020). Reasonable assumptions about the cost of biochar and the yield benefit indicate that biochar applications to crops are not economical, based on the yield increase alone, due to high cost of biochar (Bach, Wilske and Breuer, 2016).

- **No-till/reduced tillage**. Adoption of no-till alone led to a significant median decline in crop yield of 1.9 percent and no-till combined with residue retention led to a median yield increase of 6 percent (Xiao *et al*., 2021). No-till performed best under rainfed management in dry climates, where no-till matched yields from conventional tillage

for maize and legumes or exceeded yields for wheat, rapeseed and cotton (Pittelkow *et al.*, 2015a). Short-term yield losses after the adoption of no-till were often followed by yield increases over conventional in the longer term (Cusser *et al.*, 2020; Che *et al.*, 2023).

- **Organic agriculture**. Across 786 comparisons, overall yield under organic production systems was reduced by 18.4 percent compared to conventional agriculture (De La Cruz *et al.,* 2023), while Smith *et al.* (2020) found that organic sites had 18 percent lower yields than conventional sites. Average yield reductions were greatest for root and tuber crops (29 percent) followed by leguminous crops (25 percent), cereal crops (18 percent), rapeseed and oleaginous crops (16 percent) and vegetable and melon crops (15 percent) (De La Cruz *et al.,* 2023). Based on United States data, organic farms are substantially more profitable, primarily due to price premiums for organic products (Smith *et al.*, 2020).

This brief review supports the conclusion of Rietra, Heinen and Oenema (2022) that crop yield increases associated with the adoption versus the non-adoption of sustainable management practices are small. For biochar and EEFs, the yield increases are relatively high, but the high cost of the products themselves makes the widespread application of them uneconomical.

Rietra, Heinen and Oenema (2022) also concluded that while the environmental benefits of SSM techniques were relatively large compared to the observed yield increases, unfortunately the environmental benefits did not normally lead to direct economic benefits to those managing the soil. The longer-term environmental benefits were an important consideration in adoption of SSM techniques by farmers, but the lack of meaningful short-term benefits could limit adoption in the absence of incentives. The limits to adoption are perhaps most pronounced for relatively expensive products such as biochar (Bach, Wilske and Breuer, 2016) and EEFs (Li *et al.*, 2018).



6.6. Incentives for adoption

While much emphasis is placed on governments to design effective policies for SSM, ultimately it is the actions or inactions of individuals that privately own or rent land that determine the implementation of SSM and governments can only support those actions with direct or indirect incentives (Lobry de Bruyn and Prager, 2024; Bartkowski and Bartke, 2018; Montanarella *et al*., 2016).

Due to the lack of substantial yield benefits for the adoption of SSM, or the high cost of specific products such as EEFs or biochar, the availability of incentives to facilitate adoption may be required. Piñeiro *et al.* (2020) defined incentives as:

> *instruments used by the public and private sectors to encourage farmers to protect or enhance ecosystem services beneficial to them and others (for example, water quality, soil care, forestry), while simultaneously improving the productivity (yields, labour per hectare and so on) and the competitiveness (such as cost per hectare, profitability, farm incomes) of the agricultural sector (Piñeiro et al., 2020, p. 810).*

Piñeiro *et al.* (2020) also discussed four categories of incentives:

1. **market** (e.g. prices of inputs or outputs, subsidies);
2. **non-market** (technical support, technology transfer and fiscal measures, such as a price on carbon);
3. **regulatory** (e.g. certifications and environmental laws and standards, which are typically mandatory); and
4. **cross-compliance** (e.g. payments for ecosystem services or agri-environment payments).

Cross-compliance measures are commonly cited in the soil science literature as the most desirable way to foster adoption of practices that are deemed to have environmental benefit while lacking short-term economic benefit. Incentives are offered to farmers or landowners in exchange for managing their land to provide some type of ecological service, such as improved water quality, the prevention of soil erosion, Increases in C sequestration or reductions in air pollution. Cross-compliance incentives help overcome the substantial short-term economic barriers to the adoption of some SSM techniques, such as large up-front adoption costs, a lack of capital, restricted access to financial markets and the need to provide for the household's short-term economic needs (Piñeiro *et al.*, 2020).

Across the nearly 18 000 papers examined by Piñeiro *et al.* (2020), a general finding was that:

> *if the economic incentives or payment levels do not offset the costs of adoption (cover opportunity costs of changing production techniques or for the most productive land uses), farmers will rarely switch to the desired practices. However, if payment levels compensate, or overcompensate, for income losses and additional costs, then the willingness of farmers to adopt is normally high (Piñeiro et al., 2020, p. 815).*

This finding of Piñeiro *et al.* (2020) is consistent with the results of the survey of adopters of WOCAT's SLM technologies (Giger *et al*., 2018; WOCAT, ELD and UNCCD, 2022). While the long-term benefits of adoption were often clear to adopters, the high costs of the initial investment period could constitute an important barrier to adoption, preventing it from spreading and being scaled up.

Subsidies for inputs are commonly used market-based incentives, but these often have unintended consequences. For example, in South Asian countries, it is common for the use of fertilizers to be

highly subsidized (Begho, Eory and Glenk, 2022; Aryal *et al.*, 2021). The demand for fertilizers (and the environmental consequences of fertilization) have increased since the 1980s due in large part to the adoption of high-yielding crops that require large amounts of fertilizer (Begho, Eory and Glenk, 2022). The higher subsidy available on nitrogenous urea fertilizer has led to preferential use of urea that has acted as a disincentive for use of other secondary nutrients and micronutrients, limiting the yield gains achievable through unbalanced urea application alone (Aryal *et al.*, 2021).

Jayne *et al.* (2018) reviewed studies on fertilizer subsidies in South Asia and Africa, and concluded that broader supports were required if the benefits of subsidies and other incentives were to be realized:

> *it is increasingly apparent that second generation African ISPs have given too much attention to giving fertilizer to farmers and too little attention to enabling them to use it efficiently and profitably (Jayne et al., 2018, p. 12).*



6.7 Soil governance at a global scale

According to FAO and UNEP (2020):

> *For FAO, the concept of governance constitutes the formal and informal rules, organizations, and processes through which public and private actors articulate their interests; frame and prioritize issues; and make, implement, monitor, and enforce decisions (FAO and UNEP, 2020, p.28).*

Soil governance specifically refers to the interaction between different societal actors (such as land managers, government, businesses and interest groups) to affect change and to align decision-making around soil use and management with societal goals and preferences (Lobry de Bruyn and Prager, 2024; Helming *et al*., 2018). Soil governance is commonly viewed as a highly fragmented field at all scales (Juerges and Hansjürgens, 2018; Peake and Robb, 2021; Kelly *et al*., 2020). This makes the coordination of decision-making and policy formulation especially difficult, and the implementation of policies and other decisions is often flawed (Lobry de Bruyn and Prager, 2024: Helming *et al*., 2018).

In their survey of legislative approaches to natural resource governance, FAO and UNEP (2020) stated that recommended practices for SSM, being inherently technical in nature, should typically not be included in primary legislation, as these practices were frequently amended as new scientific information emerged. Therefore, the recommendations on practices were best placed in lower-level instruments. Nonetheless, the adoption of global, regional or national targets related to soils create enabling conditions for the adoption of SSM (Piñeiro *et al.* (2020).

At the global scale, there have been several major developments in soil governance since 2015 (Peake and Robb, 2021; Lal, 2024). First, the GSP have continued to produce foundational and guidance documents for SSM, including the *Revised World Soil Charter* (FAO, 2015), the *Voluntary Guidelines for Sustainable Soil Management* (FAO, 2017) and *The international Code of Conduct for the sustainable use and management of fertilizers* (the Fertilizer Code) (FAO, 2019). These have been supported with many technical documents and meetings (FAO, 2025a) to facilitate the adoption of the approaches outlined in the previously mentioned GSP documents. The scientific body formed to support the work of the GSP, the ITPS, has produced summaries (soil letters) on a range of topics including soil health, salt-affected soils, urbanization and soil sealing, and biochar (FAO, 2025b).

Since 2015 all the scientific support bodies for the Rio conventions (the United Nations Framework Convention on Climate Change [UNFCCC], the Convention on Biological Diversity [CBD] and the United Nations Convention to Combat Desertification [UNCCD]) have also produced major documents that include appreciable soil content. The restoration, maintenance and enhancement of nature's contribution to people, including soil health, is included in Target 11 of the Kunming-Montreal Global Biodiversity Framework adopted by the CBD in 2022 (CBD Secretariat, 2022). The 28th Conference of the Parties (COP) to the UNFCCC adopted the COP28 Declaration on Sustainable Agriculture, Resilient Food Systems and Climate Action that included an objective to maximize the climate and environmental benefits by, in part, enhancing soil health and biodiversity (Lal, 2024).

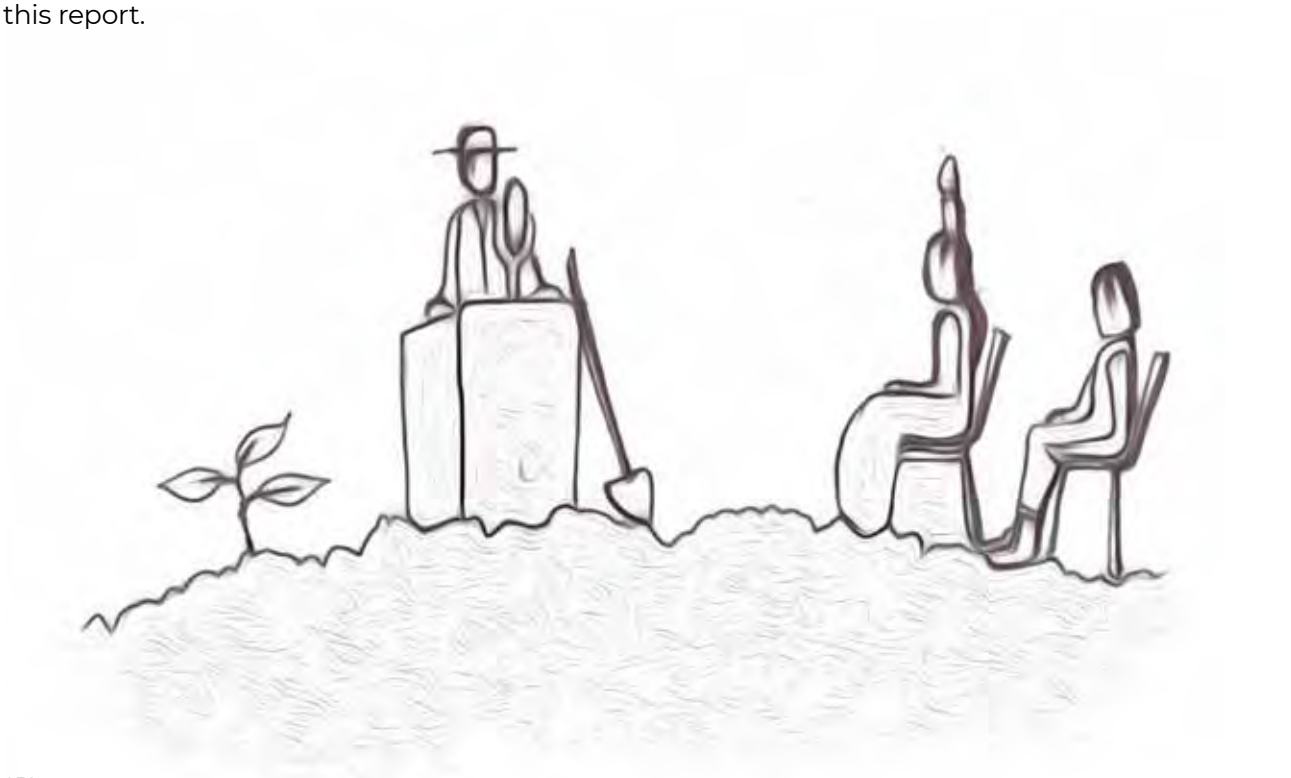
Soil health has also been included in major regional developments such as the Nairobi Declaration on the Africa Fertilizer and Soil Health Summit held in Nairobi, Kenya in May 2024 (African Union, 2024). This document made commitments to create an enabling governance environment to address fertilizer and soil health issues in Africa, including the development of continent-wide, context-specific guidelines for the formulation and implementation of relevant and effective fertilizer and soil health policies, and harmonization of national and regional policies and regulatory frameworks to ensure coherence and promote regional and continental trade. The European Union also released an EU Soil Strategy (EC, 2022) as a component of the European Green Deal.

Although international commitments are important, the FAO and UNEP (2020) survey acknowledged that the implementation of international commitments could be difficult to enforce and monitor as they were aspirational rather than legally binding (Peake and Robb, 2021). The uneven conversion of these aspirations into national soil legislation, and a lack of enforcement and governance on the ground remain major issues.

The survey of nearly 200 countries by Peake and Robb (2021) using FAOLEX (FAO's online database of policies and laws) (FAO, n.d.) found that only 7 percent of countries had explicit soil policies in place at the time of the survey. Laws were in place to explicitly address specific soil issues such as conservation (34 percent) of countries) and pollution (27 percent of countries) and soil sealing as a result of urban expansion (3 percent of countries). Legal instruments to preserve agricultural land (rather than soil services *per se*) were in place in approximately one-third of countries, ranging from policies advocating for the avoidance of urbanization (sometimes known as land take) (24 percent of countries) to laws prohibiting or strongly restricting loss of prime farmland (21 percent).

A further complication concerning governance is the reality that for many countries, the ownership of land (especially farmland) is in private hands, and often public goods (such as clean water and reducing harmful N emissions from soils) conflict with private rights pertaining to land holdings (Juerges and Hansjürgens, 2018). The existing soil laws primarily address threats with significant and well-established offsite consequences (such as pollution and soil erosion) or overall societal concerns (such as the preservation of farmland) (Peake and Robb, 2021). Decisions about the adoption of SSM practices are made by individual landowners and attempts to regulate these often conflict with their private rights (Bartkowski *et al.*, 2018).

Increasingly, the right of Indigenous Peoples to participate or lead governance decisions is being recognized. In some countries, Indigenous Peoples have a legislated right to participate in land-use planning, including the adoption of SSM. FAO and UNEP (2020) highlighted UNDRIP (UN, 2007) as the most positive global affirmation of the rights of Indigenous Peoples. The text of this agreement enshrined the rights of Indigenous Peoples over the lands, territories, resources, culture and knowledge which they had traditionally owned, occupied or otherwise used or acquired, and furthermore encouraged states to legally recognize such rights, with due respect to the customs, traditions and land tenure systems of the Indigenous Peoples concerned. These inherent rights of Indigenous Peoples are exemplified in the major influence that Māori perspectives have had on the governance of soils and other natural resources in New Zealand, as documented in Section 13.4.1 of this report.



6.8. Conclusions

Many of the previously cited articles contain general recommendations for policymakers to facilitate the adoption of SSM (e.g. Piñeiro *et al.,* 2020; Peake and Robb, 2022; Lobry de Bruyn and Ingram, 2019). The major recommendations across these studies include:

- ***Know your potential adopters and their behavioural preferences***: A lack of understanding of Indigenous Peoples' knowledge systems, and other traditional knowledge and practices has been a major factor in the failure of many initiatives. Approaches such as farmer-to-farmer information transfer or genuine attempts to engage in dialogue with potential adopters are essential in the design of future programmes.
- ***Provide accessible information on soil management***: The information for potential adopters and their advisors must be readily available and in a language and format that is intelligible to those people, while respecting the rights of Indigenous Peoples to self-determination and to FPIC. It must also be relevant to the specific conditions relevant to their operations.
- ***Balance the incentives and outcomes***: Incentives must be high enough to motivate a change in practices, as often the short-term productivity and profitability gains are insufficient to balance the costs of adoption.
- ***Target incentives at the sticking points for implementation***: Barriers to adoption are often location-specific, and targeted incentives to remove these barriers must be adopted. For example, support for initial equipment purchase would differ from ongoing supports need for high-cost inputs such as biochar or EEFs.
- ***Complement economic incentives with soil information and training***: A combination of policy instruments is required. For example, the provision of technical assistance and extension services contributes to the understanding of farmers and helps them adopt proposed practices.
- ***Develop national soil-related legislation and policies***: International organizations such as FAO, IPCC and UNCCD have developed important voluntary guidelines, but these need to be translated into national-level action. National-level soil policies that protect soils from degradation and promote soil health, protect prime farmland from urbanization and protect natural ecosystems need to be developed to put the voluntary guidelines into practice.

This combination of good governance, respectful dialogue with potential adopters, scale-appropriate soil information, effective transmission of this knowledge, and sufficient and targeted incentives is by no means simple to develop but should be the starting point for any programme aimed at facilitating the adoption of SSM.



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Chapter 7 | Criteria for assessment of state, trend and confidence for the regional assessments

7.1 Introduction

This chapter presents the methodological framework used to conduct the regional assessments of the state and trend of threats to soil functions, informed by a management-based approach. It outlines the criteria applied in the evaluation and the approach used to determine confidence levels. The soil threats considered are soil erosion, soil organic carbon (SOC) loss, soil biodiversity change, nutrient mismanagement, salt accumulation, soil pollution and soil sealing and urbanization.

Note that the area of land being considered in the assessment is the portion of the total land area in a country, subregion or region where a specific threat is believed to be operating.

7.2 Development of the criteria for assessment of state and trend

Management, in the context of this report, includes both land-use change and the management systems applied within a given land use. Changes in management can appreciably impact the threats to soil functions in a decade, which is the period used in this report.

The relationship between management and threats to soil functions is well established. For example, the state of management is a major controlling variable in the following:

- the modelling of SOC change (e.g. Karstens *et al.*, 2022);
- soil erosion by water (e.g. the land cover and management factor and the soil conservation or prevention practices factor in the Universal Soil Loss Equation [USLE]) (Alewell *et al.*, 2019);
- tillage erosion (inherently related to the management decisions about the tillage system implemented by farmers) (Öttl *et al.*, 2022);
- vegetation ground cover in managed lands (affecting processes such as salt accumulation [e.g. Hassani, Azapagic and Shokri, 2020]);
- wind and water erosion (e.g. Chappell *et al.*, 2019);
- nutrient mismanagement (including both the application of nutrients in excess of crop requirements and insufficient nutrient input to replace nutrients removed in crop harvests) (e.g. Bouwman *et al.*, 2017); and
- soil sealing through the conversion of land to urban uses.

7.3 Specific classes used for state and trend

In the regional assessments, for agricultural lands, urban areas and managed forestlands, soil is judged to be in a very good state if the soil management meets the criteria for sustainable soil management (SSM) outlined in the *Voluntary Guidelines for Sustainable Soil Management* (VGSSM) (FAO, 2017), as follows:

1. minimal rates of soil erosion;
2. soil structure that is not degraded (e.g. by soil compaction, erosion or soil crusting) and provides a stable physical environment for movement of air, water and heat, as well as root growth;
3. sufficient surface cover present to protect the soil (e.g. from growing plants or plant residues);
4. a store of soil organic matter that is stable or increasing and ideally close to the optimal level for the local environment;
5. availability and flows of nutrients that are appropriate to maintain or improve soil fertility and productivity, and to reduce nutrient losses to the environment;
6. soil salinization, sodification and alkalinization are minimal;
7. water (e.g. from precipitation and supplementary water sources such as irrigation) that is efficiently infiltrated and stored to meet the requirements of plants and ensure the drainage of any excess;
8. contaminants are below toxic levels that could cause harm to plants, animals, humans and the environment;
9. soil biodiversity that provides a full range of biological functions;
10. soil management systems for producing food, feed, fuel, timber and fibre that rely on optimized and safe use of inputs; and
11. soil sealing due to urban expansion, that is minimized through responsible land-use planning.

The definitions of the classes used in the assessment for the **state** of SSM practices with respect to a specific threat to soil functions specify the broad factors to be considered in the assessment. For each threat, a five-point scale ranging from *very poor* to *very good* describes the effectiveness of management practices in mitigating the soil threat, with an additional category (*not assessed*) for regions where data were unavailable (Table 7.1). The *very poor* category indicates that existing practices not only failed to prevent degradation but also contributed to severe impacts on soil functions and, in many cases, negatively affected water and air quality. Conversely, *very good* denotes that management practices effectively prevented or reversed degradation to the greatest extent possible in that region, in some cases achieving optimal conditions with no further improvements feasible. Intermediate categories (*poor*, *fair* and *good*) reflect progressive improvements in management effectiveness, from partial mitigation to substantial progress toward sustainable management. This classification provided a consistent basis for comparing the status of SSM practices between regions and identifying priority areas for improvement.



Table 7.1. Criteria for assessing the state of sustainable soil management practices to address specific soil threats as of 2025

Soil erosion	
State	**State of soil management**
Very poor	Management practices in place led to severe soil loss and substantial impacts on air and/or water quality.
Poor	Management practices in place led to moderate soil loss and marked impacts on air and/or water quality.
Fair	Management practices in place led to low soil loss.
Good	Management practices in place led to very low soil loss.
Very good	Management practices in place led to soil loss at the lowest level possible for that region.
Not assessed	Threat was not assessed for this region.
Soil organic carbon (SOC) loss	
State	**State of soil management**
Very poor	Management practices in place led to severe loss of SOC.
Poor	Management practices in place led to moderate loss of SOC.
Fair	Management practices in place led to stable levels of SOC.
Good	The management practices in place led to minor gain in SOC.
Very good	Management practices in place led to substantial gain in soil SOC or the state of SOC was good or very good and no further gains in SOC were possible.
Not assessed	Threat was not assessed for this region.
Soil biodiversity loss	
State	**State of soil management**
Very poor	Management practices in place led to severe soil biodiversity loss.
Poor	Management practices in place led to moderate loss of soil biodiversity.
Fair	Management practices in place led to stable levels of soil biodiversity.
Good	Management practices in place led to minor increases in soil biodiversity.
Very good	Management practices in place led to marked increases in soil biodiversity.
Not assessed	Threat was not assessed for this region.
Salt accumulation	
State	**State of soil management**
Very poor	Management practices in place led to major increases in saline or sodic soils.
Poor	Management practices in place led to small increases in saline or sodic soils.
Fair	Management practices in place led to little or no change in saline and sodic soils.
Good	Management practices in place led to minor decreases in saline and sodic soils.
Very good	Management practices in place led to marked decreases in saline and sodic soils.
Not assessed	Threat was not assessed for this region.
Nutrient mismanagement	
State	**State of soil management**
Very poor	Management practices in place led to substantial excess nutrient inputs, nutrient mining and/or soil acidification.
Poor	Management practices in place led to moderate excess nutrient inputs, nutrient mining and/or soil acidification.

Fair	Management practices in place led to some excess nutrient inputs, nutrient mining or soil acidification.
Good	Management practices in place led to sufficient nutrient inputs but with potential for some excess nutrients and no adverse impacts on soil pH.
Very good	Management practices in place led to sufficient nutrient inputs within sustainable limits and protection of water bodies from nutrient pollution in most areas.
Not assessed	Threat was not assessed for this region.
Soil pollution	
State	**State of soil management**
Very poor	Substantial pollution of soil continued throughout the assessment period.
Poor	Incidents of pollution of soil continued throughout the assessment period.
Fair	Any soil pollution was managed in a timely manner and legacy polluted sites were identified.
Good	Any soil pollution was remediated in a timely manner and reclamation of legacy polluted sites was begun.
Very good	Any soil pollution was remediated in a timely manner and significant reclamation of legacy polluted sites was undertaken.
Not assessed	Threat was not assessed for this region.
Urbanization and soil sealing	
State	**State of soil management**
Very poor	Substantial urban expansion occurred without consideration of soil functions or effects of soil sealing.
Poor	Some urban expansion occurred without consideration of soil functions or effects of soil sealing.
Fair	Soil functions were considered but not preserved in urban expansion and no measures to limit soil sealing were implemented.
Good	Soil functions were considered and partially preserved in urban expansion and some measures to limit soil sealing were implemented.
Very good	Soil functions were preserved during urban land expansion and measures were undertaken to minimize soil sealing.
Not assessed	Threat was not assessed for this region.
Other issues	
State	**State of soil management**
Very poor	Management practices in place led to substantial impacts on soil, air and/or water quality.
Poor	Management practices in place led to moderate impacts on soil, air and/or water quality.
Fair	Management practices in place led to low soil impacts and minimal offsite impacts.
Good	Management practices in place led to minimal adverse soil impacts.
Very good	Management practices in place led to soil impacts at the lowest level possible for that region.
Not assessed	Threat was not assessed for this region.

The **trend** captures the direction of change in SSM adoption from 2015 to the time of finalization of the regional chapters (late 2024). An *improving* trend indicates that the adoption of SSM is becoming more widespread since 2015 and hence the threats to soil functions are being reduced. A *deteriorating* trend indicates that either SSM has not been widely adopted (and hence deterioration continues) or that the area under SSM is decreasing. A *stable* rating indicates that the initial state of SSM has not appreciably changed from 2015 to the time of the assessment. A *variable* trend indicates that no clear direction of change is evident across the area under consideration.

7.4 Methodology for regional assessments, graphical summaries and confidence rating

While there have been excellent reviews of the state of soils or of specific soil threats published in the peer-reviewed scientific literature, the global representation of scientists in the Intergovernmental Technical Panel on Soils (ITPS) allowed this report to draw upon an extensive community of experts from different regions. Thus, for the 2026 *Status of the world's soil resources* (SWSR) report (FAO and ITPS, 2015), the ITPS decided to focus on the regional assessments.

The regional chapters include two different sections:

1. a summary of available information on the state of the various threats to soil functions and on the implementation of SSM practices to address them, based on literature published since 2015; and
2. the ITPS expert assessments of the state and trend in the adoption of SSM practices, based on the criteria defined previously.

The regional summaries of the state of soil threats were developed by over 270 scientists. The state and trend assessments for each region were completed by ITPS members for that region, based on the information gathered in the regional chapter. Input was sought from the lead authors of each chapter and additionally from the Chair of the Regional Soil Partnership of the region, if they had been involved with the creation of the content.

Three sources of information informed the assessment by the ITPS. First, authors drew on the global and regional modelling of the threats presented in Chapter 3. Second, each regional assessment incorporated information on the drivers of soil change and the national summaries of management-related data (including land-use change) available from international organizations and the peer-reviewed research literature. Third, each regional chapter compiled additional national and subnational information from national institutions and research literature, including data on both the threats to soil functions and the progress in managing them.

The authors' degree of confidence in their regional or subregional assessment of trend and state is expressed in a rating with three classes (*high*, *moderate* and *low*) (Table 7.2). The assessment of confidence is based on the type, amount, quality and consistency of evidence available and the degree of scientific agreement within the group of assessors and in the literature, following the confidence terms of Kause *et al*. (2022) used in the Intergovernmental Panel on Climate Change (IPCC) reports. *High* confidence indicates multiple independent sources with strong agreement. *Medium* confidence indicates either robust evidence with mixed agreement or limited evidence with strong agreement, while *low* confidence indicates limited evidence and low agreement.

Confidence levels are defined at the regional or subregional level and are not comparable with each other. Some regions may have had limited information, but there was strong agreement between this information and regional assessors, while in other regions with greater data availability, agreement among assessors was lower and, therefore, so was the confidence level.



Table 7.2. Categories of agreement and evidence used in the confidence assessment of state and trend

High agreement Limited evidence **Medium**	High agreement Medium evidence **High**	High agreement Robust evidence **High**
Medium agreement Limited evidence **Low**	Medium agreement Medium evidence **Medium**	Medium agreement Robust evidence **High**
Low agreement Limited evidence **Low**	Low agreement Medium evidence **Low**	Low agreement Robust evidence **Medium**

Note: The labels of the confidence classes are in bold.

Source: **Kause, A., Bruine De Bruin, W., Persson, J., Thorén, H., Olsson, L., Wallin, A., Dessai, S. & Vareman, N.** 2022. Confidence levels and likelihood terms in IPCC reports: a survey of experts from different scientific disciplines. *Climatic Change*, 173(1–2): 2. https://doi.org/10.1007/s10584-022-03382-3



7.5 Differences between the 2015 and 2026 *Status of the world's soil resources* reports

This section outlines the methodological differences between the 2015 SWSR and the current assessment. In the 2015 SWSR, the ITPS aimed "to provide a global scientific assessment of current and projected soil conditions built on regional data analysis and expertise" (FAO and ITPS, 2015, p. 11). This information was presented in seven regional assessment chapters and then used to create a graphical summary of the condition and trend of the soil threats and the confidence in the information that was available to make these assessments.

In 2020, when the preparation of the 2026 SWSR started, the ITPS had hoped to base its assessment of state and trend on estimates of the individual soil threats (e.g. soil erosion or SOC stock change) directly. Despite the considerable progress since 2015 in the production of global maps of estimated soil properties and in modelling processes associated with soil threats (as reviewed in Chapter 3 of this report), the ITPS concluded that the information available did not allow an assessment of state and trend of biophysical condition of soils for each threat based solely on the estimated or modelled data. This decision was primarily made due to concerns about the ability of the estimates or models to produce reliable estimates of decadal-scale changes due to issues with scale and with the lack of adequate ground truthing of estimates, together with a lack of widely recognized globally relevant, acceptable levels or thresholds for soil properties or processes related to soil threats. Therefore, in the 2026 SWSR report, the ITPS assessed the degree of adoption of SSM since 2015 and the effectiveness of these management practices in preventing, controlling, and mitigating each threat to soil functions, including whether their status has worsened or improved.

This report also includes a more standardized and structured approach for the regional assessments. The major methodological differences with the 2015 SWSR report are as follows:

- The high-level soil threats considered in the 2015 report vary slightly from those considered in the current assessment (Table 7.3). Regional assessment authors were asked to include information related to any threat that was relevant to their region (where information was available).
- A list of specific processes was added to the high-level soil threats for the 2026 report (Table 7.3).
- A set of management-focused criteria (Table 7.1) were developed and agreed to by ITPS and were used in the assessment process.
- Assessments were carried out for subregions within Asia, Latin America and the Caribbean, the Southwest Pacific and sub-Saharan Africa. The decision to divide these regions into subregions was made by the respective ITPS members, who considered that the differences within regions were substantial enough that a single, aggregated assessment would cause the loss of critical information from the subregions. In contrast, the 2015 SWSR report did not employ formal subregional assessments.



Table 7.3. Threats to soil functions included in the 2015 and 2026 Status of the world's soil resources reports

2015 report	2026 report	Specific processes or conditions included in the 2026 report
Soil erosion	Soil erosion	Water erosion Wind erosion Tillage erosion Mass movement Root crop harvesting
Organic carbon change	Soil organic carbon loss	Mineral soils Organic soils including wetlands and permafrost Soil inorganic carbon
Loss of soil biodiversity	Soil biodiversity change	
Nutrient imbalance	Nutrient mismanagement	Nutrient surplus Nutrient mining Acidification
Salinization and sodification	Salt accumulation	Irrigation and drainage Alteration of hydrological conditions Coastal waterlogging
Contamination	Soil pollution	Inorganic pollutants Organic pollutants Emerging pollutants
Sealing and land take	Soil sealing and urbanization	
Compaction	Physical degradation	
Waterlogging	Not included	
Soil acidification		Included under nutrient mismanagement

◊ *Source:* **FAO & ITPS**. 2015. *Status of the world's soil resources* – Main Report. Rome. https://openknowledge.fao.org/handle/20.500.14283/i5199e



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Chapter 8 | Regional assessment for Asia

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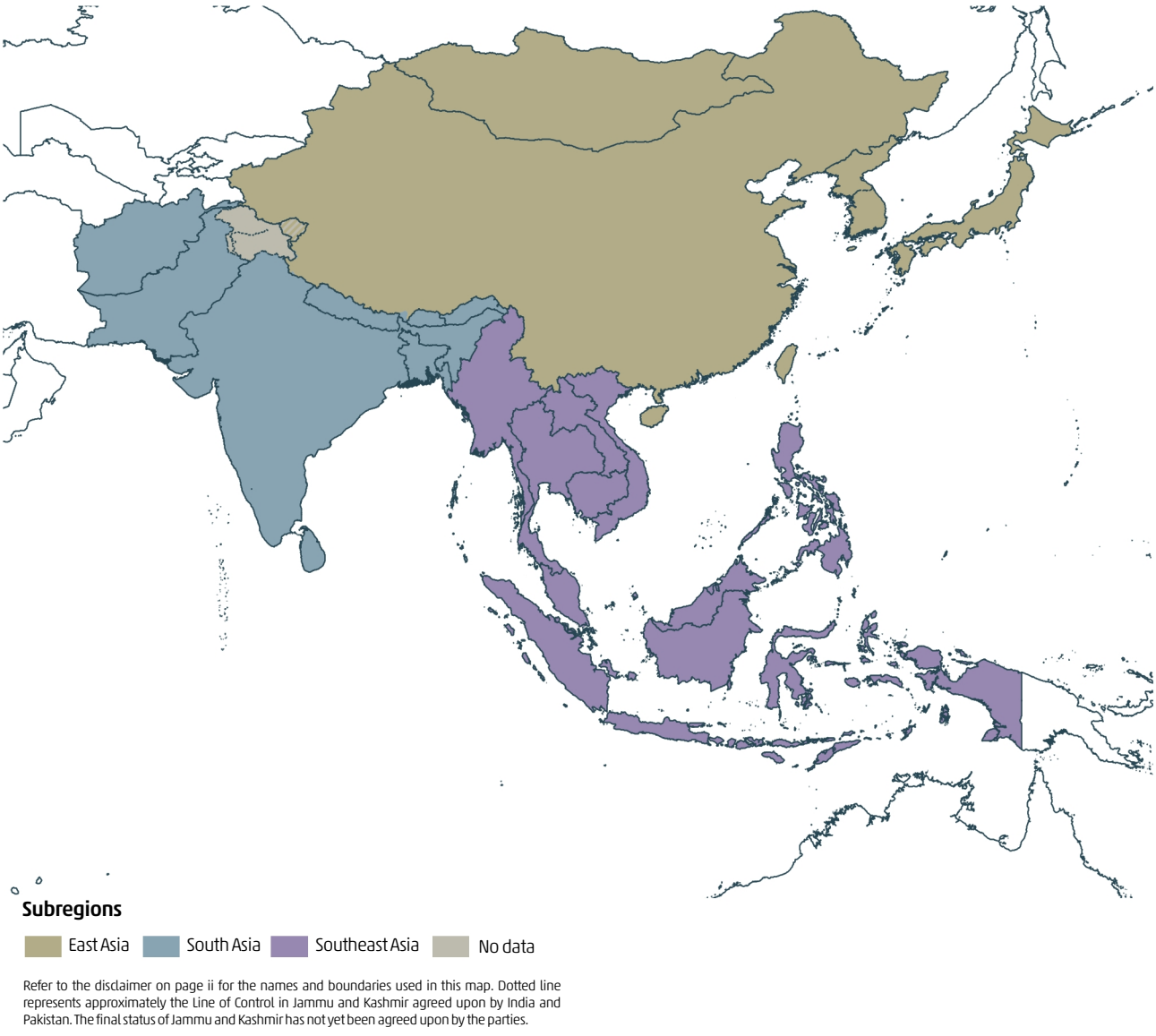


8.1 Regional overview

8.1.1 Characteristics of region and subregions

Asia is the largest and the most populous continent in the world, covering 4.1 percent of the Earth's total surface area and 16 percent of the total land area. For this report, the three subregions of East Asia, South Asia and Southeast Asia are recognized (Figure 8.1), comprising of 25 countries (Table 8.1). These subregions exhibit a wide range of climatic and environmental conditions governed by tropical, subtropical and temperate climates. The region has a predominantly warm and seasonally humid climate, favourable for agricultural practices.

Figure 8.1 Map of the Asia region showing countries and subregions



Refer to the disclaimer on page ii for the names and boundaries used in this map. Dotted line represents approximately the Line of Control in Jammu and Kashmir agreed upon by India and Pakistan. The final status of Jammu and Kashmir has not yet been agreed upon by the parties.

Table 8.1. Division of subregions

Subregions	Countries
South Asia	Afghanistan, Bangladesh, Bhutan, India, Maldives, Nepal, Pakistan, and Sri Lanka
East Asia	China, the Democratic People's Republic of Korea, Japan, Mongolia, the Republic of Korea
Southeast Asia	Brunei Darussalam, Cambodia, Indonesia, the Lao People's Democratic Republic, Malaysia, Myanmar, the Philippines, Singapore, Thailand, Timor-Leste, and Viet Nam

The total population of Asia in 2021 was 4.23 billion people, with 1.66 billion people in East Asia, 1.9 billion people in South Asia and 0.68 billion people in Southeast Asia (UN DESA, 2023). China (East Asia) and India (South Asia) are the most populous countries in the world, with approximately 1.4 billion people each.

Population growth means that there is increased pressure and impact on land and soil resources to ensure food security as well as to support human settlements. On-going rapid economic development and urbanization continue to transform the landscape and soil resources in many countries. This report discusses both existing and emerging threats to SSM, including the growing challenges posed by climate change.

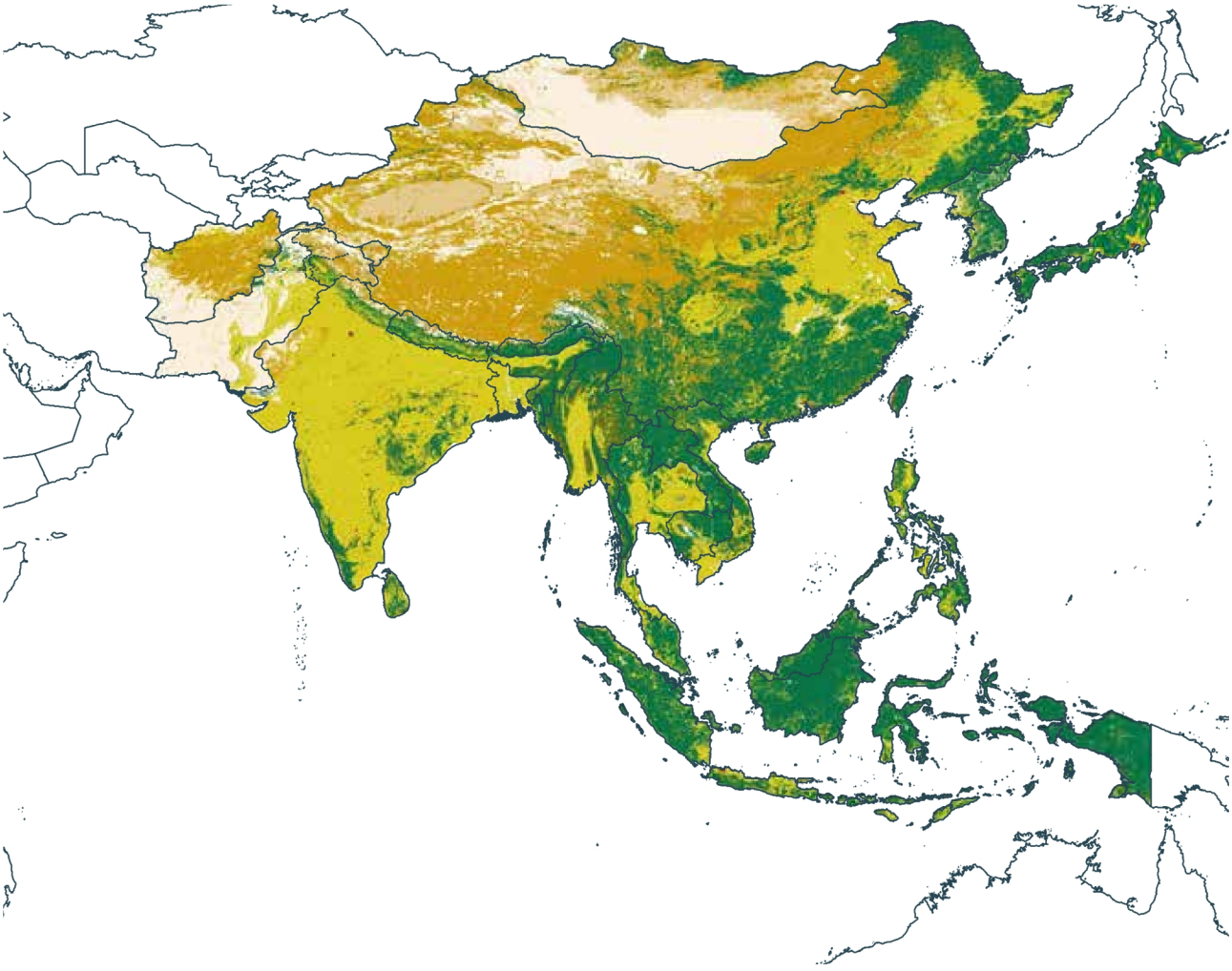
Asia has a diverse range of climates and landscapes with a complex interaction between climate and agroecology (JRC and FAO, 2023). From the frozen mountain peaks of the Himalayas to the tropical rainforests of Southeast Asia, the region's agricultural systems are influenced by diverse climatic factors, including topography, soil types and human activities. Asia has many river tributaries, large alluvial plains and deltas including the Brahmaputra, Chao, Phraya, Ganges, Huang He, Indus, Irrawaddy, Mekong, and Yangtze rivers. Several island chains lie off the eastern and southeastern shores of the continent, including Indonesia, Japan, the Philippines and Sri Lanka, some of which are characterized by mountainous landscapes and volcanic activity.

The Asia region is influenced by seasonal monsoon winds that bring heavy rainfall (Loo, Billa and Singh, 2015). During the summer months, the land heats up more quickly than the adjacent oceans, creating a low-pressure area over the land. As a result, moist air from the oceans is drawn inland, leading to the formation of clouds and rainfall. The monsoon season is particularly significant in South and Southeast Asia, where it plays a crucial role in agriculture, water resources and overall climate patterns (Loo, Billa and Singh, 2015). The monsoon season generally occurs from June to September, although the timing and intensity can vary across regions and may occasionally be erratic. The monsoon season brings heavy rainfall to countries like Bangladesh, India and Nepal, in South Asia and parts of Southeast Asia. The monsoon rains are essential for replenishing water sources, irrigating crops and supporting ecosystems. However, excessive rainfall can also lead to flooding, landslides and other natural disasters, posing challenges for communities and infrastructure (Loo, Billa and Singh, 2015).

Besides summer monsoons, there are also so-called winter monsoons, which affect the more continental areas in Central Asia and a large part of western China. The winter monsoon has a strong impact on regional climate, and can lead to strong westerlies and harsh winters. The winds often cause erosion and carry heavy dust to the coastal regions in eastern China, and even Japan and the Korean peninsula, impacting soil properties.

The Indian subcontinent has diversity in climate and agroecology (JRC and FAO, 2023). The fertile plains of the Indo-Gangetic region support intensive agriculture, with rice and wheat as major crops and are the largest area of contiguous cropland in Asia (Figure 8.2). The arid regions of Gujarat and Rajasthan rely on drought-resistant crops like millet and pulses, while the Himalayan foothills support terrace farming and the cultivation of temperate crops such as apple (*Malus* sp.) and tea (*Camellia sinensis*).

Figure 8.2. Land cover of Asia

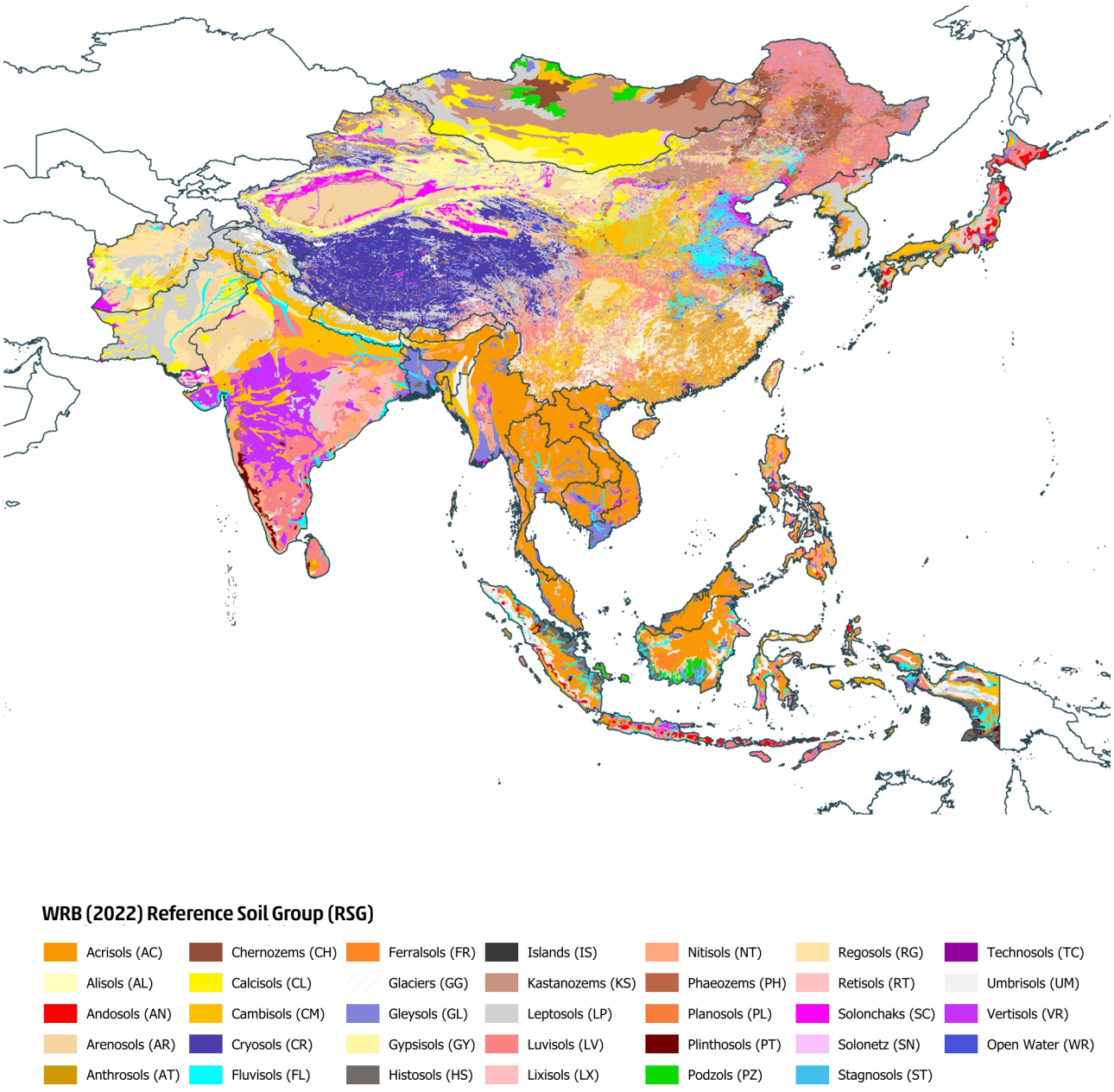


Refer to the disclaimer on page ii for the names and boundaries used in this map. Dotted line represents approximately the Line of Control in Jammu and Kashmir agreed upon by India and Pakistan. The final status of Jammu and Kashmir has not yet been agreed upon by the parties.

◊ *Source:* **FAO**. 2024. *GAEZ: Land and Water Resources.* [Accessed 24 October 2024] https://gaez-services.fao.org/apps/theme-1/. Licence: CC-BY-4.0.

South Asia has a great range of soils (JRC and FAO, 2023), due to the complex pattern of terrain and climate (Figure 8.3). Southern India is dominated by Ferralsols, Plinthosols and black Vertisols on the Deccan Plateau. In the north, soils grade to younger Cambisols and Fluvisols in river valleys. The mountains systems in northern India, Pakistan, Bhutan and Nepal have thinner Regosols, Leptosols and frozen Cryosols. Afghanistan has soils associated with drier climates, with Regosols, Leptosols and Cambisols in mountainous areas and grassland soils (Kastanozems and Chernozems) in the central highlands. The Indus basin in Pakistan has Fluvisols and sandy Arenosols. Sri Lanka dominantly has Luvisols in the drier lowland plains and Acrisols and Plinthosols in areas with high precipitation.

Figure 8.3. Soil map of Asia



Source: **FAO**. 2024. *GAEZ: Harmonized World Soils Database version 2.0.* [Accessed 24 October 2024] https://gaez.fao.org/pages/hwsd. Licence: CC-BY-4.0.

The tropical areas in Southeast Asia have high temperatures, humidity, and monsoon rains which support high biodiversity in countries such as Indonesia, Malaysia, and the Philippines, as well as diverse agroecosystems, including rice paddies, fruit orchards, and oil palm (*Elaeis guineensis*), sugarcane and rubber tree (*Hevea brasiliensis*) plantations. The major cropland areas are in the river valleys of central Myanmar, the lowlands of Cambodia, the Mekong and Hong river deltas of Viet Nam, the Chao Phraya river valley of Thailand, and in the Philippines and Indonesia (Figure 8.2).

Several broad patterns in soil distribution occur in Southeast Asia (Figure 8.3). In the mainland countries and in Indonesia, highlands areas are commonly associated with Ferralsols and Acrisols, which have been leached of cations and often have acidic pH levels due to chemical weathering through time (JRC and FAO, 2023). Highland areas with drier conditions and less weathering may

have more fertile Luvisols or Cambisols. The extensive river valleys have Fluvisols formed in alluvial sediments with textures ranging from clay to sand depending on the depositional environment. The alluvial deposits often have localized areas of wetland Gleysols or organic Histosols associated with them. Coastal soils, including those with mangroves, are often saline.

The island countries of Southeast Asia have complex patterns of soils due to volcanism. Volcanically derived Andosols are common in some areas of Indonesia. In the Philippines, Vertisols and Cambisols are the most common soils, with Acrisols in more heavily weathered soils.

The temperate climates of East Asia, including China, Japan and the Republic of Korea, support diverse agricultural practices. Rice cultivation dominates in the wetter regions, while wheat, maize and soybeans are prevalent in drier areas with intensive farming techniques. The largest cropland areas are in the North China Plain, the Manchurian Plain and the Sichuan Basin of China and coastal areas in Japan and the Korean Peninsula (Figure 8.2).

The soils of East Asia show several broad patterns (Figure 8.3). Deserts in the western areas are characterised by sandy Arenosols, Calcisols, Gypsisols and saline Solonchaks (JRC and FAO, 2023). The croplands of the Sichuan basin are dominated by Luvisols and Cambisols. Southeast China has a soil pattern similar to that of mainland Southeast Asia, with a mix of Ferralsols, Acrisols and Cambisols in highland areas, and Fluvisols and Gleysols in river valleys and deltas. The cropland area of northeast China is dominantly Fluvisols in alluvial deposits, interspersed with wetland Gleysols and some saline Solonchaks. Mongolia is dominated by soils of grassland biomes (Phaeozems and Kastanozems).

Both the Korean peninsula and Japan have significant mountainous areas with thin Leptosols (JRC and FAO, 2023) (Figure 8.3). In the Korean peninsula, the western coastal area has a mixture of Luvisols in foothills and better-drained locations and Fluvisols and Gleysols in river valleys. Japan has a complex mix of soils with fertile Andosols developed in volcanic materials and Cambisols formed other parent materials. Again Fluvisols are interspersed with Gleysols and Histosols occur in lowlands and river valleys.

Climate change poses challenges to agriculture in Asia, with rising temperatures, changing precipitation patterns and extreme weather events threatening food security and livelihoods. Adaptation strategies such as crop diversification, water management and sustainable farming practices are increasingly important to mitigate risks and ensure the resilience of Asia's agroecosystems in the face of a changing climate.

Rice is the staple food for over half of the global population. About 90 percent of all rice is produced and consumed in Asia. Thus, global food security is highly impacted by Asian rice production, mainly in China and India, which together account for 37 percent of the world's population and 49 percent of the world's rice production (Bandumula, 2017). A climate modelling study looking at the Sumatra–Java archipelago and northern Australia (Marjuki *et al.,* 2016) predicted that by 2050, the onset of the wet season may be delayed and the duration shortened by approximately 10 percent, risking rice production. The increasing scarcity of land and water resources, environmental degradation and loss of biodiversity has begun to limit the expansion of food production in both developed and developing countries. Thus, the shift in supply and demand in Asia will impact global rice prices and global food security.

8.1.2 Summary of status from 2015 Status of the world's soil resources report

According to the 2015 SWSR report (FAO and ITPS, 2015), the main threats to soil functions (including erosion, SOC loss, salinization and sodification, and pollution) were all rated as being in a “poor” state in Asia. The key points of the 2015 report are summarized in this section.

Soil erosion was judged to be the most important threat. Water erosion was dominant in regions with alternating wet and dry seasons throughout South Asia and East Asia. Wind erosion was concentrated in the semi-arid areas of Afghanistan, China, India and Pakistan.

Soil organic carbon loss, the second-ranked threat, was significantly impacted by regional management practices. Increases in crop yields were believed to have increased SOC levels in the soils of East and Southeast Asia whereas the use of crop residues as fuel and fodder was causing continuous losses of SOC in South Asia.

Salinization and sodification was the third-ranked threat and affected two widely separated areas. Salt-affected soils were widely distributed throughout the semi-arid and arid zones of Central Asia and were an increasing concern in coastal areas of South and Southeast Asia, due to saltwater intrusion.

Severe deficiencies in N, P and K were reported for countries in South Asia, whereas large surpluses of N were present in some countries of East Asia. Soil acidification associated with excessive chemical N fertilizer use and atmospheric deposition were reported throughout Asia.

Rapid urbanization across Asia had led to soil contamination and the removal of soil via soil sealing in many countries. Asia was reported to have the largest area of impervious surface area due to urban development for any region, with profound effects on the water cycle, which was contributing to the loss of cropland and forestland adjacent to urban areas.

Assessing soil biodiversity changes in 2015 was challenging due to the limited available data at the time.



8.2 Changes in drivers of threats to soil functions

8.2.1 Population growth

Population growth is the main indirect driver of threats to SSM. The three subregions of Asia have different predicted population growth pathways over the next few decades (UN DESA, 2023). East Asia is projected to have population growth rates well below the global average, with declines in population by 2035 in China, Japan and the Republic of Korea. South Asia is projected to have population growth at higher rates than the global average, although with substantial deceases in the future compared to the 1.6 percent change per year in the 2015–2021 period. Afghanistan is projected to have high growth rates through to 2025. The countries of Southeast Asia are predicted to have growth rates matching the global averages, with less variation between countries than in South Asia.

The population in Asia is projected to reach up to 5.2 billion by 2050 (Rao *et al.*, 2019). It is therefore challenging to meet the food demands and ensure food security in Asia, and SSM will remain an important issue.

The impact of population pressure, combined with rapid economic development, has led to the conversion of forestland to agriculture and agricultural areas converted to residential and urban uses with the associated impacts on ecosystem services. Change in land use and land cover has impacted forest resources, biodiversity, regional climate, biogeochemical cycles, water resources and other ecosystem services (Justice *et al.*, 2015). Asia is in transition from largely agrarian to urban societies due to increased industrialization. The biggest direct cause of deforestation is the conversion to cropland and pasture (agricultural area expansion), mostly for subsistence needs (Vadrevu, Ohara and Justice, 2017).

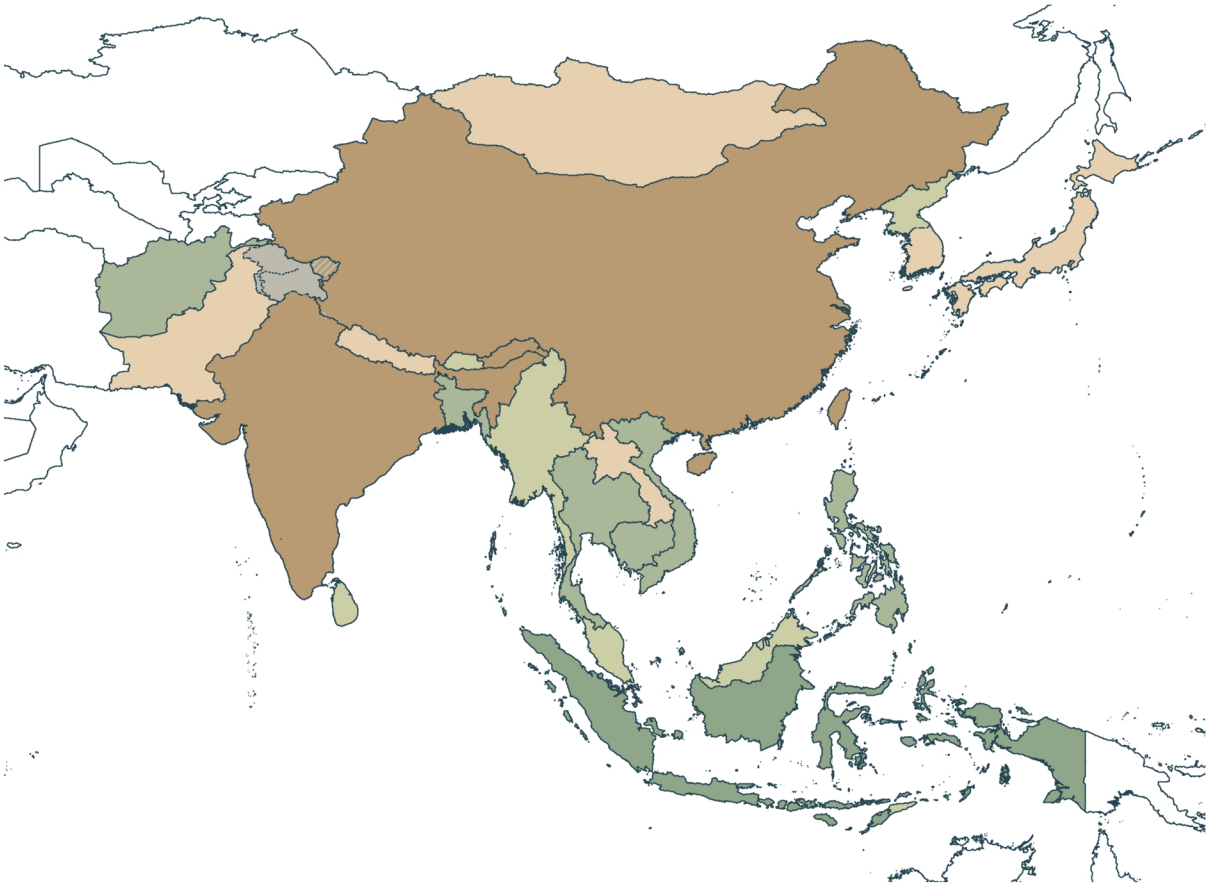
8.2.2 Land-use change

Land-use change may be a direct threat to various soil functions, as discussed in this section, although some changes may benefit SSM.

Cropland area change

Southeast Asia has had substantial increases in cropland area from 2015 to 2022 (FAO, 2024a) (Figure 8.4). The main countries driving the increase in cropland area in Southeast Asia are Indonesia (+4.4 million ha), Thailand (+0.65 million ha) and Cambodia (+0.47 million ha) (FAO, 2024a). Outside of Southeast Asia, only Bangladesh (+0.26 million ha) has experienced comparable growth. Both India and China have experienced substantial decreases in cropland area (−1.4 million ha and −3.3 million ha respectively), but in both cases the changes are only a small percentage decrease (−0.8 percent for India and −2.5 percent for China) and in the 2015 cropland area (FAO, 2024a).

Figure 8.4. Country map of change in cropland area from 2015 to 2022

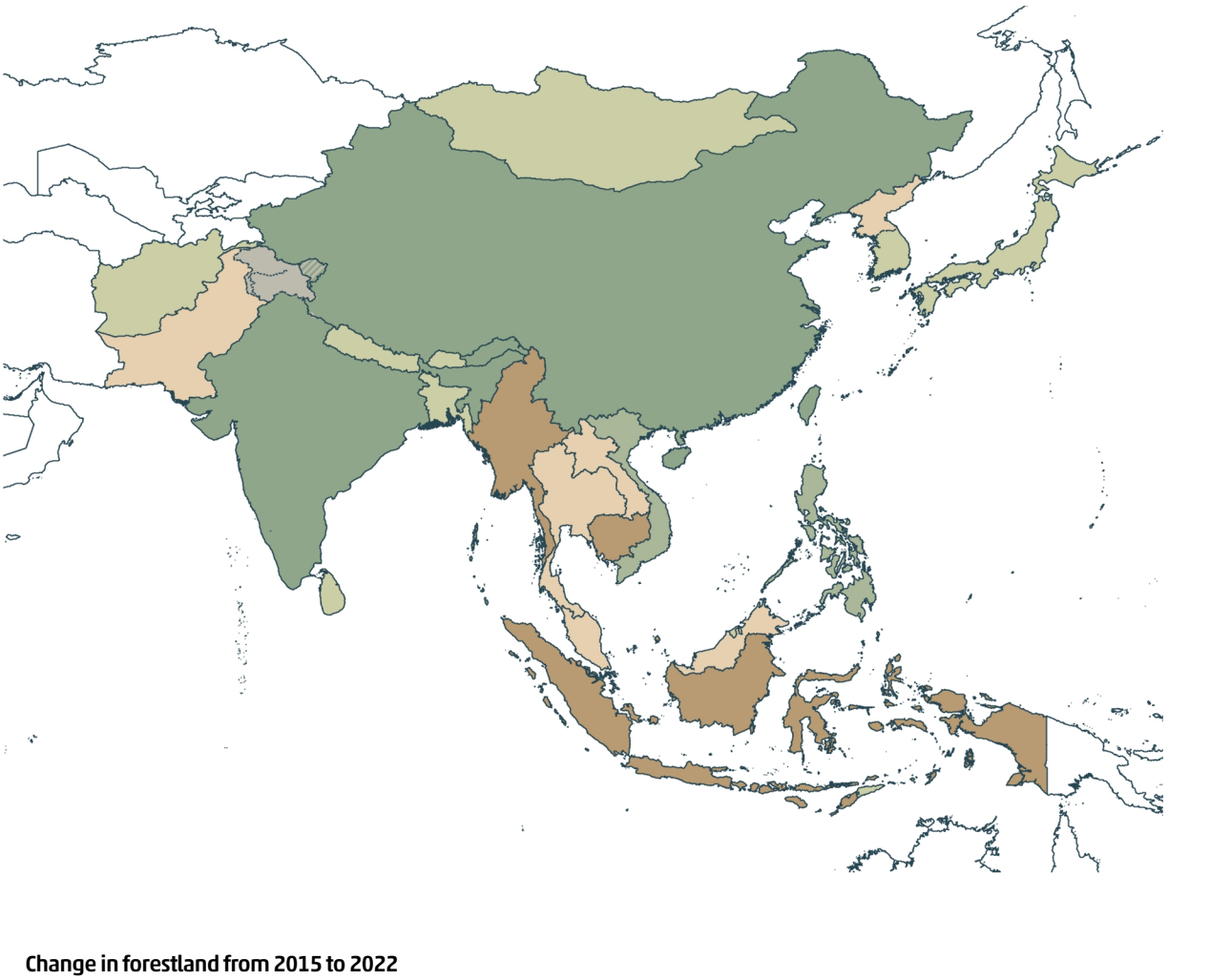


Source: **FAO**. 2024. *FAOSTAT: Land Use*. [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

Forestland change

Changes in forest area had a greater range than for cropland in the 2015–2022 period, with East Asia and South Asia experiencing a substantial increase, whereas Southeast Asia experiencing substantial deforestation (FAO, 2024a) (Figure 8.5). The increases in East Asia are driven by the substantial increase in forest area in China of 13.4 million ha (a 6.4 percent expansion) (FAO, 2024a). India is the major contributor to the forest area increase in South Asia of 1.9 million ha (a 2.6 percent area increase). Substantial gains in forestland area also occurred in the Philippines (0.24 million ha) and Viet Nam (0.73 million ha) in this time period (FAO, 2024a).

Figure 8.5. Country map of change in forestland area from 2015 and 2022



Refer to the disclaimer on page ii for the names and boundaries used in this map. Dotted line represents approximately the Line of Control in Jammu and Kashmir agreed upon by India and Pakistan. The final status of Jammu and Kashmir has not yet been agreed upon by the parties.

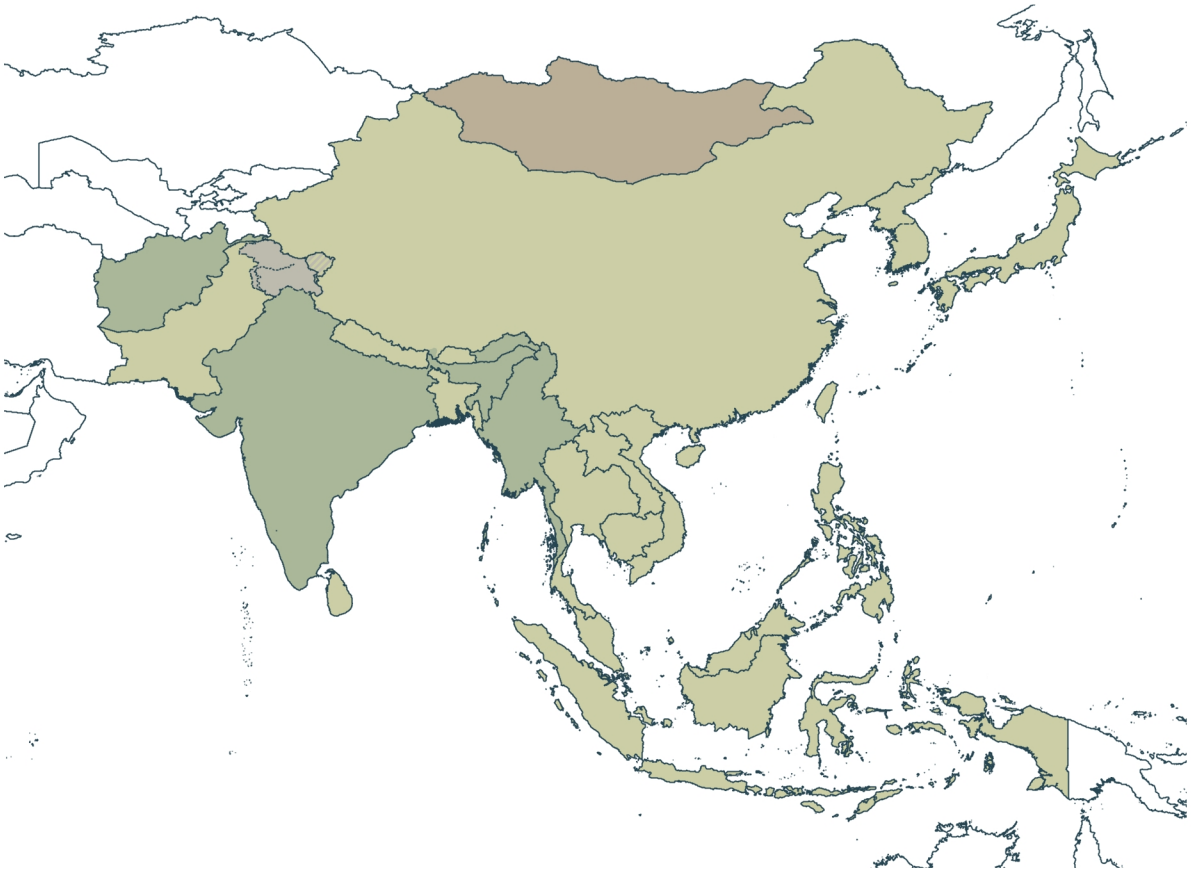
◊ *Source:* **FAO**. 2024. FAOSTAT: Land Use. [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

Deforestation in the 2015–2022 period has been greatest in Indonesia, with a 4.3 percent decrease in forest area (−4.1 million ha of forestland) and Myanmar, with a 6.8 percent decrease (−2 million ha of forestland) (FAO, 2024a) (Figure 8.5). Percentage losses have been highest in Cambodia, with a 12.3 percent decrease in the 2010 forest area (−1.1 million ha of forestland). Substantial area losses of forestland also occurred in the Democratic People's Republic of Korea (−0.15 million ha of forestland), the Lao People's Democratic Republic (−0.24 million ha of forestland), Malaysia (−0.45 million ha of forestland), Pakistan (−0.29 million ha of forestland), and Thailand (−0.26 million ha of forestland) (FAO, 2024a).

Permanent pastures and meadows

The area of permanent meadows and pastures in Asia is dominated by China (393 million ha in 2022) and Mongolia (111 million ha in 2022). Mongolia had a decrease of 1.5 million ha in pasture and permanent meadows between 2015 and 2022, but this represents a percentage change of only −1.3 percent (FAO, 2024a) (Figure 8.6).

Figure 8.6. Country map of change in permanent meadows and pastureland area between 2015 and 2022



Refer to the disclaimer on page ii for the names and boundaries used in this map. Dotted line represents approximately the Line of Control in Jammu and Kashmir agreed upon by India and Pakistan. The final status of Jammu and Kashmir has not yet been agreed upon by the parties.

◊ *Source:* **FAO**. 2024. *FAOSTAT: Land Use*. [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

Wetlands extent and change

The long-term analysis (since 1700) by Fluet-Chouinard *et al.* (2023) revealed significant wetland loss (exceeding 50 percent) in East Asia and specific countries of South and Southeast Asia, including Bangladesh, Malaysia, Pakistan, the Philippines and Viet Nam. There have been major impacts experienced due to land-use conversion and fires in the extensive peatlands of Indonesia (Warren *et al.*, 2017; Kiely *et al.*, 2021) as well as in the mangrove forests in Southeast Asia (Richards and Freiss, 2016). These trends are discussed in more detail in the Southeast Asia section.

Conservation agriculture

Kassam, Freidrich and Derpsch (2022) estimated that in the Asia region, 13.8 million ha were under conservation agriculture in 2018/2019. China has had the greatest adoption of conservation agriculture in the world, with an estimated 9 million ha under conservation agriculture in that period, followed by India (estimated at 3.5 million ha) and Pakistan (1.3 million ha). The Democratic People's Republic

of Korea is estimated to have 23 000 ha in conservation agriculture, and Cambodia, the Lao People's Democratic Republic, Malaysia, Myanmar, the Philippines, Sri Lanka, Timor-Leste, and Viet Nam, also have small areas in conservation agriculture. In East Asia, the adoption of conservation agriculture is being promoted by the Conservation Agriculture Alliance for Asia-Pacific (CAAAP), hosted by the Conservation Tillage Research Centre (CTRC) at the China Agriculture University, Beijing.

Agroforestry and palms

Asia leads globally in land allocated to agroforestry and oil palms, with South and Southeast Asia collectively having 30.7 million ha in agroforestry in 2020, representing 68 percent of the global total (FAO, 2020). South and Southeast Asia have 7.7 million ha in oil palms, 99.6 percent of the Asian total and 65 percent of the global total, with palm oil production being concentrated in Malaysia and Indonesia. The harvested area of oil palm fruit in Malaysia has been relatively stable (increasing from 4.8 million ha in 2010 to 5.1 million ha in 2022) (FAO, 2024a). In Indonesia, the harvested area increased from 8.3 million ha in 2010 to 14.6 million ha in 2020 and increasing to 15 million ha in 2022 (FAO, 2024a).

Urbanization

The three subregions of Asia have different patterns of urban population and urban growth. All countries in East Asia in 2015 had an urban population percentage greater than the global average of 53.9 percent (UN DESA, 2023). All countries in South Asia have urban populations well below the global average, ranging from Nepal at 18.6 percent of its total population, through to Bhutan at 38.7 percent. In Southeast Asia, Cambodia has the lowest percentage (22.2 percent) whereas the population of Singapore is entirely urbanized.

Fourteen countries in Asia are projected to have urban population increases of greater than 10 percent between 2015 and 2025 (substantially greater than the global average growth of 4.4 percent in this period). The proportion of urban population is predicted to continue to grow through to 2050. In East Asia, China shows substantial growth in urban population through to 2050, whereas growth is limited for Japan and the Republic of Korea.

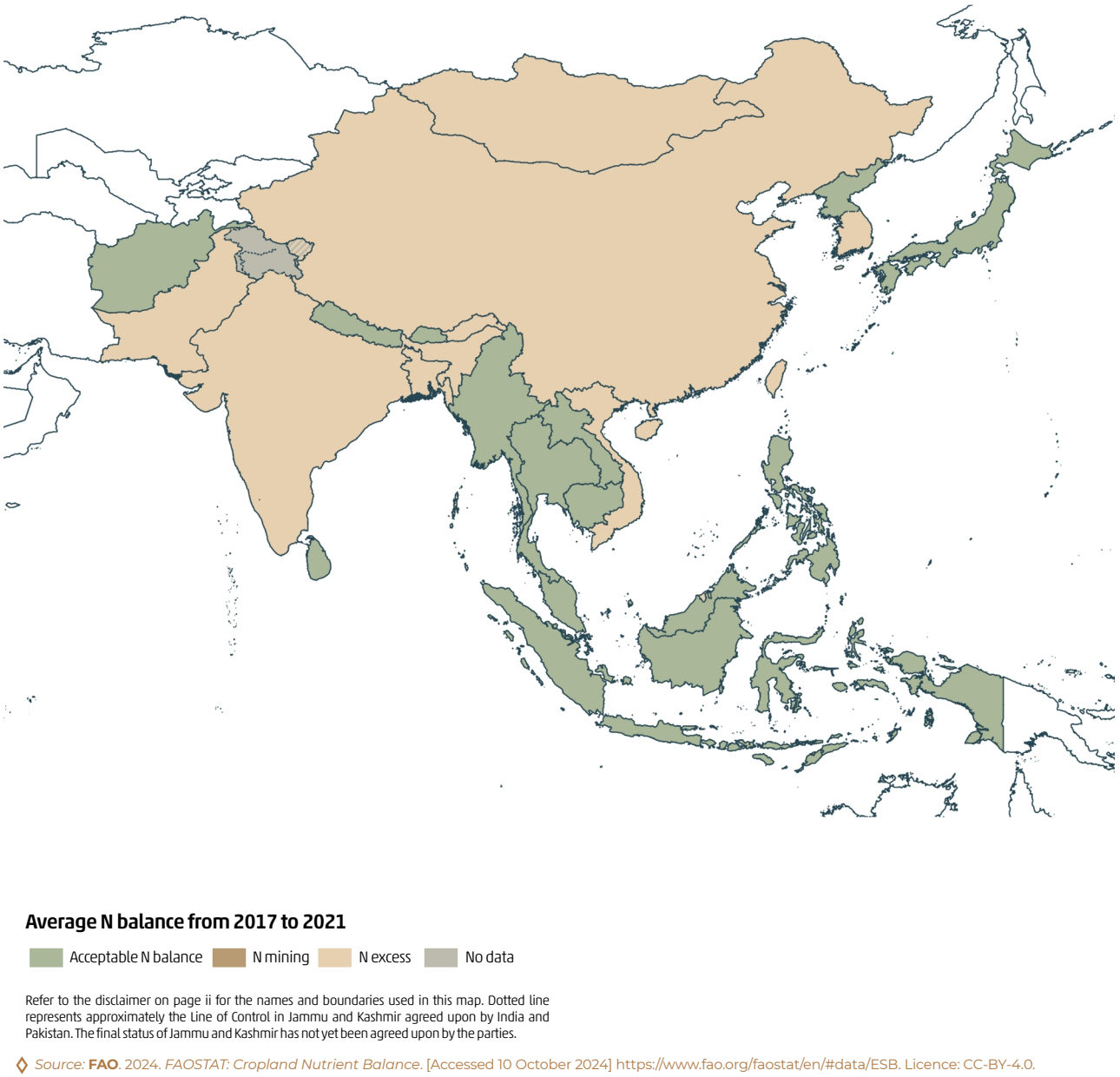
In addition, in the case of Indigenous Peoples, pressures from external actors, including the expansion of commercial agriculture and livestock operations, are causing the size of Indigenous Peoples' territories to be reduced. These drivers are transforming ancestral practices, altering sustainable food systems, and accelerating deforestation and ecosystem degradation (FAO, Alliance of Bioversity International and CIAT, 2021, p. 30–35).



8.2.3 Nutrient management

Three of the six countries in East Asia (China, Mongolia and the Republic of Korea) show evidence of unsustainably high N (FAO, 2024b) (Figure 8.7) (see Section 3.5.1 for the criteria used to identify nutrient mining and excess). In South Asia, Bangladesh, India and Pakistan also have unsustainably high annual N (FAO, 2024b). In Southeast Asia, Brunei Darussalam, Singapore and Viet Nam have unsustainably high N surpluses (FAO, 2024b), although Brunei Darussalam (10 000 ha of cropland) and Singapore (1 000 ha of cropland) (FAO, 2024a) both have small areas of cropland. No countries have negative average N balances (indicative of nutrient mining).

Figure 8.7. Countries with nitrogen excess or nitrogen mining based on average nitrogen balances for 2017 to 2021

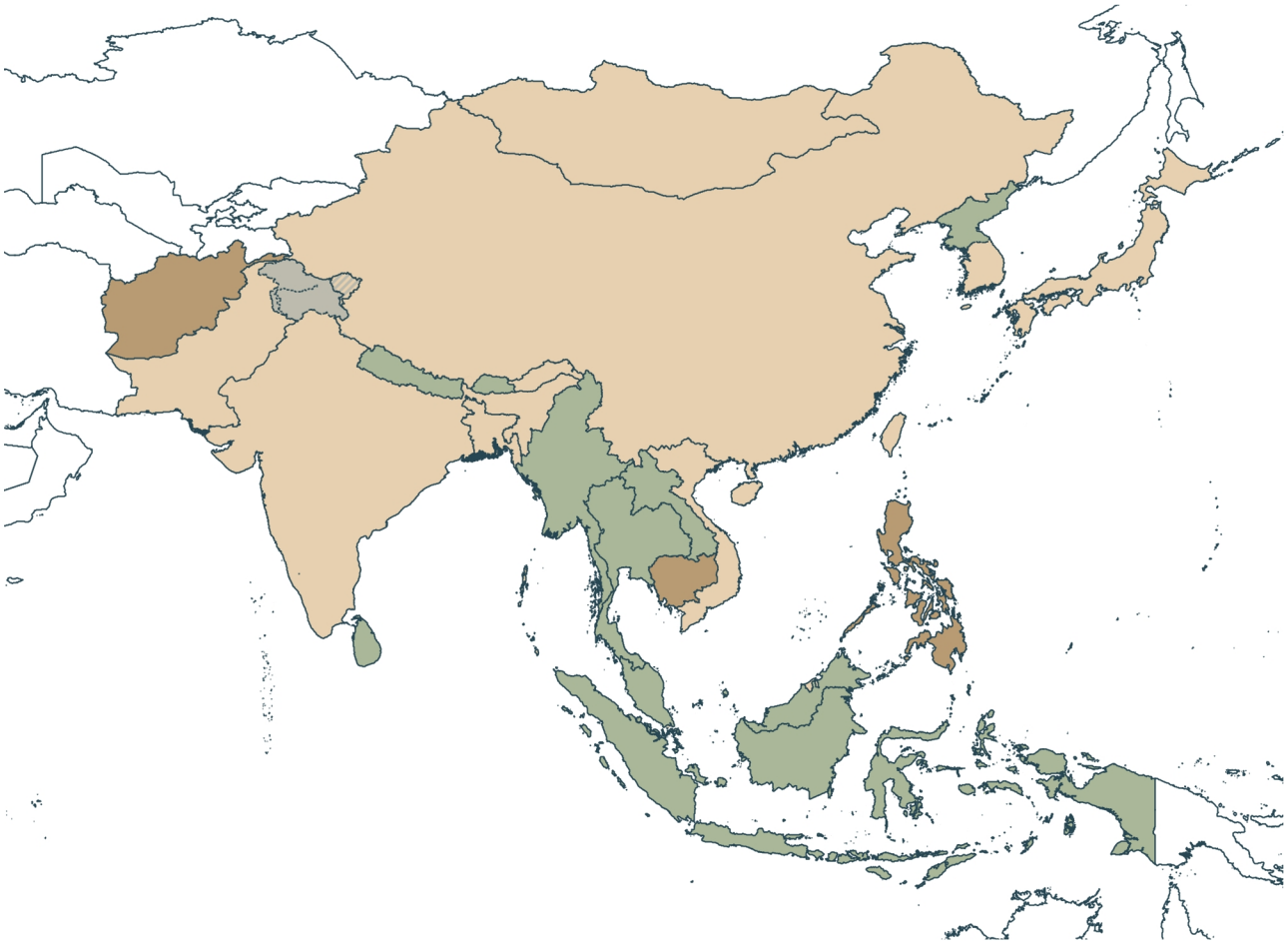


◊ *Source:* **FAO**. 2024. *FAOSTAT: Cropland Nutrient Balance*. [Accessed 10 October 2024] https://www.fao.org/faostat/en/#data/ESB. Licence: CC-BY-4.0.

In East Asia, China, Japan, Mongolia and the Republic of Korea show evidence of excess P application in the 2017–2021 period (FAO, 2024b; Zou, Zhang and Davidson, 2022) (Figure 8.8). Japan, Mongolia and the Republic of Korea all have accumulated soil P residuals from 1961 to 2019 greater than 1 000 kg P/ha (Zou, Zhang and Davidson, 2022). In South Asia, Bangladesh, India and Pakistan are all classified as showing N excess, while the accumulated soil P from 1961 to 2019 is relatively low (62 kg

P/ha for Bangladesh, 119 kg P/ha for India and 153 kg P/ha for Pakistan [Zou, Zhang and Davidson, 2022]). Afghanistan has probable P mining and a high cumulative loss of soil from 1961 to 2019 (911 kg P/ha [Zou, Zhang and Davidson, 2022]). In Southeast Asia, Cambodia and the Philippines show evidence of P mining, while Brunei Darussalam, Singapore and Viet Nam have levels of P inputs indicative of P excess.

Figure 8.8. Countries with phosphorus excess or phosphorus mining based on average phosphorus balances for 2017 to 2021 and residual soil phosphorus balances from 1961 to 2019



◊ *Sources:* **FAO**. 2024. *FAOSTAT: Fertilizers by Nutrient*. [Accessed on 9 January 2024]. https://www.fao.org/faostat/en/#data/RFN. Licence: CC-BY-4.0.

◊ **Zou, T., Zhang, X. & Davidson, E.A.** 2022. Global trends of cropland phosphorus use and sustainability challenges. *Nature*, 611(7934): 81–87. https://doi.org/10.1038/s41586-022-05220-z

◊ **EU Nitrogen Expert Panel**. 2015. *Nitrogen Use Efficiency (NUE) - an indicator for the utilization of nitrogen in agriculture and food systems*. Wageningen, Kingdom of the Netherlands, Wageningen University. https://research.wur.nl/en/publications/nitrogen-use-efficiency-nue-an-indicator-for-the-utilisation-of-n

For Sri Lanka, the average values used in Figure 8.7 and Figure 8.8 mask a major decrease in nutrient use that occurred between 2020 and 2021 due to government policies, where nutrient use on a kg/ha basis fell by 49 percent for N, 73 percent for phosphate (P_2O_5) and 35 percent for potash (K_2O) (FAO, 2024b).

Three countries in Asia show evidence of small average deficits in K balance between 2017 and 2021: the Democratic People's Republic of Korea (−6 kg K/ha), India (−6 kg K/ha) and the Philippines (−2.1 kg K/ha) (FAO, 2024b).

8.2.4 Climate change

Climate change is a driver of threats to soil functions. Climate-driven extremes, such as drought, heat waves, erratic and intense rainfall patterns, storms, floods, and emerging insect pests have adversely affected the agricultural industry in Asia (Habib-ur-Rahman *et al.*, 2022). Climate predictions suggest increases in temperature, and more erratic rainfall with higher intensity and variability in extreme climate events. Climate models predict that the global mean temperature will increase by 1.5 °C between 2030 and 2050 if changes continue at the current rate (IPCC, 2019). Changes are predicted to lead to hotter climates in the arid regions of China, India and Pakistan, along with more erratic rainfall.

In South and Southeast Asia, an increase in aridity is predicted due to a reduction in winter rainfall. Where heat waves, hot and dry weather, and variable rainfall patterns occur, it will be difficult to maintain substantial agricultural yield. Natural disasters are the main reason behind agricultural productivity (crops and livestock) losses in Asia, including extreme temperature, storms and wildfires (23 percent), floods (37 percent), drought (19 percent), and pest and animal diseases infestation (9 percent) (FAO, IFAD and WFP, 2015).



8.3. Regional status and trends of threats to soil functions for the East Asia subregion

8.3.1 Introduction

The East Asia subregion (as defined by this report), is comprised of China, Japan, the Republic of Korea, the Democratic People's Republic of Korea, and Mongolia (Table 8.1) with a total population of 1.6 billion people (UN DESA, 2023).

8.3.2 Soil erosion in the East Asia subregion

In the coastal region of East Asia, water erosion is an important issue, particularly due to the Asian monsoon, which results in torrential summer rains. Without erosion prevention measures, land use significantly impacts soil erosion rates, with exposed soil surfaces on cropland fields susceptible to erosion. In the Republic of Korea, annual soil erosion ranges between 1 t/ha in rice paddies, 3.5 t/ha in forest and grasslands, 11.1 t/ha in orchards and 41 t/ha in upland crops (Jeon *et al.,* 2022). In the Democratic People's Republic of Korea, the water erosion potential in arable lands (without erosion control measures), is approximately 30 t/ha annually (Jeon *et al.*, 2022; Jung *et al.,* 2004a; Jung *et al.*, 2019), and on steep slopes may exceed 50 t/ha.

Wind erosion is less important in coastal areas than water erosion. However, wind erosion is of concern in arid and semi-arid regions and in reclaimed tidelands. For instance, according to Jung *et al.* (2004b) the annual wind erosion of the reclaimed land in Saemangeum in the Republic of Korea, ranged from 1.8 t/ha for silty clay loam to 23.6 t/ha for sandy soil, with a mean of 7.5 t/ha. In China, 1.56 million km^2 is subject to wind erosion, with a mean annual loss of <2 t/ha, rising to 50 t/ha in severe cases (Liu *et al.,* 2020).

In Japan, deriving estimates based on USLE, and calculated with a 1 km^2 grid, it was shown that the annual erosion soil loss in the country was greater than 10 kg/ha on 31 percent of cropland, with severe to very severe levels (>30 kg/ha) on 19 percent of cropland (Kohyama *et al.*, 2012). Moderate erosion (between 1 and 10 kg/ha per year) was estimated to occur on 27 percent of cropland, with slight erosion (<1 kg/ha) on 43 percent of cropland. Highly erodible zones, mainly distributed in western Japan, characterized by complex topography and heavy rainfall, are expected to further expand in the future with more intensified typhoons and increased rainstorm events, if no prevention measures are implemented. A regional study (Takata *et al.*, 2023), suggested that severe soil loss and redeposition processes have impacted the thickness and humus content of A-horizons in a sloping vegetable fields area of Andosols. Moreover, the slope length and steepness (LS) factor of USLE calculated with a 1-m resolution suggested that low-humic soil subgroups were mainly distributed on steep slopes with a high LS factor and they were the most susceptible to soil erosion.

In the Okinawa Prefecture (which consists of subtropical islands), the Red Soil Erosion Prevention Ordinance was enforced in 1995 to control soil erosion that was leading to a decline in crop productivity and the degradation of coral reef ecosystems. The red soil erosion, of which 81 percent was attributed to croplands, was halved from 521 000 kg per year in 1993 to 246 000 kg per year in 2021 (Okinawa Prefecture, 2023). Sugarcane cultivation from soil erosion-susceptible summer planting was reduced from 47 to 26 percent, with more resistant styles being encouraged in its place, such as ratooning (sugarcane shoot establishment), increasing from 43 to 64 percent, contributed to reducing soil erosion.

In forest ecosystems, soil erosion is generally prevented by soil surface coverage with litter or

understory vegetation. Recently, however, soil erosion has become more apparent in densely planted artificial forests such as Japanese cypress (*Chamaecyparis obtusa*) plantations, due to a lack of management (Sato *et al.*, 2020), and in beech forests due to predation damage of understory vegetation by wild sika deer (*Cervus nippon*) (Katayama *et al.*, 2023). Deer numbers have increased greatly, due to the decrease in hunting by humans, rather than by wolf extinction (Iijima *et al.*, 2023).

In China, according to national monitoring of soil and water loss, nearly 28 percent of mainland China is affected by soil erosion, with 35 percent of the soil erosion-prone area at moderate to severe levels (soil erosion rate >25 t/ha annually) (MWR, 2023). China has been successful in reducing the overall area that is subject to soil erosion from 3.67 million km^2 in 1980 to 2.95 million km^2 in 2011, reducing further to 2.65 million km^2 in 2022, with a further reduction to 2.15 million km^2 possible in the long term (Cao, Qin and Wang, 2023). Liu *et al.* (2020) reported that 1.09 million km^2 in China was subject to water erosion, with a mean annual loss between 5 t/ha and 18 t/ha.

Sloping cropland, overgrazed grassland and sparsely forested areas suffer from the most intensive water erosion (Liu *et al.*, 2020). Efforts in the Loess Plateau to restore the environment have transformed the landscape (He and Zhang, 2022), and annual sediment yields from the Loess Plateau to the Yellow River have decreased from 1 600 to 200 million tonnes.

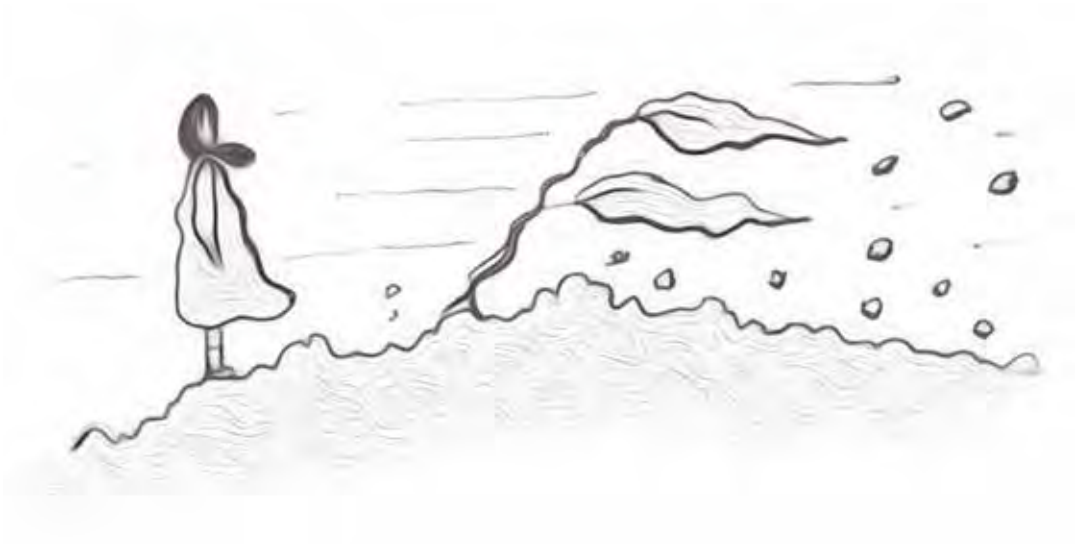
Wind erosion is severe in arid and semi-arid regions, including the Qinghai-Xizang Plateau and northeastern China (Table 8.2). Since the start of the twenty-first century, the degree of wind erosion has decreased, primarily due to afforestation (Han *et al.*, 2023).

Chinese agriculture has recently focused on reducing the planting area for root crops, particularly potato. From 2014 to 2021, there was a 3.6 percent decrease in root crop planting areas as part of broader agricultural adjustments (NBS, 2022).

Table 8.2. Water and wind erosion in China

Water erosion			Wind erosion		
Area (km^2)	Average (t/ha)	severe erosion rate (t/ha)	Area (km^2)	Average (t/ha)	severe erosion rate (t/ha)
1 090 000	5	18.20	1 560 000	<2	>50

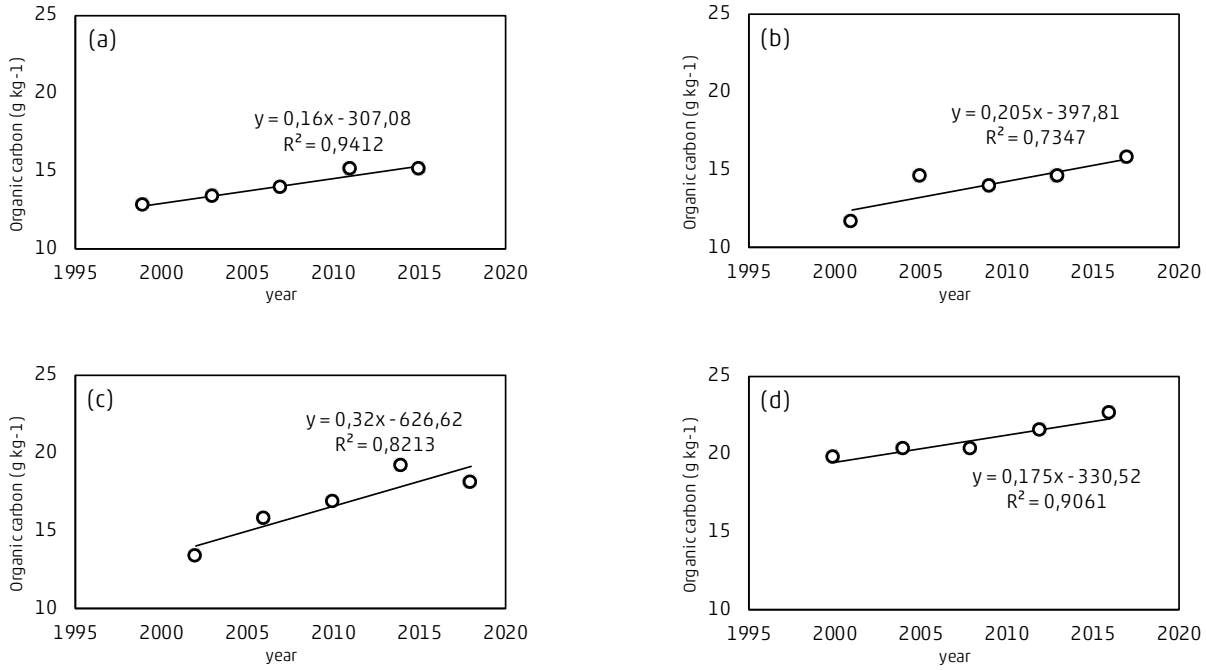
◊ *Source:* **Liu, B.Y., Xie, Y., Li, Z.G., Liang, Y., Zhang, W.B., Fu, S.H., Yin, S.Q.** *et al.* 2020. The assessment of soil loss by water erosion in China. *International Soil and Water Conservation Research*, 8(4): 430–439. https://doi.org/10.1016/j.iswcr.2020.07.002



8.3.3 Loss of soil organic carbon in East Asia subregion

In the Republic of Korea, SOC has increased across all land uses (NAS, 2020) (Figure 8.9). Carbon storage in agricultural soils (0–30 cm depth) is estimated to hold an annual storage of 1.29 million tonnes of C in the top 0–30 cm of agricultural soil, equivalent to approximately 20 percent of the 24.7 million tonnes of CO_2-equivalent emissions produced by the Republic of Korea's agriculture, livestock and fisheries sector.

Figure 8.9. Increased soil carbon content in arable lands in the Republic of Korea



Note: (a) paddy field, (b) upland crop field, (c) orchard, and (d) protected cultivation (under clear plastic or greenhouse).

◊ *Source:* **NAS (National Institute of Agricultural Science).** 2020. *Monitoring Project on Agri-environmental Quality in Korea* (5th project). Jeonju, Republic of Korea. https://scienceon.kisti.re.kr/commons/util/originalView.do?cn=TRKO202100009605&dbt=TRKO&rn=

In China, SOC storage varies significantly across ecosystems, with forestlands holding the highest levels, followed by grasslands, croplands and shrublands (Tang *et al.*, 2018) (Table 8.3). Zhao *et al.* (2018) reported that the average SOC density in topsoil (0–20 cm) of Chinese croplands increased from 28.6 t/ha C in 1980 to 32.9 t/ha C in 2011, representing a net annual increase of 140 kg/ha C. The SOC increase of Chinese croplands has been largely attributed to the increased organic inputs (such as crop roots and straw or stover) induced by economic and policy incentives (Zhao *et al.*, 2018).

Table 8.3. Soil organic carbon (SOC) pool for mineral soils of different ecosystem in China

Ecosystem		Forest	Shrubland	Grassland	Cropland	Cropland	
Sampling year		2011–2015	2011–2015	2011–2015	2011	1980 and 2011	
Depth	(cm)	0~100	0~100	0~100	0~100	0~20	
SOC stock	(Pg C)	19.98 ± 2.41	5.91 ± 0.43	24.03 ± 2.52	15.77 ± 0.57	–	
SOC density	(Mg C/ha)	–	–	–	–	28.6 (1980)	32.9 (2011)

◊ *Source:* **Tang, X.L., Zhao, X., Bai, Y.F., Tang, Z.Y., Wang, W.T., Zhao, Y.C., Wan, H.W.** *et al.* 2018. Carbon pools in China's terrestrial ecosystems: new estimates based on an intensive field survey. *Proceedings of the National Academy of Sciences*, 115(16): 4021–4026. https://doi.org/10.1073/pnas.1700291115

Soil organic carbon in cultivated land in Japan was monitored at five-year intervals with 20 000 fixed points from 1979 to 1998. There was a decline in SOC stock in cultivated land (−2.3 million tonnes of C per year) from 1989 to 1998, mainly due to the reduced area of cultivated land (Shinjo and Takata, 2021). Since 2008, the monitoring programme has continued at a reduced scale, covering 3 500 sites. According to Matsui *et al.* (2021), Japan's total SOC stock was 379 ± 17.9 million tonnes of C, including rice paddies (176 ± 12.8 million tonnes of C), croplands (100 ± 8.6 million tonnes of C), orchards (25 ± 1.7 million tonnes of C) and grasslands (78 ± 8.9 million tonnes of C). In turn, conversions from forestland to cropland or grassland across Japan resulted in a decline in soil carbon stock (Koga *et al.*, 2020) (Table 8.4).

Table 8.4. Changes in soil carbon stocks (mean ± standard deviation) in Japan after land-use conversions from forests to agricultural land

Land uses	Upland field	Orchard	Grazing land	Pasture
Number of sites	25	3	14	4
Before conversion (Mg C/ha)	151.1 ± 90.3	93.8 ± 69.7	120.8 ± 55.1	124.8 ± 28.2
After conversion (Mg C/ha)	117.3 ± 72.0	47.5 ± 29.5	95.3 ± 46.4	119.9 ± 18.5
Decrease in soil carbon stock (%)	23.8 ± 25.3	43.4 ± 29.2	17.7 ± 21.2	1.8 ± 18.5

◊ *Source:* **Koga, N., Shimoda, S., Shirato, Y., Kusaba, T., Shima, T., Niimi, H., Yamane, T.** *et al.* 2020. Assessing changes in soil carbon stocks after land use conversion from forest land to agricultural land in Japan. *Geoderma*, 377: 114487. https://doi.org/10.1016/j.geoderma.2020.114487

The area of wetland in China was estimated as 189 120 km^2 in 2000 and 162 800 km^2 in 2020 (Ren *et al.*, 2023). Soil organic carbon density within a 1 m depth increased from 25.03 ± 3.11 kg/m^2 in 2000 to 26.57 ± 3.73 kg/m^2 in 2020, a net increase of 6.15 percent (Ren *et al.*, 2023). However, total wetland SOC storage decreased from 4.73 ± 0.58 billion tonnes of C in 2000 to 4.35 ± 0.61 billion tonnes of C in 2020, representing a net decrease of 8 percent. The decrease of wetland SOC storage was mainly due to an approximately 13.6 percent decline of wetland area from 2000 to 2020, despite the increase in SOC density over the same period. In wetlands without land-use changes, the SOC accumulation rate was 107.5 ± 12.4 g/m^2 of C annually since 2000 (Ren *et al.*, 2023).

The ecosystem C stock of Japanese mangroves has a relatively high SOC stock (250 t/ha of C) and above ground C biomass (80 t/ha of C) (Iimura *et al.*, 2019) supported by high primary production (Ohtsuka *et al.*, 2019). Mangroves and other wetlands in Japan, largely designated as national preserves, are generally not threatened by human activities such as deforestation. Intrusion of terrestrial soils from nearby catchment occurs in many Japanese mangrove forests, but the effect on mangrove SOC stock remains to be studied.

The mean density of soil inorganic carbon (SIC) in topsoil (0–30 cm) in China decreased from 3.55 kg/m^2 in the 1980s to 3.21 kg/m^2 in the 2010s (Song *et al.*, 2022). Total SIC stocks have decreased by 1.37 ± 0.37 billion tonnes of C (Song *et al.*, 2022). The average annual SIC losses across China (0.046 billion tonnes of C) and in cropland (0.016 billion tonnes of C), account for 17.6 to 24 percent of the terrestrial annual C sink (0.19 to 0.26 billion tonnes of C) and 57.1 percent of the annual SOC sink in cropland (0.028 billion tonnes of C) in China. Nitrogen deposition and climate change influence SIC cycling in China, and an estimated further 19.12 to 19.47 percent of SIC stocks in China might be lost by 2100 (Song *et al.*, 2022). A recent study of the global SIC dynamics showed that in other parts of Asia, SIC was declining (Huang *et al.*, 2024).

Soil inorganic carbon is not naturally precipitated in Japan, due to slightly acidic soil conditions (pH <5–7) (Zamanian, Zarebanadkouki and Kuzyakov, 2018), so depletion is not a concern. Conversely, anthropogenic SIC accumulates in urban soils, where asphalt supplies calcium carbonate ($CaCO_3$) to underlying mineral soils (Kida and Kawahigashi, 2015). However, this increment should be carefully evaluated in C accounting, as it is offset by the CO_2 emissions throughout the industrial processes including mining, transport and construction.

8.3.4 Loss of soil biodiversity in East Asia subregion

Soil biodiversity in the Republic of Korea varies across land use. Uplands have higher soil plant and insect biodiversity than paddy fields or orchards (NAS, 2016) (Table 8.5). Based on the Simpson and Shannon indices (Simpson, 1949; Barnes *et al*, 1998), the biodiversity of the soil microbial community was the greatest in orchards as an ecotone (a transition area) between arable lands and forests. The relative dominance of soil microorganisms are as follows: proteobacteria (33.6 percent), acidobacteria (13.9 percent), actinobacteria (9.4 percent), chloroflexi (7.8 percent), bacteroidetes (7.5 percent), and firmicutes (5.3 percent). Invertebrates in paddy fields comprised of 125 species from 58 families across 20 orders, while birds included 74 species from 34 families (NAS, 2016).

Table 8.5. Biodiversity of plants and insects in the Republic of Korea soil in different land uses

Plants	Paddy	Upland	Orchard
Families	60	103	90
Genera	210	320	279
Species	339	448	422
Subspecies	3	2	1
Varieties	28	74	41
Insects			
Order	9	13	11
Families	56	108	78
Species	175	487	268

◊ *Source:* **NAS**. 2016. *Monitoring Project on Agri-environmental Quality in Korea* (4th project). Jeonju, Republic of Korea. https://data.doi.or.kr/10.23000/TRKO201700006248

The diversity of soil organisms is generally lower under conventional tillage in China, compared to conservation tillage which can have higher species richness (1 to 8 percent), density (25 to 57 percent), and biomass (30 to 50 percent) in the entire soil assemblages, including microorganisms, nematodes, Collembola, mites and earthworms. Conservation tillage can also improve the connectivity of the soil food web (14 to ~32 percent), further promoting the performance of soil biotic function in soil structure formation, soil carbon sequestration, N-efficient utilization and crop yield stability (Zhang *et al.*, 2022a). No-tillage and cover crop (rye [*Secale cereale*]) practices in andosols increased C stocks by 12 percent and microbial diversity (using Chao1) by 20 percent for bacteria, but not for fungi (Gong *et al.*, 2022). The expansion into Japan of *Fusarium oxysporum* f. sp. *cubense*, which causes Panama disease (Fusarium wilt), was observed in the banana plantation soils of Miyakojima Island, Okinawa (Nitani *et al.*, 2018). Fungi can be one of the main N_2O producers in cropland soils, but the predation by fungivorous mites reduced N_2O emissions (Shen *et al.*, 2021).

In forest ecosystems, the activities of nitrifying bacteria were limited by soil acidification (Isobe *et al.*, 2020). An increase in air temperature was shown to accelerate organic matter decomposition through enhanced lignin-degrading enzyme activity (Fujii *et al.*, 2020). The deer population has also increased rapidly, by up to 3 million individuals in Japan (ten times the deer population in 1990) due to decreases in hunting by humans, rather than wolf extinction (Iijima *et al.*, 2023). The increase in deer population has caused the overgrazing of understory vegetation in forests, accelerating soil erosion and the loss of SOM, decreasing soil heterotrophic respiration and microbial biomass (Chen *et al.*, 2023; Katayama *et al.*, 2023) as well as soil erosion (as noted previously).

8.3.5 Nutrient mismanagement in East Asia subregion

Starting in the 2000s, the Government of the Republic of Korea has implemented a range of policies aimed at reducing the use of inorganic fertilizers (Jeon *et al*., 2022). The measures include cutting subsidies for fertilizers, mandating participation in soil testing programmes to receive fertilizer recommendations, requiring farmers' compliance with the direct payment system, and offering financial incentives for participation in agricultural environmental conservation programmes. Nevertheless, annual N and P balances in the Republic of Korea continue to rank among the highest within OECD countries, with levels reaching 147 kg/ha of N and 23 kg/ha of P (Jeon *et al*., 2022). Despite the reduction in chemical fertilizer use, there has been a substantial increase in the annual application of organic manure, which has grown from 78 kg/ha of N in 1990 to 157 kg/ha in 2017 (Lim *et al*., 2021). Consequently, there has been growing interest in exploring new approaches to recycle waste from livestock, such as through bioenergy and biochar research.

Excess N and P fertilizers are applied in many Chinese croplands, causing surpluses. Annual N input ranges from 206 to 532 kg/ha and annual N output from 104 to 259 kg/ha under conventional N management across main cropping systems of China, resulting in 59 to 349 kg/ha of annual N surplus (Zhang *et al.,* 2019). Phosphorus is easily fixed by soils and so little is lost into the environment, which leads to P accumulation in arable land soils, especially during vegetable production. The annual input of P ranges from 772 to 2 458 kg/ha versus an annual P output ranging from 48 to 155 kg/ha in the various greenhouse vegetable systems, resulting in 711 to 2 303 kg/ha of annual P surplus (Tian *et al*., 2022). However, Song *et al.* (2024) showed that even under high P input conditions, the total P in Chinese surface soils is still declining, mainly due to soil erosion. The Chinese government developed an action plan for zero increase in fertilizer use by 2020, which is expected to lower nutrient surplus and its environmental effects (MOA, 2015).

The nutrient surplus in East Asia from 1996 to 2020 varies by nutrient and country (Figure 8.10). In China, the annual N and P surpluses have declined since 2015. In Japan, the surplus P has declined steadily, though it remains high compared to other countries, partly due to the inherent P-deficient soil condition (high P retention), that occurs in Andosols. It is expected to decrease due to the national policy known as the MIDORI Strategy for Sustainable Food Systems, launched by the Government of Japan in 2021 (MAFF, 2023), promoting conservation (or organic) agriculture by improving P use efficiency (Sugiura *et al.,* 2021; Yamamoto *et al.,* 2023).

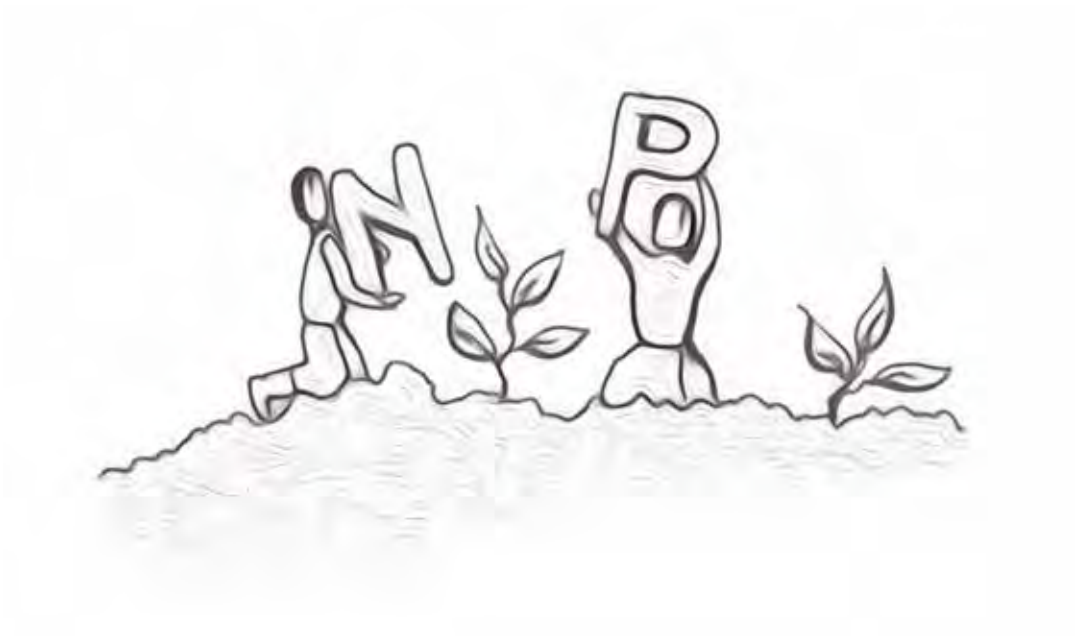
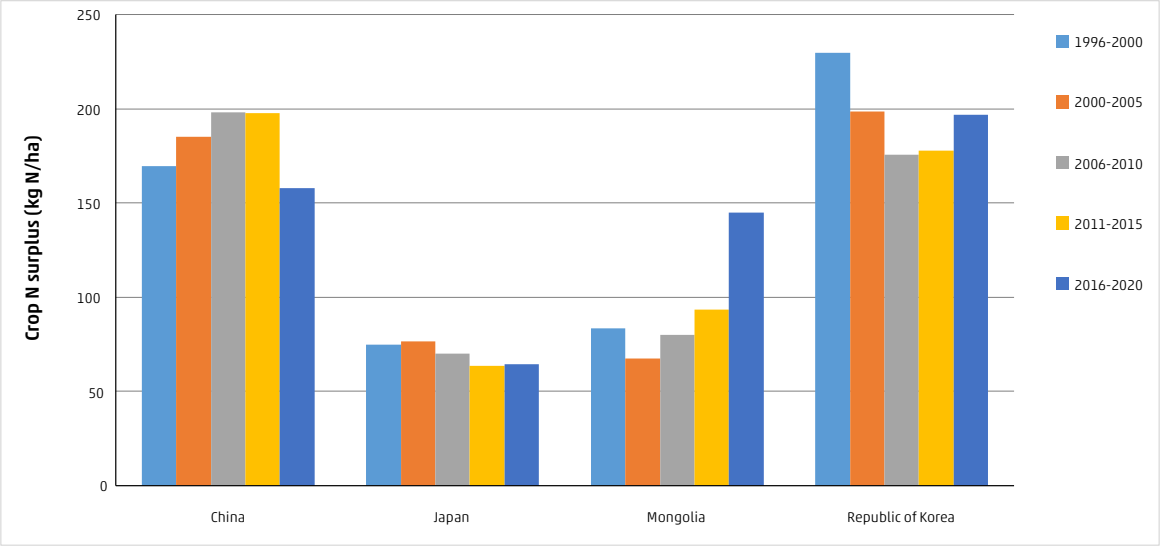
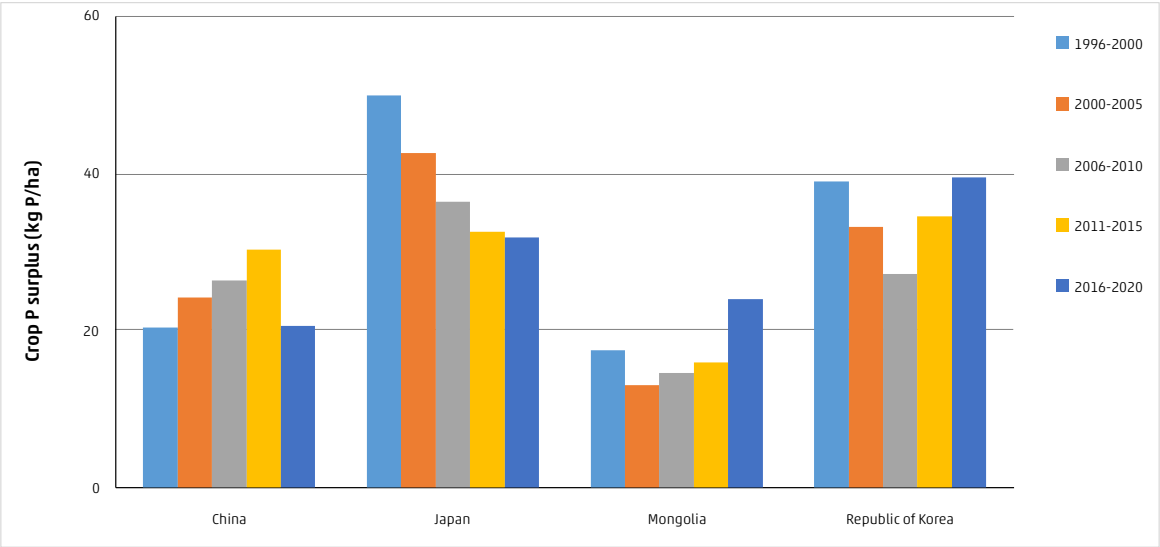
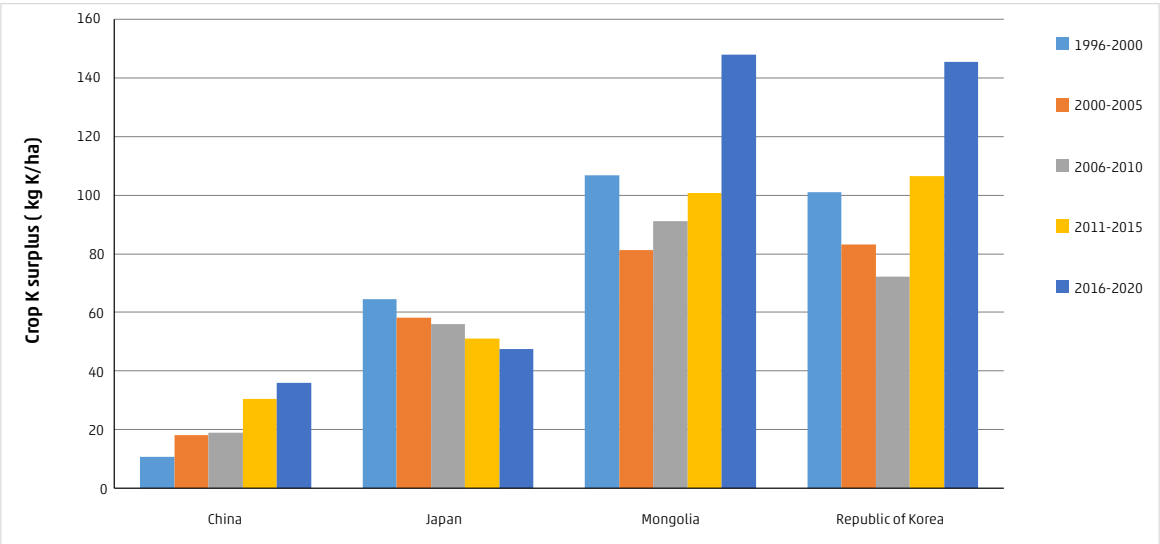


Figure 8.10. Time course of nitrogen, phosphorus and potassium nutrient surplus in East Asia between 1996 and 2020







Note: The figure excludes the Democratic People's Republic of Korea, due to no information being available.

Source: Adapted from **FAO**. 2023. *FAOSTAT: Cropland Nutrient Balance*. [Accessed on 23 November 2023]. https://www.fao.org/faostat/en/#data/ESB. Licence: CC-BY-4.0.

Micronutrients are removed through crop harvesting every year without sufficient replenishment through micronutrient fertilization. This issue is pronounced in the northwest of China, where maize cultivation contributes to declining soil micronutrient levels, underscoring the need for micronutrient fertilization in the region (Peng *et al.*, 2023). Approximately 55 percent of arable lands in China are now Mg-deficient, increasing in severity from northern to southern China (Ishfaq *et al.*, 2022).

Based on the national survey of agricultural soils in the Republic of Korea, soil acidity has decreased over time in arable lands due to the supply of soil neutralizers such as $CaCO_3$ (NAS, 2020). Soil pH increased from 5.9 in 2001 to 6.4 in 2017 for upland crop fields (NAS, 2020). The increase in soil pH was smaller in paddy soil than in upland soils, with paddy soil pH at 5.8 in 2003 and 6 in 2019.

Since the start of the twenty-first century, a significant acidification trend has impacted Chinese soils, with approximately 0.45 million km² – or 4.7 percent of China's total land area – now classified as acidic (pH <5.5) (Zhao *et al.*, 2023). The majority (75 percent) of these acidic soils are in the southern tropical and subtropical red and yellow soil regions while 17 percent are in the northeast region. Among China's arable lands, approximately 21 310 km² have a pH below 5.0, while 45 530 km² have a soil pH of 5 to 5.5 and 545 720 km² fall between pH 5.5 and 6.5.

8.3.6 Salt accumulation in East Asia subregion

There is approximately 36.9 million ha of concentrated and continuous salt-affected soils in China, of which 2.1 million ha are salt-affected farmland and 10.2 million ha are salt-affected uncultivated land with agricultural potential (Li, 2023; Wang *et al.*, 2024) (Table 8.6). There are about 1 million ha of salt-affected farmland in inland river irrigation areas and oasis farming areas in Xinjiang and upper and middle reaches of the Yellow River. About 0.38 million ha of farmland suffer from soil salinization and sodification in the Songnen Plain of Northeast China due to the unique hydrological conditions, including enclosed relief, sodium (Na)-rich parent materials and slow groundwater flow. In the coastal regions of Eastern China and in the North China Plain, approximately 0.72 million ha of farmland is salinized due to waterlogging and supplementary irrigation with saline water. In the North China Plain, the area of salt-affected soils has decreased substantially since the 1980s, due to lowering of the groundwater table (Wang *et al.*, 2024).

Table 8.6. Distribution area of salt-affected farmland and uncultivated land in China

Distribution area	Salt-affected uncultivated land (ha)	Salt-affected farmland (ha)
Total	13 500 000	7 6000 000
Northeast China	2 600 000	1 400 000
Xinjiang area	7 700 000	2 500 000
Upper and middle reaches of the Yellow River	3 100 000	1 200 000
Coastal region and North China	100 000	2 500 000

◊ *Sources:* **Li, Y.Y.** 2023. Systematically promoting the comprehensive utilization of saline-alkali land in China. *Guangming Daily*, 25 July 2023. Beijing. [Cited 4 November 2024]. https://epaper.gmw.cn/gmrb/html/2023-07/25/nw.D110000gmrb_20230725_2-07.htm

◊ **Wang, G., Ni, G., Feng, G., Burrill HM., Li, J., Zhang, J. & Zhang, F.** 2024. Saline-alkali soil reclamation and utilization in China: progress and prospects. *Frontiers of Agricultural Science and Engineering*, 11(2): 216–228. https://doi.org/10.15302/J-FASE-2024551

In the Republic of Korea, reclaimed tidal land has been developed for rice paddies (Lim *et al.*, 2020). While being used as a rice paddy, water from artificial freshwater lakes has been used to overcome soil salt damage through water exchange and leaching. Irrigation canals are mainly made of concrete structures, while drainage canals are made of earth. However, reclaimed tidal land that blocks tidal mud flats from the sea has an inherent limitation of being vulnerable to salt damage.

Land-use plans in the Republic of Korea for unsold reclaimed tidal lands are updated every five years. To expand usage beyond rice paddies and support upland crops, effective salt management and consistent high-quality water supply are essential to prevent resalinization. Supplying irrigation water to upland crops using pipelines is being considered, and efforts are underway to provide more desalinized water for field crops (Republic of Korea, 2022). Seawater infiltration in coastal areas due to the use of inland groundwater has become a problem, so a seawater infiltration monitoring network (with 257 sites as of 2022), was established and has operated continuously since 1998 (MAFRA and KRC, 2022).

8.3.7 Soil pollution in East Asia subregion

Persistent organic pollutants (POPs) have contaminated soils in some industrial sites in China (Dong *et al.*, 2021). High molecular weight polycyclic aromatic hydrocarbons (PAH) are the main residual components in the soils of coking-contaminated sites. The concentrations of phthalic acid esters (PAEs) were higher in North and Central China owing to concentrated agricultural activities. The levels of polychlorinated biphenyls (PCBs) were high in East and South China primarily because of past industrial operations and improper electronic waste processing (Sun *et al.*, 2018).

China is one of the largest producers and consumers of antibiotics, leading to a high antibiotic detection rate in soil (Lyu *et al.*, 2020). Soil antibiotic concentrations can range from 10^{-3} to 10^{3} parts per billion (µg/kg), with the ecological risk varying across different regions of China (Qiao *et al.*, 2018). The concentration of antibiotics in soil decreased between 2008 and 2020 (Zhao *et al.*, 2023). Pharmaceutical and personal care products (PPCPs), environmental hormones or endocrine disruptors, and new POPs are present in farmland soils of China due to wastewater irrigation (Lyu *et al.*, 2022).

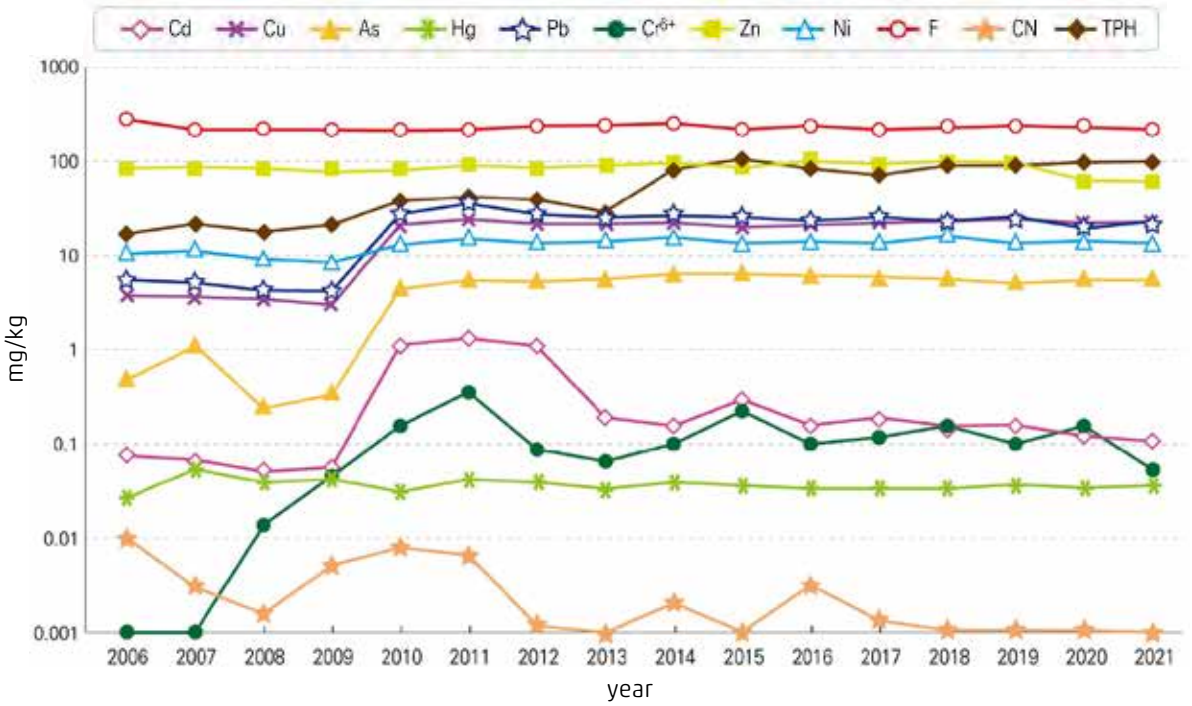
Attention should be paid to pesticide residues in soil in agricultural areas in China (Tang *et al.*, 2021). Pesticide risk in soil might last longer due to the previous legacy, although the usage has declined since 2014 (Zhang *et al.*, 2022c; Zhou *et al.*, 2023).

Pollution of soils by microplastics is increasingly being detected in East Asia, with soils functioning as a microplastics reservoir in the environment, as well as the ocean. The concentration of microplastics in soil ranges from 0 to 690 000 particles per kg of soil (Khan *et al.*, 2023). However, the concentration of microplastics in agricultural soil tends to be especially high and is often highly contaminated. Kedzierski *et al.* (2023) estimated the global stock of microplastics in agricultural soils to range between 1.5 and 6.6 million tonnes, one to two orders of magnitude higher than the quantity of microplastics present at the ocean surface. Microplastics in farmland are derived from mulching film, sludge and wastewater irrigation, and are persistent contaminants (Zhou *et al.*, 2020; He *et al.*, 2023). The microplastic content in farmland soil in some areas of China can reach up to 4 537 items per kg of dry soil (Ren *et al.*, 2023).

In Japan, microplastics derived from polymer-coated fertilizers (microcapsules) are becoming increasingly evident in soil. Katsumi *et al.* (2021a) found that the annual outflow of microcapsules from the Japanese rice paddies accounted for approximately 10 percent of the annual inflow, indicating that most of the applied microcapsules accumulated in the rice soils. Microcapsules ranging from 6 to 369 mg/kg of soil (with a mean of 144 mg/kg) have been reported in paddy soils (0–20 cm) in the Tedori River fan area, in Ishikawa Prefecture, Japan (Katsumi *et al.*, 2021b).

The Soil Environment Conservation Act in the Republic of Korea requires that a national basic plan be established and announced every ten years (Republic of Korea, 2015). Monitoring of 23 soil pollutants is being conducted at 1 000 designated sites nationwide in the country. In the 1 000 sites investigated, all measured parameters were within the soil pollution guideline standards, and the average pollution level for most pollutants was similar to the previous year (MOE, 2022) (Figure 8.11).

Figure 8.11. Concentrations of major soil pollutants measured at 1 000 sites in Republic of Korea from 2006 to 2021



Note: Cd = cadmium, Cu = copper, As = arsenic, Hg = mercury, Pb = lead, Cr = chromium, Zn = zinc, Ni = nickel, F = fluorine, Cn = cyanide, and TPH = total petroleum hydrocarbons.

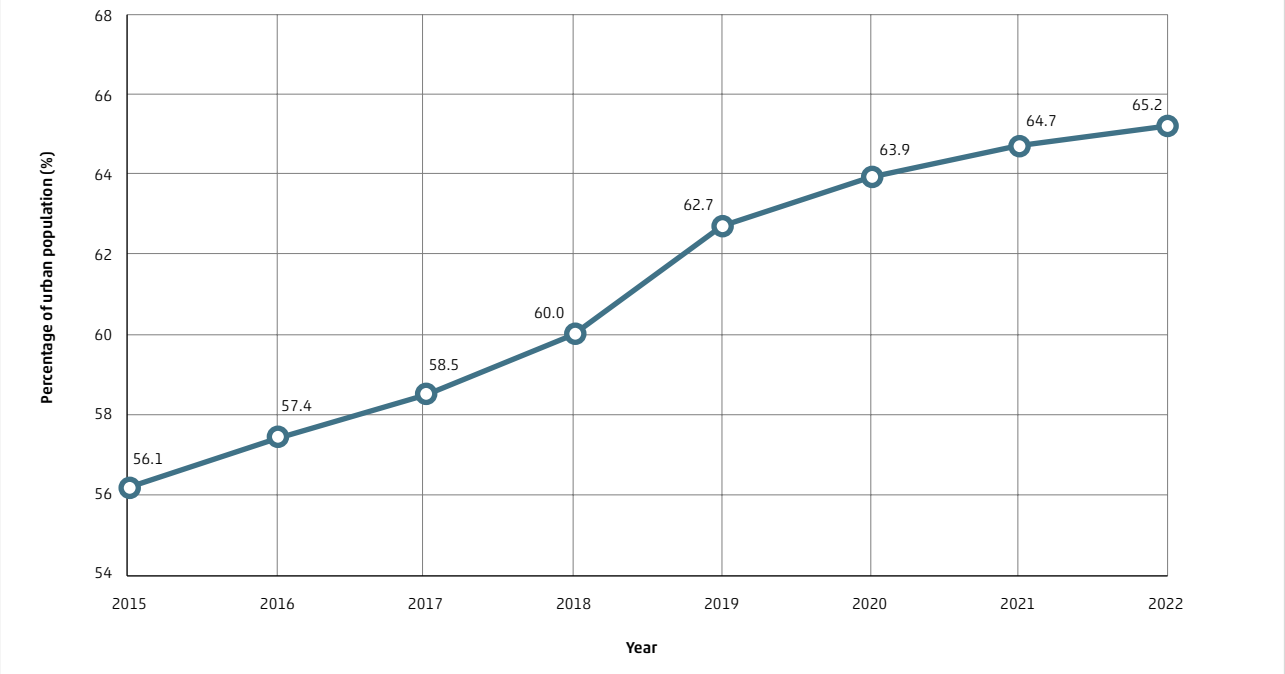
◊ *Source:* **MOE (Ministry of Environment)**. 2022. *2021 Soil Measuring Network and Soil Pollution Survey Results Report*. Sejong, Republic of Korea. https://www.me.go.kr/home/web/public_info/read.do?pagerOffset=30&maxPageItems=10&maxIndexPages=10&searchKey=&searchValue=&menuId=10123&orgCd=&condition.publicInfoMasterId=4&condition.deleteYn=N&publicInfoId=132&menuId=10123

8.3.8 Urban expansion and soil sealing in East Asia subregion

The percentage of urban population in China has increased by 9.1 percent in China between 2015 and 2022 (NBS, 2020, 2023) (Figure 8.12). According to a system dynamic model, Chinese urbanization levels will reach approximately 75 percent in 2050 (Gu *et al.*, 2017). Urbanization takes up agricultural land, especially rice paddies in southern China, and affects agricultural production. Impervious surface areas cover 164 000 km^2 in China (Zhang *et al.*, 2020), with runoff of the sealed surfaces accounting for about 95 percent of urban flood volume (Zhang and Yang, 2023).



Figure 8.12. Percentage of urban population in China



◊ *Sources:* **NBS (National Bureau of Statistics)**. 2020. 中华人民共和国2019年国民经济和社会发展统计公报 [Statistical Bulletin of the People's Republic of China on National Economic and Social Development 2019]. Beijing. https://www.gov.cn/xinwen/2020-02/28/content_5484361.htm

◊ **NBS**. 2023. 中华人民共和国2022年国民经济和社会发展统计公报 [*Statistical Bulletin of the People's Republic of China on National Economic and Social Development 2022*]. Beijing. https://www.gov.cn/xinwen/2023-02/28/content_5743623.htm

Urbanization and soil sealing notably affect SOC storage. Research across 22 Chinese cities revealed that highly urbanized areas had SOC increases of 104.4 percent in topsoil since the start of the twenty-first century (Zhang *et al.*, 2022d). China has a SOC stock of 1 037 million tonnes to 100 cm depth under the impervious surface, with a mean density of 6.21 ± 3.90 kg/m^2 (Ding *et al.*, 2022). In recent years, the design of porous sidewalks and parking lots (Figure 8.13) has reduced sealing surfaces in cities (Zhang and Yang, 2023).



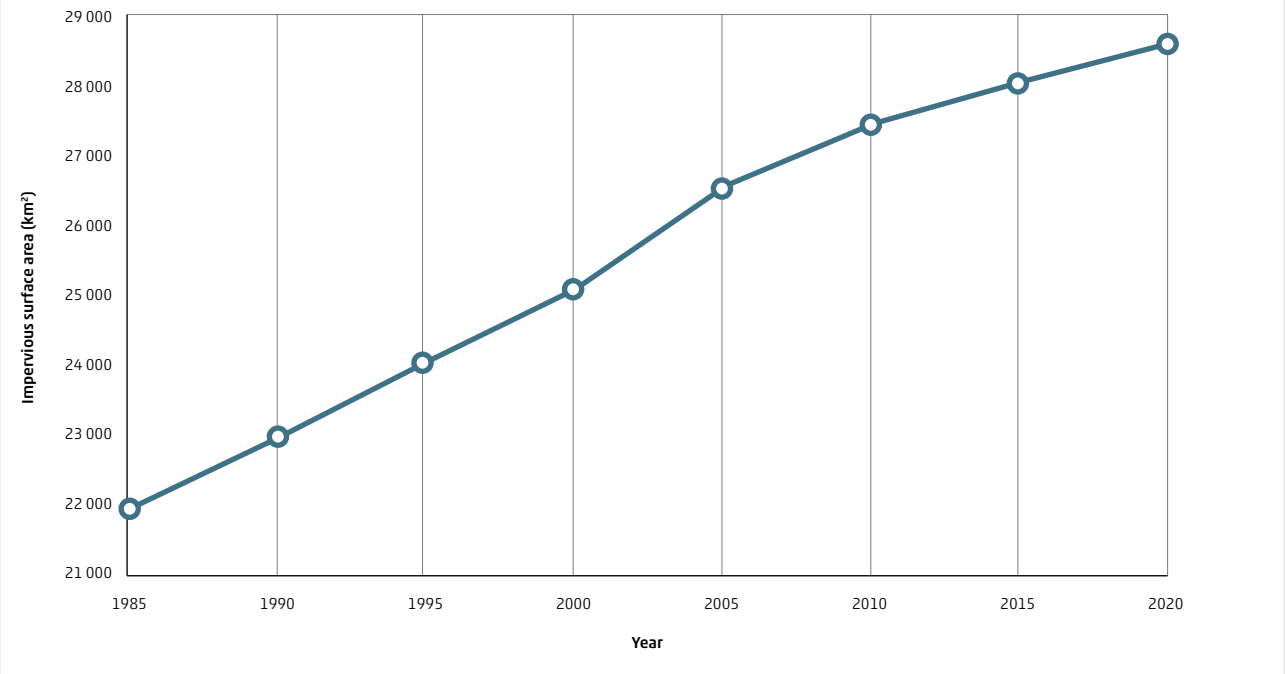
Figure 8.13. Porous surface in Chinese city



According to Zhang *et al.* (2022b), the expansion rate of impervious surface areas declined since 2005 in Japan (Figure 8.14). In response to environmental challenges presented by impervious surfaces, the Japanese government has promoted green infrastructure through the Ministry of Land, Infrastructure, Transport and Tourism (MLIT)'s 2023 guidelines, which included 27 governmental support programmes for green infrastructure installation (MLIT, 2023). Although there are some case studies about spatial analysis of green infrastructure (e.g. Ishikawa, Yamamoto and Yokoyama, 2021), the advantage is not well understood nationally in Japan.



Figure 8.14. Area of impervious (urbanized) surfaces in Japan over 1985–2020 period



◊ *Source:* Authors' own elaboration and **Zhang, X., Liu, L.Y., Zhao, T.T., Gao, Y., Chen, X.D. & Mi, J.** 2022. GISD30: global 30 m impervious-surface dynamic dataset from 1985 to 2020 using time-series Landsat imagery on the Google Earth Engine platform. *Earth System Science Data*, 14: 1831–1856. https://doi.org/10.5194/essd-14-1831-2022

8.3.9 Other threats in East Asia subregion

Soil compaction

China has experienced an increase in mechanization for cropping, rising from 63 to 72 percent from 2015 to 2021. Even for single-season crops, mechanical operations (including ploughing, planting, harvesting and field management) can occur up to 8 to 15 times. Up to 90 percent of the cropped land may be compacted, with between 10 and 80 percent of the area experiencing multiple instances of mechanical grinding (Fu *et al.*, 2022). Soil compaction has been identified in the Guanzhong Plain and the North China Plain (Ren *et al.*, 2023). The black soil region in Northeast China has a higher risk of compaction, although the southern, red soil area is also affected, with low-yield fields exhibiting an average soil dry bulk density of 1.4 g/cm^3, 14 percent higher than that of medium-yield fields (Jin *et al.*, 2018). However, adopting rational soil tillage practices can potentially reduce the compaction pressure exerted by machinery on soil structure (Tian *et al.*, 2023).

Soil water availability

Between 2015 and 2021, the average annual area affected by floods in China reached 6.5 million ha, while droughts impacted an annual average of 8.4 million ha (MWR, 2023). Droughts resulted in an annual loss of approximately 14.78 billion kg of grain (MWR, 2023). The high temperatures and droughts experienced across the Yangtze River Basin in 2022 serve as a reminder of the intensified impacts of extreme weather on agricultural production and agroecosystems within the context of climate warming (Xu *et al.*, 2023).

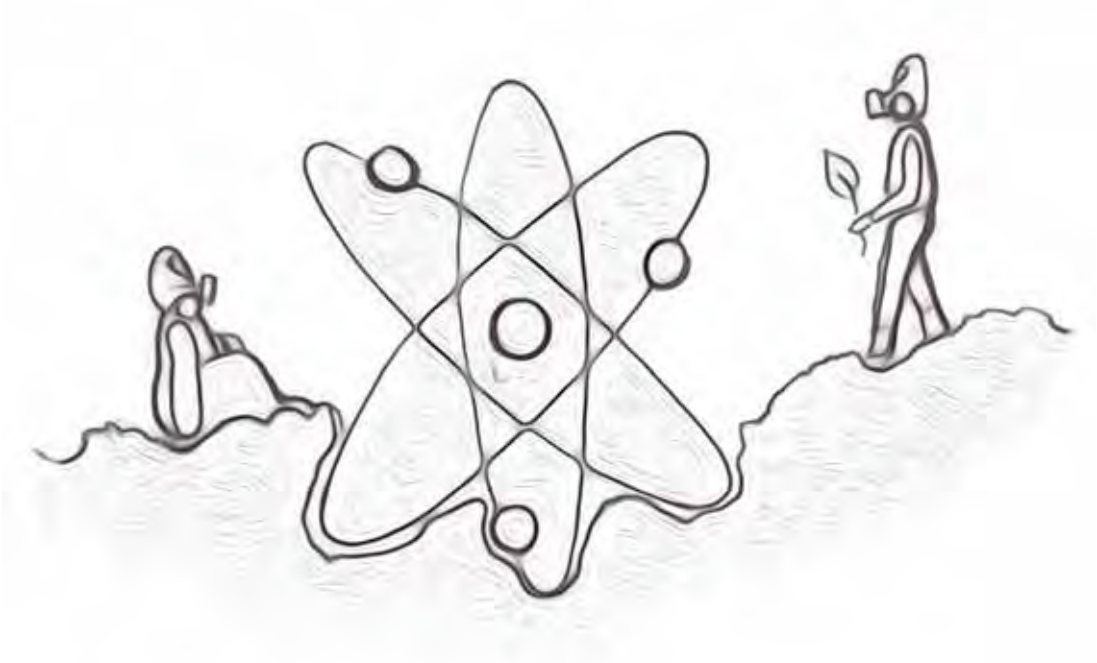
Soil reuse

The Government of the Republic of Korea has prepared guidelines to establish the safe use of by-product fertilizers through crop and soil environmental impact assessments when continuously applied to agricultural land (NAS, 2014, 2022). In addition, related research is in progress to respond to the adverse effects on the soil environment caused to the addition of stone dust and recycled soil materials generated during construction and industrialization processes to agricultural soils (Lee *et al*., 2021).

Caesium-137 derived from the Fukushima Daiichi Power Plant accident in 2011

The accident at the Fukushima Daiichi Power Plant led to deposition of 2.7 quadrillion becquerels (Bq) of caesium-137 (^{137}Cs) on the terrestrial environment around Fukushima Prefecture, Japan. Approximately 3 900 km^2 was contaminated with at least 100 000 Bq/m^2 of ^{137}Cs, and 254 km^2 was severely contaminated with more than 1 480 000 Bq/m^2 of ^{137}Cs (Onda *et al*., 2020). By 2017, 413 km^2 of agricultural fields and 126 km^2 of forests had been decontaminated by removing surface soil layers or through inversion tillage, and approximately 14 million m^3 of the removed soils were stored in an interim storage facility (Evrard, Laceby and Nakao, 2019). Ninety-eight percent of the affected forestland has not yet been remediated. Between 85 and 95 percent of the deposited ^{137}Cs is contained in the 0–5 cm soil depth, with a minimal increase in ^{137}Cs below 10 cm depth (Takahashi *et al*., 2022). Global fallout is the major source of ^{137}Cs in other regions including East Asian countries, where ^{137}Cs inventories ranged from 1 000 to 3 700 Bq/m^2 and were positively correlated with latitude (Tagami, Tsukada and Uchida, 2019).

To reduce the transfer of ^{137}Cs from soil to plants in agricultural fields, the maintenance of exchangeable K levels higher than 20–25 mg K_2O/100 g of soil in contaminated soils is recommended. This level of K reduces ^{137}Cs absorption by plants, minimizing the contamination of crops. (Yamaguchi *et al*., 2016). To manage soils contaminated by radiation, a technology development strategy was formulated by Japan's Ministry of the Environment in 2016 for volume reduction and recycling of removed soils with less than 8 000 Bq/km, accounting for 80 percent of the stored soil. While the removed soil is a valuable resource that could be reused, no designated site for recycling this soil has yet been decided, due to limited public understanding (IAEA, 2023).



8.4 Regional status and trends of threats to soil functions for the South Asia subregion

8.4.1 Introduction

This section summarizes new information on major threats to soil functions in the South Asia subregion. The subregion (as defined for this report) is comprised of Afghanistan, Bangladesh, Bhutan, India, Maldives, Nepal, Pakistan, and Sri Lanka (Table 8.1) with a total area of 400 million ha (Table 8.1) and a total population of 1.9 billion people.

8.4.2 Soil erosion in South Asia subregion

Soil erosion is the greatest threat to soil functions in the South Asia subregion. India, the largest country in the South Asia, has identified soil erosion as the major cause affecting soil functions. Bhattacharyya *et al.* (2023) reported that according to the most recent harmonized database on land degradation in India (ICAR and NAAS, 2010), out of 328.7 million ha, nearly 36.7 percent (121 million ha) of the total geographical area (329 million ha) was suffering from various forms of land degradation. The *Desertification and Land Degradation Atlas of India* (SAC, 2021) reported that 97.85 million ha, or 29.8 percent of the total geographic area of the country, was undergoing land degradation in the 2018–2019 period. This is a cumulative increase of 1.45 million ha (0.44 percent of the total geographic area) as compared to the status between 2011 to 2013. Mandal *et al.* (2020) estimated that 83 million ha of India was currently affected by water erosion, and classified 23 percent of the land area in either a very severe erosion category (a loss of >40 t/ha annually) or a severe erosion category (a loss of 20 to 40 t/ha annually). Satellite imagery analysis has revealed that 83 million ha (11.01 percent of the total geographical area) of India is experiencing water erosion. This is a cumulative increase of 0.59 million ha compared to the 2011–2013 period (SAC, 2021). Mandel *et al.* (2023) predicted that the impacts of climate change on soil water erosion would be more pronounced in the hilly regions of India.

In India, about 4.8 to 5.3 billion tonnes of soil get eroded annually, carrying along with it about 8 million tonnes of plant nutrients (Prasad and Biswas, 2000; CSWCRTI, 2011). Approximately 61 percent of soil is moved and deposited at unwanted locations, increasing offsite costs, and nearly 29 percent is lost permanently to the sea (Mandal and Sharda, 2011). The remaining 10 percent is deposited in multipurpose reservoirs, reducing their holding capacity by 1 to 2 percent per annum. The area under water erosion (>10 t/ha annually) is highest in Andhra Pradesh, Jharkhand, Karnataka, Madhya Pradesh, and Uttar Pradesh (Maji, Reddy and Sarkar, 2010).

Soil loss due to wind erosion is limited to arid and semi-arid regions of Gujarat, Haryana, Rajasthan and Punjab states. About 12.4 million ha of India's total area are affected by various kinds of wind erosion (ICAR and NAAS, 2010; Meria *et al.*, 2010). The *Desertification and Land Degradation Atlas of India* (SAC, 2021) identifies Rajasthan as the state most vulnerable to wind erosion.

High rates of soil loss due to erosion have been observed in many studies in Pakistan. About 10 million ha of cultivable land are eroded each year, causing the loss of productive land in Pakistan (Khan, Rahman and Mahmood, 2023). For example, Abuzar *et al.* (2018) reported that in the Simly watershed (162 km^2) in the northeastern Potwar region, about 20.2 t/ha were being eroded annually from agricultural land, and 27.5 t/ha in rangelands. In the Swat district, the majority of lands are highly eroded (Khan, Rahman and Mahmood, 2023). Ullah *et al.* (2018) reported a mean annual soil loss of 19 t/ha in the Potwar region, of which the maximum erosion (70 to 208 t/ha annually) was observed near river channels and in hilly areas. A geospatial assessment revealed the highest

average annual soil loss of 78 t/ha in the Chitral district (Maqsoom *et al.*, 2020). Abuzar *et al.* (2017) reported soil erosion risk zones with high (30 to 100 t/ha) and very high (>100 t/ha) annual soil loss in the Rawal watershed of the Himalayan region. The Potohar Plateau and its surrounding areas are predominantly affected by water erosion at an annual rate of 25 to 50 t/ha, while wind erosion dominates in the Kharan Desert and the Thar Desert, with soil erosion exceeding 150 t/ha annually (Yang *et al.,* 2023).

Most soil erosion studies conducted in Sri Lanka have shown that the mean annual erosion surpasses the recognized tolerable erosion rates identified for the wet and dry zones of Sri Lanka (Piyathilake *et al*., 2021). Jayasekara, Kadupitiya, and Vitharana (2018) reported that 11.8 percent of the area in Sri Lanka was under high erosion hazard level and 4.8 percent of the area faced a very high level of hazard, both of which are intolerable for any land use with respect to sustainable land productivity. An assessment of soil erosion in Sri Lanka based on a river basin approach revealed a 14.3 percent increase in soil erosion over a three-decade interval (1989–2021), mainly attributed to the conversion of forests to agricultural land (Palliyaguru *et al.*, 2023). Recent case studies have shown that this could reach two times (Fayas *et al.*, 2019) to four times (Perera and Amarasinghe, 2020) higher than the tolerable rate of erosion.

8.4.3 Loss of soil organic carbon in South Asia subregion

The terrestrial SOC stock in Indian soils is low, and implementing strategies to enhance its build-up is important for mitigating the impacts of climate change (Minasny *et al.*, 2017). Mandal, Giri and Srivastava (2020) reported that soil erosion caused a loss of 115 million tonnes of C annually. Adoption of climate-smart agricultural techniques is recommended to restore the SOC pool in agricultural soils of India for overall improvement in agricultural sustainability (Doddabasawa, Chittapur and Murthy, 2020).

Sharma, Sharma and Sharma (2019) concluded that in the Pushkar valley region of Rajasthan, India, land-use change had a potential impact on SOC storage due to changes or loss in vegetation, leading to the alteration in the input and output of C in the soil. Importantly, the adverse effects of deforestation, desertification and soil erosion have disrupted the C cycle between the pedosphere and atmosphere, resulting in a decline of SOC stock in India, thus resulting in chemical, hydrological and biological constraints in soil.

Studies in Pakistan report different levels of SOC storage associated with different land uses. Soils of Pakistan are commonly calcareous, weakly structured and low in organic matter content (Yousra *et al.*, 2023). Ali *et al.* (2020) conducted a field-based study in the Khyber Pakhtunkhwa, the largest forested province in Pakistan, and reported that maximum SOC density was in subalpine forests (69.5 ± 7.2 t/ha) followed by moist temperate forests (68.5 ± 6.7 t/ha) and dry temperate forests (60.7 ± 6.5 t/ha). In the riverine forests of Sukkur there was a SOC stock of 31.95 t/ha and a soil CO_2 sequestration potential of 20 t/ha annually (Ali, Khan and Marwat, 2018). Across various land uses at different altitudes, forest soils in the Bagrot Valley in the Gilgit-Baltistan region of North Pakistan had significantly higher values of SOC stocks (59.35 t/ha) than pasture (42.48 t/ha) and arable land (23.63 t/ha) (Ali *et al.,* 2017). In the grasslands of the Baluchistan region of Pakistan, there were significantly higher SOC concentrations (12.83 g of SOC/kg of soil), SOC stocks (19.44 t/ha), lability index (0.89 g of SOC/kg of soil), non-labile SOC (9.56 g of SOC/kg of soil), labile SOC (0.35 g of SOC/kg of soil), and easily oxidizable C (3.27 g of SOC/kg of soil), when compared with fallow land (Ameer *et al.,* 2023). The SOC stocks vary significantly under different cropping systems in Punjab soils. The SOC levels tend to be higher in cropping systems like cotton–wheat, followed by rice–wheat, maize–wheat, and are lowest in fallow–wheat systems (Yousra *et al.*, 2023).

In Sri Lanka, SOC stocks in surface soils (0–30 cm) are estimated to total 255 million tonnes of C, as reported by Vitharana, Mishra and Mapa (2019). Among different land uses, annual croplands

exhibit the lowest average SOC stocks. Tea plantation soils have comparatively higher C stock but depletion of SOC stocks on tea lands in Sri Lanka can be severe due to soil erosion triggered by steep topography, poor soil management and poor vegetation cover in areas of seedling tea propagation. Hemamali *et al.* (2020) reported that long-term vegetable cultivation in Ultisols had depleted the surface SOC stocks by 13 percent and subsurface stocks by 17 percent. Kulasinghe and Dharmakeerthi (2022) reported significantly lower SOC in paddy lands (33 percent) compared to forestlands. There is a large C sequestration capacity in mangrove soils which deserve protection (Perera and Amarasinghe, 2019).

8.4.4 Loss of soil biodiversity in South Asia subregion

Even though the South Asia subregion has a high inherent biodiversity, factors such as a rising population, high deforestation, the conversion of agricultural fields to other uses, frequently occurring natural disasters (such as forest fires, climate change, a high reliance on forest products and an increase in the presence of invasive species), have created a high threat to biodiversity. Maximum abundance and diversity indices of soil microorganisms were observed in Inceptisols (i.e. relatively young soils) owing to their relatively higher SOC and slightly alkaline pH as compared to other rainfed areas of India (Manjunath, Yadav and Khokar, 2021).

The diverse land use in India also plays a significant impact on soil biodiversity. Forestland generally has high soil biodiversity compared to agricultural land (Srivastava, Mohapatra and Latare, 2020). Increased soil salinity has negative impacts on soil biodiversity and crop yield (Haj-Amor *et al.*, 2022). India's soil biodiversity is of concern, according to a report prepared by the World Wide Fund for Nature (WWF) (WWF, 2018). The "risk index" used in their report was based on threats from loss of above ground diversity, pollution and nutrient overloading, overgrazing, intensive agriculture, soil erosion, desertification and climate change. It shows that India is one of the countries where soil biodiversity faces the highest level of risk. However, few systematic studies on the effects of loss of soil biodiversity on soil functions have been undertaken.

8.4.5 Nutrient mismanagement in South Asia subregion

Application of excessive amounts of fertilizer is a common practice in many South Asian countries, leading to soil nutrient imbalances and environmental threats. In 2020, both India and Pakistan were among the top ten countries with the highest use of synthetic N fertilizer. The high N budget surpluses are primarily due to increases in synthetic fertilizer use in the twenty-first century.

In India, nutrient mining increased with intensive cultivation during the post Green Revolution period, and the situation was compounded by the low inherent fertility of most Indian soils. In general, fertilizer use efficiencies of Indian soils are low (30 to 45 percent N, 15 to 25 percent for P, 50 to 60 percent for K, 8 to 12 percent for S, and 2 to 5 percent for micronutrients) compared to developed countries and the global average of 59 percent (Gupta, Benbi and Abrol, 2021).

Removing crop residues from fields has accelerated nutrient depletion, especially K depletion (Goulding *et al.*, 2020). This is evident in the negative K balance for the 2017–2021 period (FAO, 2024b). Information from the national database of the Soil Health Card Scheme of 2015 revealed severe K deficiency in intensively cultivated lands of northwestern India, where K deficiency was not anticipated previously due to the abundance of micaceous minerals (Patra *et al.*, 2017).

Afghanistan is the only country in South Asia that shows evidence of nutrient undersupply and of yield gaps (Figure 8.7, 8.8). The overall N output in crops is in the "very low" class, and the P balance is slightly negative, indicating that Afghanistan has mined P from the soil through time.

Sirisena and Suriyagoda (2018) reported excessive application of P fertilizers by farmers in some

production systems of Sri Lanka, despite the availability of sustainable and up-to-date P fertilizer recommendations. Excess P application has led to the development of high P concentrations in intensive vegetable-cultivating systems (267 mg of available P/kg of soil) in Sri Lanka. In a subcatchment scale study, Nayanarangani *et al.* (2022) reported that soil-available P in intensively cultivated vegetable lands was 81 times higher than in forest soils. Through a P fractionation analysis, Hemamali *et al.* (2020) reported an enrichment of stable, moderately labile and labile fractions of P in soils of intensively cultivated vegetable lands in Sri Lanka. They also found 16 times more inorganic P in intensively cultivated soils growing vegetables compared to soils in undisturbed forestlands.

In Sri Lanka, about 64 percent of the imported urea is used in paddy cultivation, with a N content of 46 percent. Most farmers practice the overapplication of fertilizers including urea (Kulasinghe *et al.*, 2020), causing soil nutrient imbalances and posing environmental threats.

In Pakistan, the national agriculture sector primarily focuses on five major crops: wheat, cotton, maize, rice, and sugarcane. Due to low SOM and arid conditions, these crops rely on chemical fertilizers. Unfortunately, the fertilizer usage is imbalanced, with high N fertilizer inputs compared to P and K fertilizers (Figure 8.7 and Figure 8.8).

8.4.6 Salt accumulation in South Asia subregion

Soil salinization is a global and dynamic problem which is projected to increase in the future under climate change scenarios. About 52 million ha are salt-affected in South Asia (Chinchmalatpure, 2017, Mandal *et al.*, 2018). In India, around 6.7 million ha, (about 2.1 percent of the land area) is salt-affected, of which 2.9 million ha is saline and the remainder (3.7 million ha) is sodic (Chinchmalatpure, 2017). Nearly 75 percent of salt-affected soils in India are in the states of Gujarat (2.2 million ha), Uttar Pradesh (1.4 million ha), Maharashtra (0.6 million ha), West Bengal (0.4 million ha), and Rajasthan (0.4 million ha) (Table 8.7).

Table 8.7. Extent (ha) of salt-affected soils in different states of India

State	Saline (ha)	Sodic (ha)	Total (ha)
Andhra Pradesh	77 598	196 609	274 207
Andaman and Nicobar Island	77 000	0	77 000
Bihar	47 301	105 852	153 153
Gujarat	1 680 570	541 430	2 222 000
Haryana	49 157	183 399	232 556
Karnataka	1 893	148 136	150 029
Kerala	20 000	0	20 000
Madhya Pradesh	0	139 720	139 720
Maharashtra	184 089	422 670	606 759
Orissa	147 138	0	147 138
Punjab	0	151 717	151 717
Rajasthan	195 571	179 371	374 942
Tamil Nadu	13 231	354 784	368 015
Uttar Pradesh	21 989	1 346 971	1 368 960
West Bengal	441 272	0	441 272
Total	2 956 809	3 770 659	6 727 468

◊ *Source:* **Chinchmalatpure, A.R.** 2017. *Reclamation and Management of Salt Affected Soils for Increasing Farm Productivity and Farmers' Income.* Karnal, India, Indian Council of Agricultural Research (ICAR)-Central Soil Salinity Research Institute (CSSRI). https://krishi.icar.gov.in/jspui/bitstream/123456789/10792/1/Reclamation%20and%20Management.pdf

In India, estimated losses of around IND 230.20 billion occur annually through crop production loss (16.84 million tonnes annually) due to soil salinization (Mandal *et al*., 2018). To minimize the impact of land degradation on crop production and ensure food security in the country, the Indian government has attached the highest priority to policy planning for the restoration of 26 million ha of degraded lands by 2030.

More than 7 million ha are affected by soil salinity or sodicity in Pakistan with over half lying in the Canal Command Area (Ali, 2023). About 14 to 26 percent of irrigated lands have deteriorated due to salinity, with a resulting 64 percent yield loss (Syed *et al*., 2021).

Salinity adaptation strategies have positive externalities for improving crop production and the income of farmers (Mustafa and Akhtar, 2019). For instance, about 0.6 million ha of agriculture land was protected from waterlogging and salinity by the completion of the Right Bank Outfall Drain project in Sindh (Mahessar *et al.,* 2019). The low success rate in implementing salinity control measures in Pakistan is attributed to unsustainable interventions, a lack of research and participatory management approaches, and a low uptake of technologies by farmers. Rising water tables are also affecting agriculture productivity, due to an increase in salinity, particularly in uncultivable land (Malik *et al*., 2021; Rahman *et al*., 2016). Soil salinity is particularly serious in the Sindh province compared to other provinces in Pakistan, where around 70 to 80 percent of the soil is classified as moderately or severely saline.

There are indications of an increase in the land area in Sri Lanka affected by coastal salinity (Gopalakrishnan and Kumar, 2020) and about 32.8 percent of the lands in Jaffna Peninsula are affected by salinity. The groundwater salinity in the same region has increased 1.6-fold over the 1999–2019 period (Gopalakrishnan, Kumar and Hasan, 2020), and a 1.4-fold increase in the area affected by coastal salinity between 1988 and 2019 (Gopalakrishnan and Kumar, 2021). Abandoned paddy lands are more prone to developing soil salinity than active paddy lands.

8.4.7 Soil pollution in South Asia subregion

Excessive use of agrochemicals and industrial activity pose a threat of soil pollution. About 92 percent of mining activities in India are concentrated in the states of Andhra Pradesh, Chhattisgarh, Gujarat, Jharkhand, Karnataka, Madhya Pradesh, Maharashtra, Odisha, Rajasthan, Tamil Nadu, and West Bengal. A report from the Central Pollution Control Board of India suggested that the states of Andhra Pradesh, Gujarat and Maharashtra contributed to 80 percent of hazardous wastes, including heavy metals (Kumar *et al*., 2019).

Urban and peri-urban soils of India are mostly associated with intensive anthropogenic impacts, consequent ecological pollution and contamination (Mukhopadhyay *et al*., 2020). The rapid expansion of metropolitan cities has had negative impacts on the atmosphere, climate, lithosphere, biosphere, and hydrosphere, as well as land and water resources (Nouri *et al*., 2019). Along with heavy metal contamination and urban waste dumping, the occurrence of microplastics in urban soils is of increasing concern. Currently, over 35 percent of India's population live in urban areas, with Indian megacities having some of the highest population densities in the world. It is predicted that by 2050, many Indian cities will have a population greater than 5 million, with more than 50 percent of the population living in cities (Kundu, 2014). The megacities have increased their amount of compact spaces, built-up areas along roadsides and elevated corridors, resulting in the loss of avenues of trees, thereby losing vital corridors of vegetation and soil in urban areas.

Geoaccumulation patterns of heavy metal(loid)s in soils of Pakistan based on the geoaccumulation index were reported by Din, Muhammad and Rehman (2023). Very high contamination factor (CF) values were exhibited for Ni (12.1) and Cr (4.07) in Baluchistan, Zn (2.53) and Cu (1.54) in Gilgit-Baltistan, Mo (6.55) and Cd (4.28) in Khyber Pakhtunkhwa, Cd (8.38) and Co (2.91) in Punjab, and As (1.08) and Pb (0.94) in Sindh province. Higher contamination of heavy metals in the soils of the Baluchistan

and Khyber Pakhtunkhwa provinces was characterized by the natural enrichment of these metals in the bedrock hosting mafic-ultramafic ore deposits and mining activities. However, those of Sindh and Punjab provinces were mainly attributed to anthropogenic activities such as wastewater irrigation (Din, Muhammad and Rehman, 2023).

Wastewater irrigation, waste disposal, vehicular exhaust, wastewater sludge and construction or other developmental activities are key sources of local soil contamination in Pakistan (Khan *et al.*, 2016). In some areas of Pakistan, the presence of As and other potential heavy metals in subsurface aquifers and drinking water systems is a potentially serious human health hazard (Samrana *et al.*, 2017).

Several studies have investigated the pollution of soils in Sri Lanka. Balasooriya *et al.* (2022) reported instances of enrichment of Ni, Cu and Zn in paddy soils in the wetter part of the country. However, most studies show that soils in Sri Lanka are not dangerously contaminated with toxic trace elements (Rosemary *et al.*, 2015; Suriyagoda *et al.*, 2022).



8.5 Regional status and trends of threats to soil functions for the Southeast Asia subregion

8.5.1 Introduction

Southeast Asia is generally divided into "mainland" and "island" zones. The mainland (Cambodia, the Lao People's Democratic Republic, Malaysia, Myanmar, Thailand, and Viet Nam) is characterized by mountain ranges in the north, large plains and plateaus in the south, and large river systems that run from the mountains through the plains to the sea. The climate is normally determined by the monsoon. Islands include Brunei Darussalam, East Timor, Indonesia, Singapore, and the Philippines. Virtually all Southeast Asia lies within the tropics, and so they share many similarities in climate.

Malaysia is a tropical country, comprising Peninsular Malaysia, Sabah and Sarawak. Approximately 70 percent of Malaysia's land area is covered by residual soil, with the remaining 30 percent being reworked and alluvial soil (DOA, 2018; Marto and Yusoff, 2017). There are 5.39 million ha currently under agricultural land use in Peninsular Malaysia (DOA, 2022) (Table 8.8), while around 19 million ha are unsuitable for agriculture (DOA, 2023) (Table 8.9). Approximately 0.1 million ha are left idle (DOA, 2022) due to inefficient agronomic and land management practices on problematic soils (DOA, 2023; Faridah *et al.,* 2022).

Table 8.8. Agriculture land use for Peninsular Malaysia in 2018

Category	Agricultural land use	Area (ha)
Mixed agriculture	Traditional cultivation	283 431
Short-term crops	Paddy, vegetables, herbs, spices, root crops, maize (*Zea mays*), sugar cane (*Saccharum* spp.) papaya (*Carica papaya*), banana (*Musa* spp.), and watermelon (*Citrullus lanatus*)	388 605
Long-term crops	Coconut (*Cocos nucifera*), durian (*Durio* spp.), mango (*Mangifera indica*), pineapple (*Ananas comosus*), areca nut (*Areca catechu),* reeds, castor bean (*Ricinus communis*), coffee (*Coffea* spp.), tea (*Camellia sinensis*), guava (*Psidium guajava),* carambolas (*Averrhoa carambola*), dragon fruit (*Stenocereus* spp.), longan (*Dimocarpus longan*), rambutan (*Nephelium lappaceum*), pummelos (*Citrus maxima*), sweet orange (*Citrus sinensis*), salak palm (*Salacca edulis*), mixed orchards, jackfruit (*Artocarpus heterophyllus*), citrus, soursop (*Annona muricata*), mangosteen (*Garcinia mangostana*), and other fruit crops	267 070
Other crops	Other crops	4 443 162
Decorative plants	Floriculture and ornamental plants	4 600
Total (ha)		5 386 868

◊ *Source:* **DOA (Department of Agriculture).** 2022. *Crop statistics booklet (Food crop sub-sector).* Putrajaya, Malaysia.



Table 8.9. Extent and distribution of land use in Malaysia

Category	Area (ha)
Total land area	33 000 000
Land areas unsuitable for agriculture	19 000 000
Land areas suitable for agriculture	12 690 000
Idle land left to fallow and suitable for agriculture	100 000
Marginal land areas	
Acid sulphate soils	570 000
Beach ridges interspersed with swales (BRIS)	140 000
Peat soils	2 600 000
Tin-tailings	120 000
Steep land	18 840 000

◊ *Sources:* **Shamsuddin, J., Alia Farhana, J. & Enio Kang, M.S.K.** 2021. *Acid Sulfate Soils in Southeast Asia: Occurrence, Properties and Management*. Serdang, Malaysia, Universiti Putra Malaysia (UPM) Press.

◊ **DOA**. 2020. *Agricultural Development Guidelines on Steep Land*. Putrajaya, Malaysia, Land Resource Management Division.

◊ **DOA**. 2023. *Degradation of Soil Quality and Agricultural Water Resources: Issues and Challenges*. Putrajaya, Malaysia, Land Resource Management Division.

Viet Nam has two major agricultural challenges: soil erosion by water and low soil water availability. Soil erosion is extensive and exacerbated by the country's topography, with a significant portion of cropped land being on slopes. Erosion is intensified by some agricultural practices, such as monocropping, as well as heavy rainfall. Drought conditions are also of concern, affecting water availability, particularly in the south central coast and the Mekong Delta.

The Philippines has a diverse topography, distinctive climate patterns and extensive human influence. The northern islands of the Philippines are especially affected by storms paths from typhoons and severe tropical storms. The country has soil degradation issues, which threaten not only agricultural productivity, but also the overall sustainability of its natural resources needed to support its growing population. Approximately 20 percent of Philippine land is devoted to annual crops, 22 percent to perennial crops and 23 percent to closed and open forests (Table 8.10). Approximately 20 percent are classified as brush (shrubland), and these land areas are the most likely to transition to agriculture (NAMRIA, 2020).

Table 8.10. Land use/land cover estimates for the Philippines in 2020

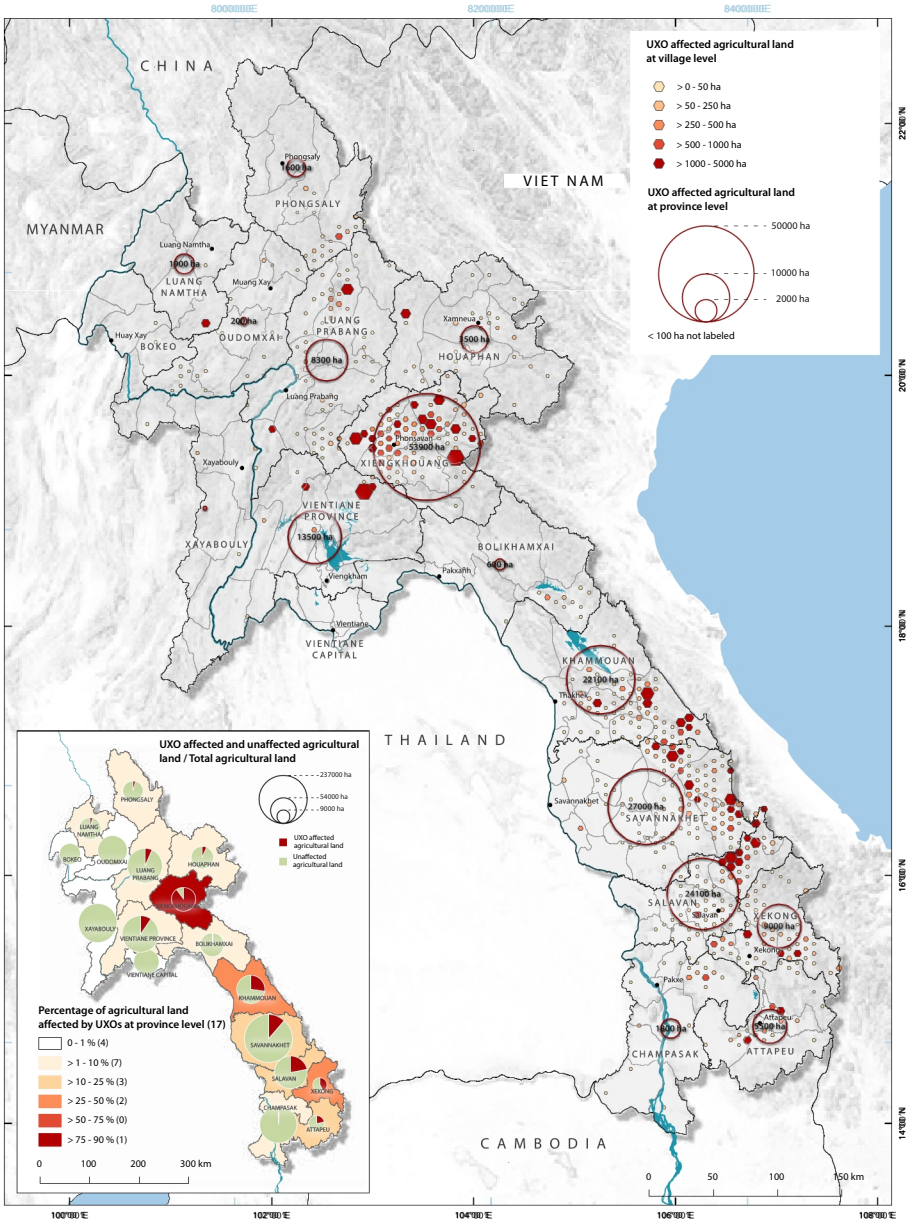
Land use/land cover	Area (ha)	%
Annual crop	5 938 798	20.1
Brush/shrubs	5 792 210	19.6
Built-up area	1 034 850	3.5
Closed forest	2 217 369	7.5
Fishpond	228 376	0.8
Grassland	1 956 548	6.6
Inland water	492 244	1.7
Mangrove forest	311 394	1.1
Marshland/swamp	142 362	0.5
Open forest	4 685 793	15.9
Open/barren	159 108	0.5
Perennial crop	6 563 179	22.2
Total	29 522 228	100

◊ *Source:* **NAMRIA (National Mapping and Resource Information Authority)**. 2020. *Geoportal Philippines Land Cover Map*. [Accessed on 3 July 2024]. https://www.geoportal.gov.ph/. Licence: CC-BY-4.0.

Thailand has 51.3 million ha of land, with 55.6 percent of the land used for agriculture, followed by 32.6 percent as forest area and 11.8 percent as miscellaneous areas (such as abandoned areas, marsh areas, community and buildings, and water resource areas). In 2006, marginal soils covered 10.6 million ha of the country, which declined to 9.6 million ha in 2018 due to the adoption of SSM practices (LDD, 2006, 2018).

Agriculture in the Lao People's Democratic Republic is limited, due it being the most heavily bombed country per capita in the world (Figure 8.15). The most heavily targeted regions are located along the Annamite Mountains, with Xieng Khouang Province having the highest number of unexploded ordnances (UXOs) (bombs that were dropped but did not explode) although Khammouane, Saravan, Savannakhet, and Sekong are also significantly affected.

Figure 8.15. Agricultural land area in the Lao People's Democratic Republic affected by unexploded ordnances



Refer to the disclaimer on page ii for the names and boundaries used in this map.

◊ *Source:* **Epprecht, M., Weber, A., Bernhard, R., Keoka, K., Saphangthong, T., Manivong, V., Ingxay, P**. *et al.* 2018. *Atlas of Agriculture in the Lao PDR, Patterns and trends between 1999 and 2011.* Bern, University of Bern, Switzerland, Centre for Development and Environment (CDE) & Vientiane, Lao People's Democratic Republic, Ministry of Agriculture and Forestry (MAF). https://doi.org/10.7892/boris.116598

Indonesia is an archipelago consisting of 17 499 islands with a land area of 1.9 million km^2, a high annual rainfall of over 1 500 mm, and a population of over 270 million. Sixty percent of Indonesia's topography consists of undulating to mountainous relief, and soils have originated mostly from sedimentary rock and alluvium (Subagyo, Suharta and Siswanto, 2004). Forests cover 47 percent of the land and 32 percent is cultivated as paddy fields, plantation, dryland agriculture, and mixed gardens (Table 8.11). Paddy fields (7.1 million ha), peatland (13.4 million ha) and mangroves (3.36 million ha) (Table 8.12) are essential for national food security, development and the global environment. For national agricultural planning, the Indonesian government uses a paddy field area benchmark of 7.49 million ha as a national reference site for decisions on key initiatives, including fertilizer and seed subsidy (Christopher Teh Boon Sung, personal communication, 10 September 2024). Soils are predominantly Inceptisols and Ultisols, with smaller areas classified under various other United States Department of Agriculture (USDA) soil taxonomy orders (Subagyo, Suharta and Siswanto, 2004).

Table 8.11. Land use/land cover of Indonesia in 2019

Kind of land use	Area (ha)	%
1. Dryland forest (primary)	40 263 800	21.45
2. Dryland forest (secondary)	34 307 600	18.27
3. Swamp forest (primary)	5 003 700	2.67
4. Swamp forest (secondary)	6 517 500	3.47
5. Mangrove forest (primary)	1 501 100	0.80
6. Mangrove forest (secondary)	1 411 100	0.75
7. Industrial timber plantation	5 109 400	2.72
8. Dryland shrub and bush	12 663 600	6.74
9. Swamp shrub and bush	7 600 600	4.05
10. Savannah	2 910 800	1.55
11. Plantation/gardens	18 007 900	9.59
12. Dryland agriculture	8 798 800	4.69
13. Mixed dryland agriculture with bush/mixed gardens	26 297 500	14.01
14. Paddy fields	7 884 000	4.20
15. Brackish fishpond	971 500	0.52
16. Mining area	810 400	0.43
17. Swampland	1 304 000	0.69
18. Open ground	2 469 900	1.32
19. Transmigration area	224 600	0.12
20. Settlement/building area	3 669 400	1.95
21. Airport/seaport area	24 900	0.01

◊ *Source:* **KLHK (Ministry of Environment and Forestry)**. 2020. *Basisdata Geospasial Lingkungan Hidup dan Kehutanan [Geospatial Database of Forestry and Environment]*. Jakarta. https://phl.menlhk.go.id/static/file/publikasi/1612239357-BUKUBASISDATAGEOSPASIAL2020.pdf

Table 8.12. Distribution of paddy fields, critical land, mangrove, and peatland based on georegion In Indonesia

Georegion	Paddy fields (2019)[1] (ha)	Critical land (2022)[2] (ha)	Mangrove (2021)[3] (ha)	Peatland (2019)[4] (ha)	Acid soils (2015)[5] (ha)
Sumatra	1 529 720	3 926 834	660 445	5 850 561	30 934 790
Java	3 473 033	1 684 136	56 499		8 340 527
Nusa Tenggara	442 935	1 112 462	34 974		168 792
Kalimantan	646 462	2 350 302	688 025	4 543 362	39 094 313
Sulawesi	964 556	1 803 381	137 185	24 783	7 466 479
Maluku	22 701	970 516	224 046		1 999 401
Papua	25 737	699 665	1 562 906	3 011 811	19 353 331
Total	7 105 144	12 744 925	3 364 080	13 430 517	107 357 632

◊ *Sources:* **1BPN (National Land Agency)**. 2019. Luas Lahan Baku Sawah Nasional Tahun 2019 [The Coverage of National Paddy Fields in 2019]. Jakarta, Ministry for Agrarian Affairs and Spatial Planning (Kementerian ATR)/BPN.

◊ **[2]KLHK**. 2022. *Keputusan Direktur Jenderal Pengelolaan Daerah Aliran Sungai dan Rehabilitasi Hutan nomor SK.49/PDASRH/PPPDAS/DAS.0/12/2022 tentang Peta dan Data Lahan Kritis Nasional Tahun 2022* [Decision of Director General of Watershed Management and Forest Rehabilitation Number SK.49/PDASRH/PPPDAS/DAS.0/12/2022 about National Critical Land Map and Data, 2022]. Jakarta. https://jdih.menlhk.go.id/new/uploads/files/2022skmenlhk1295_menlhk_01172023134804.pdf

◊ [3]**Direktorat KTA (Directorate of Soil and Water Conservation)**. 2021. *Peta Mangrove Nasional Tahun 2021* [National Mangrove Map, Year 2021]. Jakarta, KLHK.

◊ **[4]Kementerian Pertanian**. 2020. *Peta Lahan Gambut Indonesia Skala 1:50.000. Edisi Desember 2019* (Indonesian Peatland Map, scale of 1:50.000, December 2019 edition). Jakarta, Kementerian Pertanian

◊ [5]**BBSDLP (Centre for Research and Development of Agricultural Land Resources)**. *2015. Sumber Daya Lahan Pertanian Indonesia: Luas, Penyebaran dan Potensi Ketersediaan [Indonesian Agricultural Land Resources: Coverage, Distribution, and Availability Potential]*. Bogor, Indonesia.

8.5.2 Soil erosion in Southeast Asia subregion

Soil loss due to splash, sheet, rill, and gully erosion is a problem in Malaysia (DOA, 2020), partly because of the high annual rainfall (between 1 500 and 5 400 mm) (MetMalaysia, 2021). Steep land regions (covering 18.84 million ha) are vulnerable, particularly the Cameron Highlands and Pahang (Zainal Abidin *et al.,* 2023) (Table 8.13 and Table 8.14). Approximately 29 percent of land in the Cameron Highlands have steep slopes between 25 to 30 degrees, and 66 percent of its catchment area has gradients exceeding 20 degrees (Razad *et al.,* 2018; Razali *et al.,* 2018). There is an annual soil loss of approximately 125 kg/ha in the Cameron Highlands catchment area (Teh *et al.,* 2017), with an annual sedimentation rate ranging from 120 000 to 200 000 m^3 (Razad *et al.,* 2018). Vegetable farms in the catchment area, particularly open field cultivation, experience a high annual soil loss (82 t/ha) whereas under rain shelter cultivation, annual soil loss is only 1 t/ha (DOA, 2020; Razali *et al.*, 2018). About 6.7 percent of the land area has annual soil losses of >25 t/ha, while 84 percent of the catchment area has lower soil loss (Zainal Abidin *et al.*, 2023).



Table 8.13. Current land use in the Cameron Highlands, Pahang, Malaysia

Land use	Area (ha)
Forest	59 788.65
Agriculture	7 212.93
Residential	351.82
Water body	263.39
Public amenities	162.84
Commercial	61.48
Recreational space	57.64
Transportation	43.28
Vacant land	43.28
Infrastructure	28.39
Industry	20.99
Mixed development	1.36
Total area	68 036.03

◊ *Source:* **Zainal Abidin, R.Z., Mahamud, M.A., Saad, N.A., Zakaria, N.A. & Noh, M.M**. 2023. Erosion Risk Level at Cameron Highlands with Regards to Soil Erosion Loss. *ASM Science Journal*, 18: 1–11. https://doi.org/10.32802/asmscj.2023.1078

Table 8.14. Annual soil loss risk level distribution area for Cameron Highlands, Pahang, Malaysia

Soil loss level	Risk (t/ha)	Area (ha)	%
Low	≤10	57 104.87	84.37
Moderate	11 to 25	6 034.52	8.92
High	26 to 50	3 945.03	5.83
Very high	51 to 100	589.66	0.87
Critical	≥101	12.65	0.02

◊ *Source:* **Zainal Abidin, R.Z., Mahamud, M.A., Saad, N.A., Zakaria, N.A. & Noh, M.M.** 2023. Erosion Risk Level at Cameron Highlands with Regards to Soil Erosion Loss. *ASM Science Journal*, 18: 1–11. https://doi.org/10.32802/asmscj.2023.1078

Soil erosion in Malaysia is generally controlled via agronomic, soil management, and mechanical methods (DOA, 2020; Teh *et al.,* 2017) (Table 8.15). Agricultural development on steep land must follow the Malaysian Certificate Scheme for Good Agricultural Practice (myGAP) and the Agricultural Development Guideline on Steep Land (Razali *et al.,* 2018; DOA, 2020).

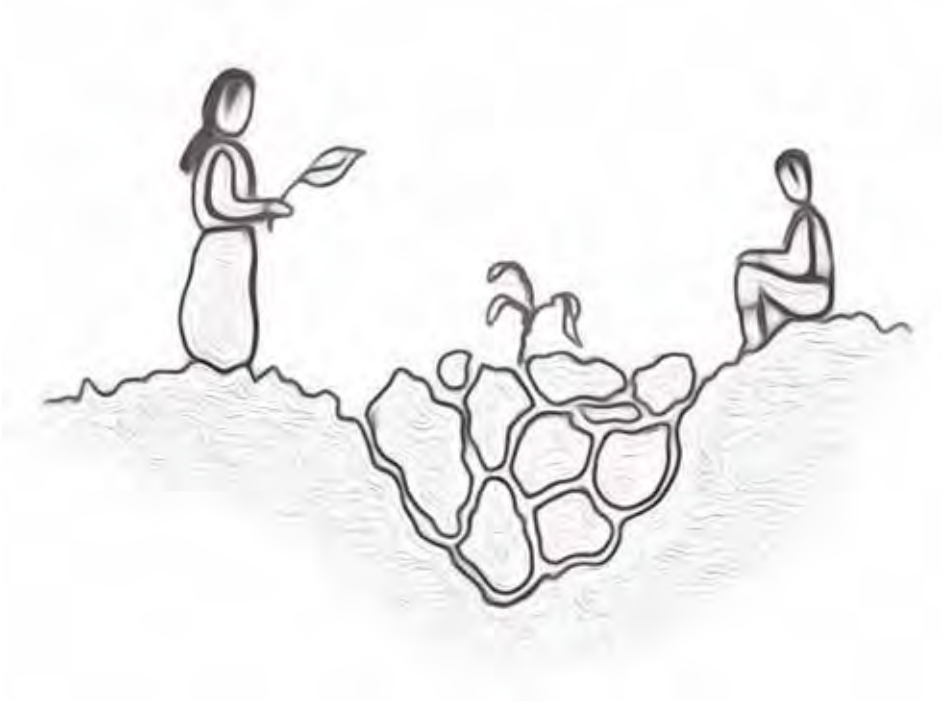


Table 8.15. Soil erosion control measures in Malaysia

Methods	Measures
Agronomic	Use of legumes as cover crops to protect the soil as it is fast-growing, capable of fixing N, and could provide complete ground cover. Popular cover crops include *Pueraria phaseoloides*, *Calopogonium mucunoides*, *Centrosema pubescens*, *Mucuna bracteata, Arachis pintoi, Sechium edule,* and liverworts. Use of natural (straws) and synthetic (silvershine plastic) mulching. Use of vetiver grass (*Vetiveria zizaniodies*) and Kim Chiam or daylily (*Hemerocallis fulva*) as slope stabilizers. Shifting cultivation practices with a fallow period between 10 and 15 years on steep land are reported to be ecologically stable and sustainable. Minimal tillage, contour planting, and high-density planting.
Soil management	Addition of organic matter to increase water retention capacity and aggregate stability.
Mechanical	The use of structures such as concrete walls, gabions, terraces, field drains, hillside ditches, silt fences, silt traps, check dams, culverts, and plastic rain shelters.
Specific guidelines and requirements durian cultivation (25 to 30 degrees slope)	Soil rooting depth: at least 1 m depth is required. Soil clay content >35% is required. Manual and semi-manual land clearing method is required for clearing of new agriculture areas. Clearing of new land areas exceeding 10 ha must be executed in stages. Implement appropriate soil conservation measures (drainage, erosion and silt control systems). Expedite planting of cover crops once land clearing is completed. Implement disaster risk management whenever relevant. Cultivation of durian for commercial purposes only because durian has been earmarked as a premium crop for international trade.

◊ *Sources:* **DOA**. 2020. *Agricultural Development Guidelines on Steep Land*. Putrajaya, Malaysia, Land Resource Management Division

◊ **Razali, A., Syed Ismail, S.N., Awang, S., Praveena, S.M. & Zainal Abidin, E.** 2018. Land use change in highland area and its impact on river water quality: a review of case studies in Malaysia. *Ecological Processes*, 7(1): 1–17. http://dx.doi.org/10.1186/s13717-018-0126-8

◊ **Teh, C.B.S., Ishak, C.F., Abdullah, R., Othman, R., Panhwar, Q.A. & Aziz, Md. M.A.** 2017. Soil Properties (Physical, Chemical, Biological, Mechanical). In: M.A. Ashraf, R. Othman & C.F. Ishak, eds. *Soils of Malaysia*, pp. 103–154. Boca Raton, USA, CRC Press.

Leaching losses of applied N, K and Mg fertilizers are very high on Oxisols when palm trees are newly established but losses at lower levels continue past the establishment stage (Foong, Syed Sofi and Tan, 1983; Foong, 1993) (Table 8.16). Annual erosion losses can also be high, as the example for Ultisols in Malaysian oil palm plantations shows (Maena *et al.*, 1979; Kee and Chew, 1996) (Table 8.17). The greatest losses come from uncovered and compacted harvesting rows, while the least occur under covered frond heaps (Maena *et al.*, 1979).



Table 8.16. Nutrient losses via leaching in Malaysian oil palm plantations

Soil order	Annual rainfall	Age of palm	Leached (% of applied fertilizer)			
	(mm)	(years)	N	P	K	Mg
Oxisols[1]	1 909	1	26.5	trace	19.5	169.4
	1 495	2	10.9	trace	3.4	8.4
	2 729	3	12.2	1.4	10.4	53.6
	2 787	4	16.8	5.8	5.6	47.6
	2 391	5	2.7	1.7	1.9	5.4
	2 193	6	4.8	1.4	3.3	6.6
Oxisols[2]	n/a	1–4	16.6	1.8	9.7	69.8
		5–8	1.2	1.6	2.5	11.5
		9–14	3.0	1.5	2.9	15.5

Sources: [1] **Foong, S.F., Syed Sofi, S.O. & Tan, P.Y.** 1983. A lysimetric simulation of leaching losses from an oil palm field. Presentation at the Seminar on Fertilizers in Malaysian Agriculture, 1983. Kula Lumpur, Malaysian Society of Soil Science.

[2] **Foong, S.F.** 1993. Potential evaporation, potential yields and leaching losses of oil palm. Conference presentation at Progress, Prospects and Challenges Towards the 21st Century (Agriculture). 9-14 September 1993. Kuala Lumpur, Palm Oil Research Institute (PORIM).

Table 8.17. Nutrient losses via runoff and eroded sediments in Malaysian oil palm plantations

Soil order		**Annual losses by erosion (kg/ha)**				
	Transport	**N**	**P**	**K**	**Mg**	**Ca**
Ultisols[1]	Runoff	9.93	1.43	10.40	1.82	4.04
	Sediments	5.57	3.63	8.79	21.10	7.40
Ultisols[2]	Runoff	5.85	0.90	26.90	5.20	n/a
	Sediments	0.65	0.90	trace	0.10	n/a

Sources: [1] **Maena, L.M., Thong, K.C., Ong, T.S. & Mokhtaruddin, A.M.** 1979. Surface wash under mature oil palm. Presentation at the Symposium on Water in Malaysian Agriculture, 1979. Kuala Lumpur, Malaysian Society of Soil Science.

[2] **Kee, K.K. & Chew, P.S.** 1996. Nutrient losses through surface runoff and soil erosion – implications for improved fertilizer efficiency in mature oil palms. Presentation at Competitiveness for the 21st Century. The PORIM International Palm Oil Congress, 23-28 September 1996. Kuala Lumpur, PORIM.

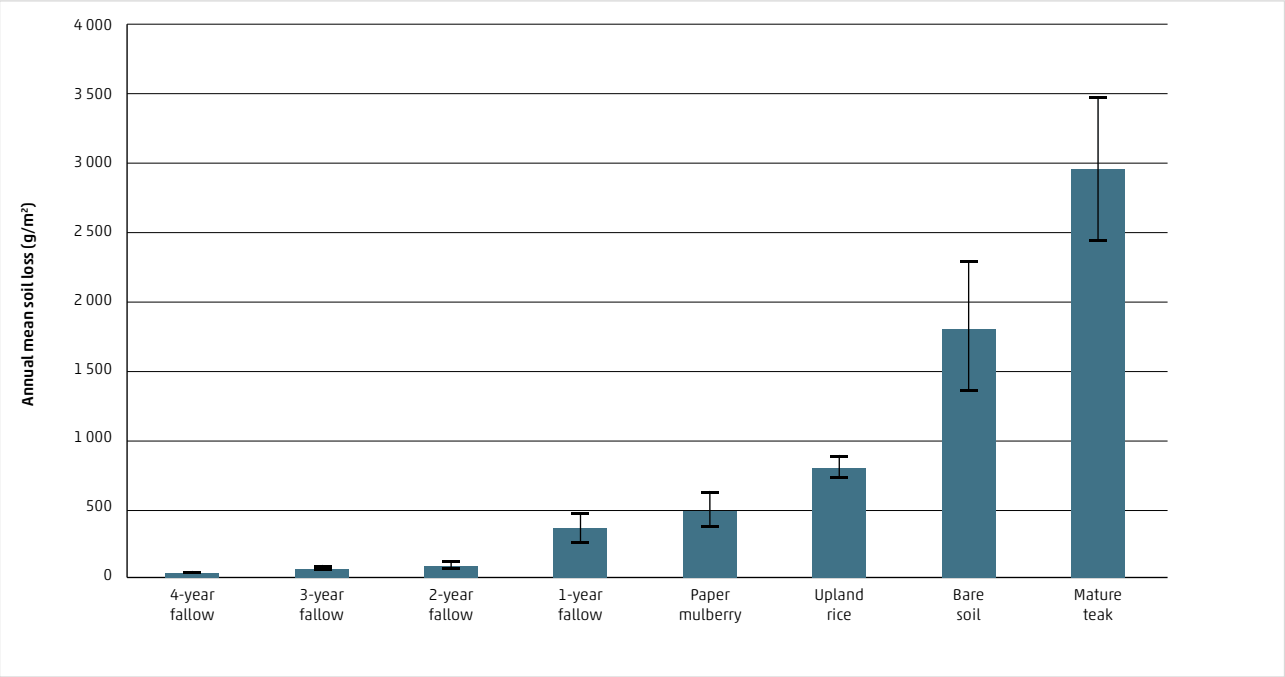
In upland and plateau areas of the Lao People's Democratic Republic (primarily in the mountainous northern and eastern regions of the south), shifting cultivation is common and leads to declining soil fertility, increased water runoff and soil erosion. Annual crops cause field-scale erosion on the steep slopes (Kohmoto *et al.*, 2021) (Figure 8.16). However, under shifting cultivation, erosion is spatially and temporally distributed, limiting watershed-level soil loss (Ribolzi *et al.*, 2017). To combat deforestation and provide steady income to farmers, the Government of the Lao People's Democratic Republic has promoted tree plantations. However, for rubber trees, price volatility has affected producers, and erosion in rubber plantations is high (Patin *et al.*, 2018; Lacombe *et al.*, 2018; Phasouysaingam, 2020).

Between 2010 and 2014, the transition from slash-and-burn agriculture to a land-use system that included the establishment of teak *(Tectona grandis)* plantations covering 30 percent of a 62 ha catchment in the north of the country resulted in an increase in sediment runoff, ranging from 98 to 609 t/km^2 (Ribolzi *et al.*, 2017) (Figure 8.16).

The teak plantation soils are exposed to a higher kinetic energy from raindrops than bare soils (Lacombe *et al.*, 2018) due to the teak tree being a tall species (up to 40 m in height) with large leaves that intercept rainfall. Raindrops accumulate on the leaves and coalesce into larger drops, which then fall from a greater height and reach terminal velocity before striking the soil surface. This results in a high-impact force that can significantly increase soil erosion. Downstream dispersion of weed seeds (De Rouw *et al.*, 2018), pathogens such as *Escherichia coli* (Nakhle *et al.*, 2021) and

sedimentation also occur in reservoirs (Shrestha, Hurley and Wemple, 2018). The use of crop residues such as mulch appears to be the most effective approach for reducing soil losses from continuous maize cultivation (Kohmoto *et al.*, 2021), although the planting of broom grass (*Thysanolaena maxima)* under teak is also effective (Song *et al.*, 2020).

Figure 8.16. Annual mean (±SD) soil loss in different land uses in the north of the Lao People's Democratic Republic



◊ *Source*: Authors' own elaboration and **Patin, J., Mouche, E., Ribolzi, O., Sengtahevanghoung, O., Latsachak, K.O., Soulileuth, B., Chaplot, V. & Valentin, C.** 2018. Effect of land use on interrill erosion in a montane catchment of Northern Laos: An analysis based on a pluri-annual runoff and soil loss database. *Journal of Hydrology*, 563: 480–494. https://doi.org/10.1016/j.jhydrol.2018.05.044

Soil erosion has been identified as the dominant form of land degradation in the Philippines. Approximately 45 percent or 13.7 million ha (Figure 8.17) of the total land area of roughly 30 million ha were classified by the Bureau of Soils and Water Management (BSWM) in 1993 as moderately to severely eroded (Contreras, 2018). About 38 percent (± 11.5 million ha) were considered vulnerable to soil erosion by BSWM (2013). Climate change, flooding and drought pose a threat to different parts of the Philippines.

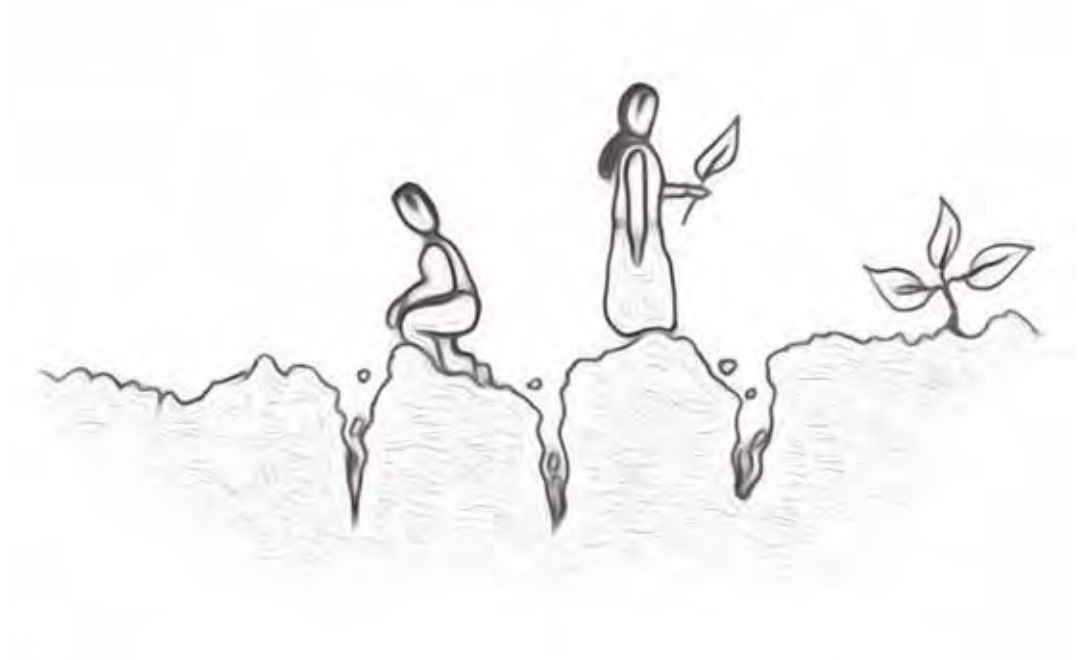
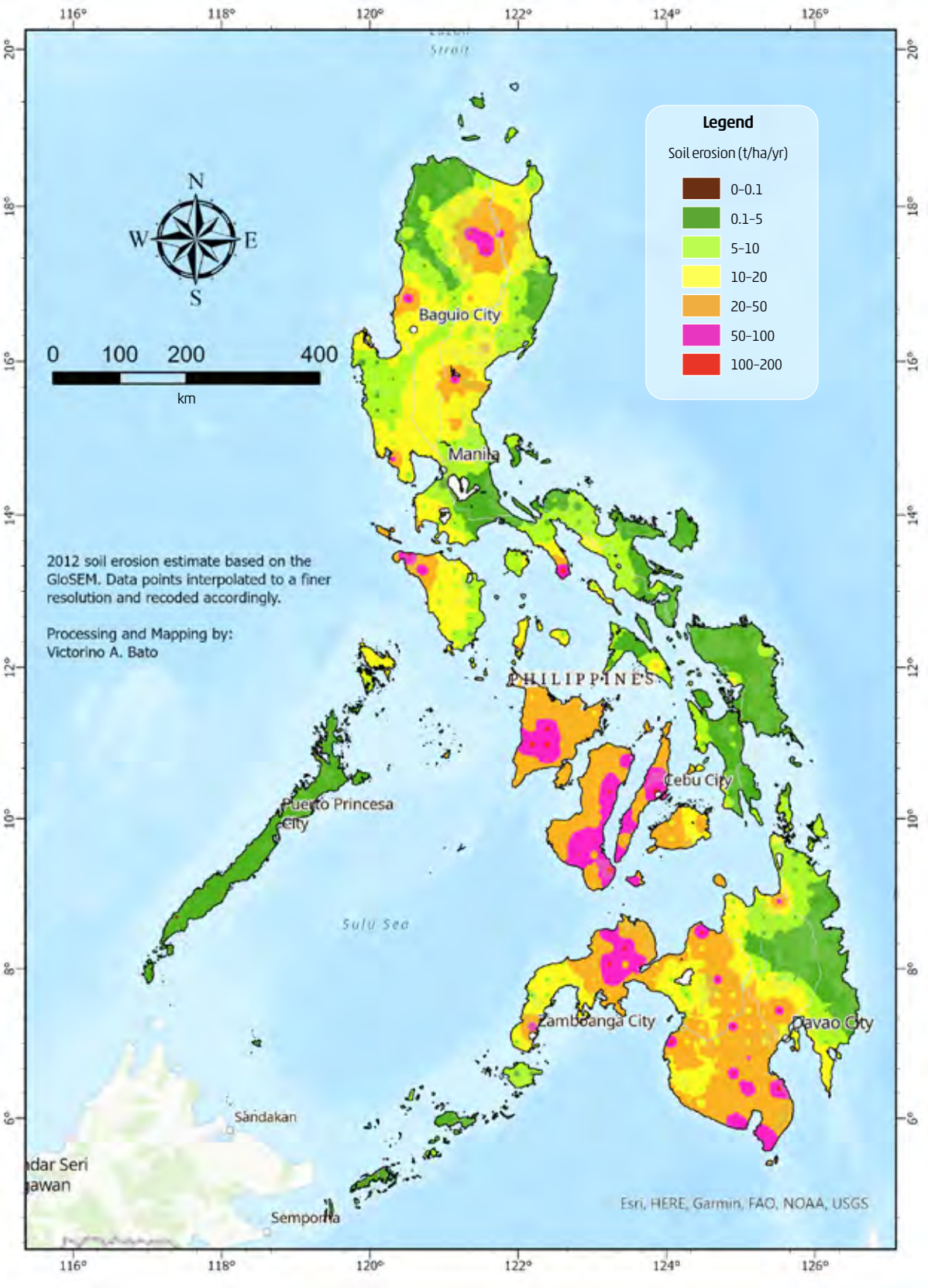


Figure 8.17. Soil erosion estimates of the Philippines, 2012 Annual soil erosion



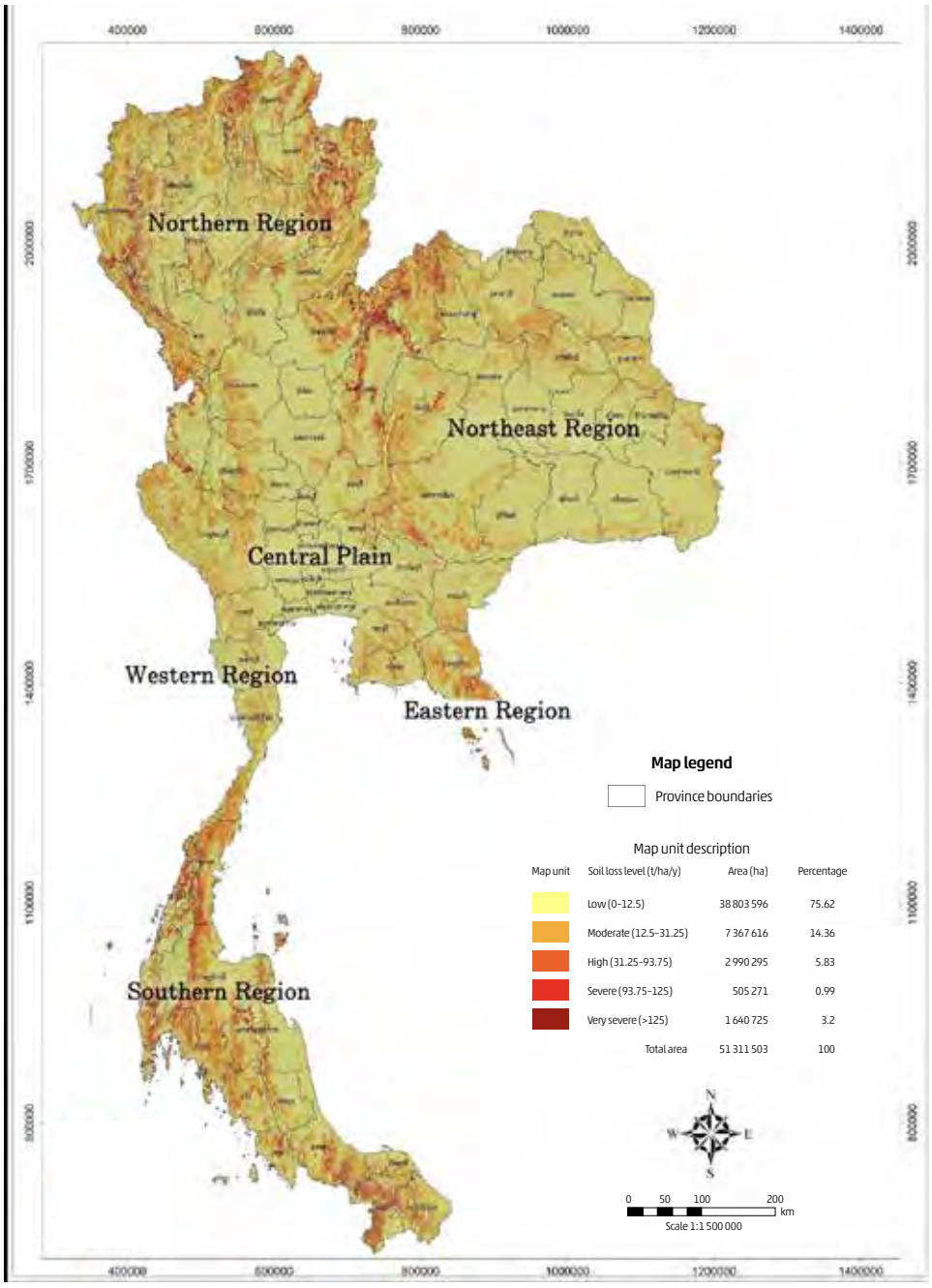
Refer to the disclaimer on page ii for the names and boundaries used in this map.

◊ *Sources of data*: GloSEM data grid - interpolated to a finer resolution and recoded accordingly, NAMRIA – Philippine Boundary, ESRI, TomTom, Garmin, FAO, NOAA, and USGS. Processed by VA, Bato.

Soil erosion has led to a decline in soil fertility. The FAO-BSWM initiative for land degradation assessment has also recognized a decline in soil fertility, especially in intensive agricultural areas (BSWM, 2013).

Soil erosion is widespread in Thailand (Figure 8.18). Crops such as rainfed rice and maize cultivated on steep slopes have led to severe water erosion (Rochelle-Newall, 2015; Punyalue *et al.*, 2018; Wicharuck *et al.*, 2023). In rubber plantations, erosion rates can reach up to 5.7 kg/m^2 annually, owing to the absence of undergrowth (controlled using herbicide). Intercropping maize with leguminous plants, avoiding residue burning and reducing herbicide use in mature rubber plantations can mitigate soil losses (Punyalue *et al.*, 2018; Neyret *et al.*, 2020).

Figure 8.18. Soil loss levels and their distribution in Thailand, 2018



Refer to the disclaimer on page ii for the names and boundaries used in this map.

◊ *Source:* **LDD**. 2018. แผนบริหารจัดการทรัพยากรดินปัญหาของประเทศไทย ระยะ 20 ปี (พ.ศ. 2561-2580) [*Thailand's 20-year soil resource management plan (2018-2037)*]. Bangkok.

In Viet Nam, areas with slopes between 8 and 15 degrees occupy around 16 percent of arable land, and the rest are steep sloping lands with slopes greater than 15 degrees (approximately 62 percent of the total cultivation area) (GSO, 2019). In addition to intensive cultivation and heavy rainfall, widespread continuous monocropping contributes to soil erosion (Tuan *et al.*, 2014; Trung *et al.*, 2021) (Figure 8.19). About 20 percent of Viet Nam has moderate to very high erosion (Table 8.18).

Figure 8.19. Soil erosion in the central highlands (a) and northern mountainous (b) areas of Viet Nam in 2023



Table 8.18. Area of eroded soil in Viet Nam in 2019 (ha)

Region	Erosion level		
	Weak (ha)	Moderate (ha)	Very high (ha)
Northern midland and mountainous	2 018 000	677 000	2 064 000
Red River delta	151 000	189 000	22 000
North central and south-central coast	4 050 000	846 000	761 000
Central highlands	1 060 000	858 000	247 000
Southeast	246 000	165 000	4 000
Total	7 525 000	2 735 000	3 098 000
Erosion level compared to the total investigated area (%)	25.83	9.39	10.63

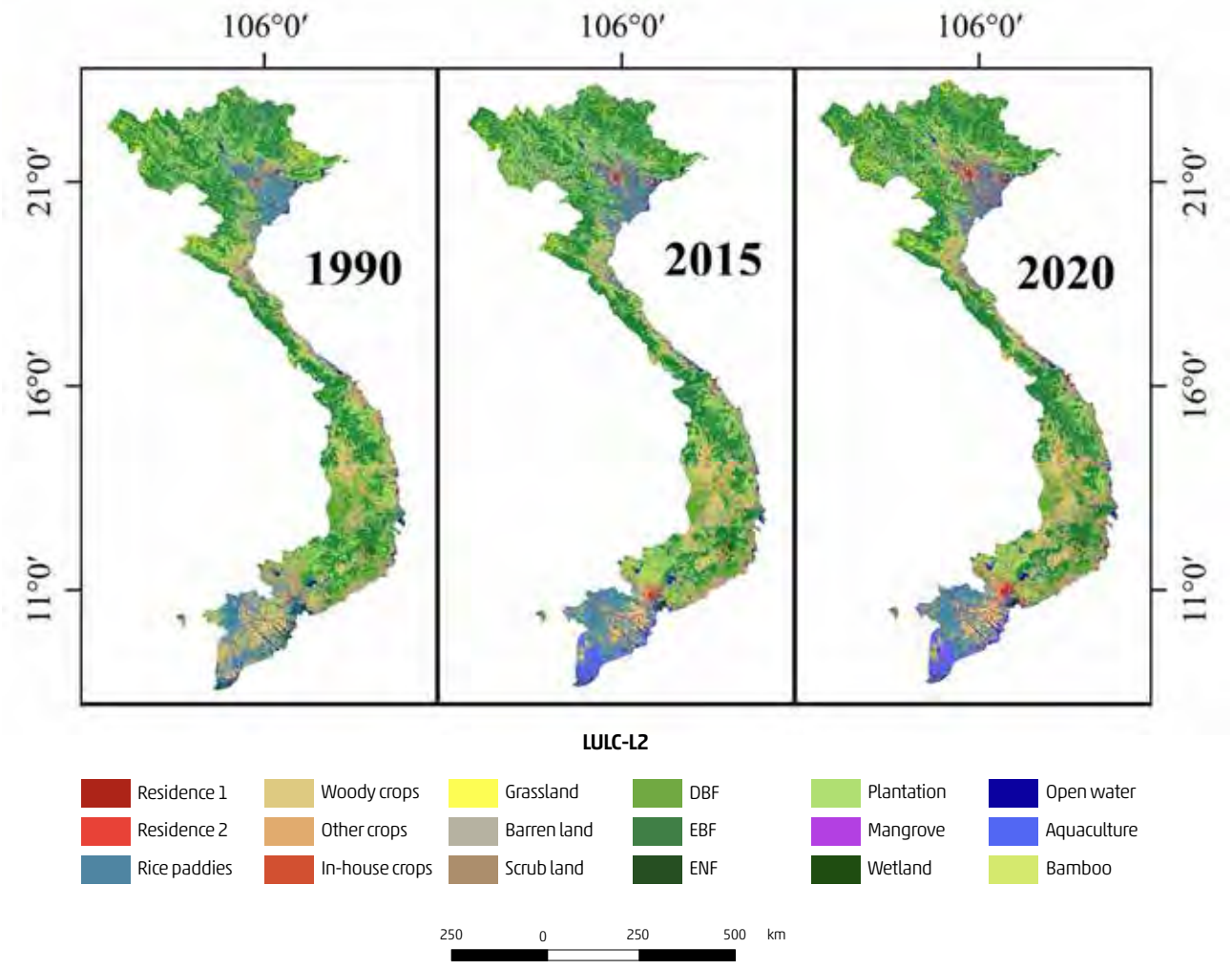
◊ *Source:* **GDLA (General Department of Land Administration).** 2021. *Summary report "Investigation and assessment of land degradation in Vietnam".* Hanoi, Ministry of Natural Resources and Environment. https://monre.gov.vn/VanBan/Pages/ChiTietVanBanChiDao.aspx?pID=2624

Most farmers that farm on sloping land in Viet Nam are aware of the significance of soil erosion and its implications for their agricultural productivity (Huynh *et al.*, 2021, 2022). Erosion is primarily attributed to deforestation (Ngo, Nguyen and Rajendra, 2015; Wang, Ning and Khujanazarov, 2018), low vegetation cover (Trung *et al.*, 2021; Nguyen, Whelan and Nichols, 2022) and the cultivation of cassava (*Manihot esculenta*) and maize (Brandt *et al.*, 2016; Zimmer *et al.*, 2018; Nguyen and Pham, 2018; Huynh *et al.*, 2022), which leads to a reduction in soil structural stability (Schweizer *et al.*, 2017). Erosion can also be substantial in plantations of coffee (Nguyen and Sarker 2018), acacia (*Vacellia* spp.) (Kolyan *et al.,* 2019) and teak (Brandt *et al.,* 2016). According to Trung *et al.* (2021), in the northern mountainous area of Viet Nam, soil loss from cassava monocropping was 3.2 t/ha on gentle slopes and 8.3 t/ha on steep slopes, leading to high nutrient losses (15.1 to 35.4 kg/ha of N and 16.9 to 24.3 kg/ha of K_2O).

The application of vermicompost (Doan *et al.*, 2015), grass strips and reduced soil tillage (known as intercropping [Pham, Degener and Kappas, 2018]) have been shown to be effective erosion control

methods in the country. Intercropping cassava with cowpea (*Vigna unguiculata*) on both gentle and steep sloped fields improved soil cover levels (22 percent on gentle slopes and 17 percent on steep slopes), reduced soil erosion by 40 percent on gentle slopes and 58 percent on steep slopes, and decreased nutrient losses (between 16 and 36 percent for N, between 0 and 19 percent for P, and between 15 and 23 percent for K) (Trung *et al.* (2021). Surface crusting and runoff, and therefore soil loss, were lower at sites that had natural regeneration after cultivation than in teak plantations, and highest under maize cropping (Lacombe *et al.*, 2015; Rochelle-Newall *et al.*, 2016): Simulation models that assess land-use and land-cover change and predict potential soil erosion and runoff would be of use for better control of soil erosion (Ngo *et al.*, 2015; Le Huong and Son, 2020) (Figure 8.20). In addition, an updated soil map for Viet Nam is necessary for soil erosion assessment and prediction.

Figure 8.20. Land-use and land-cover maps in Viet Nam in 1990, 2015 and 2020



Refer to the disclaimer on page ii for the names and boundaries used in this map.

Note: LULC = land use and land cover, DBF = deciduous broadleaved forest, EBF = evergreen broadleaved forest and ENF = evergreen needleleaf forest

Source: **Phan, D.C., Trung, T.H., Truong, V.T., Sasagawa, T., Vu, T.P.T., Bui, D.T., Hayashi M., Tadono T. & Nasahara, K.N.** 2021. First comprehensive quantification of annual land use/cover from 1990 to 2020 across mainland Vietnam. *Scientific Reports*, 11(1): 9979. https://www.nature.com/articles/s41598-021-89034-5

In Indonesia, sloping areas (>4.5 degree slope) cover about 61 percent of the land area. Sloping land combined with either high rainfall (>1 500 mm annually) or low but erratic rainfall, can trigger soil erosion, which can also be exacerbated by mismanagement (Sumiahadi and Acar, 2018). Soil erosion occurs at rates ranging from 0.3 to 1 607 t/ha annually (Table 8.19), depending upon the assessment method, province and landscape position.

Table 8.19. Annual erosion rate from selected studies, Indonesia

Location	Methods	Land use	t/ha	References
Dolago watershed Central Sulawesi province	USLE	Primary forest	52.5	Naharuddin, Malik and Ahyauddin (2021)
		Secondary forest	86.3	
		Shrubland	26.3	
		Mixed dryland farming	291.3	
		Dryland farming	577.9	
		Rice field	2.2	
Samin subwatershed, Central Java province	SWAT	Average across all land uses	57.4	Syahdiba and Kusumandari (2021)
Krueng Mane watershed, Aceh province	USLE	Forest	0.3	Safira *et al.* (2023)
		Open land	1 606.7	
Krueng Jreu subbasin, Aceh province	Gavrilović	Savannah	282.6	Devianti *et al.*, (2023)
		Shrubland	214.2	
		Agricultural dryland	46.6	
		Open land	493.3	
Penet and Beratan watershed, Bali province	USLE	Primary forest	1.3–6.7	Trigunasih and Saifulloh (2023)
		Secondary forest	2.5–3.6	
		Mix dryland farming	74.9–290.8	
		Dryland farming	147.4–1 535.3	
		Rice field	0.3–1.7	
Upper Cikapundung subwatershed, West Java province	RUSLE	Vegetable farming	22.6–141.2	Sutrisna *et al.* (2021)
Cilebak–Cirasea microwatershed, West Java province	USLE	Upland agriculture	45.4	Yustika, Ariani and Mandaya (2021)
Pineapple plantation, Lampung province	USLE	Pineapple farming	1.4–15.0	Afandi, Wiharso and Novpriansyah (2020)
Cipunagara watershed, West Java province	SWAT		1.1–654.4	Fitriyana, Supratman and Mardiani (2020)
Community forest of Krajan Hamlet, Blitar, East Java province	Runoff plot	Agroforestry	1.5–4.1	Pratiwi, Mindawati and Tyasmoro (2020)
Sekampung Hulu watershed, Lampung province	USLE	Coffee plantation	12.5	Yustika *et al.* (2019)
Sangharus watershed, Lampung province	USLE	Coffee plantation	5.6	Yustika *et al.* (2019)
Endanga watershed, Southeast Sulawesi P province	Runoff plot	Forest	0.9	Leomo *et al.* (2018)
		Bush	3.4	
		Oil palm trees	18.6	
		Pepper	6.3	
		Upland rice	48.3	
		Maize	14.6	
		Intercropping (maize and peanut)	8.8	

Location	Methods	Land use	t/ha	References
Jenawi district of Karanganyar, Central Java province	USLE	Forest	7.9–27.6	Herawati *et al.* (2018)
		Garden	32.1–1 600.2	
		Dry fields	20.9–1 514.3	
		Rainfed rice field	6.5–478.2	
		Irrigated rice field	0.2–3.2	
Manna watershed, Bengkulu province	USLE	Coffee and rubber plantations	170.7	Amri *et al.* (2017)
Kintamani district of Bangli, Bali province	USLE	Forest	4.8–32.9	Sumarniasih and Antara (2017)
		Shrubland	17.2–248.9	
		Dryland	70.1–370.6	
		Mixed garden	15.9–250.3	

Note: USLE = Universal Soil Loss Equation, SWAT = Soil and Water Assessment Tool, RUSLE = Revised Universal Soil Loss Equation.

Sources: See References.

There are 12.7 million ha of Indonesian land mapped as "critically" degraded (based on erosion hazard), mainly in Kalimantan and Sumatra. Guided by the *National Critical Land Map* (KLHK, 2022) governments and communities are rehabilitating land and forests by implementing water conservation measures through financial support, community empowerment and national policies.

In addition to technical and social measures, policies and various programmes have been developed to address soil-related problems, namely implementing an Enhanced Nationally Determined Contribution (Government of Indonesia, 2022), forest and land rehabilitation (Indonesia, 2021), and a low carbon and climate resilience strategy (Government of Indonesia, 2021).

8.5.3 Loss of soil organic carbon in Southeast Asia subregion

Changes in forest area are a driver of SOC change. In the 2015–2022 period, Malaysia, the Philippines and Viet Nam have experienced substantial increases in forested area (Figure 8.5). During the same period, Cambodia, Indonesia, the Lao People's Democratic Republic, Myanmar, and Thailand have experienced decreases in forest area.

Tropical peatlands are important as a reservoir for SOC. Warren *et al.* (2017) cited figures showing that Southeast Asia had 41 percent of the global tropical peatland area and 65 percent of tropical peat stores. The Southeast Asia peatlands typically occur as forested peat swamps in the low-lying areas between river valleys in the coastal plains of Borneo and Sumatra. Land-use change on the peatlands is typically driven by conversion to industrial and smallholder plantations for cash crops, such as oil palm trees and acacia trees for pulp and paper production. Vijayanathan *et al.* (2021) reported that peat soil carbon stocks were 1 596 t/ha in peat swamp forests and 970 t/ha in peatlands used for mixed cash crops in the Selangor peatlands of Malaysia, measured down to the mineral substratum.

Mangrove forests are also of concern for SOC storage in Southeast Asia. From 2000 to 2012, the mangrove forest cover was estimated to decline at an annual average rate of 0.18 percent (Richards and Friess, 2016) due to agricultural land expansion, urbanization and aquaculture industries (Friess *et al.*, 2019). Nevertheless, active restoration measures were undertaken since the 1990s, to increase the below- and above ground C pool in Southeast Asia (Gerona-Daga and Salmoll, 2022).

Loss of biodiversity and depletion of SOC are manifestations of soil degradation in the Philippines (Asio *et al.*, 2009). The loss of SOC is mostly due to unfavourable farming practices such as the indiscriminate use of pesticides, biomass removal and absence of organic-based fertilizers.

Over half of Thailand's terrain has low to moderate SOC levels (Figure 8.21), with the northeast region particularly C-deficient, as more than 70 percent has less than 31.25 t/ha of SOC. Rapid decomposition of organic material contributes to low SOC (Jaiarree *et al.*, 2011a; Saree *et al.*, 2012; Chaun *et al.*, 2017). However, high compost application rates can help sustain SOC (Jaiarree *et al.*, 2011b). In Northeast Thailand, SOC variation in rice fields is correlated with yields, with a 1 g SOC/kg of soil increase correlating with a 302 kg/ha rise in rice production (Arunrat *et al.*, 2020). Additionally, annual C sequestration in soil ranges widely, from 0.04 to 14.1 percent, depending on practices and C inputs, with rice straw contributing 1.43 t/ha of C annually (Oechaiyaphum *et al.*, 2020). Studies by Thammasom *et al.* (2016) and Arunrat *et al.* (2020) underscore that biochar application can effectively boost SOC content, though the extent of SOC improvement can vary significantly based on soil type and the feedstock used to produce the biochar.

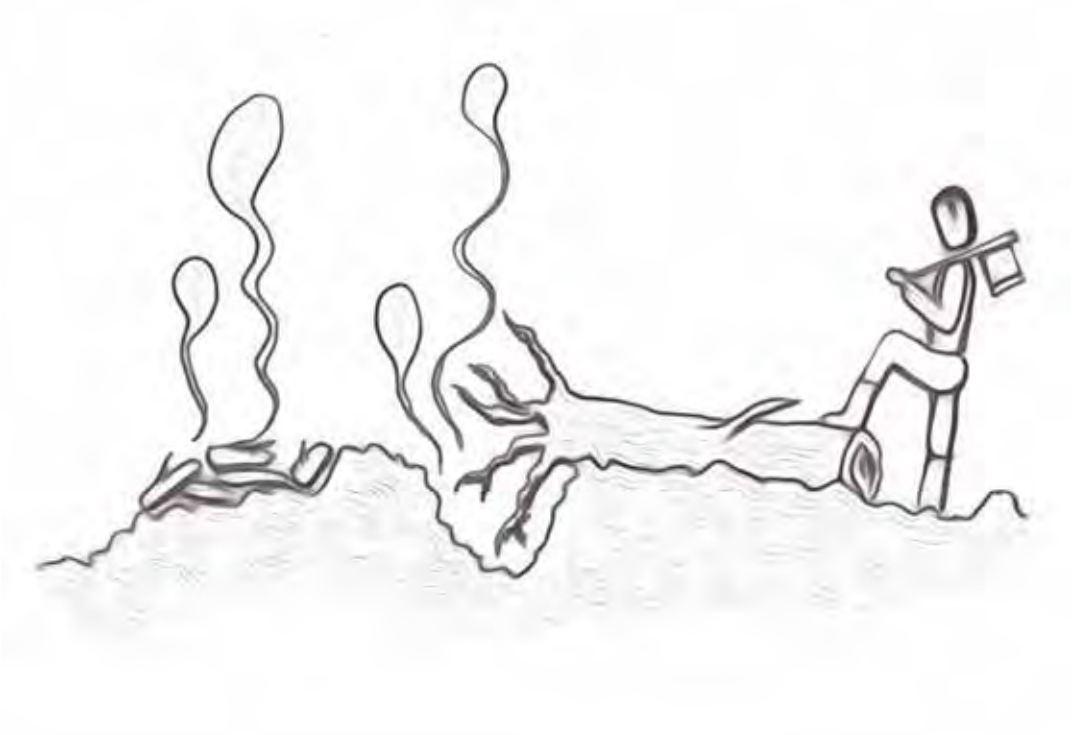
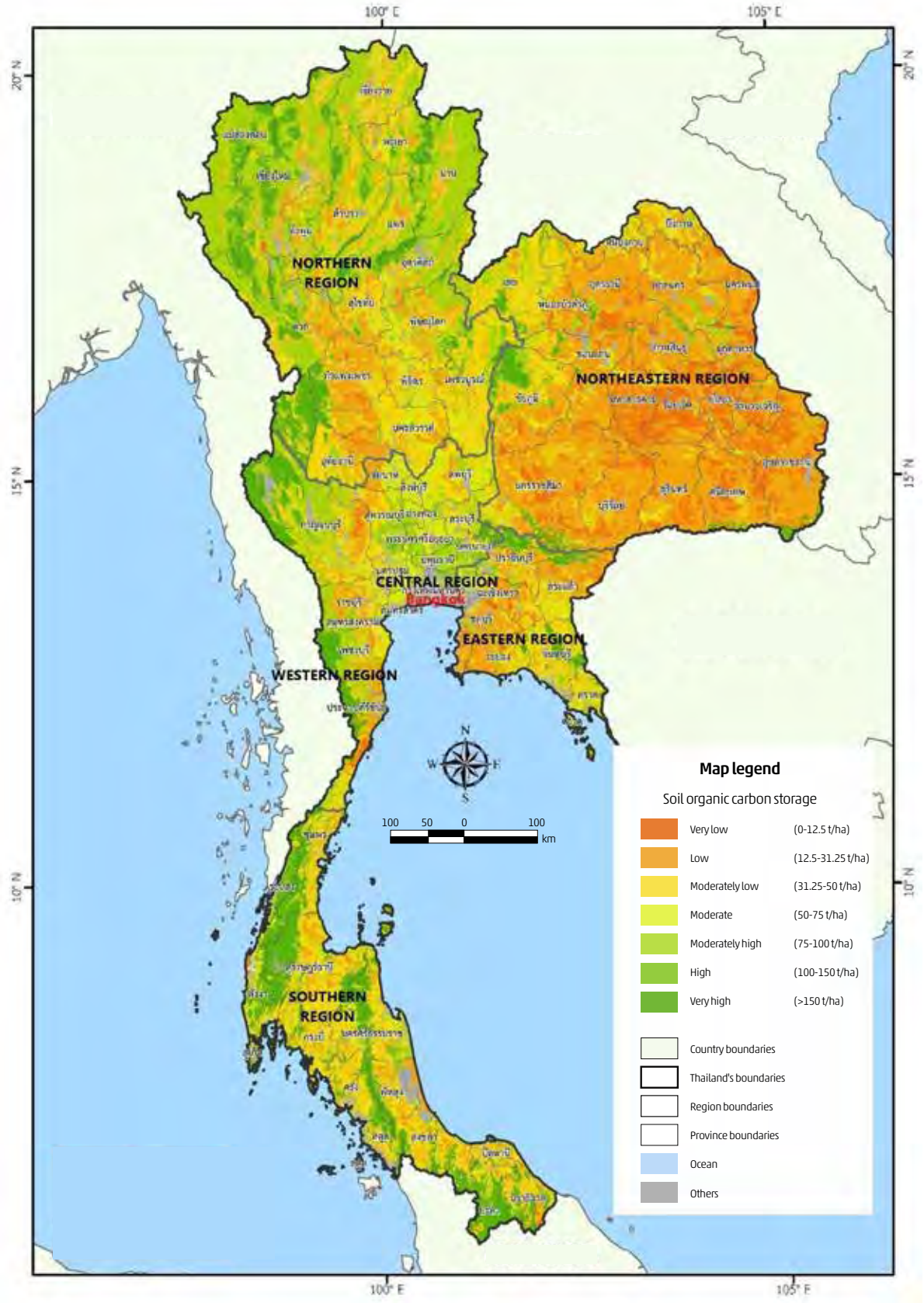


Figure 8.21. Soil carbon map of Thailand to 1 m depth, 2018



Refer to the disclaimer on page ii for the names and boundaries used in this map.

◊ *Source:* **LDD**. 2018. แผนบริหารจัดการทรัพยากรดินปัญหาของประเทศไทย ระยะ 20 ปี (พ.ศ. 2561-2580) *[Thailand's 20-year soil resource management plan (2018-2037)]*. Bangkok.

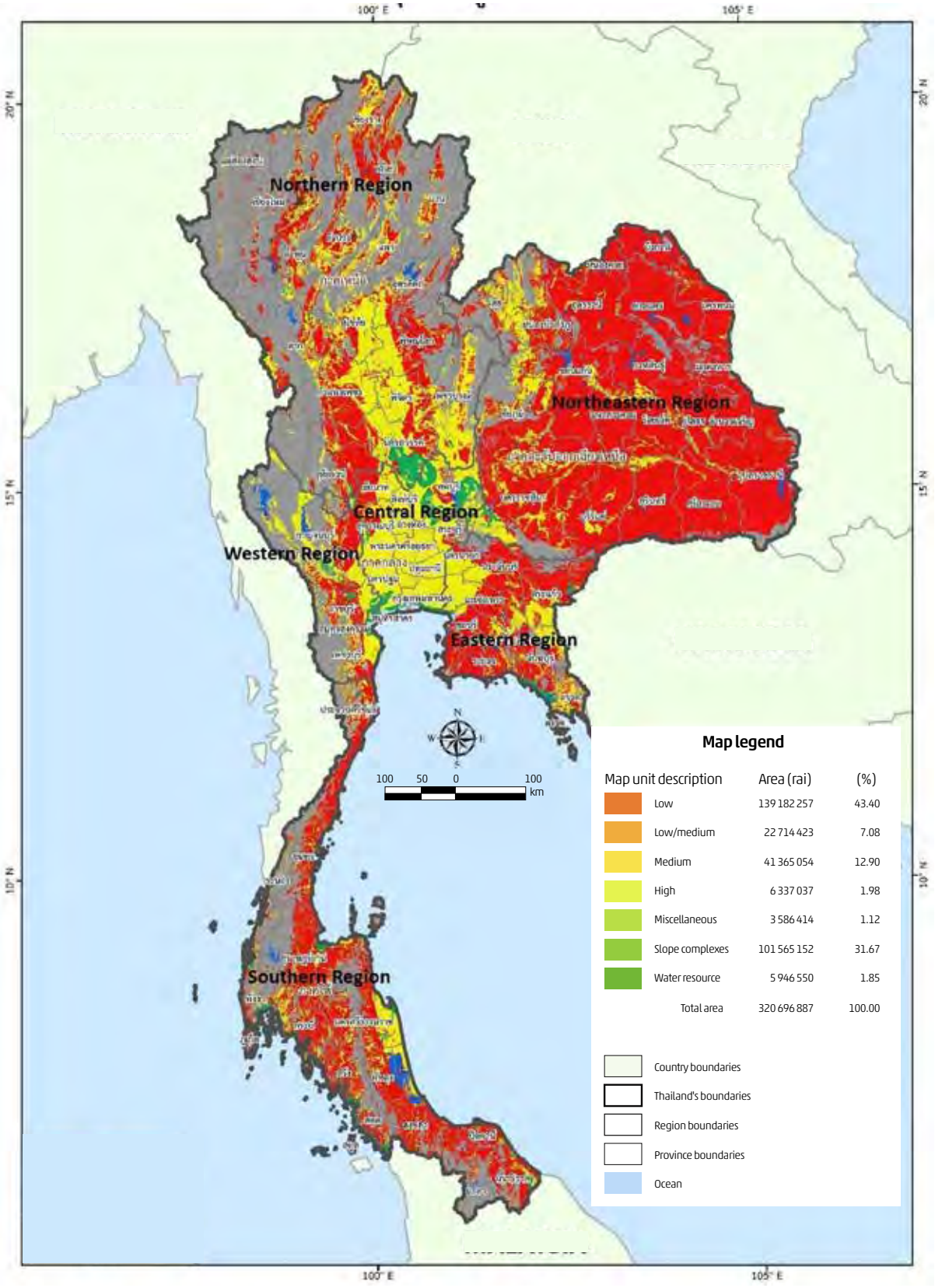
8.5.4 Nutrient mismanagement in Southeast Asia subregion

Only Brunei Darussalam and Viet Nam show evidence of nutrient surpluses of N and P in the 2017–2021 period (Figure 8.7 and Figure 8.8). Timor-Leste has very low N outputs, possibly indicating yield gaps in crop production due to N deficiency. The remainder of the countries (at the national level) have nutrient use budgets within a sustainable range (Figure 8.7 and Figure 8.8).

Soil fertility in Thailand is generally low, with approximately 43 percent of agricultural soil with low fertility (Figure 8.22). Site-specific nutrient management (SSNM) is recommended to address nutrient imbalances, which require soil analysis, plant genetics, climate, and soil type to determine the nutrient requirements of crops (Attanandana and Verapattananirund, 2015). Site-specific nutrient management can reduce fertilizer use by 30 to 50 percent and increase yield by 10 to 20 percent for crops such as rice, maize, cassava, and sugar cane (Attanandana *et al.*, 2006, 2010). Site-specific nutrient management is also recommended to reduce greenhouse gas emissions in rice cultivation (Thai-German Cooperation, 2023) and is considered as a low-cost technology that can be easily followed by farmers. Online soil analysis using a web application is available with minimal data requirements (Komkaew, 2005). In the Philippines, BSWM (2021) mapped 82 percent (2.2 million ha) of the total paddy rice areas as low to moderately low soil fertility.



Figure 8.22. Map of soil fertility in Thailand



Refer to the disclaimer on page ii for the names and boundaries used in this map.

Note: **The areas highlighted in red are considered low-fertility soils. 1 hectare is the equivalent to 6.25 rai.**

◊ *Source:* **LDD**. 2021. แผนบริหารจัดการทรัพยากรดินปัญหาของประเทศไทย ระยะ 20 ปี (พ.ศ. 2561-2580) [*Thailand's 20-year soil resource management plan (2018-2037)*]. Bangkok.http://oss101.ldd.go.th/soilr/Soil_Fer.jpg/Picture/map%20Soil%20Fer.jpg

To provide enough food, boosting crop production is necessary in Indonesia, leading to the intensive use of paddy and cropland soils. Both high and low soil nutrient imbalances were observed, as fertilizer was being applied by farmers regardless of soil nutrient status and crop needs amid high fertilizer prices and limited fertilizer subsidies (Table 8.20) (Kementerian Pertanian, 2020). Nutrient imbalance was observed in organic rice farming where P was surplus but N and K were deficient (Setyorini and Hartatik, 2021). Nutrient imbalance was also reported in smallholder plantations (Woittiez *et al.*, 2018, 2019; Lim *et al.*, 2023; Sugianto *et al.*, 2023), with soil erosion worsening the imbalance in sloping cropland (Nugroho, 2022). However, the Indonesian government have developed strategies that promote balanced fertilizer applications, and have developed soil nutrient maps of P and K to assist fertilizer requests (Agus *et al.*, 2019). Organic fertilizers, such as compost and manure, are promoted, and fertilizer subsidies are provided to smallholder farmers.

Table 8.20. Nutrient phosphorus and potassium status of paddy fields in 23 rice-producing provinces in Indonesia

Province	Nutrient P status (ha)			Nutrient K status (ha)		
	Low	Medium	High	Low	Medium	High
1. Aceh	42 204	91 303	79 787	14 509	59 513	139 272
2. North Sumatra	24 560	184 078	99 603	12 174	156 343	139 724
3. Riau	25 830	34 033	2 826	1 894	43 976	16 819
4. Jambi	24 224	33 705	10 385	2 939	41 757	23 618
5. West Sumatra	33 586	84 275	76 409	68 979	95 591	29 700
6. Bengkulu	13 613	15 275	21 849	10 198	28 231	12 329
7. South Sumatera	325 799	122 898	22 041	35 749	214 708	220 281
8. Lampung	205 521	127 226	27 460	158 173	172 397	29 637
9. Banten	21 219	136 002	47 117	93 112	56 716	54 510
10. West Java	105 955	196 568	62 532	129 536	396 297	402 010
11. Central Java	133 475	519 792	393 563	196 836	375 541	476 453
12. DI Yogyakarta.	41 560	13 290	21 462	18 828	24 794	32 690
13. East Java	127 416	472 613	615 099	178 908	408 794	627 426
14. West Kalimantan	96 179	124 464	22 345	75 307	112 838	54 843
15. South Kalimantan	109 951	109 024	72 101	126 273	93,755	71 048
16. Bali	1 767	11 808	57 409	–	214	70 770
17. West Nusa Tenggara (Lombok)	2 667	8 502	100 179	–	–	111 348
18. South Sulawesi	81 638	225 586	328 781	25 292	72 787	537 926
19. West Sulawesi	3 429	19 565	16 462	9 108	8 724	21 624
20. Central Sulawesi	2 766	49 446	64 375	5 299	38 308	72 980
21. Southeast Sulawesi	7 251	55 515	19 351	15 630	52 550	13 937
22. North Sulawesi	824	24 134	22 084	1 070	24 090	21 882
23. Gorontalo	6 210	12 077	14 760	802	11 581	20 664
Total	1 437 644	2 671 200	2 762 768	1 180 616	2 489 505	3 210 491
Percentage of nutrient	21%	39%	40%	17%	36%	47%

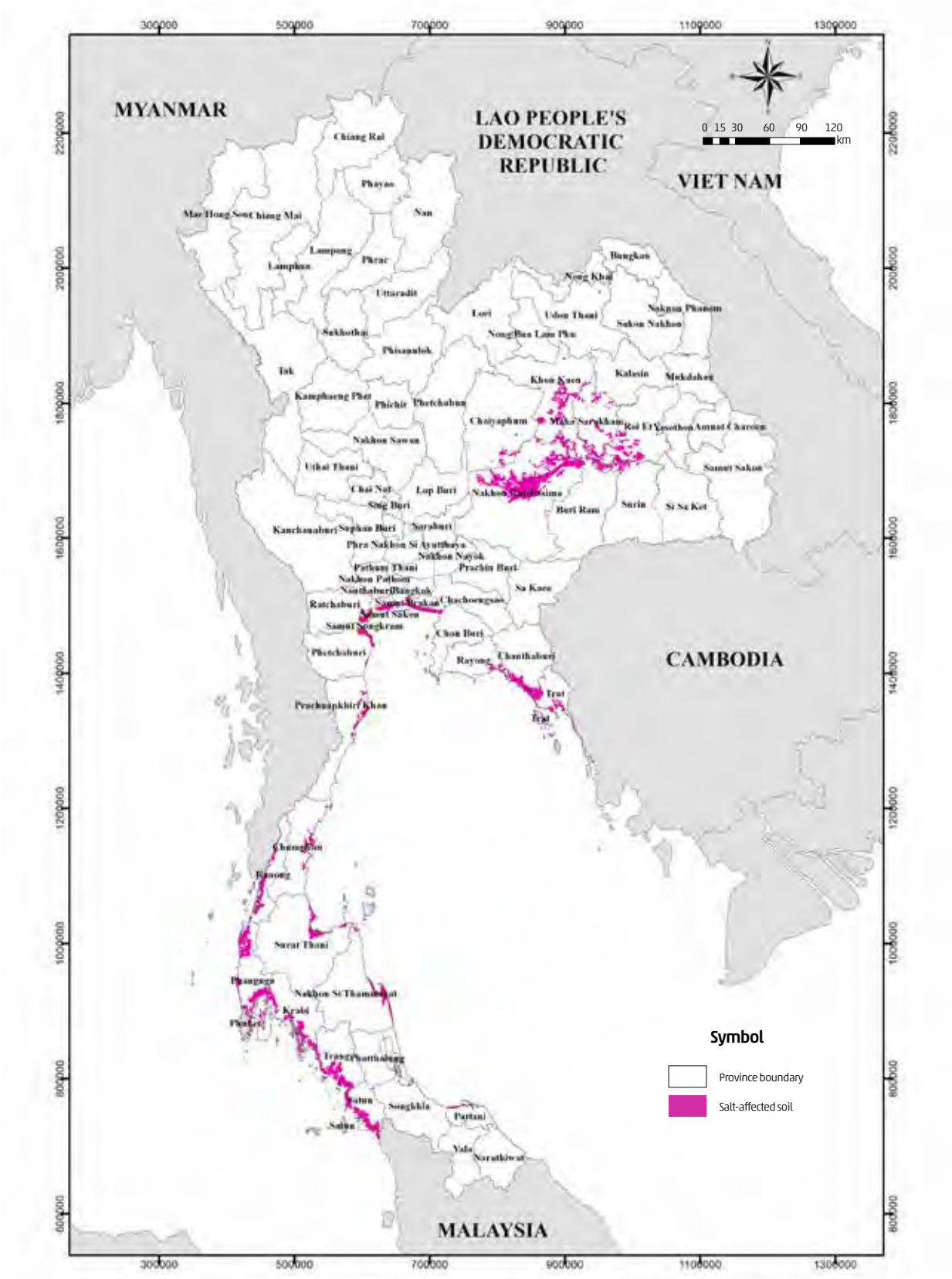
◊ *Source:* **Kementerian Pertanian**. 2020. *Rekomendasi Pupuk N, P, and K Spesifik Lokasi Untuk Tanaman Padi, Jagung, Kedelai Pada Lahan Sawah (per Kecamatan).Buku I: Padi [Site Specific Fertilizer Recommendation of N, P, and K for rice, maize, and soybean in paddy fields (at district level).* Book I: Rice]. Jakarta, Badan Penelitian dan Pengembangan Pertanian

8.5.5 Salt accumulation in Southeast Asia subregion

In 2018, Thailand had around 700 000 ha of salt-affected soils, a decrease from 720 000 ha in 2006 (Figure 8.23) (LDD, 2018). Salt-affected soils occur in coastal areas with potential acid sulphate soil from tidal activity, and in inland areas with underlying rock salt. To combat salinization and enhance land use, Thailand's Land Development Department (LDD) introduced measures such as groundwater management, salt-resistant crops, soil moisture conservation, and organic matter enrichment with green manure, cow manure and compost (LDD, 2006). A study by Petmuenwai *et al.* (2024) suggested that incorporating vermicompost and organic amendments would be a feasible and economical method for enhancing the quality of salt-affected soils in a sustainable manner and improving yields in severely salt-affected areas.



Figure 8.23. Distribution of salt-affected soil in Thailand in 2018



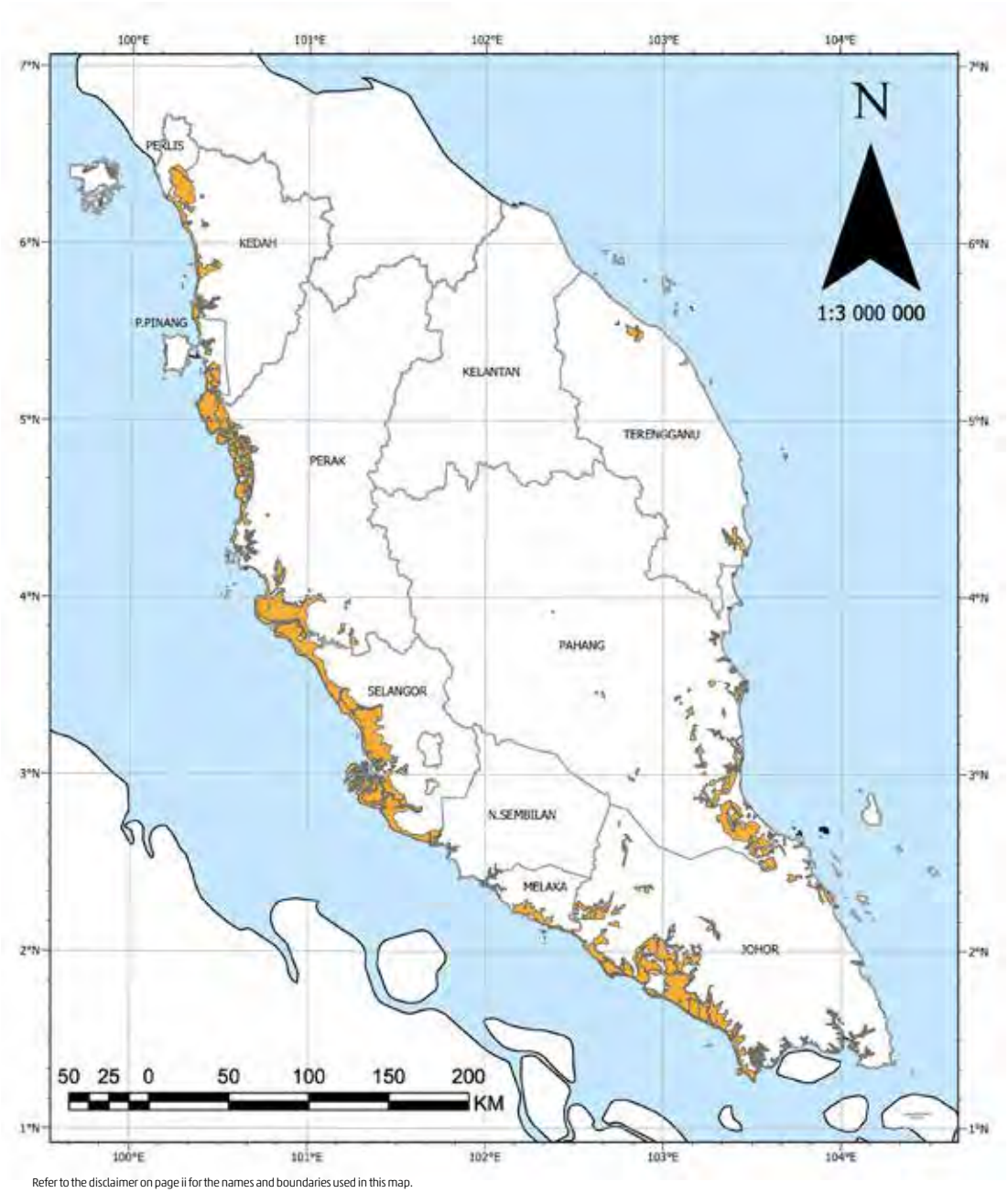
Refer to the disclaimer on page ii for the names and boundaries used in this map.

◊ *Source:* **LDD**. 2018. แผนบริหารจัดการทรัพยากรดินปัญหาของประเทศไทย ระยะ 20 ปี (พ.ศ. 2561-2580) [*Thailand's 20-year soil resource management plan (2018-2037)*]. Bangkok.

8.5.6 Soil acidification in Southeast Asia subregion

Soil acidification is a major cause of chemical land degradation in Malaysia, affecting 72 percent of highly weathered acidic soils and acid sulphate soils. The natural ecosystem processes of pyrite oxidation and organic matter deposition are primary causes of soil acidification in the country (Shamsuddin *et al.*, 2021; Rahman *et al.*, 2017; Teh *et al.*, 2017). Acidic soils have a low pH (3–5), low nutrient availability of P, Ca and Mg, and micronutrient deficiencies. Approximately 6.8 percent of the soil in Malaysia is comprised of pyrite-bearing sediments, with acid sulphate soils being mostly found in coastal plains (DOA, 2023; Shamsuddin *et al.*, 2021) (Figure 8.24). Oil palm trees, rubber trees, rice, cacao trees and rice plants are cultivated in the soils, although high Al^{3+} toxicity has been reported for rice cultivation on acid sulphate soils due to pyrite oxidation (Shamsuddin *et al.,* 2021).

Figure 8.24. Distribution of acid sulphate soils in Peninsular Malaysia



◊ *Source*: Adapted from **Shamsuddin, J., Alia Farhana, J. & Enio Kang, M.S.K.** 2021. *Acid Sulfate Soils in Southeast Asia: Occurrence, Properties and Management*. Serdang, Malaysia, Universiti Putra Malaysia (UPM) Press

Methods to reduce acidity include the use of ground magnesium limestones (GMLs) or basalt (Shamsuddin *et al.*, 2021; Rahman *et al.*, 2017) and controlling the water table levels via the systematic construction of drains (flap gates) to maintain water levels above the pyritic layer. For rice production on acid sulphate soils, the application of GML or basalt in combination with biofertilizer fortified by beneficial microorganisms, or GML with fused magnesium phosphate ($Mg_3[PO_4]_2$) is recommended (Shamsuddin *et al.,* 2021; Panhwar *et al.,* 2020; Panhwar *et al.,* 2016).

Most soils in the Lao People's Democratic Republic are acidic and low in organic matter. Acrisols, Alisols, Histosols and Lixisols are the major soil types, and dominate approximately 74 percent of the land. In general, these major soils are acidic and shallow, with comparatively poor productivity. Low-lying lowland areas, specifically the Vientiane Plain, are acidic, deficient in exchangeable Ca, Mg and available P, and have a risk of Al toxicity (Matsuo *et al*., 2015). To determine appropriate soil management, the effects of liming on non-rice crops and rice plants were studied by Kazuyuki *et al.* (2015) in pot experiments. Liming is usually effective in enhancing the growth of non-rice crops tested. Under submerged conditions, soil pH naturally increases despite strong pre-flooding acidity (Kazuyuki *et al*., 2015).

However, there are also approximately 4.5 million ha of agricultural soils in lowland areas along the Mekong River and its tributaries, (NLMP, 2018), which are dominated by Luvisols, Cambisols, Fluvisols, Gleysols, and Leptosols. These soils are neutral to alkaline and fertile, with a favourable organic matter content.

Almost 27 percent of Philippine soils are Ultisols which are inherently acidic (Carating *et al.*, 2014). Due to the use of synthetic fertilizers and other unsustainable practices in the management of agricultural lands, soil acidification poses a major soil threat. Being archipelagic in nature and where large swaths of low-lying crop lands are situated along the coast, sizeable paddy rice areas are seasonally affected by soil salinization due to saltwater intrusion into paddy fields (Asio *et al.*, 2009). The salt impacts soil properties, which leads to poor crop yields.

Thailand has approximately 15.5 million ha of acid soil and 0.9 million ha of acid sulphate soil that are at risk of further acidification (LDD, 2018). Acidic soils occur in the central region, as well as the southeastern and eastern coast of the southern region of Thailand (Figure 8.25) (LDD, 2018) and are characterized by the presence of yellowish jarosite. The soil becomes increasingly acidic when exposed to the atmosphere, with pH levels falling below 3.5 (Figure 8.26). Aluminium toxicity is common in these soils and can be reduced by applying liming materials, such as dolomite (Suddhiprakan *et al.*, 2014). Additionally, trenching and maintaining proper water and drainage levels can prevent the exposure of sulphide materials to air.

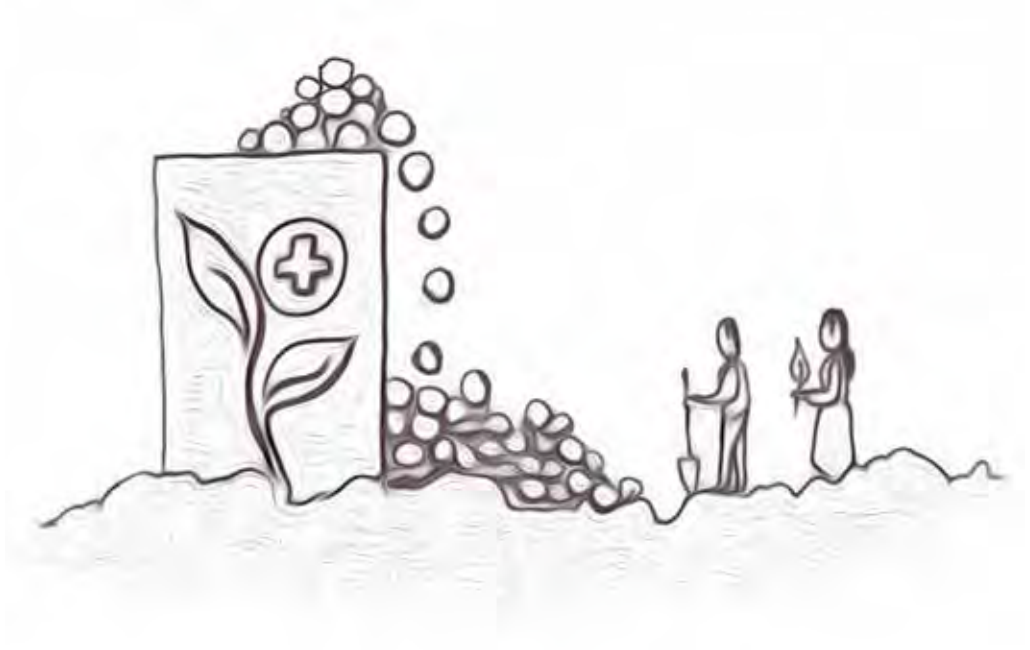
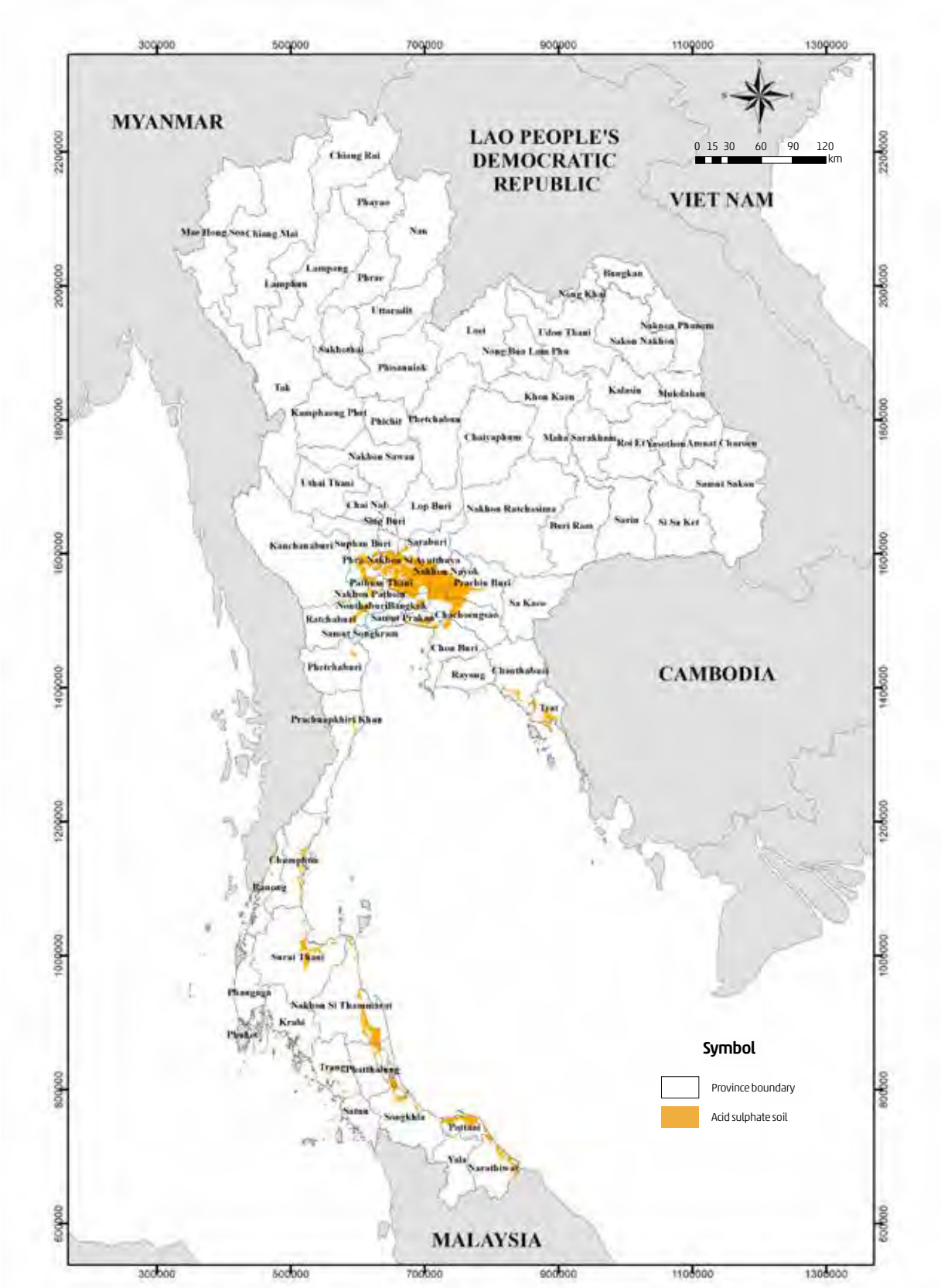


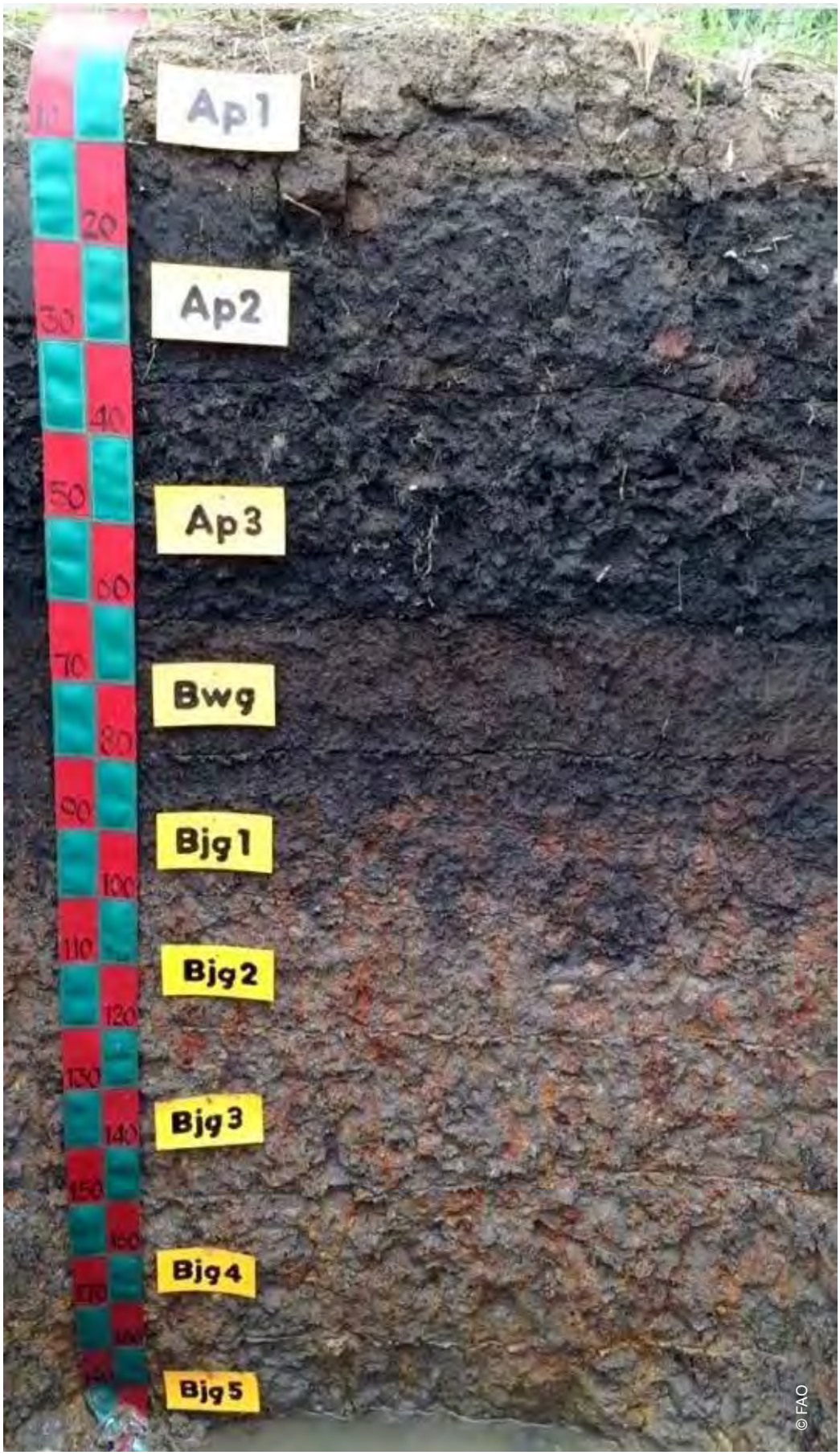
Figure 8.25. Distribution of acid sulphate soils in Thailand in 2008.



Refer to the disclaimer on page ii for the names and boundaries used in this map.

◊ *Source:* **LDD**. 2018. แผนบริหารจัดการทรัพยากรดินปัญหาของประเทศไทย ระยะ 20 ปี *(พ.ศ. 2561-2580) [Thailand's 20-year soil resource management plan (2018-2037)].* Bangkok.

Figure 8.26. Typical profile of acid sulphate soil in Rangsit, Thailand (showing a soil series of Typic Sulfaquepts)



Indonesian soils are mostly naturally acidic, with intensive agrochemical applications such as found in coffee and vegetable cultivation leading to further soil acidification (Manson *et al.*, 2022; Kashyap *et al.*, 2023). Nitrification controls soil acidification in cultivated soils (Fujii *et al.*, 2021) and liming is recommended, with its rate is adjusted to Al saturation and crop type.

Poor water management and pyrite oxidation intensify the soil acidification in tidal paddies and wet cropland soils. Liming and water management are recommended, where the rate is based on pyrite depth and soil pH. Liming at the rate of 2 t/ha is recommended for acid sulphate soils and 1 t/ha for others. Water management to ensure good water circulation can be undertaken by maintaining channels and water-gates.

8.5.7 Urban expansion and soil sealing in Southeast Asia subregion

Urban encroachment on productive land in the Philippines is becoming a substantial threat because of the competing use of prime land for non-agricultural development projects (BSWM, 2022).

The Department of Agriculture, through the BSWM, leads the conservation and sustainable management of the country's soil and water resources, primarily for agriculture. The top priority is the National Soil Health Program, which aims to promote a systematic and holistic approach to addressing soil health to ensure the sustainable use and management of soil resources (Nilo, 2020). Among the important activities currently being implemented are enabling soil laboratories across the country, promoting an adaptive and balanced fertilization strategy for paddy rice production, assembling databases, and mapping soil reference and soil monitoring sites.

8.5.8 Soil pollution in Southeast Asia subregion

Soil pollution in Indonesia is an environmental concern and is primarily caused by industrial waste and mining activities due to heavy metals such as Hg, Cd, Pb, and As due to gold mining and textile industries (Anggriawan, 2022). Gold mining has led to Hg concentrations of up to 240 parts per million (ppm), while textile industries contribute approximately 296 500 tonnes of heavy metals annually that has spread to agricultural lands, including rice fields in Rancaekek, Bandung, and along the northern coast of Java, most likely through untreated industrial effluent, contaminated irrigation water, sediment deposition, atmospheric deposition, flooding, and soil erosion. Phytoremediation using plants such as vetiver grass (*Vetiveria zizanioides*) is being investigated as a potential remediation technique.

Studies by Brindha *et al.* (2019) and Vongdala *et al.* (2019) focusing on the municipal solid waste (MSW) landfill (receiving 300 tonnes of waste daily) in Vientiane, in the Lao People's Democratic Republic found levels of heavy metals (such as Cd, Cr, Cu, Pb, and Zn) in soil that exceeded national and international safety standards, indicating a serious risk to both the environment and public health. The accumulation of heavy metals in edible plant parts of water spinach (*Ipomoea aquatica*), a widely consumed vegetable, may cause health problems for villagers living in and around MSW landfills (Vongdala *et al.*, 2019).

Microplastics have been detected in soils at multiple landfill sites (Tun *et al.*, 2022). Additionally, urban canals, particularly along the Mark-Hiao subriver in Vientiane, have substantial levels of microplastics in both water and sediment (Noudeng *et al.*, 2023).

The cultivation of boom crops such as cassava, maize, sugar cane, rubber, banana, watermelon and soybean has intensified in the Lao People's Democratic Republic, frequently involving monoculture practices with substantial inputs of fertilizers and pesticides, which can lead to various

environmental concerns, such as reliance on high nutrient and pesticide inputs that can degrade soil quality over time and pose soil erosion threats. Organochlorine and organophosphate pesticides have been frequently detected in regions engaged in intensive rice cultivation, shifting cultivation and boom crop growth. Between 2015 and 2020, approximately 610 000 tonnes of pesticides and 5 240 000 tonnes of fertilizers were imported into the country, highlighting the escalating reliance on chemical inputs in agricultural practices (DNREI, 2022).

Soil contamination in Malaysian agricultural soils is attributed to the excessive use of synthetic fertilizers and agrochemicals and the reuse of untreated industrial wastewater for agricultural activities that contain heavy metals, pesticides and nitrates (Che Lat *et al.,* 2023; Zulkafflee *et al.,* 2021; Kamarudzaman *et al.,* 2018). Urban soils show elevated levels of metals such as Pb and Cr (Praveena *et al*., 2015). The concentrations of heavy metals (Cu, Cd, Cr, Pb, and As) detected in soils from rice granary areas in Kedah were reported to be lower than the maximum permissible limit recommended by the European standards for agricultural soils (Zulkafflee *et al.,* 2021), indicating that the soils were safe for paddy cultivation. Surface soil microplastic pollution in agricultural farms in Klang Valley (with an average particle soil microplastic size ranging between 2 822.31 and 3 260.76 µ), was studied by Shanmugam *et al.* (2024), identifying the dominant soil microplastics as polyvinyl chloride, high-density polyethylene, polycarbonate, and polystyrene. Waterbodies in the Cameron Highlands are also contaminated by fertilizers such as chicken manure and organochlorine pesticide residues such as methoxychlor, endrin ketone, and 4,4'DDE (Razali *et al.*, 2018). Mitigation efforts include strict environmental regulations, integrated pest management and waste reduction initiatives (Khanif and Salmijah, 1996). However, comprehensive soil quality monitoring and research are still lacking, highlighting the need for coordinated efforts to address soil pollution in the country.

Soil pollution in the Philippines primarily stems from excessive chemical fertilizer and pesticide use in intensive farming areas, as well as improper handling of agricultural chemicals and inappropriate waste management (Magcale-Macandog, 2016). Pesticide application is widespread, with 80 percent of Central Luzon farmers using insecticides at least once per season. The agricultural revolution during the 1970s to the 1980s has brought about a four-fold increase in pesticide application (Moya *et al.*, 2015).

The problem extends beyond soil contamination, as rice cropping systems contribute 23 706 tonnes of N annually to Manila Bay in the Philippines (Samar, 2012). Agricultural GHG emissions increased by 38 percent between 2000 and 2012, reaching 51.256 billion kg CO_2 equivalent (FAO, 2013).

Mitigation efforts include promoting integrated pest management (Escalada and Heong, 2012), organic farming practices and improved fertilizer management (Velasco *et al.*, 2012). Biofertilizers and biopesticides are being developed (Javier and Brown, 2009), and crop rotation and diversification are encouraged to improve soil health.

Levels of pesticide residues in the Thachin, Chao Phraya and Bangpakong rivers in Thailand were above the safety limit set by the European Union of 0.1 µg/l (Chatsantiprapha, 2002). Organochlorine pesticides were detected in 40.62 percent of the samples (0.01 to 1.21 µg/l) and 20.62 percent of the samples had organophosphate pesticides (0.01 to 5.74 µg/l in the river waters (Tirado *et al.,* 2008).

The World Bank (2016) reported that in Viet Nam, the overuse of fertilizers was widespread, with rice and coffee farmers applying 20 to 30 percent more fertilizer than recommended. This has resulted in an estimated 140 000 tonnes of N, 82 000 tonnes of P and 66 000 tonnes of K released annually into soil and water from rice farming in the Mekong Delta alone. Pesticide usage has increased from 6 500 to 9 000 tonnes in the 1981–1986 period to approximately 100 000 tonnes per year in 2015. It is estimated that up to 30 000 tonnes of pesticides are released into the environment annually (excessive N use leads to soil acidification and micronutrient imbalances).

Sediments also cause the siltation of reservoirs, consequently reducing their lifespan (Wang *et al.*, 2018), and contribute to water pollution with heavy metals (Mai, Maeght and Valentin, 2020; Nguyen *et al.*, 2020) and pathogens (Rochelle-Newall *et al.*, 2016).

8.5.9 Soil water availability

In the Lao People's Democratic Republic, droughts affect rainfall patterns and soil moisture, which, in turn, affect agricultural yields. Palanisamy *et al.* (2023) found that areas with forests had shorter periods of reduced river flow (by at least two months) when compared to agricultural regions, indicating that forests may mitigate drought effects on streamflow and thus drought severity.

Drought can affect large areas, with a lack of rain occurring twice a year during the early and late monsoon seasons, with the late monsoon season especially impacting the growth of crops. Flooding is more localized, with flash floods tending to occur more frequently in areas with a high degree of damaged forest or heavy land exploitation (Eppretch *et al.*, 2018). Increased precipitation affects crops through increased flooding, soil waterlogging and a higher incidence of fungal diseases and pests. Large rainfall events are expected to occur more frequently in the future, potentially triggering floods and landslides (GoI, 2018).

In Viet Nam, drought frequently occurs in many regions, from the midlands and uplands to the coastal plains, causing damage to the national economy, especially in agricultural production. Viet Nam's south-central coast suffers from acute droughts and water scarcity in the dry season, owing to its rugged terrain, short rivers, sandy soils, shallow coastal aquifers and low rainfall. Since 2004, rainfall has decreased, often dropping 20 to 30 percent below the long-term average, leading to increasingly severe dry season droughts in these areas (Thanh *et al.*, 2020).

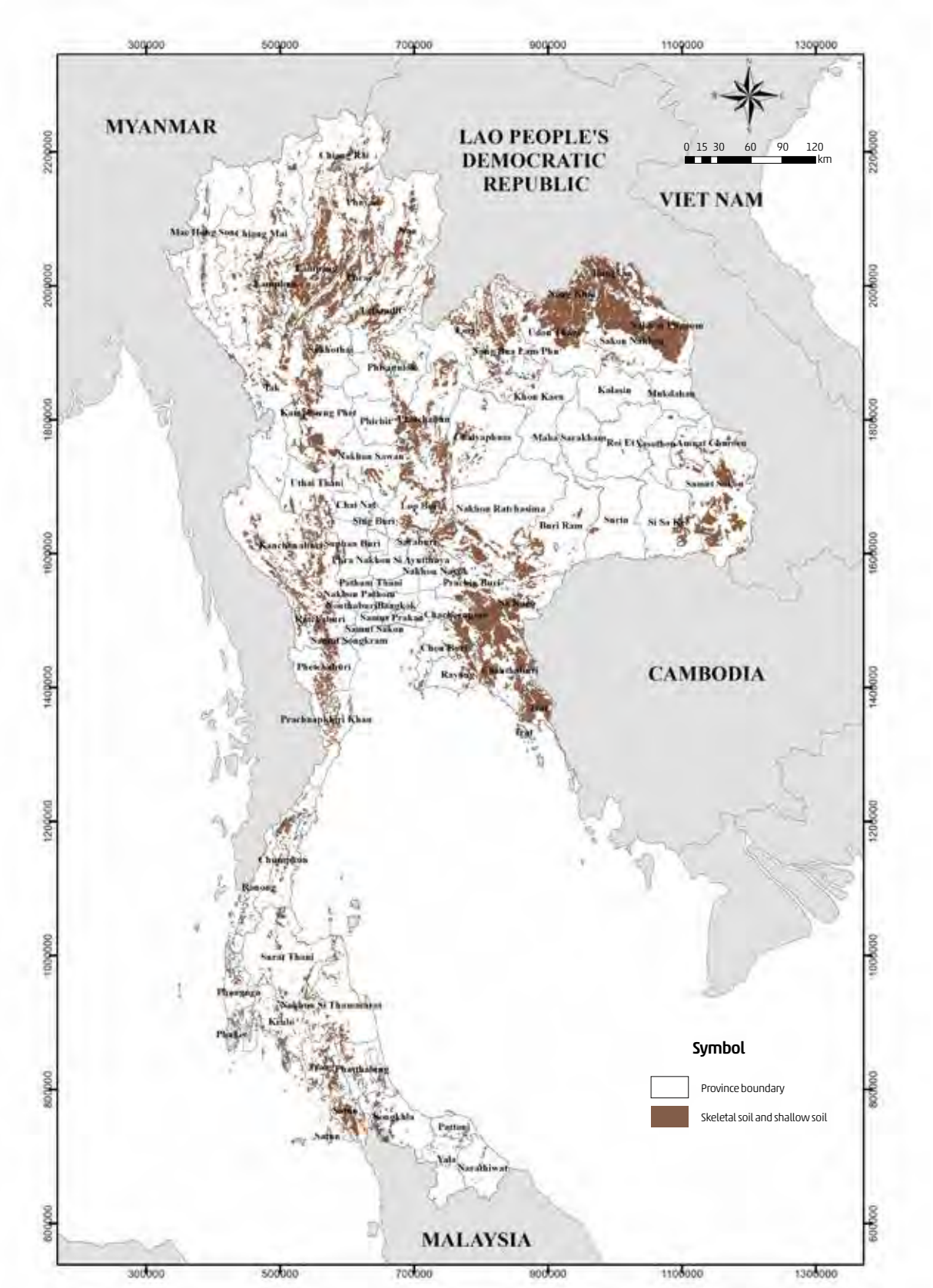
In the Mekong Delta region, water shortage and drought occur in all cropping seasons in the southwestern area, leading to saltwater intrusion, which has impacted agricultural production and people's livelihood. The 2019–2020 drought was the most severe disaster event in history, affecting 58 000 ha of rice, 6 650 ha of fruit trees, 1 241 ha of vegetables, and 8 715 ha of aquaculture. Up to 96 000 households or 430 000 people faced a shortage in daily water use (VDDMA, 2020).

Recently, the Vietnamese government has actively developed drought management strategies such as constructing more reservoirs, enhancing irrigation efficiency, expanding protected forests and watersheds, modifying crops, and adopting farming practices to boost SOM and moisture (Thanh *et al.*, 2020). Additionally, employing predictive simulation models for rainfall has become crucial for improving drought response (Le Huong and Son, 2020).

8.5.10 Other issues

In 2018, 61 percent of Thailand's 6.1 million ha of skeletal soil was used for agriculture (LDD, 2018) (Figure 8.27). Skeletal soil, containing at least 35 percent by volume of laterite, gravel and fragmented rock, with a layer of marl clay or rock near the surface, restricts plant growth and reduces crop yields (LDD, 2018). Sandy soils, which cover 1.9 million ha (LDD, 2018), have low fertility and are prone to acidification from intense farming (76 percent used for agriculture in 2018). While cassava and sugar cane are suitable crops, sugar cane's high nutrient uptake can cause acidification. However, the natural breakdown of organic matter in sandy soils can largely offset acidity from sugarcane harvesting, even with low organic matter levels (Fujii *et al.*, 2018).

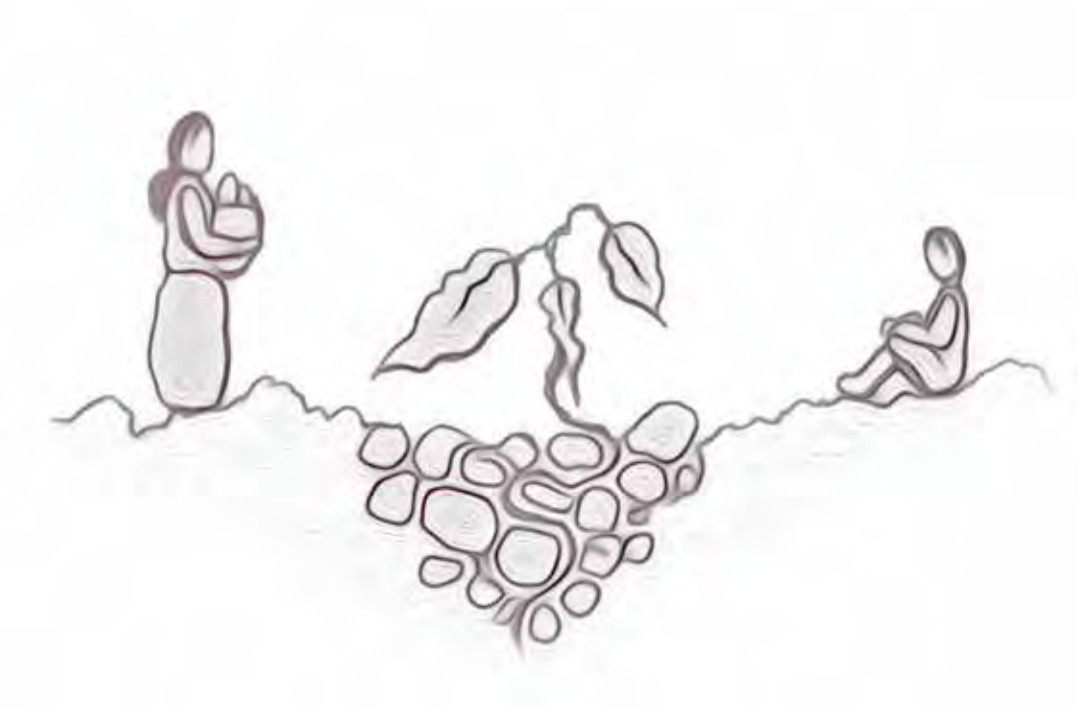
Figure 8.27. Distribution of skeletal soil areas in Thailand in 2018



Refer to the disclaimer on page ii for the names and boundaries used in this map.

◊ *Source:* **LDD**. 2018. แผนบริหารจัดการทรัพยากรดินปัญหาของประเทศไทย ระยะ 20 ปี (พ.ศ. 2561-2580) *[Thailand's 20-year soil resource management plan (2018-2037)]*. Bangkok.

Soil compaction has also been observed in the Philippines because of intensive farm mechanization being implemented in recent years. The use of agricultural machinery can damage the soil structure, compacting the soil aggregates and consequently destroying other soil properties, such as its water-holding capacity, thereby affecting the availability of water and nutrients to crops. Waterlogging in compacted areas is an unintended consequence of soil compaction.



8.6 Trends in sustainable soil management

8.6.1 Summary of state and trend in sustainable soil management

The information presented in this chapter was used by the ITPS members of this region to assess the state and trend of threats to soil functions through a management-based approach. Instead of directly measuring or modelling the physical severity of each soil threat, as was done in the 2015 SWSR report (FAO and ITPS, 2015), the 2026 report assesses the level of adoption of SSM practices known to mitigate each soil threat.

The state reflects the effectiveness of management practices in place as of 2015 in preventing or reducing a specific soil threat. At the regional or subregional scale, changes in management occur at decadal or longer time scales and hence the state as of 2015 is relevant for the 2015–2024 assessment period.

The trend indicates the direction of change in the adoption of SSM practices between 2015 and the finalization of this report (late 2024). In short, the trend reflects the how SSM adoption has progressed over time.

Table 8.21 summarizes the conditions and trends in soil management of threats to SSM for the Asian region according to their severity, with Figure 8.28 exploring the conditions and trends in more detail. Soil erosion is ubiquitous throughout the region and particularly when combined with heavy rainfall during the monsoon season in the hilly and mountainous landscapes, as well as agricultural lands of the region. Wind erosion is more pronounced in arid regions of China, India and Pakistan, whereas water erosion occurs more severely in humid regions throughout Asia.

In South and Southeast Asia, the decline in SOC is notable in areas where soil erosion and land-use change exacerbate SOC depletion, particularly the conversion of natural land to cropland. Conversely, countries like China, the Republic of Korea and Thailand showed increased amounts of SOC in croplands, with interventions such as the reintroduction of harvesting by-products into the soil, the utilization of biochar and SSM practices.

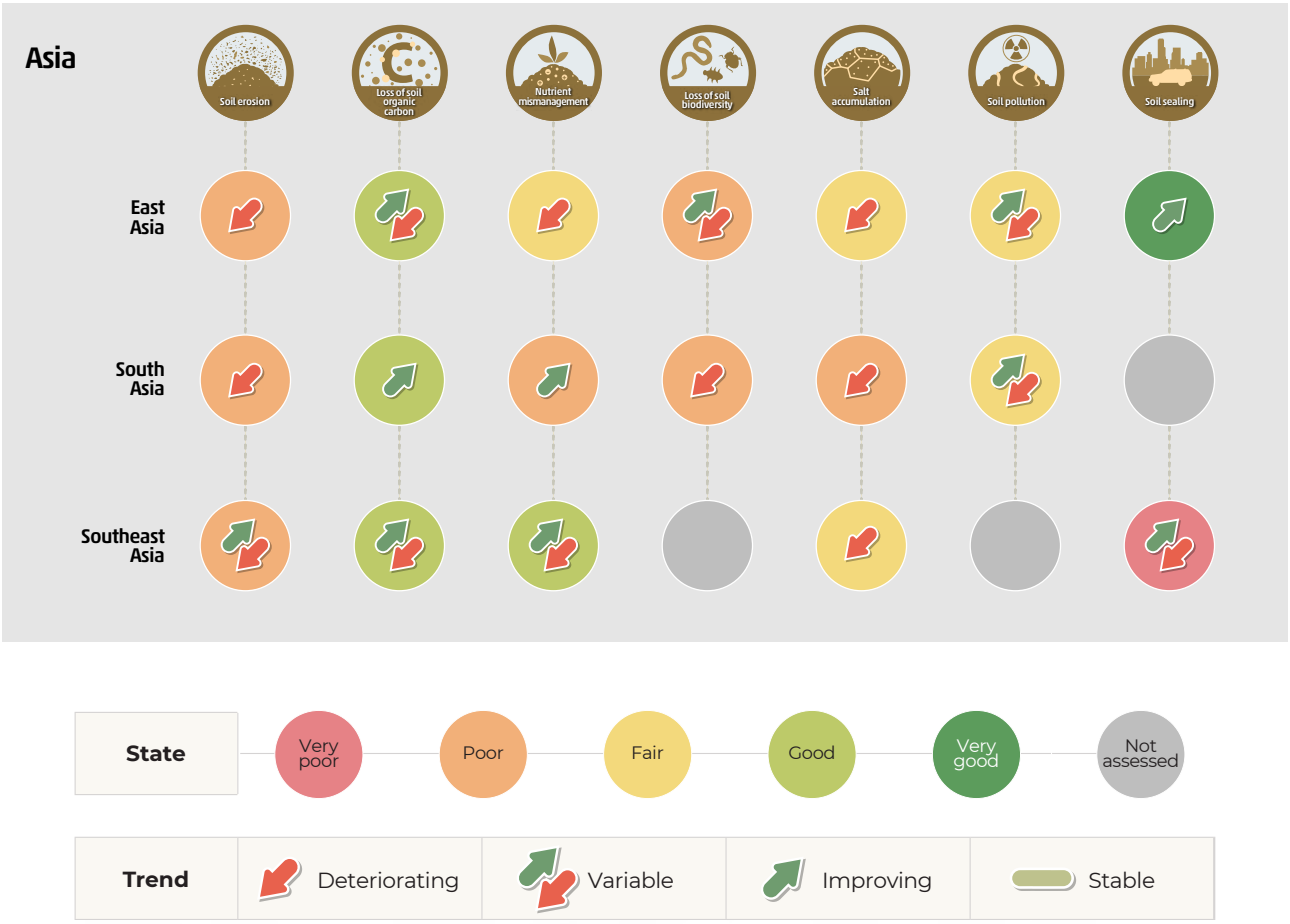
The recent increase of fertilizer prices due to global crisis such as war and the COVID-19 pandemic has caused a radical change in the Asian region, where the promotion of balanced fertilizer application is being advocated in Indonesia, the Republic of Korea and Thailand to increase crop yield and simultaneously reduce chemical soil contamination. Since 2015, the Implementation Plan for Zero Growth Action of Chemical Fertilizer Use in China is gradually reducing N and P overuse. However, the overfertilization of N and P – which can disrupt crop yield, nutrient leaching and the formation of either acidic or sodic soil environments – was indicated in Brunei Darussalam, Pakistan, Sri Lanka and Viet Nam.

Increasing sea level rise triggers saltwater intrusion and threatens coastal agriculture throughout the region. Both China and India, with the highest populations in the region, have reported the widespread threat of salinization and sodification and there is a need to strategize management guidelines to reduce the threat.

Other major threats which were identified to cause tangible losses in terms of crop yield and soil degradation are soil acidification (particularly in Southeast Asia), soil pollution in East Asia and parts of India and Pakistan, urbanization and soil sealing in China and the Philippines, and soil biodiversity change in China. Despite measures are being undertaken to mitigate these threats via liming for acidic soils, monitoring schedules for pollutants, the introduction of green infrastructure in cities, and conservation tillage for maintaining soil biodiversity, there is a need for an improvement of strategies to further sustain soil resources.

Some emerging threats are a reduced availability of soil water due to droughts and changing climates in parts of China, the Lao People's Democratic Republic, Thailand, and Viet Nam, which negatively affects crop production and the livelihood of farmers. The mechanization of land management has increased the incidence of soil compaction in cropland, leading to waterlogging. Lateritic soils and their impenetrable layer have caused plant growth restriction and yield in Thailand. The Republic of Korea is proactively establishing the need to research recycled soils for agricultural use and Japan is pioneering the research on the promotion of decontaminated soils from ^{137}Cs for alternative soil uses.

Figure 8.28. Assessment of state and trend of threats to soil functions in Asia through a management-based approach



Source: Authors' own evaluation.



Table 8.21. Summary of threats to soil functions in Asia and the confidence in the assessment of their state and trends

Threat to soil functions/sequence by relevance of threats	Confidence	
	Condition	Trend
Soil erosion. Erosion is the main threat throughout Southeast Asia, East Asia and South Asia, particularly in the mountainous landscapes and agricultural areas. Lack of soil cover in agricultural areas in East Asia and deforestation in South Asia and Southeast Asia are major drivers. Increased forest cover in China and India has reduced erosion.	**High** East Asia **High** South Asia **High** Southeast Asia	**High** East Asia **High** South Asia **High** Southeast Asia
Loss of soil organic carbon. Change of land use and soil erosion has been a major driver for SOC decline in South Asia and Southeast Asia. Increases of SOC in cropland were reported for China, the Republic of Korea and Thailand with good management practices.	**High** East Asia **High** South Asia **High** Southeast Asia	**High** East Asia **High** South Asia **High** Southeast Asia
Salt accumulation. Most salt-affected soils are found in the coastal regions due to land reclamation for agriculture and in isolated paddy fields in South and East Asia. Both China and India reported widespread threat. There is a need to develop SSM strategies and use of fresh water for cropland affected by saline soils.	**Medium** East Asia **Medium** South Asia **Medium** Southeast Asia	**Medium** East Asia **Medium** South Asia **Medium** Southeast Asia
Nutrient mismanagement. A surplus of N and P was indicated in Brunei Darussalam, China, Pakistan, Sri Lanka, and Viet Nam. Nutrient mining of N, P, K, Mg and micronutrients are evident in some parts of South Asia and East Asia. Balanced fertilizer application and promotion of organic fertilizers are being introduced in Indonesia, the Republic of Korea and Thailand to fill in yield gaps.	**Medium** East Asia **Medium** South Asia **Medium** Southeast Asia	**Medium** East Asia **Medium** South Asia **Medium** Southeast Asia
Soil acidification. Soil acidification was reported throughout in Southeast Asia due to highly weathered soils, intensive agrochemical usage and improper management of water resources. Ameliorations with liming and judicious water table control are recommended to reduce this threat.	**High** East Asia **High** Southeast Asia	**Medium** East Asia **Medium** Southeast Asia
Soil pollution. Excessive use of agrochemicals, urbanization and wastewater from industries affect China, India, Japan, Pakistan, and the Republic of Korea. Emerging pollutants such as antibiotics, microplastics and polycyclic aromatic hydrocarbons are being detected in East Asia and heavy metals and pose risk to human health.	**Medium** East Asia **Medium** South Asia	**Medium** East Asia **Medium** South Asia
Urban expansion and soil sealing. Rapid urbanization and development of mega-cities significantly increased the rate of impervious surface area in China and the Philippines. Efforts are underway in Japan and the Republic of Korea to mitigate sealing by installing green infrastructures and proper guidelines in urban planning.	**Medium** East Asia **Medium** Southeast Asia	**Low** East Asia **Low** Southeast Asia
Soil biodiversity change: Forests generally show high soil biodiversity compared to agricultural land in Asia. The importance of soil biodiversity is less studied but the main drivers that pose risks are loss of above-ground diversity, pollution and nutrient over-loading, over-grazing, intensive agriculture, soil erosion, desertification and climate change. Conservation tillage was reported to increase soil biodiversity in China.	**Medium** East Asia **Medium** South Asia	**Low** East Asia **Low** South Asia
Soil compaction. Mechanization of land management has caused soil compaction in agricultural land in China and the Philippines leading to waterlogging issues.	**Medium** East Asia **Medium** Southeast Asia	**Low** East Asia **Low** Southeast Asia

Note: **The threats are presented in the order of most severe threat (erosion) to least severe (soil compaction).**

8.6.2 Progress in and impediments to adoption of sustainable soil management

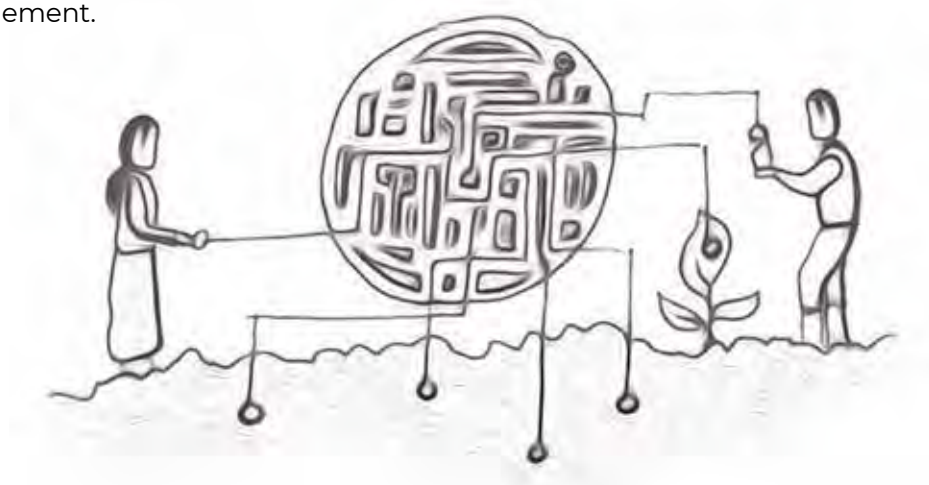
The importance of SSM in Asia has become more significant, due to its crucial role in ensuring food security, mitigating climate change and preserving ecosystem health. Several factors contribute to the progress in adopting SSM practices in the region, alongside impediments that hinder widespread implementation. India is implementing strategies for soil conservation that promote chemical-free soil application, protect soil biodiversity, enhance SOM and soil moisture, mitigate soil degradation, and prevent soil erosion. Sri Lanka introduced the Good Agricultural Practices (GAP) certification system in 2016 (Bamunuarachchi *et al.*, 2019), aimed at ensuring sustainable agricultural practices. It intentionally targeted the entire value chain, from the judicious application of fertilizers, pesticide and chemical application, and mulching (reducing soil erosion), to water quality testing, leading to the sustainable use of soil resources. The introduction of mulches and organic wastes has shown positive effects of increasing SOC in China, the Republic of Korea and Thailand. Viet Nam has actively developed drought management, with increasing reservoirs, enhancing irrigation efficiency and adopting farming practices to boost SOM and moisture (Thanh *et al.*, 2020).

The current Global Soil Doctors Programme developed by FAO has also become crucial for countries like Thailand and Malaysia who are engaging farmers at the grassroot level and teaching SSM practices (FAO, 2025c).

Some of the highlighted impediments related to SSM include the challenge to get farmers involved in the soil monitoring process in their farms after implementing new practices. Their commitment to adopt and adapt to SSM tools varies, depending on farm conditions, soil types, environmental factors, crop type, gender inclusivity, age, socioeconomic status, government policies and the technical knowledge on SSM practices. There are also issues related to logistics, operational costs, risks, and uncertainties related to projected yieldΩ.

8.6.3 Recommendations for future priorities for action

The sustainable management of land and water resources is direly needed in the Asia region to increase the food security requirements in the region and beyond. The livelihood of farmers needs to be reassessed and given priorities such as incentivization, the introduction of artificial intelligence for smart farming, the promotion of awareness about SSM tools with a real-time support system, and concerted efforts at local and regional scale. With the conflicting interests of soils for industrialization, agriculture, environment and the ever-growing effects of climate change, strategies for a balanced use of soil are much needed. Policymakers should work hand-in-hand with research and development authorities in finding ways to increase land productivity, reverse land degradation and pollution, use water resources sustainably, enhance environmental health, promote agriculture and natural resources studies, and natural disaster risk reduction planning. Development of a regional implementation plan may be the best step forward to safeguard the future prospects of soil management.



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Chapter 9 | Regional assessment for Europe and Central Asia

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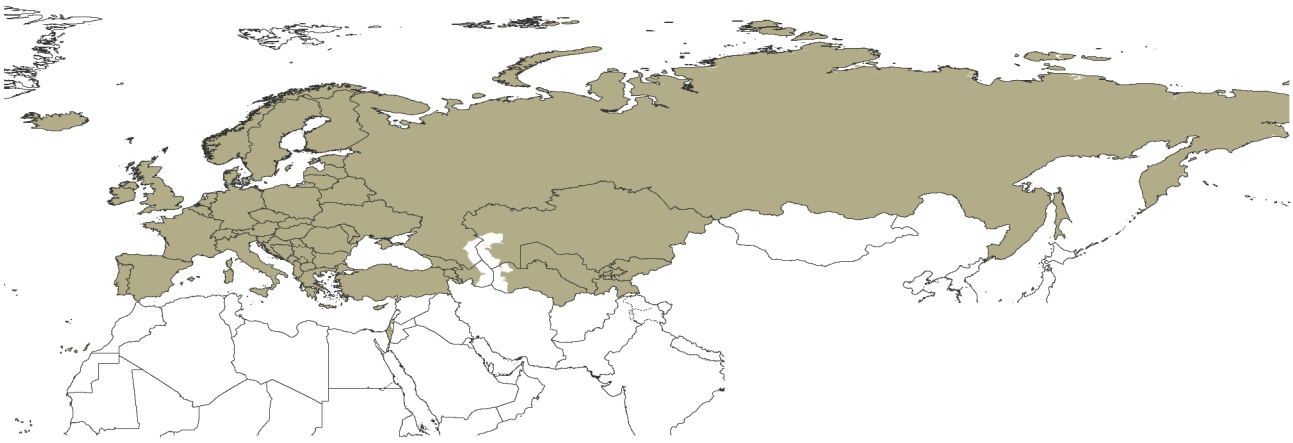


9.1 Regional overview

9.1.1 Characteristics of the region

The Europe and Central Asia region covers 50 countries, as considered in this report and since the establishment of the European Soil Partnership in 2013 (Figure 9.1). Thus, the region includes a wide diversity of bioclimatic and soil resources, as well as land uses, characterized by a mosaic of ecosystems, from the Mediterranean in the south, to the Arctic in the north, with a heterogeneity of soil types and land-use patterns. Moreover, the legacy of ancient civilizations and modern nation states is deeply intertwined with the landscape. Traditional farming methods coexist with modern agro-industrial complexes, reflecting a blend of heritage and innovation in soil management.

Figure 9.1 Map of the countries of Europe and Central Asia region



Eleven natural biomes are recognized in the Europe and Central Asia region (Figure 9.2). The natural biomes are strongly related to latitude and are discussed in the following sections. The main Reference Soil Groups (RSGs) (IUSS Working Group WRB, 2022) indicated in each biome are taken from the *Soil Atlas of Europe* (Jones, Montanarella and Jones, 2005) and the *Soil Atlas of Asia* (EC, JRC and FAO, 2023).

The *polar and montane zones* extend over the highest latitudes and high mountains, occupied by a treeless ecosystem. Although the demographic pressure is low, some areas are affected by mining and petroleum extraction, and climate change is altering the thermal regimes of the soil, causing permafrost melting (Smith *et al.*, 2022). The main soils are Cryosols (with permafrost within the top 1 m of soil) and Retisols.

The *boreal forest/taiga zone* spans vast areas at northern latitudes, characterized by low mean annual temperatures, ranging from just above freezing to as low as −10 °C or colder. This biome is predominantly covered by coniferous forests and experiences minimal agricultural pressure. It is supported by a variety of RSGs, including Gleysols, Retisols, Histosols and Podzols, with some occurrences of Andosols, particularly in Iceland.

The *temperate forest zone* extends across the region from the British Isles to Western Siberia and includes deciduous, coniferous and mixed forests. Most of the area is under cultivation or supports industrial land use and urbanization, although there are also large areas occupied by deciduous forests. Cambisols, Chernozems, Phaeozems, Retisols and Luvisols are the main soils of this large area. Temperate coniferous forests occur in coastal areas and islands with warm summers and cool winters. The natural vegetation consists of a mix of needle-leafed and broadleaved evergreen trees, or a combination of both, with lower anthropogenic pressure compared to surrounding regions. This biome exhibits high diversity in soil types due to its fragmented distribution, but Leptosols and Andosols are among the most representative soil types.

The montane forests and shrublands zone has similar vegetation and soils to the temperate coniferous forests.

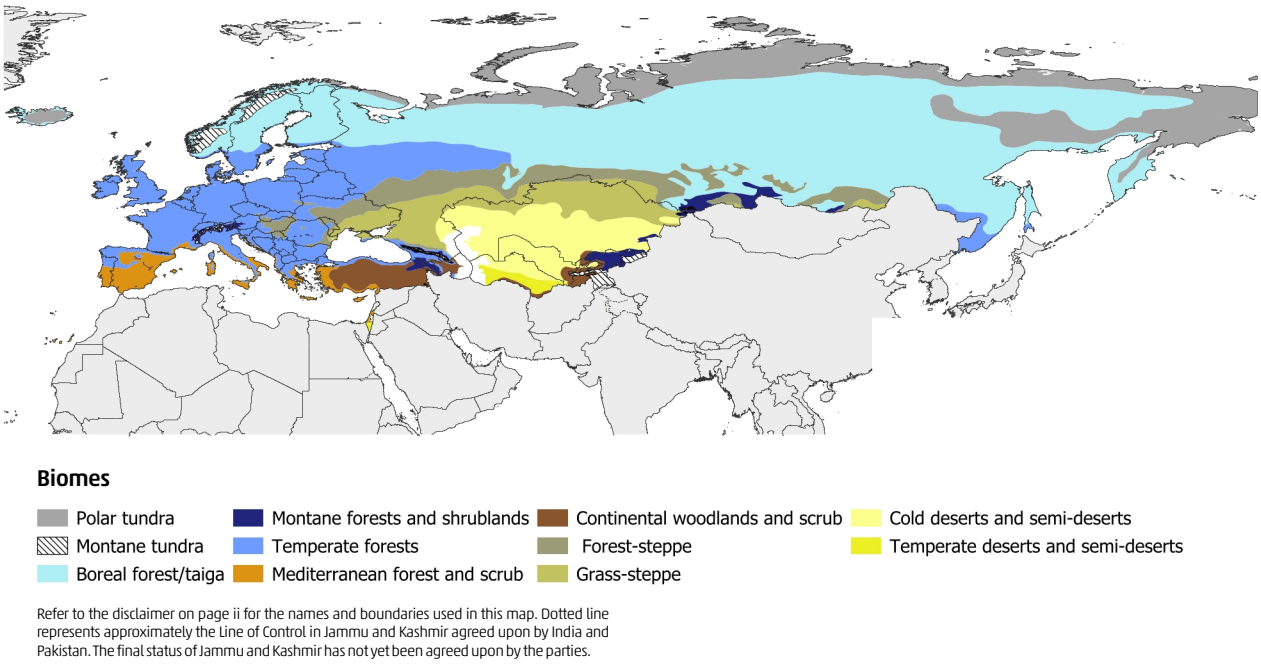
The *forest-steppe and grass-steppe zones* have large areas of black soils, comprising of Chernozems, Kastanozems and Phaeozems, although other groups such as Luvisols and Cambisols are also found which are subject to intensive agricultural use. The main threats to SSM are water and wind erosion leading to the loss of soil, and thus OC. Other issues are high population density, urban expansion and industrial development, leading to soil surface sealing and contamination, with soil salinization possibly occurring, especially in endorheic valleys.

The *Mediterranean forest and scrub zone* is characterized by hot, dry summers and cool, moist winters, with precipitation occurring mainly in winter. Xerophytic natural vegetation has to compete with high population pressure, a long history of agricultural land use and fire. A dry climate and shallow limestone soils make ecosystems and soils vulnerable, leading to erosion, carbon loss and biodiversity decline. Soil salinity can also limit agriculture in some areas. Calcisols, Cambisols, Leptosols and Regosols are the most important soils, covering large areas.

The *sub-arid and arid zone* encompasses both the *desert and semi-desert zone* and *the continental woodland and scrub zone,* primarily in Central Asia but also in Anatolia (Asia Minor) and Israel. Vegetation is sparse, and desertification affects all ecosystems. Soil threats include salinization, sodification and wind erosion, with irrigation-induced salinization posing the most significant risk to soil fertility. A wide variety of soils, including Kastanozems, Cambisols, Calcisols, Leptosols, Regosols, Arenosols and salt-affected soils (Solonchacks and Solonetzs) are the main soils in this biome.



Figure 9.2. Terrestrial ecoregions of the European region



◊ *Source:* **Loidi, J., Navarro-Sánchez, G. & Vynokurov, D.** 2023. A vector map of the world's terrestrial biotic units: subbiomes, biomes, ecozones and domains. *Vegetation Classification and Survey*, 4: 59–61. https://doi.org/10.3897/VCS.99167

9.1.2 Summary of status from the 2015 Status of the world's soil resources report

The 2015 SWSR report (FAO and ITPS, 2015) highlighted significant differences in threats to soil functions between Europe and Eurasia. Overall, approximately half of the threats were assessed as being in poor condition, while the other half showed signs of improvement.

In Europe, soil sealing was considered the most serious threat to soils. The countries of Europe are highly urbanized and pressure on forest and agricultural land is high and still increasing, since urban sprawl occurs largely at the cost of arable land. Europe was the only global region where urban expansion and soil sealing was judged to be a top concern in 2015.

Salinization and sodification were judged to be the most serious threats in the Central Asian countries as well as in areas of Hungary, the Russian Federation, Spain and Türkiye.

Soil contamination was a serious threat to soil functions and human societies in Europe. In Europe, many of the contamination problems were legacy issues associated with the long history of industrialization in the region, and the situation was improving in many areas. However, pollutants were widespread over large parts of agricultural land, where metals, organic pollutants and some emerging contaminants were further accumulating. In Eurasia, the legacy of the Soviet Union had been the widespread contamination from many sources in several of the former Soviet republics (FAO and ITPS, 2015).

Loss of SOC was the fourth-ranked threat and was associated in part with widespread losses of SOC from cropland throughout the region, particularly in managed organic soils. Gains of SOC were noted in some areas of the former Soviet Union due to the abandonment of agricultural land that had occurred. Peatland drainage and utilization of peat as a fuel and soil amendment had led to high losses of SOC in some countries.

Excessive nutrient inputs in some countries of Western Europe were noted, along with significant nutrient mining in some countries of Eastern Europe and Eurasia (FAO and ITPS, 2015).

9.2 Changes in the drivers of threats to soil functions

9.2.1 Overall population and urban population growth

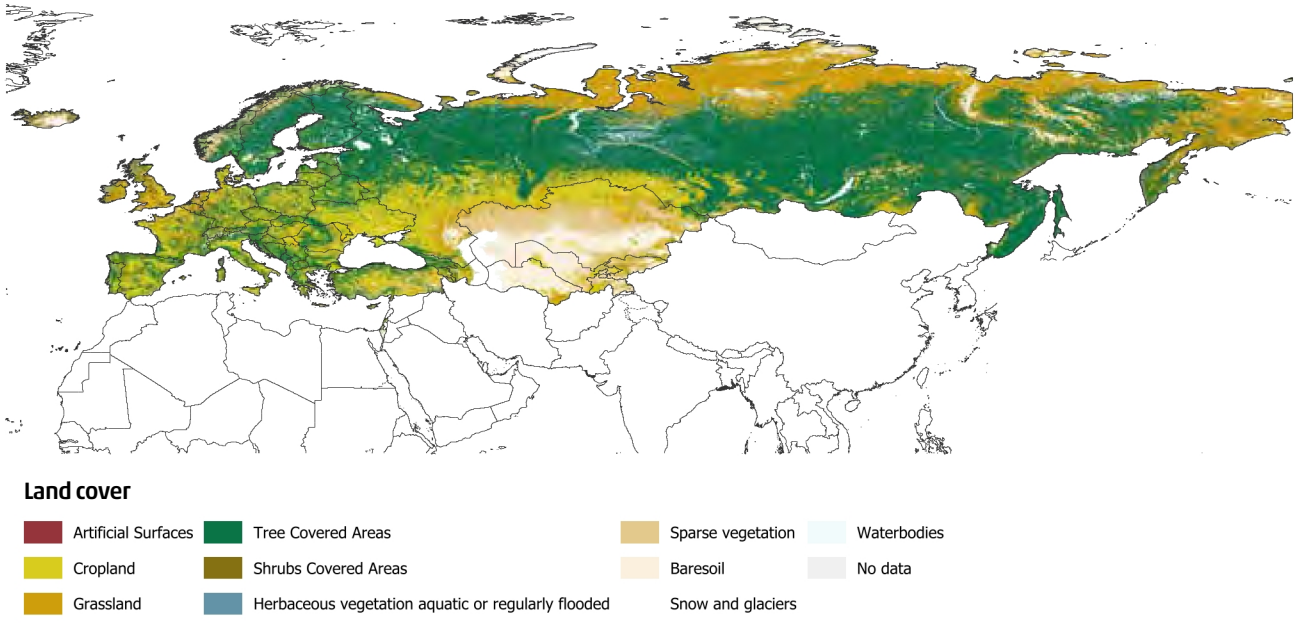
Europe and Central Asia have projected population increases below the global mean through to 2035 (UN DESA, 2023). Central Asian countries are projected to have higher population growth rates in the region, with Kyrgyzstan, Tajikistan and Uzbekistan having the highest growth rates overall. Half of the countries in the region are projected to have small decreases in population through to 2035.

Generally, the countries in the eastern part of the region have a lower percentage of the total population residing in urban areas than those in the west of the region in 2015 (UN DESA, 2018). The countries in the west of the region are highly urbanized with projected lower rates of growth in their urban populations through to 2035 and beyond to 2050. The highest growth rates in Europe and Central Asia region are projected to occur in Albania, Bosnia and Herzegovina, Kyrgyzstan, Tajikistan and Turkmenistan.

9.2.2 Land-use changes

Cropland in Europe and Central Asia is widely distributed throughout Western and Central Europe and in the southern areas of Eastern Europe and Central Asia (Figure 9.3). The cropland area in Eastern Europe and Central Asia grades in grasslands to the south of the cropland zone (Figure 9.3).

Figure 9.3. Land cover in Europe and Central Asia

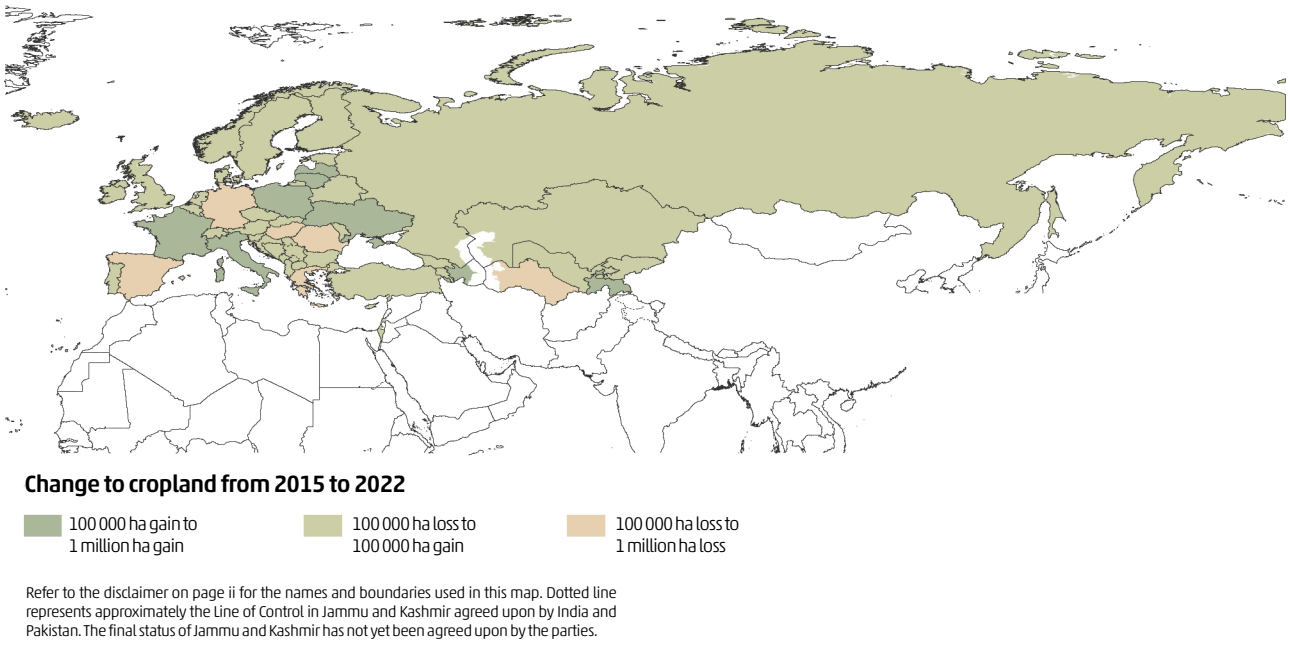


Refer to the disclaimer on page ii for the names and boundaries used in this map. Dotted line represents approximately the Line of Control in Jammu and Kashmir agreed upon by India and Pakistan. The final status of Jammu and Kashmir has not yet been agreed upon by the parties.

◊ *Source*: **FAO.** 2024. *FAOSTAT: Land Use.* [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

Most countries only experienced small changes in agricultural and cropland area between 2015 and 2022 (FAO, 2024a) (Figure 9.4). The largest percentage increases in cropland occurred in Azerbaijan (8.8 percent), Israel (20.5 percent), Latvia (10.6 percent), and Tajikistan (11.3 percent) (FAO, 2024a). The largest percentage decreases were in Greece (13 percent decrease) and Malta (–14.5 percent decrease) (FAO, 2024a).

Figure 9.4. Change in area of cropland in Europe and Central Asia between 2015 and 2022

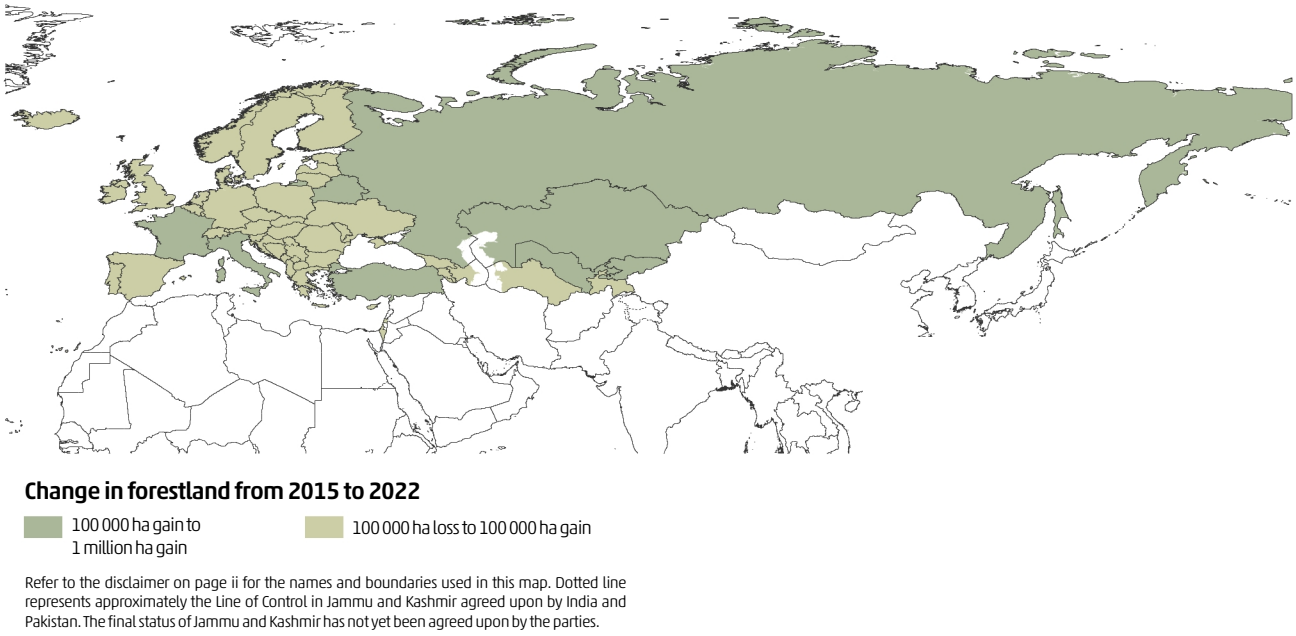


Source: **FAO.** 2024. *FAOSTAT: Land Use.* [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

Most countries in Europe and Central Asia experienced reforestation rather than deforestation in the 2015–2022 period (FAO, 2024a) (Figure 9.5), with the highest areal increases in forestland in Türkiye (902 000 ha gain), France (584 000 ha gain), the Russian Federation (381 000 ha gain), Italy (377 000 ha gain), Kazakhstan (205 000 ha gain) (FAO, 2024a). Only Hungary (–11 000 ha loss), Israel (–25 000 ha loss) and Slovenia (–14 000 ha loss) experienced significant deforestation in the same period (FAO, 2024a).



Figure 9.5. Change in area of forestland in Europe and Central Asia between 2015 and 2022



Source: **FAO.** 2024. *FAOSTAT: Land Use.* [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

Changes in the area of permanent meadows and pastures of Central Asia were highest in Kazakhstan and Turkmenistan in the 2015 to 2022 period. Kazakhstan had a loss of 2.1 million ha, while Turkmenistan had a net gain of 6.1 million ha (FAO, 2024a). Overall, most countries of Eastern Europe, and France, Italy and the United Kingdom had losses of permanent meadows and pastures and only Spain and Israel had substantial increases (Figure 9.6) (FAO, 2024a).

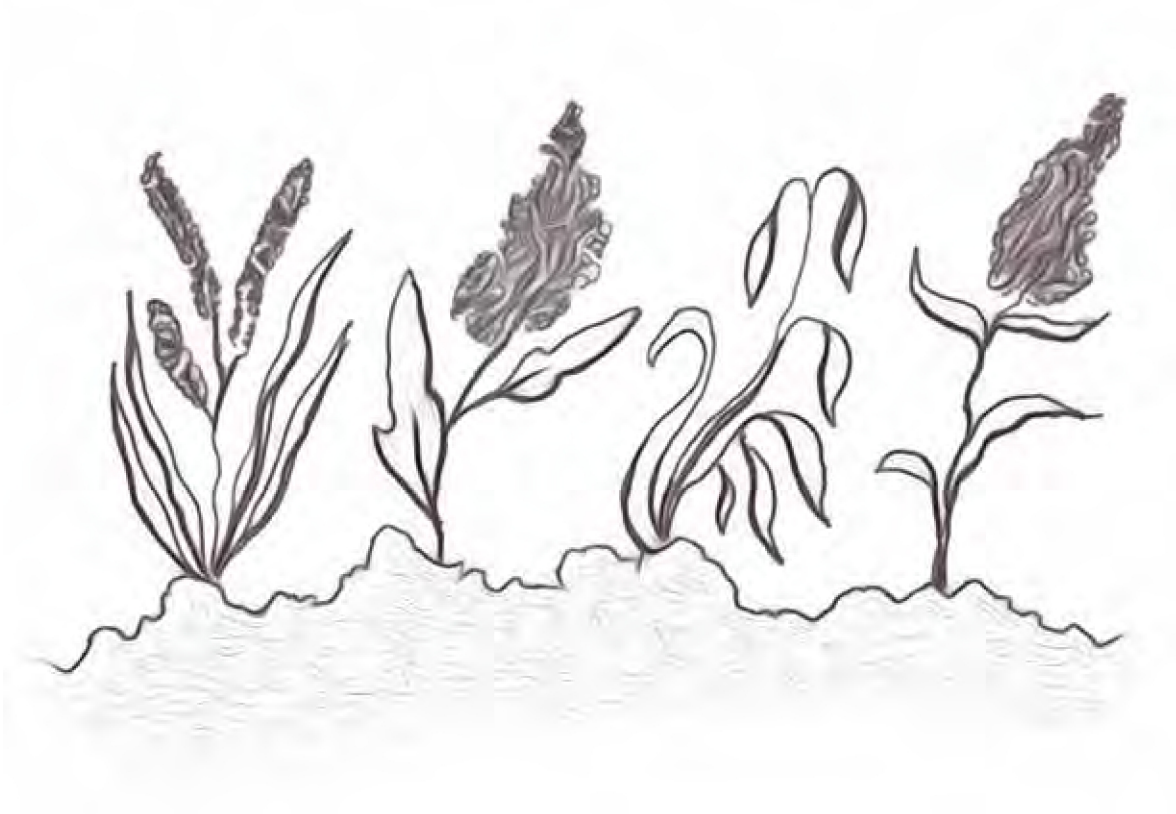
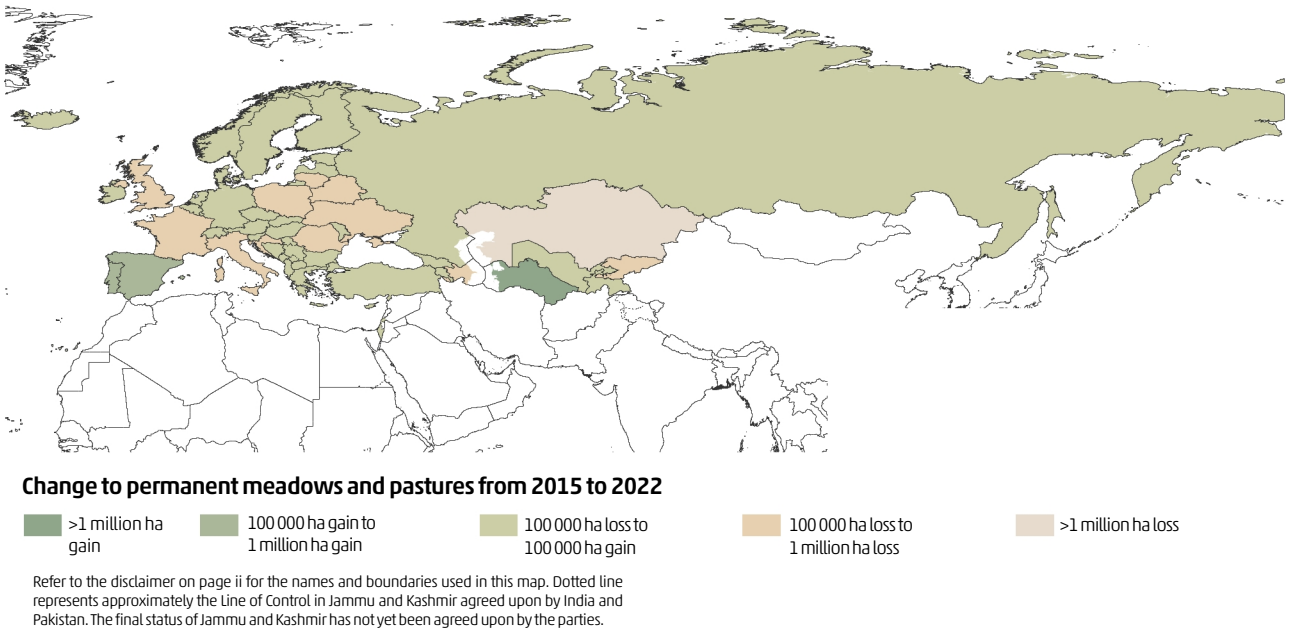


Figure 9.6. Change in area of permanent meadows and pastures in Europe and Central Asia between 2015 and 2022

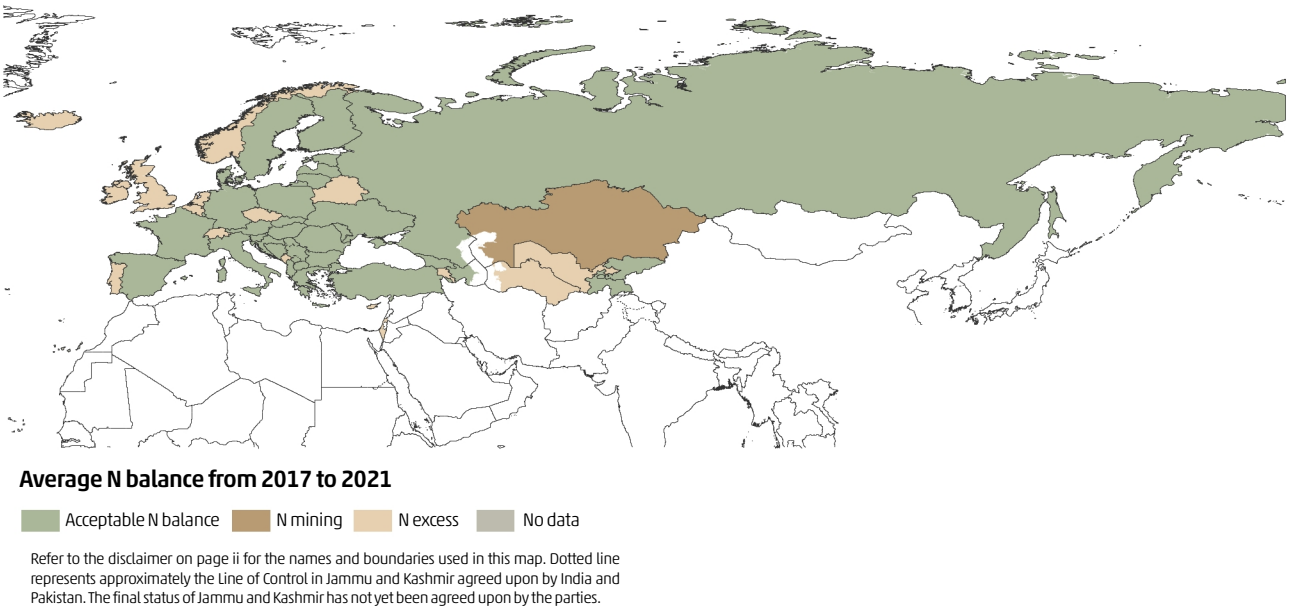


◊ *Source*: **FAO.** 2024. *FAOSTAT: Land Use.* [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

9.2.3 Nutrient management

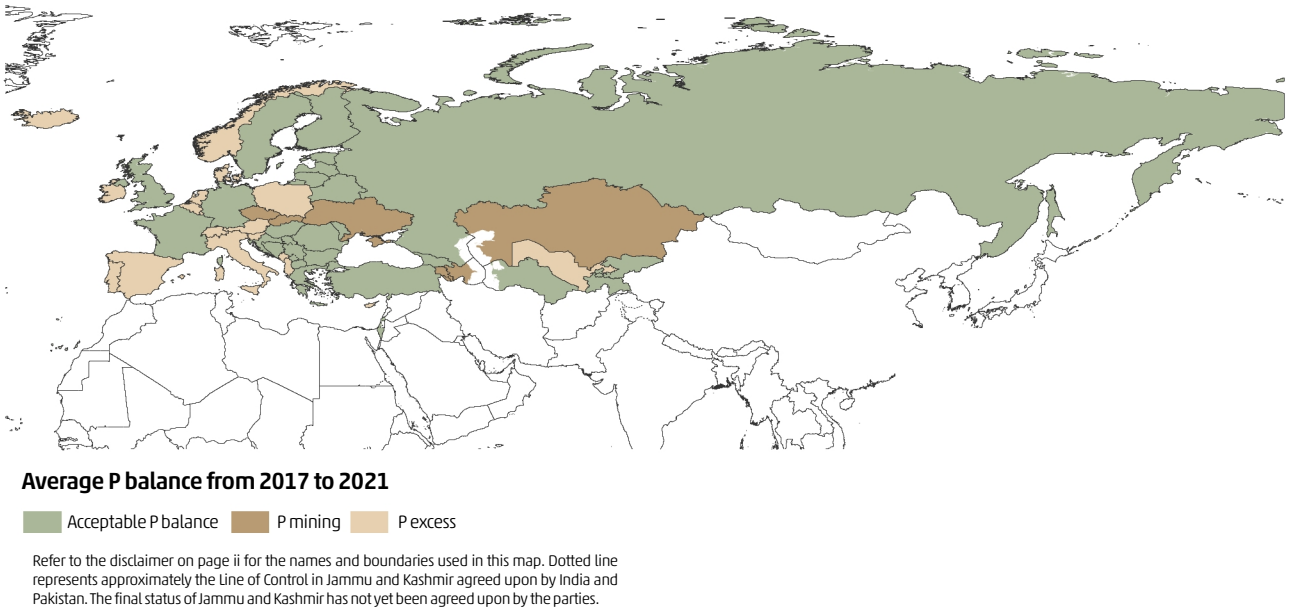
Many countries of Europe and Central Asia have nutrient imbalances in agricultural soils, to some extent affected by excessive fertilizer inputs and atmospheric deposition. Nutrient mining occurs in some parts of the region (FAO, 2024b) (see Section 3.5.1 for the criteria used to identify nutrient mining and excess). Nineteen countries have excessive N inputs (Figure 9.7) and nineteen have excessive P inputs (Figure 9.8), while twelve countries (Belgium, Cyprus, Denmark, Iceland, Ireland, Malta, Montenegro, the Kingdom of the Netherlands, Norway, Portugal, Switzerland, and Uzbekistan) have excess inputs of both (FAO, 2024b).

Figure 9.7. Average nitrogen balance for 2017–2021 for Europe and Central Asia



◊ *Source*: **FAO**. 2024. *FAOSTAT: Fertilizers by Nutrient*. [Accessed on 9 January 2024]. https://www.fao.org/faostat/en/#data/RFN. Licence: CC-BY-4.0.

Figure 9.8. Average phosphorus balance for 2017–2021 for Europe and Central Asia

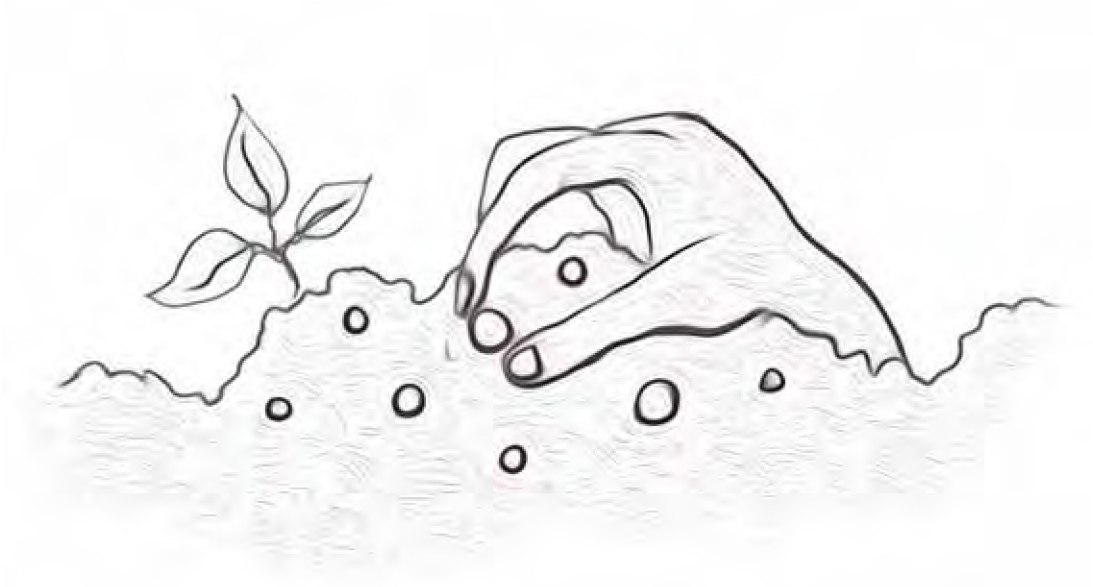


Source: **FAO.** 2024. *FAOSTAT: Fertilizers by Nutrient.* [Accessed on 9 January 2024]. https://www.fao.org/faostat/en/#data/RFN. Licence: CC-BY-4.0.

Nutrient mining occurs in some countries (FAO, 2024b) (Figure 9.7 and 9.8). Kazakhstan has balances of N and P indicative of nutrient mining (Figure 9.7 and Figure 9.8) (FAO, 2024b). Ukraine, Armenia and Azerbaijan have balances of N and P indicative of nutrient mining (FAO, 2024b).

In total, twenty countries of the region have N and P balances within acceptable limits (Figure 9.7 and Figure 9.8) (FAO, 2024b).

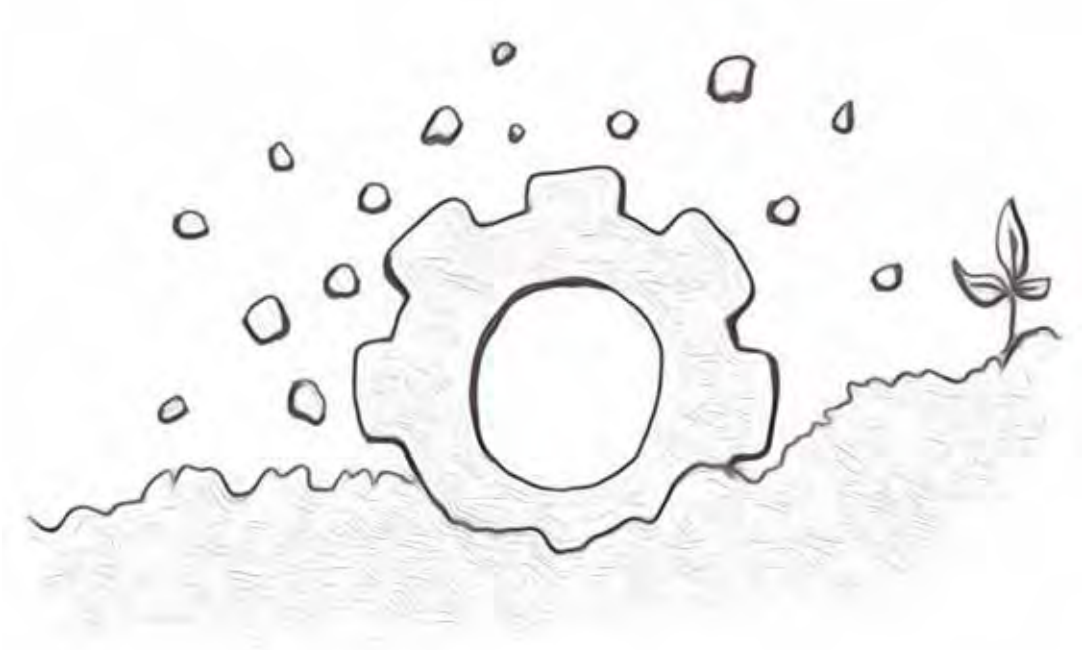
Armenia, Azerbaijan, Estonia, Türkiye and Ukraine had average negative K balances in the 2017–2021 period, which are possible indicatations of K mining.



9.3 Regional status and trend of threats to soil functions for Europe and Central Asia

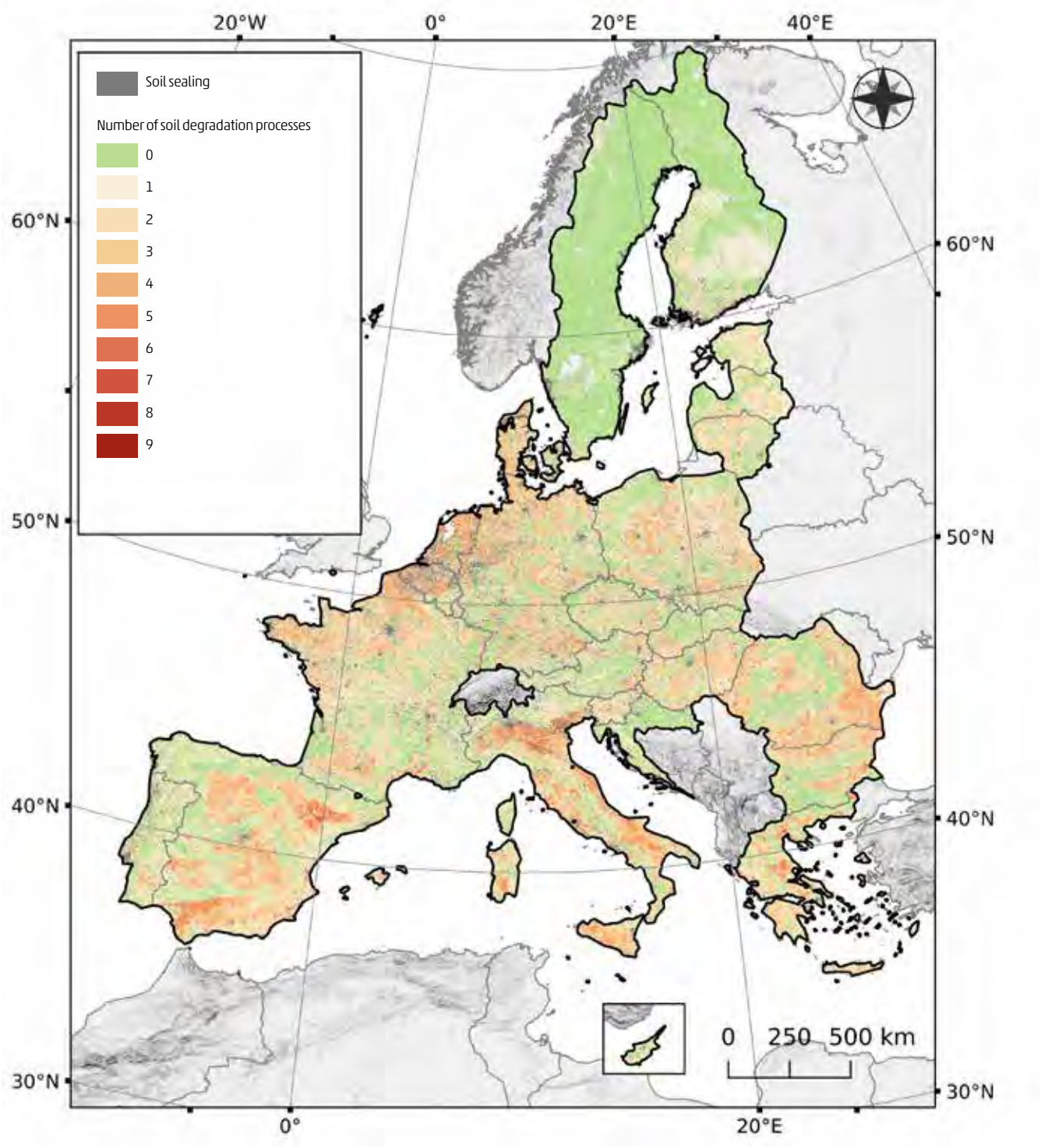
9.3.1 Introduction

Soil threats in Europe and Central Asia are complex due to diverse gradients of climate, land use and parent material. This uneven spread of pressures and soil health is determined by the large diversity of biophysical conditions, as well as varying socioeconomic and political conditions. Some countries have soil monitoring schemes in place, but they are fragmented and not harmonized. The European Environment Agency (EEA) concluded in its *State of the Environment 2020* report that "soil degradation is not well monitored, and often hidden, but it is widespread and diverse"(EEA, 2020a, p. 115). The lack of comprehensive and harmonized data on soil health has resulted in a lack of consistency and comparability between countries (Bispo *et al.*, 2021). In 2023, the Soil Degradation Dashboard[1] of the European Union Soil Observatory (EUSO) (JRC, 2023) provided a unique spatial assessment of degraded soils for the European Union, and which degradation processes were responsible for their condition (Figure 9.9). The approach serves as a role model for other countries of the European Soil Partnership. Harmonised soil datasets from the European Soil Data Centre (ESDAC) (Panagos *et al.*, 2022), as well as from the EEA and other institutions, provide a view of the state of soil health across the European Union. The novel use of a "convergence of evidence" approach – which spatially combines datasets to highlight the intensity and location of soil degradation processes (urban expansion being a different class) – provides a new interpretation. Over the European Union's 4.2 million km^2, the degradation process with the largest spatial extent is the loss of soil biodiversity, followed by soil erosion by water and SOC losses (Panagos *et al.*, 2024). The overlay of all soil degradation processes has resulted in the estimation that 62 percent of soils in the European Union are "unhealthy". In relation to the intensity of soil degradation, 24 percent of European Union soils are subject to a single degradation process, 16 percent to two degradation processes, 10 percent to three degradation processes, 5 percent to four or more degraded processes and 7 percent are sealed by urban and infrastructure development (Figure 9.9).



[1] The Soil Degradation Dashboard is a spatial assessment of where unhealthy soils may be located in the European Union, and which degradation processes affect them. It uses a convergence of evidence methodology, which spatially combines datasets to highlight the intensity and location of 15 soil degradation processes (covering the following topics: soil erosion, soil pollution, nutrients, loss of soil organic carbon, loss of soil biodiversity, soil compaction, soil salinization, the loss of organic soils and soil sealing).

Figure 9.9. Soil degradation across the European Union using the Soil Degradation Dashboard



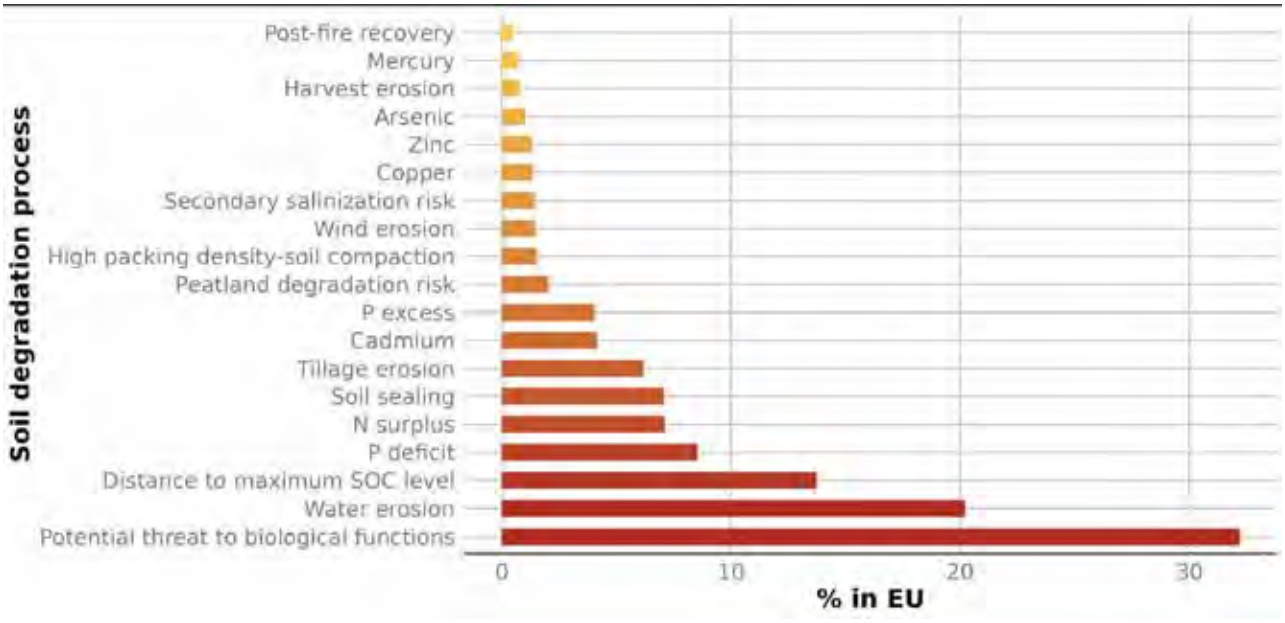
Refer to the disclaimer on page ii for the names and boundaries used in this map

◊ *Source:* Adapted from **EC.** 2024. EUSO Soil Degradation Dashboard. In: *ESDAC*. Ispra, Italy, JRC. [Cited 19 August 2024]. https://esdac.jrc.ec.europa.eu/esdacviewer/euso-dashboard/

The extent of soil degradation (Figure 9.10) (Panagos *et al*., 2024) is likely to be an underestimate, given the lack of data on many other soil degradation issues, such as soil contamination or subsoil compaction, and that some specific degradation processes of forested soils and peatlands are not fully included. In addition, the map shows that most of the degraded soils are subject to more

than one soil degradation process. The EUSO Soil Degradation Dashboard will be enriched with additional indicators as new evidence becomes available and expanded to include data on countries beyond the European Union.

Figure 9.10. Percentage of unhealthy soils in the European Union, by soil degradation process



◊ *Source*: **Panagos, P., Broothaerts, N., Ballabio, C., Orgiazzi, A., De Rosa, D., Borrelli, P., Liakos, L.** *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

The total surface area of Azerbaijan is 8.66 million ha, with 4.78 million ha being used for agriculture. Of that, 1.89 million ha is cropland, including 1.45 million ha under irrigated cropland (RASCPI, 2019). The land reform that started in Azerbaijan in 1997 has led to changes in land tenure. At present, the land belongs to either the state (57 percent), municipalities (23 percent) or private landowners (20 percent) (Ismayilov, 2013). The transfer of land to private ownership, and the establishment of private farms has transformed the attitude towards soil, with greater care being shown in soil cultivation and the use of soil resources. As a result, the processes of soil degradation, salinization and erosion have started to decrease in privately owned lands (Ismayilov, 2013).

The Russian Federation has 197.7 million ha of agricultural land (MARF, 2022). From 2017 to 2022, 51.6 million ha (or 26 percent of the total agricultural land), has been sampled under a soil monitoring programme, with coverage varying from 13.5 to 42.6 percent of each administrative region (Figure 9.11) (Ministry of Agriculture of the Russian Federation, 2024). Four key properties were measured: soil organic matter, pH, and available P and K.

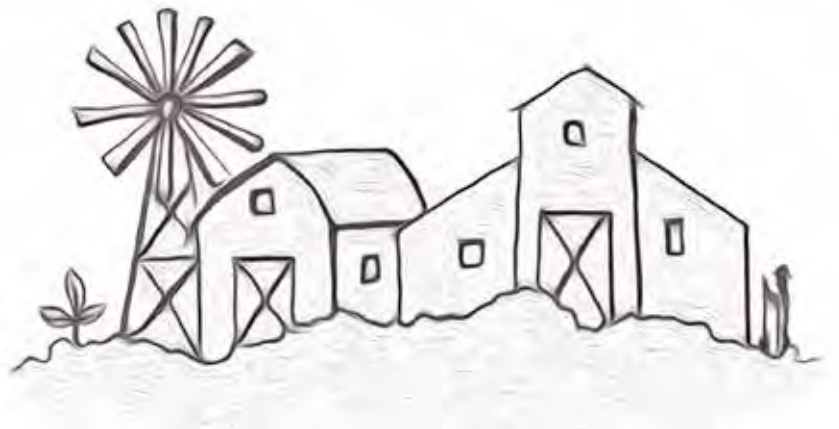
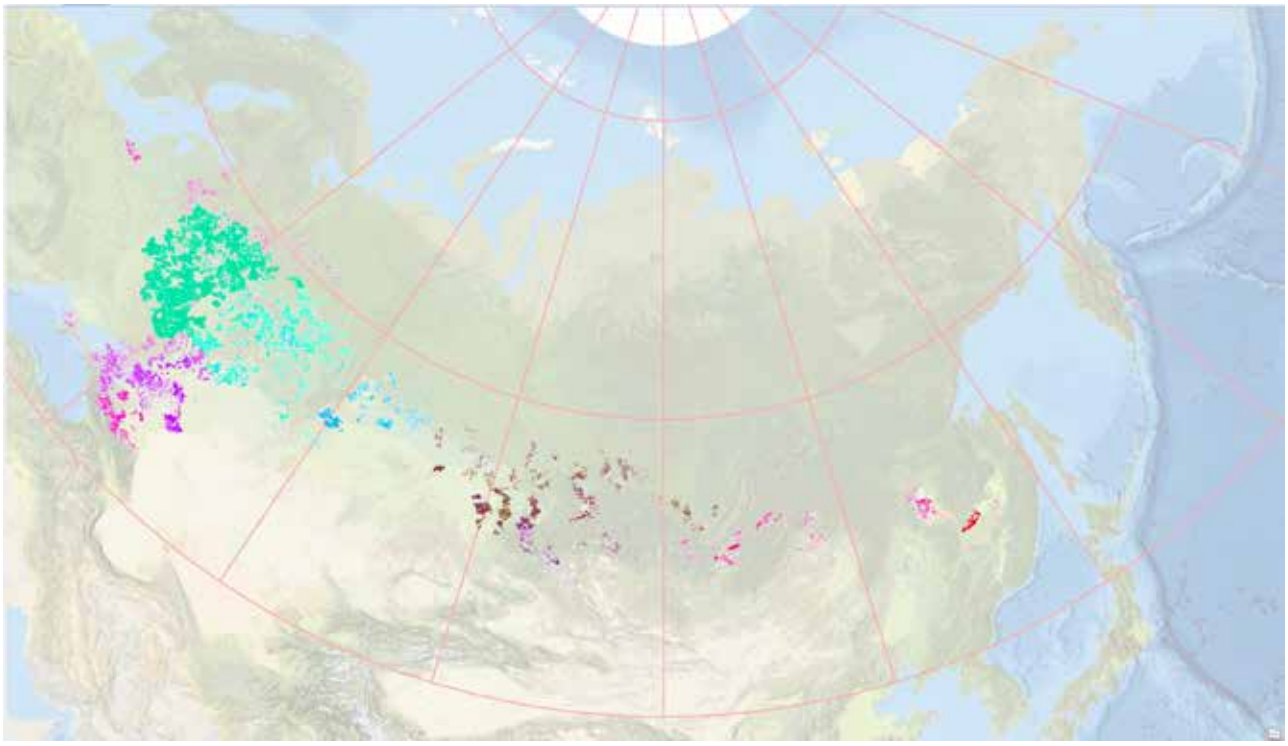


Figure 9.11. The distribution of soils of agricultural lands sampled within the 2017–2022 period in the Russian Federation



Refer to the disclaimer on page ii for the names and boundaries used in this map

Note: The colours depict different administrative regions of the Russian Federation.

◊ *Sources:* Authors' own elaboration based on Esri, USGS. National Geographic Style Base. https://tiles.arcgis.com/tiles/P3ePLMYs2RVChkJx/arcgis/rest/services/NatGeoStyleBase/MapServer

◊ **Ministry of Agriculture of the Russian Federation.** 2024. *EFIS ZSN (Edinaya Federalnaya Informatsionnaya Sistema o Zemplyah Selskokhozyaistvennogo naznachaniya [Unified Federal Information System on Agricultural Lands]*. Moscow. https://efis.mcx.ru/landing/

In Kazakhstan, 213.7 million ha, (78.6 percent of the total area of the country) is agricultural land. Over 75 percent of agricultural lands are affected by salinization and sodification, soil erosion, and stoniness (MARK, 2021) (Table 9.1)

Table 9.1. Soil properties that are limiting for agriculture and contribute to soil degradation in agricultural lands of Kazakhstan

Negative property	Area (ha)	%
Stony and pebbly	42 156 300	19.7
Saline	35 449 900	16.6
Sodic	58 164 900	27.2
Water erosion	4 950 300	2.3
Wind erosion	24 168 100	11.3
Water and wind erosion	201 700	0.1
Waterlogged	2 847 600	1.4
Peaty	1 083 400	0.5
Others	3 163 600	1.5
No negative properties	41 516 800	19.4
Total agricultural lands	213 702 600	100

◊ *Source*: **Ministry of Agriculture of the Republic of Kazakhstan**. 2021. *Summary Analytical Report on the Status and Use of Lands in the Republic of Kazakhstan in 2021*. Nursultan. https://www.gov.kz/uploads/2022/4/11/b09469de9be9cc54d2cc0e9cc7a77e84_original.7131188.pdf

As is the case in other regions, armed conflicts have historically impacted the soils of Europe and Central Asia, and continue to do so. These conflicts are discussed in more detail in Chapter 3 of this report.

9.3.2 Soil erosion in Europe and Central Asia

Continental-scale assessments provide a broad overview of soil erosion patterns across large geographical areas, offering insights into regional trends and identifying- hotspots of erosion. Borrelli *et al.* (2023) modelled soil erosion across the 27 European Union (EU27) member states and the United Kingdom by estimating the gross soil displacement caused by factors such as water, wind, tillage and crop harvesting. Unsustainable annual soil erosion rates of >2 t/ha (according to the criteria established by Panagos *et al.* [2021]), were estimated to occur on over half of the European Union and the United Kingdom arable land (54 percent or about 55 million ha). Additionally, approximately 13 percent had severe erosion (>10 t/ha annually). The economic loss due to the decline of land productivity (onsite effects) caused by water erosion is estimated at about EUR 1.25 billion per year in the European Union (Panagos *et al.*, 2018).

National circumstances and monitoring data should be considered when establishing threshold values. National data can serve as a validation tool for continental-scale assessments, helping to identify- potential discrepancies or areas where further investigation is needed. For instance, there is a marked disparity in erosion rates between Southern Europe and central Northern Europe (Table 9.2). Given the disparity, there are questions about the appropriateness of a uniform annual threshold (such as 2 t/ha) for defining soil erosion across regions. With such a low threshold in place, all Southern Europe would be classified as severely degraded, even though erosion may have occurred over centuries due to long-standing land-use practices. The issue of thresholds is further discussed in EEA (2023a).



Table 9.2. Status and trend of wind and water erosion in European Union member states

Country	Quantification		Specification	Trend	Source
Austria	2.8 t/ha annually		2017/18 (mean for all agricultural land).		Schmaltz *et al.* (2023)
Belgium	>5 t/ha annually		2015 (44% of arable land).		SPW (2017)
Bulgaria	3.3 t/ha annually		2018 (mean for arable land).		Özşahin and Eroğlu (2019)
Croatia	1 489 604 ha		2008 (46% of arable land moderately high risk).		Husnjak *et al.* (2008)
Cyprus	11.8 t/ha annually		2015 (mean for arable land).		Karydas and Panagos (2016)
Czechia	4.8 t/ha annually		National mean (7.1 t/ha annually for arable land alone).		Žížala *et al.* (2021)
Denmark	<3 t/ha annually		2003 (national mean).		Schjønning, Heckrath and Christensen (2009)
Germany	>3 t/ha annually		2007 (14% of arable land).		Lehmphul (2015)
Greece	700 000 ha		2015 (50% of arable land).		Koutalakis *et al.* (2015)
Finland	0.1–1.3 t/ha annually		Arable land erosion rate at municipality level.		Räsänen *et al.* (2023)
France	21.4 ± 3.1 t/ha annually		Case study: Mercurey vineyards (Burgundy, France).		Fressard, Cossart and Chaize (2022)
Hungary	8.74 t/ha annually		1990 to 2018 (with the exception of the period 1990–2000), the potential erosion has been continuously reduced. At least 16% of Hungary has an erosion rate higher than 5 t/ha annually.		Kertész and Gergely, (2011) and Waltner *et al.* (2020)
Italy	8.4 t/ha annually		National mean; Calabria: mean erosion in the regional territory is 30.6 t/ha annually (~13% is subject to very severe erosion (75–300 t/ha annually).		Terranova *et al.* (2009)
Latvia	430 000 ha		17% arable land.		Ministry of the Environment of the Republic of Latvia (2011)
Malta	6 102 ha		19% of natural land is at high risk (>10 t/ha annually).		Sultana (2015)
Netherlands (Kingdom of the)	194 875 ha		19% of arable has moderate to high erosion risk (wind). About 600 000 ha are potentially vulnerable for water erosion (slope >2%), where actual erosion depends on land use.		Hessel, Stolte and Riksen (2011)

Country	Quantification		Specification	Trend	Source
Norway	1 million ha		All arable land is at risk for erosion.		OECD (2024)
Poland	2 225 million ha 0.4–0.9 t/ha annually (case study)		7.1% of the natural land (case study in Trzebnickie Hills).		Wawer and Nowocień (2007)
Portugal	10 t/ha annually		2022 (estimated >30% of natural land) (post-fire). Two-thirds of the Portugal area has potential soil losses higher than 6 t/ha annually.		Parente *et al.* (2022)
Romania	>10.2 t/ha annually (case study)		1991 (42% of arable land). Case study in the Puriceni-Bahnari catchment: mean erosion rate of 15.5 t/ha annually on its agricultural land.		Lup, Miron and Alim (2017)
Slovakia	710 466 ha		29.5% of arable land >10 t/ha annually. Case study: 28% of the agricultural land was threatened by erosion.		Kobza (2015)
Spain	3.48 t/ha annually		2014–2020: 7.4% of total agricultural area. 2022: An area of 11 838 890 ha >25 t/ha annually.		MITECO (2022)
Switzerland	40% of arable land affected (0.20 t/ha annually)		Erosion rate for period 2007–2017 in agricultural fields.		Prasuhn and Blaser (2019) and Prasuhn (2020)
Türkiye	57 million ha (3.9 t/ha annually)		Estimation with PESERA model with precipitation data series for 1975–2012.		Berberoglu *et al.* (2020) and Kilic and Gunal (2021)

Note: = wind erosion, = water erosion, = increasing, = decreasing and = stable.

Sources: See References.

National monitoring of soil erosion and assessment of trends varies between the European countries (Table 9.2). Water erosion is most frequently monitored, followed by wind erosion and coastal erosion. Only Austria, the Wallonia region of Belgium and Finland provide reliable soil erosion monitoring data. No national soil monitoring information was obtained for Ireland, Lithuania, Luxembourg, Slovenia and Sweden. National studies, including regional case studies, give varied results, with large amounts of agricultural land under high erosion risk, long-term projections of high erosion rates, and lower current erosion rates. European cropland soils have had historic soil losses. However, the impact of current management and climate is uncertain without intensified and representative soil erosion monitoring (see EEA, 2022a). Although the absence of accurate national-level data may present challenges, continental-scale assessments provide valuable insights into overall trends and patterns of soil erosion across Europe and help identify broad regional disparities, prioritize areas for further investigation or intervention and inform policy decisions aimed at mitigating soil degradation at both national and European levels.

By 2050, soil erosion rates are projected to increase by 13 to 23 percent in agricultural lands of the European Union and the United Kingdom (Panagos *et al.*, 2021) with differences at a regional scale. Mean soil losses in southern European countries are predicted to either decrease (Spain) or

not change (Italy). However, in central and northern European regions, soil erosion is projected to increase by 10 to 50 percent on average between 2016 and 2050. Countries with currently low erosion rates such as Denmark, France and the Kingdom of the Netherlands are expected to experience the largest increase, mainly due to climate change. The Mediterranean climate domain is predicted to be the most vulnerable to wind erosion due to increased drought (Borrelli *et al*., 2023).

Türkiye is particularly vulnerable to erosion due to unsustainable agricultural practices (such as the cultivation of steep lands, and excessive soil tillage), hilly topography and soil conditions (Berberoglu *et al.*, 2020) with nearly 57 million ha subjected to water erosion, causing ecosystem degradation and affecting both the economic and social welfare of its people (Kilic and Gunal, 2021). The erosion of scrublands, olive groves and degraded forests is above Türkiye's mean national soil erosion rate of 3.9 t/ha annually (Kilic and Gunal, 2021). Land degradation is a severe challenge, leading to approximately 255.5 million tonnes of soil loss which threatens almost one-third of agricultural land prone to severe water erosion (Berberoglu *et al.*, 2020).

Switzerland has extensive statutory provisions for soil erosion prevention within the framework of environmental protection legislation, as well as agricultural legislation (Prasuhn *et al.*, 2013). However, up to 40 percent of arable land in Switzerland is affected by water erosion (Prasuhn and Blaser, 2019). Farming methods unsuited to the location are the main cause of erosion, with soil on slopes with little vegetation cover being particularly at risk.

Most of the Western Balkan countries have no erosion monitoring systems in place, meaning that there are no official soil erosion data available for those countries. However, it is estimated that about 30 percent of agriculture land has productivity loss due to water erosion, with about 45 percent of the total land area being affected by soil erosion (Zdruli *et al.*, 2022). North Macedonia prepared a new soil erosion map showing that about one-third of the territory was affected by soil erosion (Blinkov *et al.*, 2008), with modelling showing a mean annual soil loss on agricultural land of 4.1 t/ha. The mean annual soil loss in Albania is about 30 t/ha, with 22 percent of Albania having an annual soil loss rate >10 t/ha (accounting for 93 percent of the country's soil erosion). In Serbia, about 80 percent of agricultural land is estimated to be impacted by soil erosion. Water erosion is dominant in central and hilly, mountainous regions, while in the Vojvodina region, wind erosion predominates, affecting about 85 percent of agricultural land (Zdruli, Wojda and Jones, 2022; Vidojevic *et al*., 2022).

The estimated area of eroded soils (including all types of erosion) in Ukraine is 10.5 million ha (Agropolit, 2019). Sheet and rill erosion occurs on 17 percent of arable land, with gully erosion on 3 percent and wind erosion on 11 percent. This is due to the large proportion of land under intensive cultivation with a high percentage of row crops and a lack of contour reclamation farming, along with growing crops on steep slopes, excessive cutting of forest and shrubland, and overgrazing. Every year, 300 to 600 million tonnes of soil are lost to erosion.

In Armenia, the soils most affected by water erosion are Cambisols (brown forest soils), Chernozems and Kastanozems. About 140 000 ha (4.7 percent of total land area) are eroded. Between 2015 and 2023 the area of eroded soils became stable (Croyan *et al*., 2022).

In Belarus, eroded soils are mainly found in northern and central areas and are confined to the uplands of the Belarusian ridge and the adjacent loess plains. About 7.2 percent of agricultural land is considered eroded, mostly by water erosion, with about 115 000 ha (1.4 percent) being affected by wind erosion (Chernysh, 2016). However, the area of eroded soils has remained stable over a long period of time, indicating that efforts to prevent further soil erosion have been successful.

Shelter belts help reduce the negative effect of wind erosion in Belarus. Annually, 1 ha of forest shelter belts is capable of filtering between 35 and 70 tonnes of dust from the air and has a protective effect on 35 to 40 ha of adjacent cropland (Chernysh *et al.*, 2016). About 7 000 ha of protective forest shelter belts (94 percent of the total area of shelter belts), were constructed in the 1970s, but it has been

estimated that the area of shelter belts has decreased to 1 500 ha. A survey conducted by the Forest Institute of the National Academy of Sciences of Belarus in the Gomel region showed that about a quarter of forest belts had died or been destroyed, with approximately 30 percent requiring repair and reconstruction, and only 15 percent now perform protective functions satisfactorily (MNRB, 2014). The remaining forest belts require maintenance.

In 2011, Kazakhstan had over 90 million ha of land under erosion risk, of which 29.3 million ha were eroded (MARK, 2021). In 2023, it was estimated that there were 34.1 million ha of eroded lands in Kazakhstan (Table 9.3) according to calculations from the digital soil mapping of Kazakhstan (State Land Cadastre of the Republic of Kazakhstan, 2023). The growth occurred primarily due to a 20 percent increase in the area of eroded agricultural lands in the Abay, Turkestan and Zhambyl regions.

Table 9.3. Areas of soil erosion in agricultural lands of Kazakhstan (ha)

Land	Year	Water erosion (ha)	Wind erosion (ha)	Water and wind erosion (ha)	Total eroded soils (ha)
Agricultural land	2011	4 950 300	24 168 100	201 700	29 320 100
	2023*	5 818 800	28 075 500	233 400	34 127 700
Cropland	2011	1 220 000	451 800	96 200	1 768 000
	2023*	1 458 800	485 800	96 200	2 040 800

◊ *Sources:* **MARK (Ministry of Agriculture of Republic of Kazakhstan)**. 2021. *Summary Analytical Report on the Status and Use of Lands in the Republic of Kazakhstan in 2021*. Nursultan. https://www.gov.kz/uploads/2022/4/11/b09469de9be9cc54d2cc0e9cc7a77e84_original.7131188.pdf

◊ ***State Land Cadastre of the Republic of Kazakhstan**. 2023. *The map of the state of soils*. Astana, Department of the Land Cadastre and Automated Information System of the State Land Cadastre. https://aisgzk.kz/aisgzk/ru/content/maps?type=rst_kart

In the seasonally dry regions of the steppe and forest steppe zones of the Russian Federation, areas of Chernozem soils actively used in intensive farming systems exhibit accelerated water erosion, as well as salinization and sodification (Rosreestr, 2024). The total area of agricultural soils affected by water erosion in the country is 28 million ha (Ivanov *et al.*, 2020), with annual regional summaries of local monitoring studies (covering 10 to 12 million ha per year), showing an increase in the area of moderately and highly eroded soils by 0.9 to 1 percent per year relative to the studied area of eroded soils (Rosreestr, 2024).

Currently, about 70 percent of agricultural lands in Israel are at moderate to high risk of erosion (Crouvi, Zaidenberg and Shapiro, 2013).

9.3.3 Loss of soil organic carbon in Europe and Central Asia

The monitoring of SOC in mineral soils is diverse throughout the European countries, with results varying between countries, and even within the same country (Meurer *et al.*, 2024). Only a few countries report long-term SOC trends for the key land uses of grassland, cropland and forestland (Table 9.4). No information from national SOC monitoring was obtained for Cyprus, Greece and Italy.



Table 9.4. Monitoring of soil organic carbon content in European Union member states (plus two non-European Union countries) according to land uses, status and trend

	Grassland		Cropland		Forestland		Source
	Status	Trend	Status	Trend	Status	Trend	
Austria		✓	>15 g/kg	↗		✓	Baumgarten *et al.* (2021)
Belgium	36 g/kg	↗	13 g/kg	↘			SPW (2017)
Bulgaria	5–28 g/kg	✓	5–28 g/kg	✓		✓	Hristov and Filcheva (2017)
Croatia	0.2–0.4 g/kg	✓	0.1–0.2 g/kg	✓	0.2–0.4 g/kg	✓	Husnjak *et al.* (2014)
Czechia			15 g/kg	✓	62.6 ± 17.2 t/ha		Marek *et al.* (2020)
Denmark	56 t/ha		56 t/ha		82 t/ha		Callesen *et al.* (2015) and Harbo *et al.* (2023)
Estonia	133 t/ha	✓			156 t/ha		Kõlli, Köster, and Kauer (2007) and Kõlli, Asi and Köster (2004)
Finland			84–98 t/ha	↘	65 t/ha (100 cm)		Heikkinen *et al.* (2021) and Rantakari *et al.* (2012)
France	84,6 ±34 tC/ha		51,6 ±16,2 tC/ha		81 ± 35,4 tC/ha	(+ 0,35 tC/ha/year)	Pellerin *et al.* (2019), Jonard *et al.* (2017)
Italy	91 t/ha	✓	54 t/ha		132 t/ha	✓ ↗	Chiti *et al.* (2018)
Germany	23–46 g/kg	✓	12–23 g/kg	✓	12–23 g/kg	↗	Lehmphul (2015) and Poeplau *et al.* (2020)
Hungary		✓	6–17 g/kg	↗		✓	Németh, Michéli and Pásztor (2002)
Ireland	0.5 g/kg	✓					McGrath and Zhang (2003)
Latvia	23 g/kg	✓	2 g/kg				Bardule *et al.* (2017)
Lithuania	74 t/ha	✓	72 t/ha	✓	80 t/ha	✓	Armolaitis *et al.* (2021)
Luxembourg	36 g/kg	↔	23 g/kg	↗			Kibblewhite and Gregor (2021)
Malta			23 g/kg	✓ ↗			Sultana (2015b)
Netherlands (Kingdom of the)	27.855 g/kg	↘	17.275 g/kg	✓ ↘			Knotters *et al.* (2022)
Portugal	14 ± 4 g/kg	✓	8 ± 2 g/kg	✓			Madeira *et al.* (2004)

	Grassland		Cropland		Forestland		Source
	Status	Trend	Status	Trend	Status	Trend	
Romania			137 t/ha	✓			Dincă *et al.* (2012)
Serbia			19–22 g/kg	✓			Vidojević *et al.* (2015)
Slovakia	48 g/kg	✓ ↗	20 g/kg	✓ ↔			Skalský *et al.* (2024)
Spain	33 g/kg	✓	1 g/kg	✓	37 g/kg ✓		Rodríguez-Martin *et al.* (2016)
Sweden	77–105 t/ha	✓	27 g/kg	✓ ↗			Poeplau *et al.* (2015)
Switzerland	64–131 t/ha	✓ ↔	22–80 t/ha	✓ ↔	79.9 t/ha		Nussbaum *et al.* (2014 and Moll-Mielewczik *et al.* (2023)

Notes: The majority of the SOC stocks are assessed from 0–30 cm depth and correspond to mineral soils.

✓ = monitoring data for status and/or trend available, ↗ = increasing SOC/SOM trend, ↘ = decreasing SOC/SOM trend and ↔ = stable SOC/SOM trend.

Sources: See References.

Climate, vegetation type, soil characteristics and land use determine the spatial variability of SOC content across the European Union. Based on the SOC measurements from LUCAS in 2018, SOC content increased across southeastern and northwestern climatic zones (Fernández-Ugalde *et al.*, 2022), with the highest SOC levels (a mean of 318.1 g/kg) in the wetlands of the boreal and Atlantic zones, due to peat. SOC content was also high in woodlands (a mean of 88.1 g/kg), especially in northwestern climatic zones (boreal, Atlantic, suboceanic and northern subcontinental). The mean SOC content of grasslands was 40.2 g/kg in grasslands, rising slightly to 55.2 g/kg in shrubland. Organic carbon content was the lowest in croplands (a mean of 18.3 g/kg) and in bare land (a mean of 17.3 g/kg).

Soils are losing C as CO_2 emissions to the atmosphere (EEA, 2022b), which if not reversed, could hamper the ambition of the European Union climate targets. Soil organic carbon changes in agricultural soils across the European Union and the United Kingdom from 2009 to 2018 were assessed based on LUCAS Soil (JRC and ESDAC, 2022), and showed varied impacts, depending on land use and management practices (De Rosa *et al.*, 2023). The total SOC loss from croplands and grasslands was moderate, representing 0.75 percent of their initial stocks in the top 20 cm of soil.

In the Western Balkans region, it is estimated that the area of land with low and declining C stocks is less than 5 percent of the total land area and about 10 percent of agriculture land (Zdruli *et al.*, 2022), although the loss of SOC is evident in most agricultural soils (Vidojevic *et al.*, 2022). Albanian soils have relatively low SOC content (≤2 percent) (Ministry of Tourism and Environment, 2019), while in Bosnia and Herzegovina, the SOC content is mostly at a moderate level (3 to 4 percent). Soil organic carbon content in Kosovo ranges from 0.02 to 2.8 percent, depending on the soil type, depth, land use, slope, soil cover, and other factors. For Montenegro, there are no data on SOC change.

In Serbia, most SOC ranges from1 to 2 percent. A major form of degradation of agricultural land in Serbia comes from a loss of organic matter due to intensive agricultural production, intensive tillage, the lack of organic fertilizers, irrigation, removal or burning of crop residues, as well as other unsuitable cultivation practices (Zdruli, Wojda and Jones, 2022; Vidojevic *et al.*, 2022).

In Ukraine, the weighted average of humus content in soils has declined from 3.16 percent in 2015 to 3.07 percent in 2020 (SOC was not measured in the research) (Institute of Soil Protection of Ukraine, 2023). Ukraine's Polissya zone has the lowest humus content at 2.4 percent, whereas the forest steppe zone and the steppe zone has a humic content of 3.2 percent and 3.3 percent respectively.

In the United Kingdom, studies disagree over whether there is a trend of loss in SOC linking to changes in climate (Thomas *et al.*, 2020). However, agricultural soil monitoring data in Finland from Heikkinen *et al.*(2022) suggested that climate change resulted in SOC loss, and that summer temperatures had a greater effect than the change in precipitation. Twenty-five years of observations of SOC in Swiss croplands by Gubler *et al.* (2019) showed stability overall, but with some divergent trends. In Norway, the monitoring of SOC began in July 2023 in forests and intensive grasslands and will provide information in the future about the levels of SOC stores, and the expected changes (NIBIO, 2025).

There has been a decline in SOC content in Armenian soils associated with climate warming and land aridification (MNP, 2020), and arid zones in mountain landscapes have expanded. On the Ararat Plain, the groundwater level has decreased by 2 to 3 m due to extraction by fish farms since 2007 (MTAIRA, 2018). As a result, the area of waterlogged soils has decreased by about 10 000 ha and moist meadow brown soils have lost around 1 percent of humus, or 785 000 tonnes of C in total, released into the atmosphere.

Agrochemical soil surveys in Belarus (Bogdevich, ed, 2022) showed that the humus content in arable soils increased from 1.78 percent in 1975 to 2.04 percent in 1984, reaching a maximum of 2.28 percent in 2000, after which there was a slight decrease down to 2.23 percent by 2012, before increasing again to 2.27 percent by 2020, which was attributed to a slight increase in organic fertilizer use. The proportion of soils in agricultural lands with a humus content of less than 1.5 percent appears to have stabilized at 10 percent, indicating that the current annual application of organic fertilizers of 10 t/ha has achieved a balance in humus content. On meadow lands, SOM has also remained at the same level of 2.7 percent (Figure 9.12).

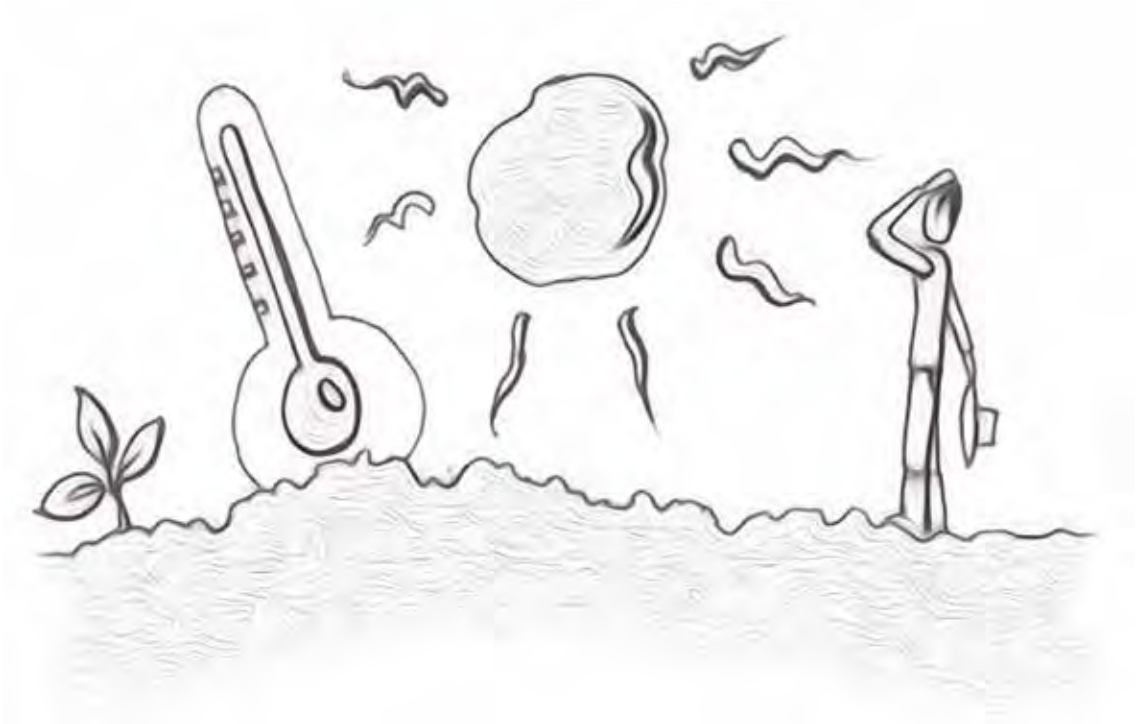
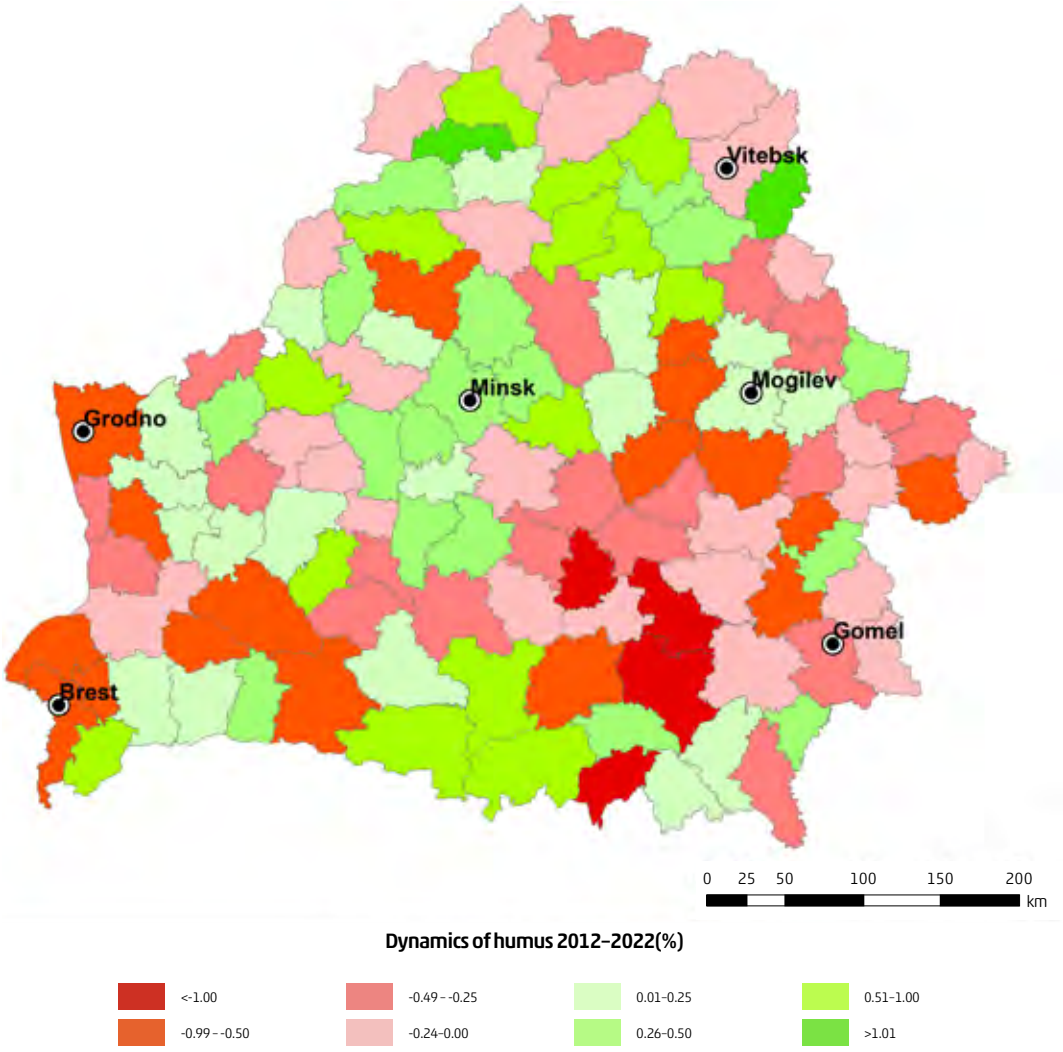


Figure 9.12. Dynamics of soil organic carbon in agricultural lands of Belarus 2012–2022



Refer to the disclaimer on page ii for the names and boundaries used in this map

◊ *Source*: Adapted from **Bogdevich, I.M.,** ed. 2022. *Agrochemical Characterization of Soils in the Agricultural Lands of the Republic of Belarus (2017–2022).* Minsk, Institute of Soil and Agrochemistry & Institute of Systemic Studies in the Agricultural Industry.

About 4.4 percent of soils in the agricultural lands of the Russian Federation have very low (<1.4 percent) SOM, varying from 0.3 percent of agricultural land in the Ural Federal District to 9.7 percent in the Southern Federal District (Table 9.5). In contrast, 47 percent of agricultural lands have very high (4 to 12 percent) SOM content (Russian Agricultural Land Monitoring, 2024).

Table 9.5. Organic matter content in soils of federal districts in the Russian Federation (2017–2022)

Administrative district	Organic matter content (%)					
	Very low (<1.4%)	Low (1.4–2%)	Medium (2–2.5%)	Elevated (2.5–3%)	High (3–4%)	Very high (4–12%)
Northwestern Federal District	7.3	11.2	20.2	17.2	25.4	17.8
South Caucasian Federal District	5.3	6.9	9.4	15.0	28.8	34.1
Privolzhskii Federal District	2.7	7.6	13.5	13.1	17.4	45.3
Siberian Federal District	1.1	2.8	4.5	6.1	14.9	66.9
Far Eastern Federal District	5.8	6.8	8.4	9.9	18.8	48.6
Ural Federal District	0.3	0.9	1.5	2.6	10.0	82.5
Southern Federal District	9.7	15.8	13.9	12.3	37.1	11.2
Central Federal District	5.8	11.2	11.2	7.8	9.7	54.1
Total	4.4	8.4	10.6	10.0	18.2	47.2

◊ Source: **Ministry of Agriculture of the Russian Federation.** 2024. *Edinaya Federalnaya Informatsionnaya Sistema o Zemplyah Selskokhozyaistvennogo naznachaniya (EFIS ZSN) [Unified Federal Information System on Agricultural Lands (EFIS ZSN)].* [Accessed on 5 August 2024]. https://efis.mcx.ru/landing/

The soils of Türkiye generally have very low organic matter contents, except for the Black Sea region, which receives higher and more regular rainfall compared to the country's other regions, and the Eastern Anatolia region, due to a lack of anthropogenic activities. In the latter region, only 14 percent of soils have >2 percent SOM, while across the whole of Türkiye, 64 percent of soils have <1 percent SOM (Saygın, Madenoglu and Erpul, 2015). The low level of SOC in agricultural areas in Türkiye is related to intensive agriculture, which relies on irrigation, fertilization, and land-use change.

Peatlands are comprised of organic soils (also referred to as Histosols) and thus, store C. Because of the typically high C content as well as the depth of organic soil horizons, peatlands hold more C per hectare than other ecosystems, making them the largest terrestrial C pool (Temmink *et al.*, 2022). The peatland map of Europe (Tanneberger *et al.*, 2017) shows that most peatlands are in Northern Europe. Almost one-third of the European peatland is in Finland, and more than a quarter is in Sweden. The remainder is in Estonia, Germany, France, Ireland, Latvia, the Kingdom of the Netherlands, Norway, Poland, and the United Kingdom. Small areas of peat and peat-topped soils also occur in the Czech Republic, Denmark, Hungary, Lithuania (Tanneberger *et al.*, 2022) and Slovakia (Barančíková and Halas, 2022). The Russian Federation also has extensive peatlands in northern latitudes, and Belarus has extensive peatland along the border to Poland (Tanneberger *et al.*, 2017). Many northern peatland areas are forested, so management heavily affects the soil water regime.

The amount of C in organic soils may be more than ten times the C stored in mineral soils, depending on the peat thickness. It is estimated that 50 percent of the European peatland area is degraded (Tanneberger *et al.*, 2022), with a higher proportion of peatland drained and used for agriculture (and thus degraded) in temperate than arctic regions. The total proportion of degraded peatlands in Europe is 46 percent, while within the European Union, it is 50 percent (12 million ha) (Tanneberger *et al.*, 2021). Globally, since 1700, Northern Europe has undergone the earliest and highest losses of peatlands (Fluet-Chouinard *et al.*, 2023). Peatland degradation is caused by artificial drainage, most often for agriculture, forestry or peat extraction. Countries such as Denmark, Germany, Ireland, the Kingdom of the Netherlands, and Poland have drained more than 80 percent of their peatlands. The cultivation and drainage of organic soil leads to CO_2 emissions as the organic matter is biodegraded.

For 2019, European Union member states reported C loss from 17.8 million ha of land with organic soil (4.2 percent of the total land area) corresponding to emissions of 108 million tonnes of CO_2 (EEA, 2022b). Without rewetting, drained peatlands will continue to lose SOC, and climate change is predicted to induce further peat loss from undrained peatlands (Tanneberger *et al.*, 2022). Mitigation options – such as rewetting peatlands – offer C sequestration potential, but the feasibility varies depending on local contexts and environmental considerations (Kløve *et al.*, 2017; de Jong *et al.*, 2021). For example, rewetting tropical peat soils provides immediate benefits, while in temperate and boreal agricultural peat soils, the positive effect is estimated to take several decades (Ojanen and Minkkinen, 2020).

The EUSO Soil Degradation Dashboard (EC, 2024) shows that European Union peatlands are likely to be degraded due to agriculture-related pressures (2 percent of the European Union total area), based on the UNEP's Global Peatlands Assessment work (UNEP, 2022).

Soil inorganic carbon is important in semi-arid and arid environments, contributing to up to half of the C stock in terrestrial ecosystems. In dry areas, the SIC stored in carbonate minerals can exceed that in the organic fraction of the soil (Suarez, 2000), with observations of up to 20 times more SIC than SOC in some regions (Wang *et al.*, 2015). Lu *et al.* (2023) summarized the data contained in the LUCAS 2015 topsoil data on SIC concentration in European topsoils and reported a wide variability. In summary, SIC distribution in Europe is geographically variable and tends to concentrate in areas with a Mediterranean climate and calcareous parent materials. Soil inorganic carbon also occurs in certain humid and subhumid temperate areas, such as the French Champagne and Charente regions.

The use of chemical fertilizers, climate warming and atmospheric acid deposition have caused the acidification of European soils (EEA, 2013) and may accelerate SIC turnover (Monger and Gallegos, 2000), leading to SIC sinks or SIC sources (Lal, 2002; Bughio *et al.*, 2016; Zamanian *et al.*, 2018; Raza *et al.*, 2020). No information is available in relation to the trends of SIC concentration or storage in European soils, although data from the more recent LUCAS surveys could be used in the future to estimate changes compared to the first LUCAS survey, such as carried out for SOC by De Rosa *et al.* (2023).

9.3.4 Loss of soil biodiversity in Europe and Central Asia

Soil organisms are a crucial part of the terrestrial biosphere (Van den Hoogen *et al.*, 2019; Guerra *et al.*, 2022; Anthony *et al.,* 2023) . Historically, soil organisms and soil biodiversity have been underrepresented in the assessment of soil health status, compared to chemical and physical soil characteristics (Orgiazzi *et al.*, 2022).The *European Atlas of Soil Biodiversity* (Montanarella *et al.*, 2010) contains maps with rather coarse information on soil biodiversity. However, a more granular evaluation of soil biodiversity has become increasingly feasible thanks to the advancements in molecular biology techniques (such as metagenomics and metabarcoding). The 2018 LUCAS survey included a soil biodiversity component (LUCAS Soil Biodiversity) (Ogiazzi *et al.*, 2022) for the first time, with 850 soil samples collected across the European Union and the United Kingdom, providing the first assessment of soil biodiversity in the EU27 and the United Kingdom (Köninger *et al.*, 2023; Labouyrie *et al.*, 2023).

Orgiazzi *et al.* (2016) estimated that several factors needed to be considered to explain the magnitude and spatial patterns of soil biodiversity loss in the European Union, such as habitat fragmentation, land-use change, soil pollution and urban expansion, all potential threats to soil biodiversity. The EUSO Soil Degradation Dashboard (European Commission, 2024) shows areas where the risk to soil biodiversity is estimated to be moderately high or high.

Beside LUCAS Soil Biodiversity, some European Union member states have national soil monitoring schemes that include measures of soil biodiversity, such as France and the Kingdom of the Netherlands (for an overview, see EEA [2022]). France has developed national maps for soil bacteria (Dequiedt *et al.*, 2011; Terrat *et al.*, 2017; Karimi *et al.*, 2020) and more recently for fungi (Djemiel *et al.*, 2023a, 2023b). Ongoing experiments include biodiversity measurements of soil fauna in the French national monitoring network (Imbert *et al.*, 2022). In Switzerland, 30 long-term monitoring sites were surveyed over five years to define and assess long-term stability of abundant, rare and scarce soil bacterial and fungal community components, and may facilitate long-term soil quality monitoring by detecting disturbances of the characteristic habitat-associated core communities (Gschwend *et al.*, 2021). An ecological survey was undertaken in Wales in the United Kingdom from 2013 to 2016 to characterize environmental status and trends and quantify the effect of agri-environment schemes. The Norwegian Agricultural Soil Monitoring Programme will be including a soil biology module as of 2024 and the Finnish Agricultural Soil Monitoring will release the first bacterial and fungal biodiversity monitoring assessments in the near future (Arias-Navarro, Barita and Jones, eds, 2024). However, soil biodiversity data is lacking in all the countries of Western Balkans, and biological indicators of soil health are not determined in the current soil monitoring programmes in Ukraine.

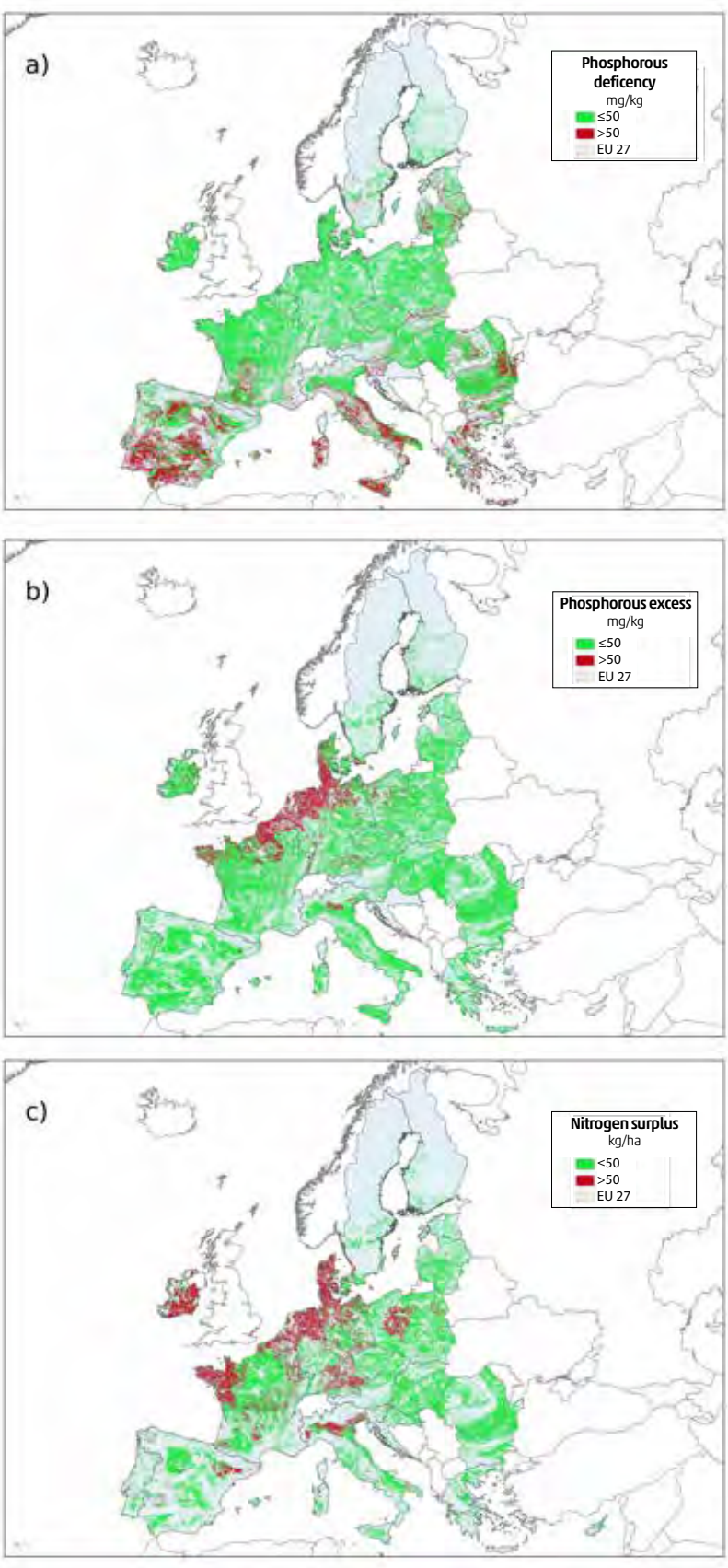
Soil biodiversity regulates the ecosystem processes essential to the functioning of the Earth's ecosystems. To determine which regions of Europe are most vulnerable to soil biodiversity loss, sufficient information must be available on the status and trends of biodiversity in soils.

9.3.5 Nutrient mismanagement in Europe and Central Asia

Nutrient overuse remains one of the most serious issues in the European Union due to the harmful impact of excess nutrients on the wider environment. Excessive N can impact drinking water safety and lower crop quality, affecting both livestock production and human food security (Lundin and Nilsson, 2021; Ishfaq *et al.*, 2023). Excessive nutrients cause eutrophication in waterbodies, leading to biodiversity loss and oxygen depletion (Smith and Schindler, 2009; Lundin and Nilsson, 2021). Excess N may also result in increased N_2O emissions to the atmosphere (Butterbach-Bahl *et al.*, 2013; Arias-Navarro *et al.*, 2017; McDonald *et al.*, 2022).

Nutrient-related pressures and concentrations in the European Union have gradually improved over the since the 1990s. The major imbalances in the N and P cycles give rise to diffuse source pollution in agriculture (EEA, 2020b). Information on long-term annual N and P budgets is needed to better inform future management strategies. About 21 percent of European Union agricultural land is considered to be at risk of P deficiency for crop production and about 10 percent is considered to have an excess in P, with possible additional environmental risks (Ballabio *et al.*, 2019) (Figure 9.13). High levels of P, compared with threshold values, were reported in some countries of Western and Central Europe, and are a likely consequence of the balance of P fertilizers in European Union soils, with less overfertilization in grasslands than in croplands (Van Dijk *et al.*, 2016) . The annual P budget at a regional scale shows potential to improve P management by both reducing inputs in regions with a high surplus and rebalancing fertilization in those at risk of soil fertility depletion. Comparable results were obtained in France where it would seem that intensive livestock production led to increased extractable soil P (Guejjoud, Curie, and Grosbois, 2024). Encouragingly, 35 years of monitoring data from Finland indicate a recent decline in the upward trend of P levels, suggesting improved P fertilizer management (Keskinen *et al.*, 2016).

Figure 9.13. Areas showing phosphorus deficiency, excessive phosphorus and nitrogen surplus



a) Areas with P deficiency (less than 20 mg/kg). This threshold is in the range of thresholds used by a number of European countries to define P deficiency.

b) Areas with P excess (more than 50 mg/kg). This threshold corresponds to the average of excessive P concentrations used by a number of European countries to define soil P excess.

c) Areas of agricultural land in the European Union subject to N surplus (N input minus N output). The layer presented shows agricultural land areas where the N surplus exceeds 50 kg/ha annually. This value can be considered as a threshold beyond which the environmental impacts of N might be significant.

Sources: Authors' own elaboration and **EC**. 2024. EUSO Soil Degradation Dashboard. In: *ESDAC*. Ispra, Italy, JRC. [Cited 19 August 2024]. https://esdac.jrc.ec.europa.eu/esdacviewer/euso-dashboard/

Refer to the disclaimer on page ii for the names and boundaries used in this map.

Batool *et al.* (2022) analysed the annual long-term N surplus for both agricultural and non-agricultural soils across Europe for more than a century (1850–2019). A latitudinal pattern was noted, with lower N surplus values in Northern Europe, higher values at mid-latitudes (particularly in Western and Central Europe), and moderate values in Southern Europe. As noted by De Vries *et al.* (2021), relatively high N surpluses occured in intensive livestock regions, including Belgium, France (Brittany), northwestern Germany, Italy (the Po Valley), Luxembourg, and the Kingdom of the Netherlands. A surplus of N on 22 percent of agricultural land was also noted, aligning with the data presented in the EUSO Soil Degradation Dashboard (EC, 2024) (Figure 9.10).

Lee *et al.* (2020) estimated soil N inputs, outputs and balances in Switzerland, based on a process-based simulation of alternative cropping systems. A negative soil N balance for conventional Swiss croplands at the recommended amount of fertilization indicates the need to improve the management of both input and output of N. In the Western Balkans, Zdruli *et al.* (2022) estimated that about 5 percent of agriculture land was affected by overfertilization In Ukraine, while analyses from a 2015–2020 soil survey indicated an increase in N content in soils by 5 percent compared to previous surveys, along with a 6 percent increase in P content (Institute of Soil Protection of Ukraine, 2023). Despite the increases, the overall balance of nutrients for agricultural soils in Ukraine remains negative. Specifically, in 2020, there was a deficit of 5 kg N/ha and 15 kg P/ha (Arias-Navarro *et al*., 2024).

About 36 percent of the total agricultural lands of the Russian Federation have very low or low available P, varying from 9 percent in the Northwestern Federal District to 89 percent in the Southern Federal District (Table 9.6). Low content of available P in the Southern Federal District is related to the increase in yields in recent years and, accordingly, the removal of P with the harvest. Most probably, the fertilizers with P may be being applied inadequately to the crops' needs. Soils in this region are also rich in carbonates, making P applied to soil unavailable to plants (due to retrogradation). However, the method of strip application of starting doses of P fertilizers works well in these areas.

Soils with very low and low available K account for 12.6 percent of the total agricultural lands of the Russian Federation, varying from 0.3 percent for the Southern Federal District to 28 percent for the Northwestern Federal District (Table 9.6), the latter being related to the inherent low content of K in the parent material.

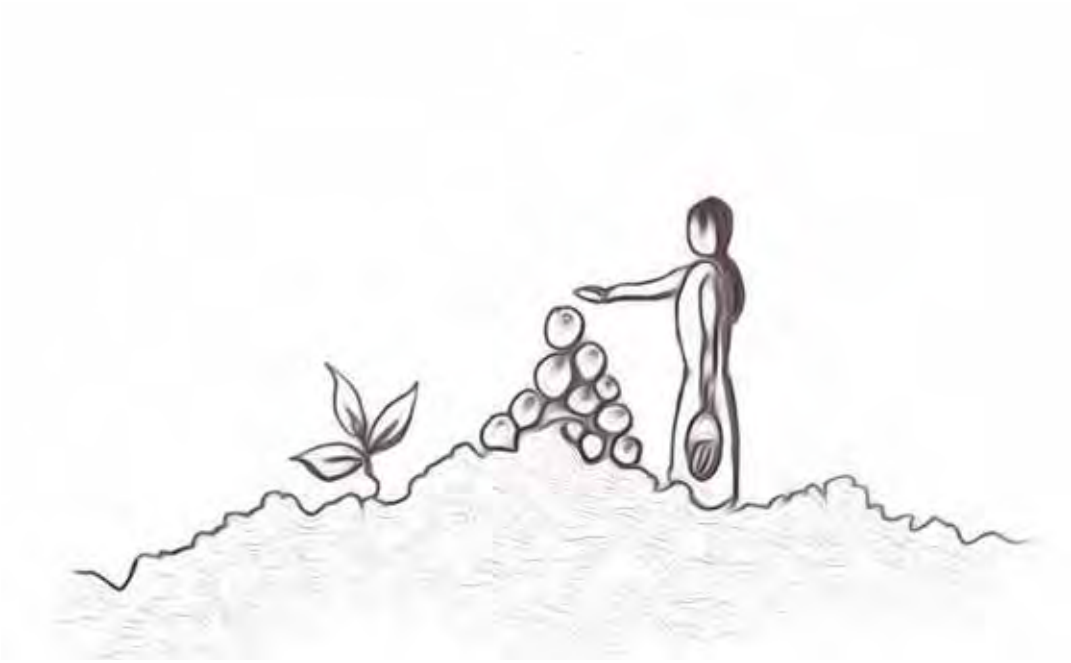


Table 9.6. Phosphorus pentoxide and potassium oxide contents of soils in the federal districts of the Russian Federation (2017–2022)

Administrative district	Phosphorus pentoxide (P_2O_5) (mg/kg)					
	Very low (0–25 mg) (%)	Low (25–50 mg) (%)	Medium (50–100 mg) (%)	Elevated (100–150 mg) (%)	High (150–250 mg) (%)	Very high (250–1 000 mg) (%)
Northwestern Federal District	1.6	7.2	21.4	20.3	30.7	18.4
South Caucasian Federal District	64.6	21.6	6.7	2.8	4.3	0.0
Privolzhskii Federal District	16.0	14.2	29.3	20.0	15.5	5.0
Siberian Federal District	14.6	9.2	24.3	25.8	20.7	5.5
Far Eastern Federal District	29.0	24.2	18.8	9.2	9.8	8.9
Ural Federal District	10.5	38.6	33.9	10.0	5.4	1.6
Southern Federal District	54.5	34.6	10.4	0.4	0.0	0.0
Central Federal District	3.2	9.0	33.5	26.1	20.3	8.0
Total	19.7	16.6	25.3	18.1	14.8	5.4

Administrative district	Potassium oxide (K_2O) (mg/kg)					
	Very low (0–40 mg) (%)	Low (40–80 mg) (%)	Medium (80–120 mg) (%)	Elevated (120–170 mg) (%)	High (170–250 mg (%)	Very high (250–1 000) mg (%)
Northwestern Federal District	5.3	23.1	26.5	21.6	16.9	6.6
South Caucasian Federal District	3.9	5.0	3.8	8.2	17.1	62.0
Privolzhskii Federal District	1.4	8.8	21.4	27.6	19.1	21.6
Siberian Federal District	0.4	6.1	20.4	21.7	18.8	32.6
Far Eastern Federal District	1.6	15.2	24.0	28.9	22.3	8.0
Ural Federal District	0.6	6.7	19.1	27.7	34.8	11.2
Southern Federal District	0.1	0.2	0.6	1.8	7.1	90.1
Central Federal District	5.2	20.2	31.5	27.7	12.4	3.1
Total	2.2	10.4	20.2	21.9	16.5	28.7

◊ *Source:* **Ministry of Agriculture of the Russian Federation.** 2024. *Edinaya Federalnaya Informatsionnaya Sistema o Zemplyah Selskokhozyaistvennogo naznachaniya (EFIS ZSN) [Unified Federal Information System on Agricultural Lands (EFIS ZSN)].* [Accessed on 5 August 2024]. https://efis.mcx.ru/landing/

9.3.6 Soil acidification in Europe and Central Asia

Acidification has impacted both forest and agricultural soils and certain semi-natural ecosystems, such as calcareous grasslands (Bolan *et al.*, 2005). Acidification through deposition from the air occurs mainly in northwest and Central Europe (EEA, 2013) but is less of an issue further south, to the east, and to the far north. Critical loads of N and S for acidification are exceeded in much of Northern Europe.

Nowadays, soil acidification mainly occurs on unlimed agricultural land, due to the application of ammonium-based fertilisers and urea, especially on naturally acidic soils such as sandy soils. Since the beginning of the twenty-first century, there have been some signs of recovery, with increasing pH being detected in the upper soil horizons (Achilles *et al.*, 2021). However, recovery from former acidification seems to be soil depth-dependent. While some studies show increasing pH values over time for forest floor and topsoil (Navrátil *et al.*, 2007; Berger *et al.*, 2016; Wellbrock and Andreas, eds, 2019), others show that the deeper mineral soil is still acidified as a legacy of past acidifying air pollutants (Berger *et al.*, 2016).

In the United Kingdom, declining sulphate deposition has allowed the recovery of topsoils from acidification (Thomas *et al.*, 2020). An increase in soil pH across all United Kingdom habitats was reported by several independent monitoring schemes. Within the Countryside Survey data (Henrys *et al.*, 2012), there was an increase of 0.5 pH units at a national scale from a mean of 5.39 to 5.87 in 30 years (1978 to 2007) (Reynolds *et al.*, 2013).

In France, Saby *et al*. (2017) studied changes in the pH of cultivated topsoil by comparing data from the 1996–2000 period with that of the 2006–2010 period. It was shown that nearly all significant changes were increases. When significant, the mean rate of pH increase was 0.025 annually. It was assumed that part of the increase could be explained by changes in N inputs to the soil, especially associated with animal breeding systems, and possible decreases in acid deposition from the atmosphere.

In Switzerland, soil acidification continues to be an issue in forests, both in terms of extent and ongoing progression (Braun, Tresch and Augustin, 2020). In Ukraine, 24.4 percent of soils are considered acidic (Institute of Soil Protection of Ukraine, 2023). Soil acidification has been identified in the Polissia zone where there is a high percentage of acidic soils (average pH of 5.69), while the largest areas of slightly acidic soils are in the Steppe zone (average pH of 6.83). The threat of soil acidification is increasing due to limited actions, including a lack of liming of soils since the beginning of the twenty-first century (Arias-Navarro *et al*., 2024).

The neutralization of soil acidity is one of the main goals of soil management in Belarus. In the 2000–2020 period, the use of lime decreased, resulting in increased soil acidity in cropland and managed meadows. The annual demand for carbonate application in Belarus is 2.5 million tonnes (Klebanovich and Vasilyuk, 2003). According to Bogdevich (2022), the demand for liming was exceeding 2.8 million tonnes per year, including 2.2 million tonnes used on cropland and 0.6 million tonnes used on managed meadowland. However, in the decade up to 2023, the amount of dolomite flour and other lime amendments was only 0.9 to 1.2 million tonnes per year (NCSRB, 2023). In Belarus, comparing data from 2013–2016 with 2017–2020, pH decreased by 0.09 (cropland) and 0.08 (meadows) which is notable for such a short period. The area of cropland in Belarus with a soil pH <5.0 (able to cause a shortfall in yield) increased from 7.1 to 9.8 percent (Bogdevich, 2022). It is recommended that the scale of liming of agricultural lands in Belarus is increased by 2.5 to 3 times, or 2.8 million tonnes of $CaCO_3$ annually, to prevent soil acidification and bring economic benefits from increased productivity of cropland and managed meadows.

About 12 percent of agricultural lands in the Russian Federation have a pH <5, varying from 0.5 percent of the Southern Federal District to 34 percent of the Far Eastern Federal District (Table 9.7).

Table 9.7. Levels of pH in soils of federal districts in the Russian Federation

Administrative district	Area of agricultural lands (ha)	Area of surveyed land (ha)	Extremely acid (pH <4.0) (%)	Very acid (pH 4.0–4.5) (%)	Moderately acid (pH 4.5–5) (%)	Slightly acid (pH 5–5.5) (%)	Very slightly acid (pH 5.5–6) (%)	Neutral (pH 6–7.5) (%)	Slightly alkaline (pH 7.5–8.5) (%)	Moderately alkaline (pH 8.5–9) (%)	Strongly alkaline (pH 9–12) (%)
Northwestern Federal District	5 565 200	2 174 762	0.9	4.2	12.7	25.5	26.4	30.2	0.1	0.0	0.0
South Caucasian Federal District	11 385 500	2 239 285	2.1	1.3	2.2	3.0	4.7	31.8	50.0	5.0	0.1
Privolzhskii Federal District	51 260 100	13 880 770	0.2	2.1	9.6	24.7	23.8	35.0	4.5	0.1	0.0
Siberian Federal District	40 978 800	8 945 409	0.2	1.5	4.2	19.0	28.3	36.5	10.1	0.2	0.0
Far Eastern Federal District	13 900 000	1 884 820	0.8	7.5	26.1	23.4	18.6	19.6	3.9	0.1	0.0
Ural Federal District	13 864 300	2 821 625	0.2	1.2	6.7	34.5	30.4	26.2	0.7	0.0	0.1
Southern Federal District	31 447 000	7 215 277	0.0	0.1	0.4	2.6	4.5	49.3	38.3	4.5	0.4
Central Federal District	29 345 500	12 463 394	0.3	3.6	17.2	36.3	23.4	19.1	0.2	0.0	0.0
Total	197 746 400	51 625 342	0.3	2.2	9.5	23.0	21.2	32.1	10.7	0.9	0.1

◊ *Source*: **Ministry of Agriculture of the Russian Federation.** 2024. *Edinaya Federalnaya Informatsionnaya Sistema o Zemplyah Selskokhozyaistvennogo naznachaniya (EFIS ZSN) [Unified Federal Information System on Agricultural Lands (EFIS ZSN)].* [Accessed on 5 August 2024]. https://efis.mcx.ru/landing/

Naturally occurring soil acidity is an issue in some regions of Türkiye, especially in the eastern Black Sea region. Standardized liming practices are widely accepted in affected regions and are often used for tea and hazelnut cultivation. Soil acidity potential is also significantly influenced by the amount of synthetic fertilizer used (Duman *et al*., 2023; Temizyurek-Arslan, 2023). Therefore, to prevent the exacerbation of soil acidity, a closely integrated process with stakeholders should be followed, and the reported drivers should be carefully controlled.

9.3.7 Soil pollution in Europe and Central Asia

Despite a common understanding of the impact of soil pollution in the European Union, comprehensive, large-scale assessments are scarce. Most of the data on soil pollution at the European Union scale are derived from the various LUCAS surveys (JRC and ESDAC, 2020, 2022).

High Cu concentrations have been identified in European Union croplands and linked to human activities (Ballabio *et al.*, 2018). High concentrations of Hg occur close to well-known mining sites like Almadén (Asturias, Spain), Mount Amiata (Italy), Idrija (Slovenia) and in Krompachy and Rudňany (Slovakia) (Ballabio *et al.*, 2021). Van Eynde *et al.* (2023) reported differences in soil Zn levels between Northern and Southern Europe, with the lowest mean concentrations in north European countries. In explaining Zn concentration in European Union countries, the distance from natural Zn deposits or Zn mines was the most important variable. Only a small fraction (1.5 percent) of the European Union had high concentrations of Zn in the soil.

Cadmium levels in soil have risen by 50 percent during the twentieth century, primarily due to the use of fertilizers, sewage sludge, local contamination from waste dumping and mining, and industrial emissions like those from Zn smelters (Ballabio, Jones and Panagos, 2024). Of 21 682 LUCAS survey topsoil samples from across the European Union and the United Kingdom, 72.6 percent had Cd levels below 0.07 mg/kg, 21.6 percent fell between 0.07 to 1 mg/kg, and 5.5 percent exceeded the 1 mg/kg threshold, which is generally considered a risk level. The average Cd concentration in European Union topsoils was 0.20 mg/kg, with grasslands having a slightly higher concentration (0.24 mg/kg) compared to croplands (0.17 mg/kg) (Ballabio, Jones and Panagos, 2024).

Data from individual surveys are often obtained using different methods which limit potential comparisons. Except for a few exceptions, there is no regular monitoring of heavy metals in soils. Some countries do not publish regional measurements from polluted regions, such as mining areas. Fourteen countries have developed national background values for some elements (Table 9.8) and have identified target values, which require action. In the year 2000, the Spanish Ministry of Agriculture, Fisheries, Food and Environment (MAPAMA) initiated a collaborative research programme to establish the presence of Cr, Ni, Pb, Cu, Zn, Hg and Cd trace elements in the soils of Spain (Rodríguez *et al.*, 2008). France has developed national statistics (INRAE, 2021) and maps for several trace elements (Saby *et al.*, 2009; Marchant *et al.*, 2010; Marchant, Saby and Arrouays, 2017) and organic contaminants (Orton *et al.*, 2013; Froger *et al.*, 2021) based on their soil monitoring network.



Table 9.8. European Union member states with available background levels for heavy metals (shadowed cells)

Country	Heavy metals														
	As	Cd	Co	Cr	Cu	Hg	Mn	Mo	Ni	Pb	Se	Sb	Ti	V	Zn
Austria															
Belgium															
Bulgaria															
Czechia															
Denmark															
Finland															
France															
Italy															
Germany															
Ireland															
Latvia															
Netherlands (Kingdom of the)															
Romania															
Spain															
Slovakia															

Note: As = arsenic, Cd = cadmium, Co = cobalt, Cr = chromium, Hg = mercury, Mn – manganese, Mo = molybdenum, Ni = nickel, Pb = lead, Se = selenium, Sb = antimony, Ti = titanium, V = vanadium, and Zn = zinc.

Pesticide residues occur in 74.5 percent of European Union agricultural soils, distributed across crops and farming systems, with 57 percent of assessed sites containing mixtures of pesticide residues (Vieira *et al.*, 2023). In France, out of 47 analysed soil samples that included grasslands and forests, 98 percent had at least one pesticide residue (Froger *et al.*, 2023). In organically managed soils, mostly long-banned substances were reported, while in conventionally managed soils there was a mix of currently used, recently used and long-banned residues. Glyphosate, its main metabolite, α-amino-3-hydroxy-5-methyl-4-isoxazolepropionic acid (AMPA), and dichlorodiphenyl-trichloroethane (DDT) were the most frequently identified soil residues (Geissen *et al.*, 2021; Riedo *et al.*, 2021). Silva *et al.* (2019) reported a higher incidence of pesticide residues in 2018, along with an increased toxicity risk, than in 2015 (Franco *et al.*, 2024).

In France, 76 percent of soil samples (n = 25/33 samples) contained microplastics (Palazot *et al.*, 2024).

Hudson-Edwards *et al.* (2023) used a global database of mine sites to identify active (75 percent) and inactive (25 percent) mine locations in the European Union as an indicator for likely point source pollution. The major commodities under extraction (18 percent Cu, 16 percent Au, 15 percent Pb, and 12 percent Zn) were also identified. Over a larger area (including 27 European Union countries and 12 neighbouring countries), the Joint Research Centre (JRC) reported around 2.8 million potentially contaminated sites (Payá Pérez and Rodríguez Eugenio, 2018).

The main contaminants associated with past industrial activities in the Western Balkan countries are mineral oils, trace elements (such as As, Cd, Pb, Ni, and Zn) and organic contaminants such as halogenated and nonhalogenated solvents, PCBs and PAHs (Arias-Navarro *et al.*, 2024). Some countries have made preliminary estimates of sites where contamination occurs (such as Serbia), but it is not possible to get aggregated data for the region. Even though some progress has been made regarding adequate waste management (due to strict legislation), recent estimates have shown that inadequate waste management is still an important source of soil pollution. Diffuse soil

contamination is one of the specific threats to soils in the Western Balkans. Agriculture is the main land-use activity in countries of the Western Balkans, comprising 45 percent of the land area. Trace elements from fertilisers (such as Cd from mineral fertilizers) and some fungicides (mainly Cu) and pesticide residues are the contaminants of major concern in agricultural soils. Contaminants of emerging concern, such as microplastics, pharmaceuticals, personal care products, and PFAS, are still poorly studied in the region, and require further attention (Vidojevic *et al.*, 2022).

In 2016, the Ministry of Ecology and Natural Resources of Ukraine (FAO and UNEP, 2021) reported that the main sources of soil pollution were:

- the residual radionuclides that originated from the accident at the Chernobyl nuclear power station;
- the metallurgic, chemical and mining industry, which release trace elements and radionuclides; and
- agricultural activities that release pesticides, fertilizers and agricultural liquid waste (FAO and UNEP, 2021).

According to Baliuk, Kucher and Nadiya (2021) soil pollution in Ukraine occurred on 9 to 11 percent of arable land. The State Sanitary Epidemiological Service of Ukraine monitors pollutant levels in the soils of urban areas. According to the surveys, the soils most contaminated with heavy metals (Cd, Mn, Pb, Cu, and Zn) were those of the Mariupol, Pavlohrad and Pervomaisk regions. More than the maximum allowable concentration (MAC) of Pb mobile forms (6 mg/kg) were detected in the Dnipro, Odesa and Mykolaiv regions. Levels above the MAC for Cd mobile forms (0.7 mg/kg) were also observed in the Dnipro, Mykolaiv and Odesa regions. In Ukraine, 5.35 million ha of territory remain radioactively contaminated, including 1.24 million ha of agricultural land with a density of over 37 kilobecquerel (kBq)/m^2 of caesium-137 (^{137}Cs) (Plyush and Biushkin, 2021; Arias-Navarro, Baritz and Jones, eds, 2024).

There are 50 500 ha of contaminated soils in Armenia. The most contaminated are Cambisols (brown forest soils) in the northeastern part of Armenia, where the Pb content is 150 mg/kg of soil (Sukiasyan *et al.,* 2021). However, there is no increasing trend in the content of contaminants in soils during the last decade, especially after the copper-molybdenum plant in Alaverdi closed in 2018.

In Belarus, the National Environmental Monitoring System (NSMOS) (NSMOS, 2023) showed that none of the 18 monitoring points had sulphates, nitrates, chlorides, petroleum products, benzopyrene, Cd, Zn, Pb, Cu, Ni, Mg or Cr exceeding the maximum permissible concentration. Thus, the concentrations of identified pollutants were lower than the maximum permissible concentration. Radionuclide pollution of soils in Belarus, which resulted from the Chernobyl catastrophe, is gradually declining (Tsybulko, 2017) (Figure 9.14).

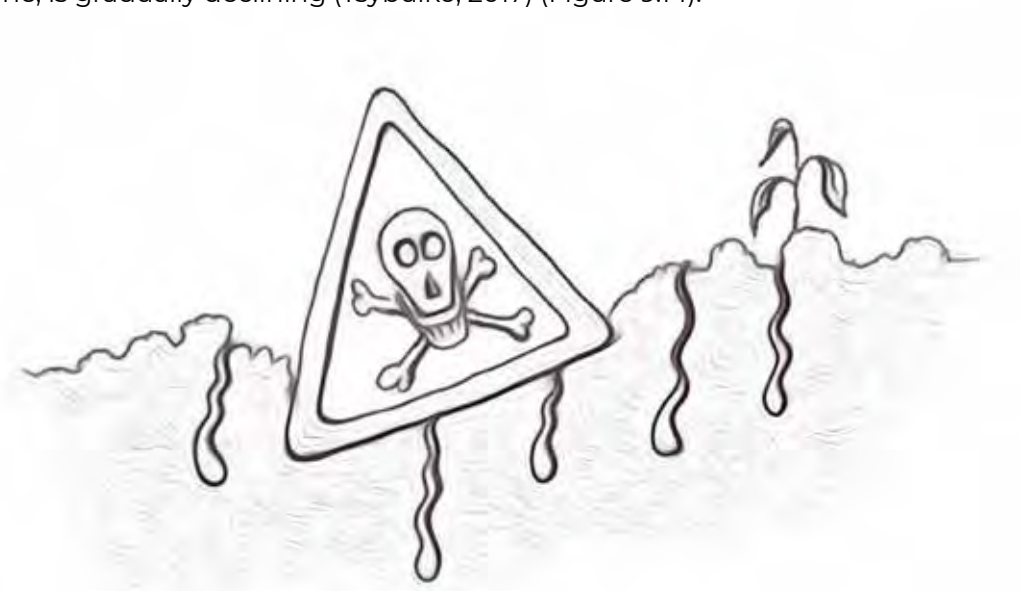
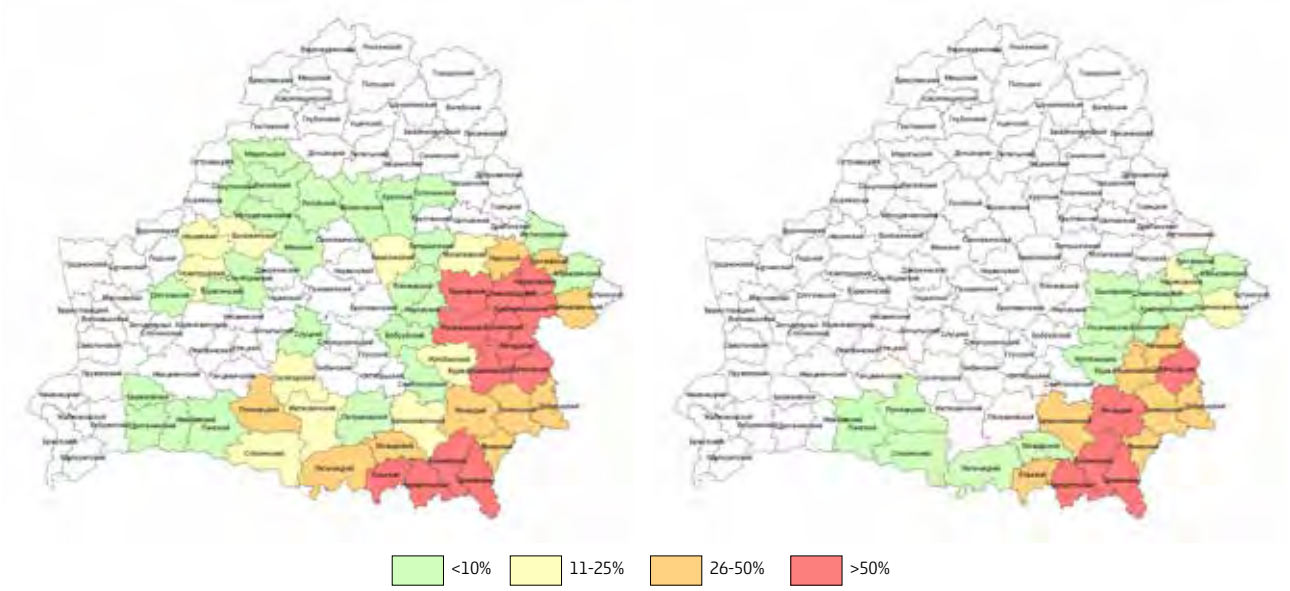


Figure 9.14. The radionuclide pollution of agricultural lands in Belarus by caesium-137 (left) and strontium-90 (right)



Refer to the disclaimer on page ii for the names and boundaries used in this map

◊ *Source:* Authors' own elaboration adapted from **Tsybulko, N.N.** 2017. Optimization of the use of agricultural lands contaminated with radionuclides and agrochemical protective measures in the post-Chernobyl accident period. Minsk, Institute of Soil Science and Agrochemistry of the National Academy of Sciences of Belarus. PhD dissertation.

Soil pollution is monitored within cities and industrial centres in the Russian Federation (Figure 9.15 and Figure 9.16). However, soil samples are usually taken in parks and lawns, where soils are often formed on a bulk layer of transported material. In areas of new buildings, a new soil profile is beginning to form in large areas occupied by soils mixed with construction waste (NPO Taifun, 2023). These aspects should be considered when interpreting the results of soil pollution in urban and industrial areas.

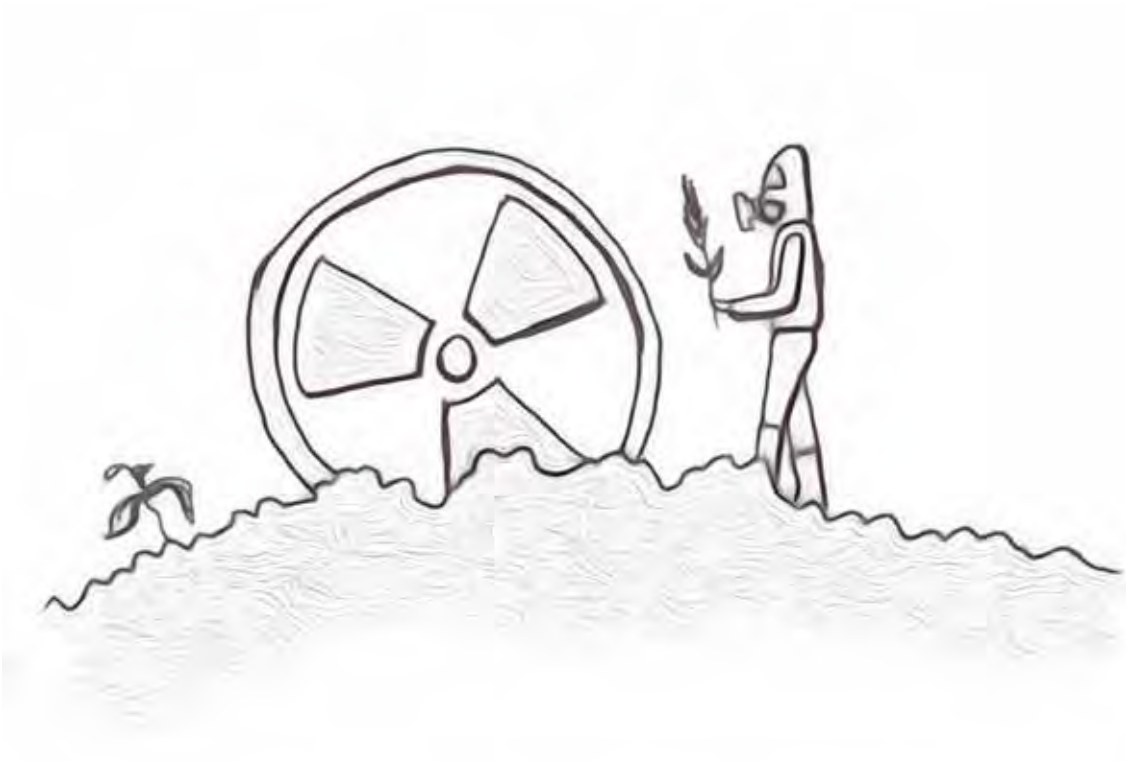
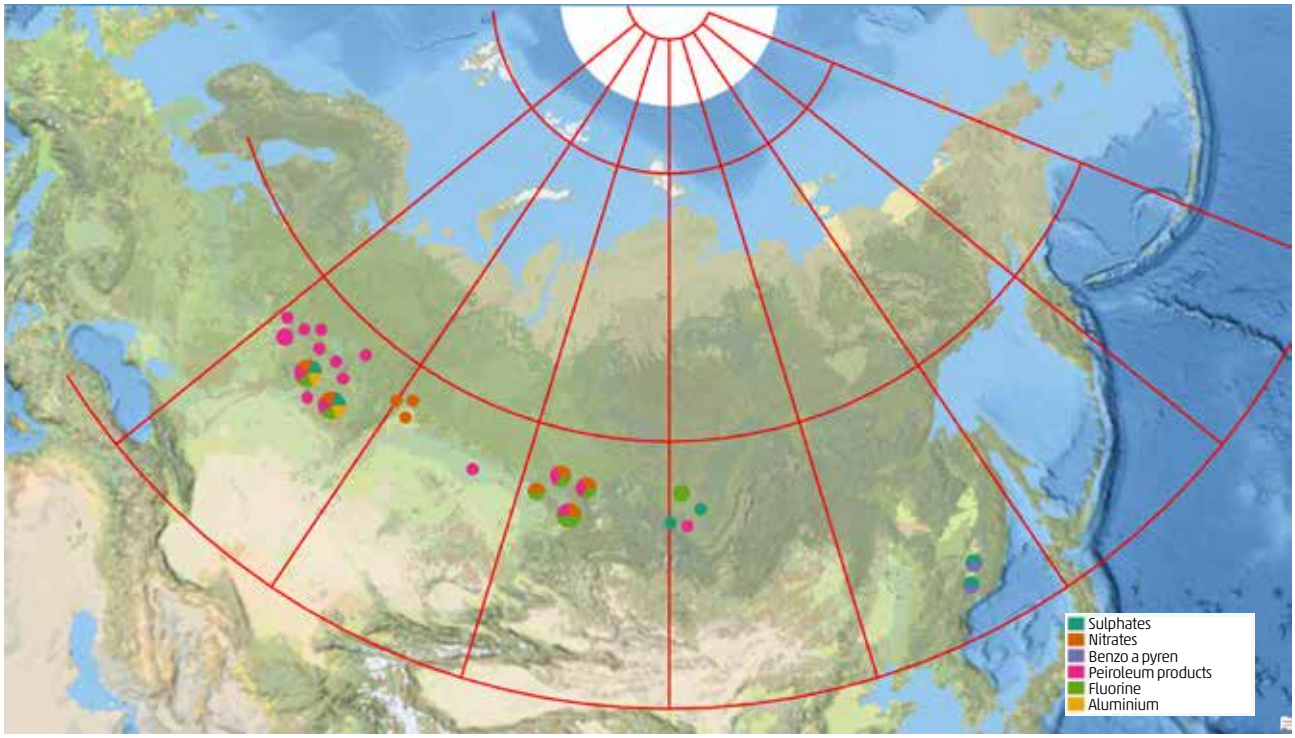


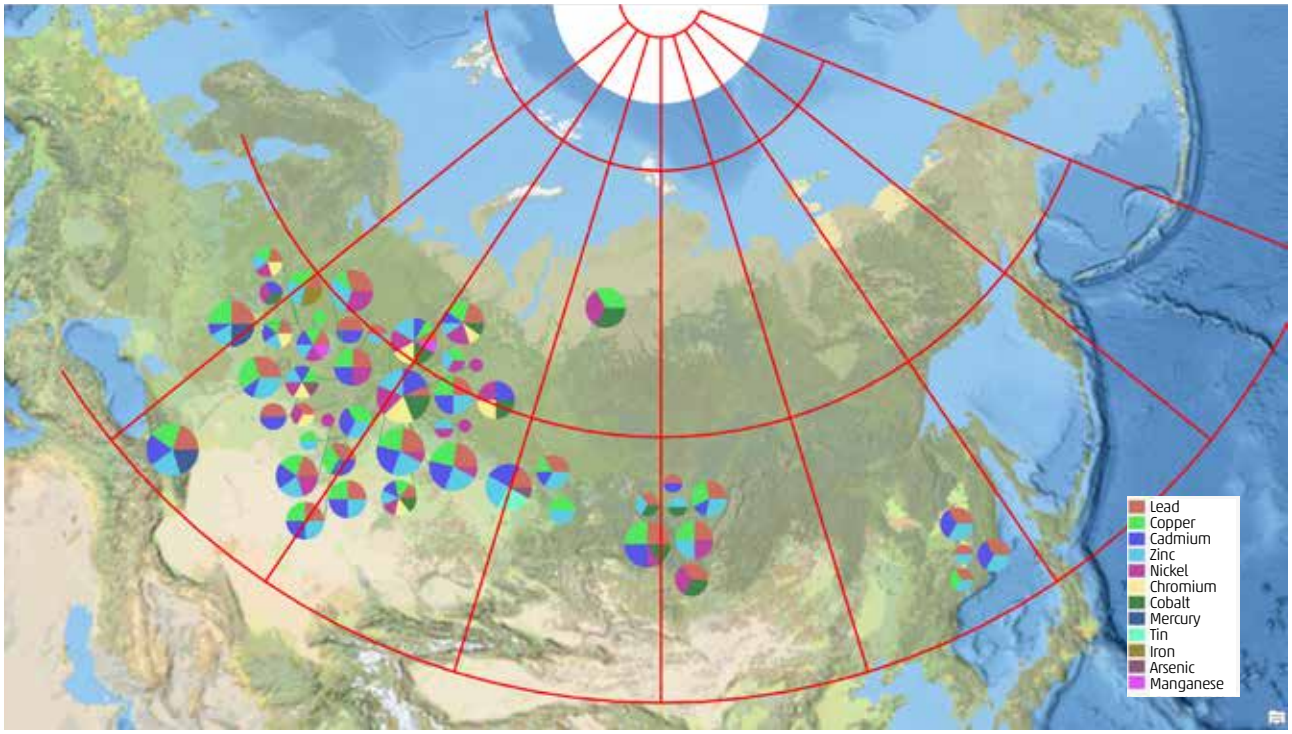
Figure 9.15. Soil pollution (sulphates, nitrates, benzo(a)pyrene, petroleum products, fluorine and aluminium) in the cities and industrial centres of the Russian Federation



Refer to the disclaimer on page ii for the names and boundaries used in this map

◊ *Source*: Author's own elaboration adapted from **NPO Taifun**. 2023. *Soil Contamination of the Russian Federation with Toxicants of Industrial Origin in 2022 – Yearbook*. Obninsk, Russian Federation, Federal State Budgetary Institution.

Figure 9.16. Soil pollution (heavy metals) in the cities and industrial centres of the Russian Federation



Refer to the disclaimer on page ii for the names and boundaries used in this map

◊ *Source*: Authors' own elaboration adapted from **NPO Taifun**. 2023. *Soil Contamination of the Russian Federation with Toxicants of Industrial Origin in 2022 – Yearbook*. Obninsk, Russian Federation, Federal State Budgetary Institution.

To compare the levels of soil contamination by industrial enterprises near sources of industrial emissions with the background mass fractions of the corresponding chemicals, soil samples are taken annually in background areas adjacent to technogenic ones. The background areas are selected to be remote from the source of pollution with soils that are similar to the soils near the source (Ministry of Nature of the Russian Federation, 2023).

Observations of heavy metal soil contamination are carried out mainly in areas with sources of industrial atmospheric emissions of metals (Ministry of Nature of the Russian Federation, 2023), where the source of pollution may be one company, a group of companies or a whole city. When choosing observation points for soil pollution with heavy metals, priority is given to areas with non-ferrous and ferrous metallurgy, mechanical engineering and metalworking, fuel and energy, chemical and petrochemical, and construction industries.

The reliable measurement of changes in the levels of mass fractions of heavy metals in soils over an observation period of five years or longer are prevented by the high heterogeneity (spotting) of soil contamination with heavy metals near sources of industrial emissions, the slow process of self-purification, a generally strong adsorption of heavy metals to soils, and other factors. In general, the soils near industrial centres and adjacent areas are contaminated with heavy metals, which accumulate due to pollutants coming from the atmosphere, spills and other discharges.

9.3.8 Salt accumulation in Europe and Central Asia

Soluble salt accumulation, due to unsustainable soil and water management, is a soil degradation threat predominantly in Southern and Central Europe. While many saline areas of primary, natural origin have inherent ecological value (geological, climatic, topographic, and hydrological), secondary salinization occurs as the result of unsustainable human activities (Daliakopoulos *et al.*, 2016). It affects around 4 million ha of European soils (VanCamp *et al.*, 2004), especially in the coastal areas of Southern Europe such as Sicily, Italy, the Ebro Valley in Spain and, more locally, in other parts of France, Greece, Hungary, Italy, Portugal, Romania, and Slovakia (Daliakopoulos *et al.*, 2016). While the precise extent remains uncertain within Europe, the Mediterranean region is most susceptible to soil salinization, with the coastline most at risk (Stolte *et al.*, 2016). The layer presented in the EUSO Soil Degradation Dashboard (EC, 2024) is a first attempt to show where secondary salinization is likely to occur, reflecting irrigation in areas where evaporation exceeds precipitation.

The global map of soil salinity change between 1986 and 2016 shows that the extent of saline, sodic and saline-sodic croplands has increased over time (Ivushkin *et al.*, 2019), resulting in accelerated land degradation and decreased agricultural productivity, consequently jeopardizing environmental and food security (Stavi *et al.*, 2021). While no systematic data on soil salinization trends are available, research indicates that salt accumulation is increasing in Italy and Spain due to large irrigated areas and inadequate drainage systems, in Portugal due to the heightened risk of saline intrusion in coastal areas caused by groundwater abstraction and rising sea levels, and in Cyprus due to mining activities (Tóth *et al.*, 2008).

In Albania, salinization and acidification altogether cover about 15 000 ha, largely due to natural conditions, but with some salinity buildup as a result of poor-quality irrigation water (Vidojevic *et al.*, 2022). In the Western Balkans, salinization and sodicity occur in Mgrich soil (a Chromic Vertisol), locally known as *smonitsa*, which is characterized by a dark colour and high clay content. Smonitsa soils cover large areas (about 10 percent) of the Western Balkans (Albania, North Macedonia and Serbia) (Zdruli *et al.*, 2022) and have been cultivated for many years, despite their poor chemical and physical properties.

Türkiye has salinity and alkalinity issues across approximately 1.5 million ha of land, with 1.1 million ha with saline soils, 390 000 ha with saline-sodic soils, and 10 000 ha with sodic soils (Okur and Örçen, 2020). Coastal waterlogging and salinity induced by seawater intrusion from the Black and Mediterranean seas are threatening coastal areas (where 55 percent of the population lives), in the

face of predicted sea level rise. In this context, coastal area plans are being prepared that consider integrated agriculture and ecosystems, including guiding strategies and objectives (MoAF, 2022).

In Ukraine, soil salinization and alkalization processes are also widespread, being found on 4.1 percent of the country's arable land (Baliuk, Kucher and Nadiya, 2021).

In Armenia (Ararat Valley), while the groundwater level has decreased, the area of salt-affected soils remained constant during the 2015–2023 period at about 40 000 ha (MTAIRA, 2023).

The area of saline and sodic soils in irrigated lands in Azerbaijan decreased between 2017 and 2019 (Table 9.9) because of modern approaches to soil management, the introduction of government projects on soil reclamation, improvements in water collection and drainage systems, and the installation of more efficient sprinkler and drip irrigation systems. As a result, the areas of Solonchak – the most strongly saline soils – have decreased substantially.

Table 9.9. The areas of saline and sodic soils in Azerbaijan in 2017 and 2019

Year	Slightly saline	Moderately saline	Strongly saline	Very strongly saline	Solonchak	Total saline soils	Slightly sodic	Moderately sodic	Strongly sodic	Total sodic soils
2017	373 300	283 100	242 600	178 100	103 700	1 180 800	1 155 800	114 700	8 500	1 278 900
2019	312 300	253 900	241 600	153 200	69 100	1 030 100	927 800	78 900	20 200	1 008 700

◊ *Sources:* **RASCPI (Republic of Azerbaijan State Committee for Property Issues).** 2017. *Information of the Unified Land Fund of the Republic of Azerbaijan.* Baku.

◊ **RASCPI.** 2019. *Information on the Unified Land Fund of the Republic of Azerbaijan.* Baku.

9.3.9 Soil compaction in Europe and Central Asia

Although adverse impacts of compaction are relatively well documented, estimates of the extent and severity of compaction in Europe remain elusive. Information is lacking on compaction risks for different soil climatic zones and cropping systems in Europe and how the risks may evolve due to climate change. The EUSO Soil Degradation Dashboard shows the natural susceptibility of agricultural soils to compaction (Houskova and Montanarella, 2008), with susceptibility to soil compaction increasing with the soil water content. Wet to moist soils are particularly susceptible to compaction when exposed to compaction risk (such as ploughing, tilling or harvesting when large tractors exert pressure on the soil surface). Subsoil compaction is also associated with the use of heavy agricultural machines, especially under wet soil conditions (Piccoli *et al.*, 2022). It is assumed that the increase in agricultural machinery loads has exacerbated the extent and severity of soil compaction (Keller *et al.*, 2019). Although dry regions are less affected by soil compaction, upper subsoil (25 to 70 cm) compaction is a concern across the Mediterranean region (Koue, Balstrøm and Breuning-Madsen, 2008), with some regions of Greece, Italy, Portugal and Spain displaying critically high soil dry bulk densities (Houskova and Montanarella, 2008).

The EEA (2022a) as well as the European Commission (EC) (EC, 2023) have both proposed indicators for monitoring soil compaction. One of the most prominent indicators of soil structure as well as soil compaction is soil dry bulk density, which LUCAS Soil included for the first time in 2018, where 9 000 points were measured. In addition, some European Union member states have undertaken pilot measurements in test regions (Table 9.10). Several countries have computed the susceptibility of agricultural soils to compaction, based on physical parameters including soil moisture, bulk density and texture. A bulk density high resolution map for the EU27 and the United Kingdom has been recently published (Panagos *et al.*, 2024) for different soil depths. In Switzerland, pilot measurements of soil dry bulk density in agricultural soils have shown that bulk density has increased since the 1980s in most measurement sites. Switzerland's national soil report (FOEN, 2017) refers to the increasing compaction of forest soils, estimating that 0.7 percent of forest soils are compacted due to the increased use of heavy vehicles and machinery in forestry operations.

Table 9.10. Available information regarding soil compaction in European Union member states and Switzerland

Country	Map or (pilot) measurements	Year	Comment	Source
Austria		2010	Susceptibility to compaction based on soil physical parameters.	AGES (2011)
Belgium		2014 2016	Wallonia: national estimate for different scenarios. Flanders: 5 pilot measurements in 5 test regions.	D'Or and Destain (2014) and Van der Bolt *et al.* (2016)
Czechia		2005	Map of physical soil degradation threat, including erosion and dryness.	Šarapatka *et al.* (2010)
Denmark		2023 2019 2010–2014	Wheel load carrying capacity for all agricultural soils. National report. Three pilot sites on arable land.	PulidoMoncada, Munkholm and Schjønning (2019), Gomes *et al.* (2023) and Schjønning *et al.* (2023)
Estonia		2008	Eighteen pilot sites including crops and grassland and different soil types	Reintam *et al.* (2010)
France		2010	Sensitivity to compaction for agricultural soils with wheat cultivation.	Lefebvre (2010)
Germany		2015	Map of soil susceptibility to compaction based on soil structure and soil moisture.	Lehmphul (2015) and Schneider and Don (2019)
Greece		2017	Several measurements on different soils regarding bulk density, porosity, compaction, and hydraulic conductivity.	Yassoglou *et al.* (2016)
Hungary		2007	Monitoring of soil compaction of 1.8 million ha of Panonnian region agricultural soils over three decades.	Birkás *et al.* (2009)
Lithuania			Economic quantification of compaction effects on agricultural soils.	Zabrodskyi *et al.* (2021)
Malta		2003 and 2013	Change of bulk density.	Sultana (2015)
Netherlands (Kingdom of the)		2018	National monitoring survey 2018 (bulk density measurements).	Van den Elsen *et al* (2020)
Poland			15% of agricultural land under strong risk of compaction.	Krasowicz *et al.* (2011)
Romania			Study on affected areas in need of mitigation measures.	Croitoru *et al.* (2014)
Slovakia		Five year cycle	Bulk density is monitored every five years.	Kobza (2015)
Switzerland			Pilot measurements of bulk density in agricultural soils in central Switzerland.	FOEN (2017)

Note: = map and = (pilot) measurements.

Sources: See References.

According to Baliuk, Kucher and Nadiya (2021), soil compaction in Ukraine was widespread (>12 million ha) on arable land. The dynamics before 2022 were characterized by an increasing trend in areas with compacted soils. After 2022, large areas were either occupied or subject to military actions and removed from agricultural land use.

In the Western Balkans, the general assessment is that soil compaction is not of great importance on most agricultural lands due to the lower use of agricultural machinery compared to developed agricultural countries. Except for sporadic case studies (such as Jakšić *et al.*, 2021), data on soil compaction in the Western Balkans are scarce. Serbia reports that soil compaction is present, particularly on agricultural land cultivated with heavy machinery (Zdruli, Wojda and Jones, 2022). Unfortunately, there is not sufficient research and data in the countries to estimate the area affected by soil compaction.

9.3.10 Urbanization and soil sealing in Europe and Central Asia

Four percent of the European Union is under urban development, predominantly at the expense of agricultural land (EEA, 2021). Soil sealing due to urban development increased by 2.6 percent between 2012 and 2018 in the EU27 and the United Kingdom, representing 3 581 km^2 of functional urban areas (cities plus commuting zones), which occupy 23 percent of the European Union territory and host 75 percent of its population. Almost 80 percent of urban expansion has taken place in commuting zones. Net change in urban area in the EU27 and the United Kingdom – calculated by subtracting the area of recultivated land from the area of land covered with new infrastructure – amounted to 3 013 km^2, mostly at the expense of croplands and pastures. The major pressure was the expansion of industrial and commercial units, residential areas, and construction sites (EEA, 2021). Soil sealing indicators are defined based on imperviousness and soil sealing data.

The EEA reports on soil surface sealing status and change in Europe triennially, measured by the high-resolution Copernicus imperviousness datasets (Figure 9.17). According to EEA (2023b), 2.72 percent of the European territory (made up of the 38 European Economic Area member countries and the United Kingdom) was sealed in 2016, increasing to 2.95 percent in 2018. The increase in sealed surface between 2016 and 2018 varied by country, with Malta at 5.95 percent, the Kingdom of the Netherlands at 5.08 percent and the United Kingdom at 3.99 percent. In Albania, the change in sealed surface was lower than in other parts of Europe, at 0.66 percent.

Urban expansion is high in Türkiye, with increased urban dwelling and an intensive rural exodus ongoing since the 1950s (Genç *et al.*, 2021), increasing by 33 percent since 1990. For Istanbul, there has been an urban expansion rate of 10 km^2 per year for more than 40 years (Kaya and Curran, 2006). In the early 1980s, the city had 4.7 million inhabitants (Kaya and Curran, 2006), rising to more than 17 million inhabitants reported in the latest census (İnal-Çekiç *et al.*, 2023).

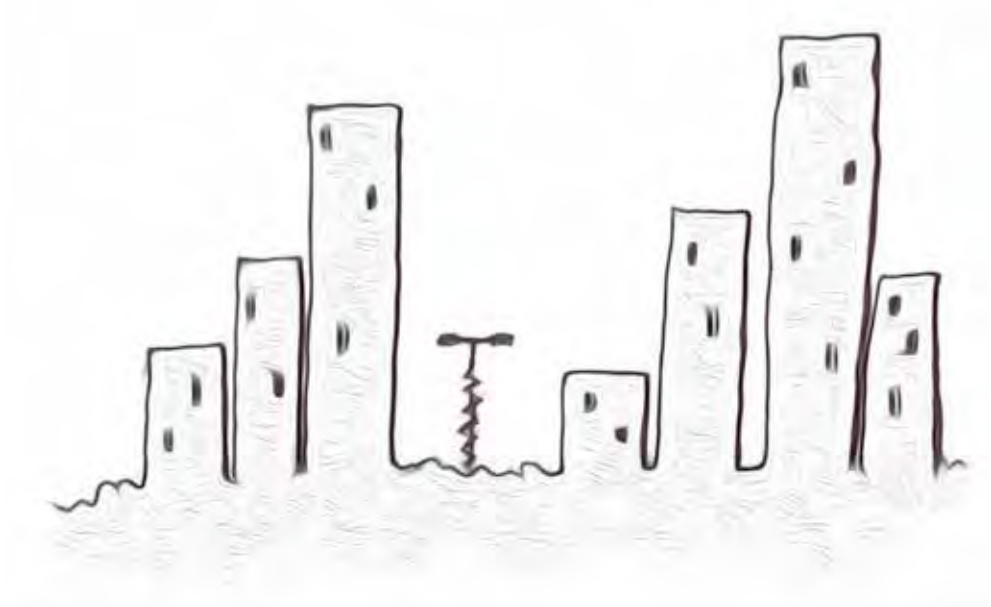
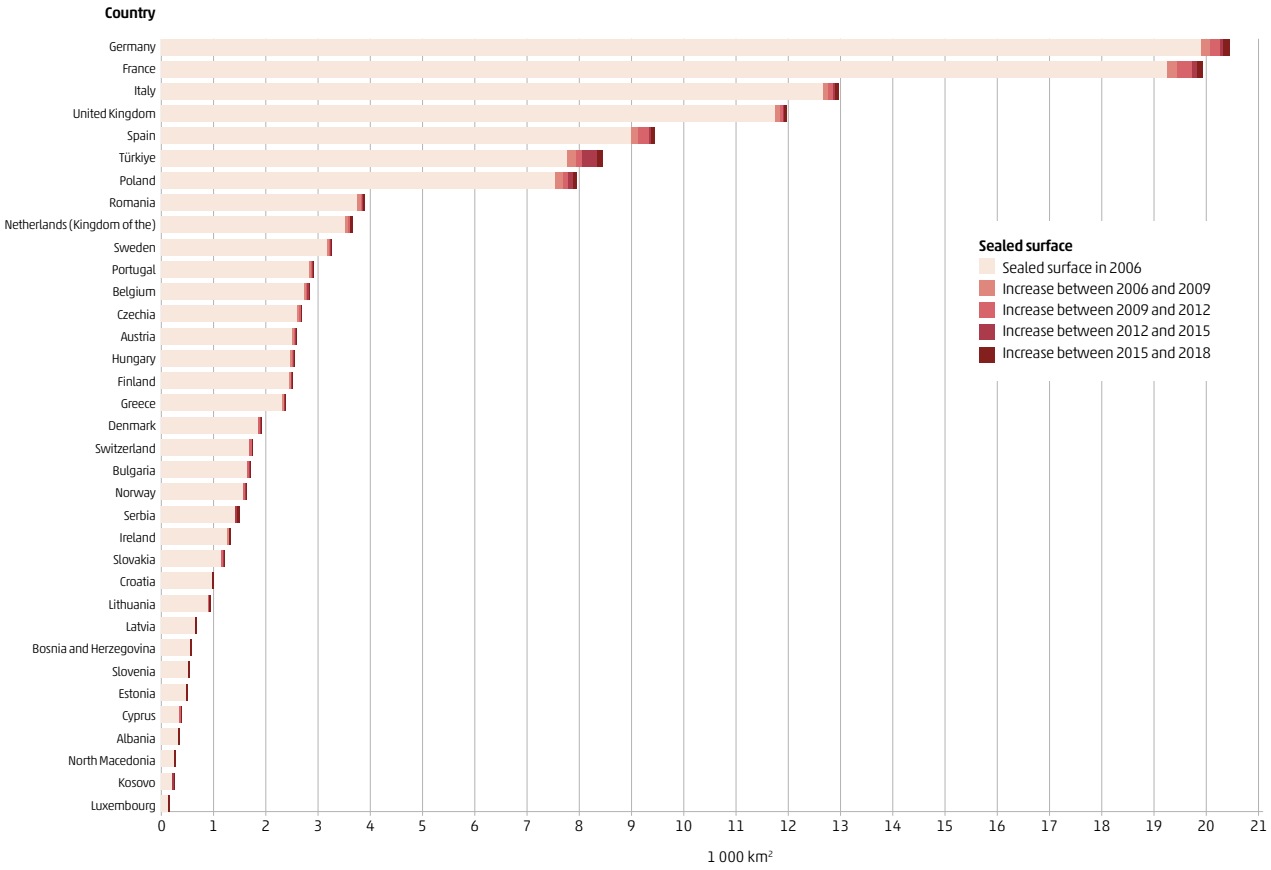


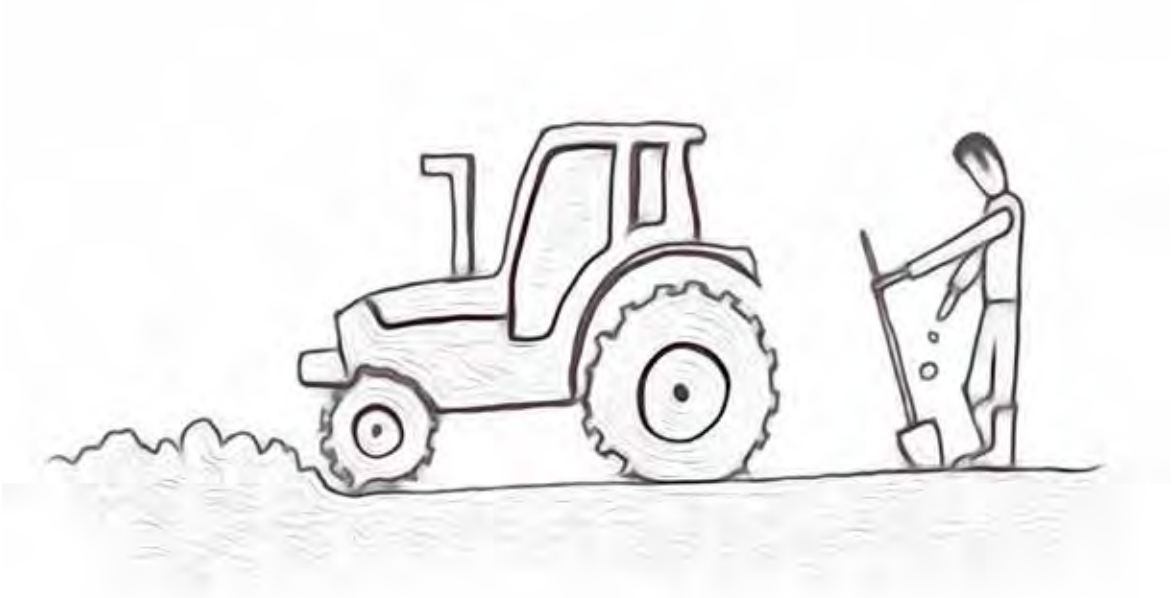
Figure 9.17. Sealed surface increment by period for the 38 European Economic Area countries and the United Kingdom (km²)



◊ *Source*: Authors' own elaboration adapted from **EEA (European Environment Agency).** 2023. *Imperviousness: accounting for the sealed land surface.* [Accessed on 24 August 2024]. https://tableau-public.discomap.eea.europa.eu/views/Imperviousness_time-series_v2023/Imperviousness. Licence: CC-BY-3.0.

For Israel, fast urbanization and the construction of infrastructure has compelled agriculture to exploit marginal erosion-prone soils (Itkin *et al*., 2022).

In Ukraine, developed land falls into several categories, including residential and public buildings, industrial, transport, communications, energy, defence and other purposes, and totals 2.5 million ha (4.1 percent of the country) (Arias-Navarro, Baritz and Jones, eds, 2024). Ukraine is one of the top thirty most urbanized countries globally, according to the UN (2018) and there was a marked increase of soils affected by soil sealing and urbanization from 2015 to 2018.



9.4 Trends in sustainable soil management in Europe and Central Asia

9.4.1 Examples of promising trends in sustainable soil management

Europe has been actively engaged in addressing soil management issues within both national and international frameworks. The adoption of policies under the Common Agricultural Policy (CAP) in European agricultural lands is directed to ensuring a sustainable agricultural sector, supporting farmers, promoting rural development and ensuring food security. The recent Proposal for a Directive of the European Parliament and of the Council on Soil Monitoring and Resilience (EC, 2023) is an important step forward to protect the soils of the European Union. Although some of the measures, like greening (maintaining soil cover by crop rotation) have contributed to protect the soil against erosion, other measures like set-aside land in some regions have increased forest fire risks.

The European Soil Partnership is a mechanism for SSM, relying on voluntary participation from member countries, support from the EC, and collaboration between national institutions. Successful implementation depends on institutionalization, funding availability and policy updates. Subregional partnerships like the Eurasian, Alpine and Pyrenean soil partnerships (EASP, AlpSP and ASPyr, respectively) have had success by developing funded projects directed to increase soil information and social awareness of the importance of soils. Regional implementation of the VGSSM (FAO, ITPS and GSP, 2017) requires cooperation between ministries at national and local levels. While voluntary approaches like that taken by the European Soil Partnership complement soil conservation efforts in Europe, a balance between voluntary and mandatory approaches is essential for effective regulation (Erdogan *et al*., 2021).

Azerbaijan had an increase in protected forest area from 794 000 ha in 2015 up to 893 000 ha in 2023 (RAMENR, 2024), with protected areas including 44 nature reserves.

Belarus has a stable structure of land uses which has developed historically, but changes are gradually taking place. The main trend in recent years is a decrease in the share of agricultural land and an increase in forestland through the transfer of unproductive lands to less intensive land use – especially based on the valuation of agricultural lands – as well as through use of land for construction.

Data on areas with organic farming in the total area of agricultural land are contradictory. According to the National Agency of Investment and Privatization of Belarus (NAIP), in 2020 there were 45 producers of organic agricultural products in Belarus (NAIP, 2021). Since 2017, this figure has increased by 2.5 times. Despite this, as of 2020, Belarus has the fewest producers and processors of organic products among the countries of the Eurasian Economic Union. Most producers are represented by private farms and household plots. According to the National Committee of Statistics of Belarus (Belstat) in 2022, the gross harvest of organic products of grain and leguminous crops (in weight after processing) in agricultural and peasant farms was 1 032.5 tonnes (159 percent increase compared to 2021) with a yield of 1.57 t/ha (0.4 t/ha increase) (Belstat, 2022). According to operational data from agricultural reports (the Ministry of Natural Resources and Environmental Protection and the Ministry of Agriculture and Food, personal communications, 01 September 2023), in 2022, precision farming was employed in cropland growing spring grain and leguminous crops, with it being applied to 17 percent of the area when sowing, 20 percent of the area when applying mineral fertilizers and 26 percent of the area when performing chemical treatments.

Likewise, for cropland growing winter grain, precision farming was applied to 17 percent of the area when sowing, 18 percent of the area when applying mineral fertilizers and in 11 percent of the area when performing chemical treatments.

In Kazakhstan, according to the Government Decree of September 1, 2023, the Republican State Enterprise, “State Institute for Land Survey Work” of the Committee for Land Resources Management of the Ministry of Agriculture was created to combat land degradation, and SOC and nutrient loss (Government of Kazakhstan, 2023). The entity will perform land accounting and carry out the digitalization of soil resources.

Since 2012, the area of agricultural land in the Russian Federation with no-till technology has increased threefold, to about 5 million ha in 2023 (Belyaev *et al*., 2022). Mulching, a transitional stage to the use of no-till, is used on 15 million ha (Belyaev *et al*., 2022). Monitoring on Chernozems with no-till has confirmed the accumulation of organic C in the soil with an average reduction in annual CO_2 emissions of about 1 t/ha (Orlova *et al*., 2023).

Progress has been done in mapping soil nutrients in Türkiye (MoAF, 2021). Outputs from collaborative public research projects (Nepiz, 2023), aimed at determining the sensitivity of waterbodies to agricultural pollution, are identifying potential adverse effects of nutrient excess. Moreover, government subsidies for soil analyses in Türkiye, which have been ongoing for approximately 20 years, are considered the most significant decision-making option to promote the sustainability of nutrient management.

Türkiye is among the top three countries with the most afforestation in the world. Therefore, there has been a decrease in soil erosion, from 500 million tonnes per year in the 1970s, to 140 million tonnes per year in the 2020s (TÜBİTAK BİLGEM, 2018).

In Israel, developing and implementing new agricultural strategies, such as conservation and regenerative agriculture, can limit soil disturbance, nurture the soil with organic C and sustain biodiversity, reducing GHG emissions and enhancing resilience to climate change (Eshel *et al*., 2023). Continuous reclaimed wastewater irrigation in soils can act as a sink for persistent wastewater-derived chemicals (Ben Mordechay *et al*., 2021, 2022).

Figure 9.18 shows the state and trend of the effectiveness of soil management practices in combating soil threats in the Europe and Central Asia region, with Table 9.11 exploring this in more detail.

The state reflects the effectiveness of management practices in place as of 2015 in preventing or reducing a specific soil threat. At the regional or subregional scale, changes in management occur at decadal or longer time scales and hence the state as of 2015 is relevant for the 2015 to 2024 assessment period.

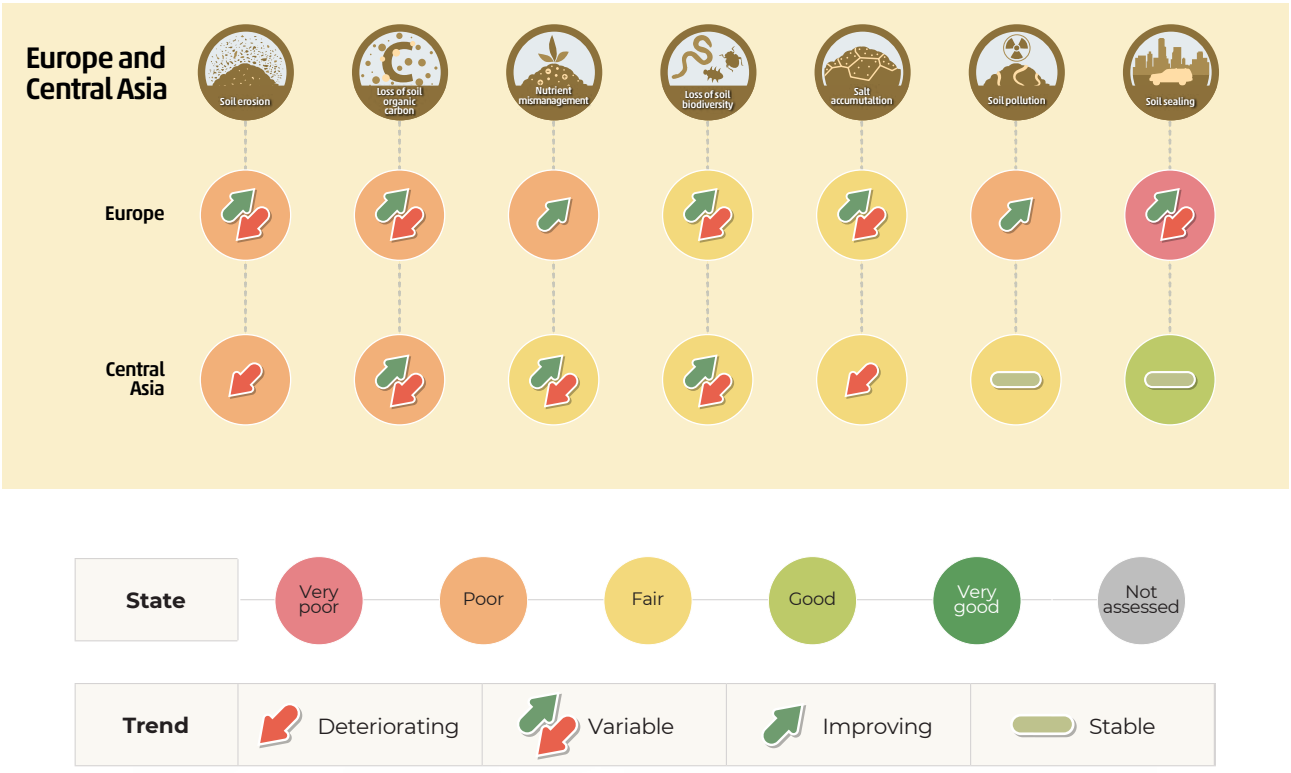
The trend indicates the direction of change in the adoption of SSM practices between 2015 and the finalization of this report (late 2024). In short, trend reflects the how SSM adoption has progressed over time.

Compared with the equivalent diagram for Europe and Eurasia in 2015 (FAO and ITPS, 2015), there is a noticeable increase in confidence regarding the soil condition and trend, owing to the generation of soil information for the region since 2015. Despite this progress, soil biodiversity remains the soil function with the least confidence due to a lack of data. Soil sealing continues to pose the greatest threat to European soils, with few to no effective measures to prevent urbanization. Despite the efforts of European Union policies to combat soil erosion, implementing soil conservation measures has led to varying effects depending on the region, sometimes offset by actions such as the removal of traditional terraces in Southern Europe to facilitate the use of machinery. This is likely contributing to an increased degree of soil compaction, although confidence in this assessment is low.

Erosion and the decline of SOC are significant threats to soils in the eastern part of the region, particularly in agricultural areas under intensive cultivation. In the driest regions, climate aridification has lowered water tables and drained meadow soils, leading to SOC loss and increased salinity. Despite these challenges, many countries have implemented soil conservation programmes that are enhancing soil conditions in the areas where they are applied.

However, threats such as changes in SOC, nutrient mismanagement and pollution remain largely stable, even with the emergence of new pollutants like bioplastics. This stability indicates that current soil management practices aimed at addressing these threats have not yet succeeded in significantly improving soil conditions.

Figure 9.18. Assessment of state and trend of threats to soil functions in Europe and Central Asia through a management-based approach



Source: Authors' own elaboration.

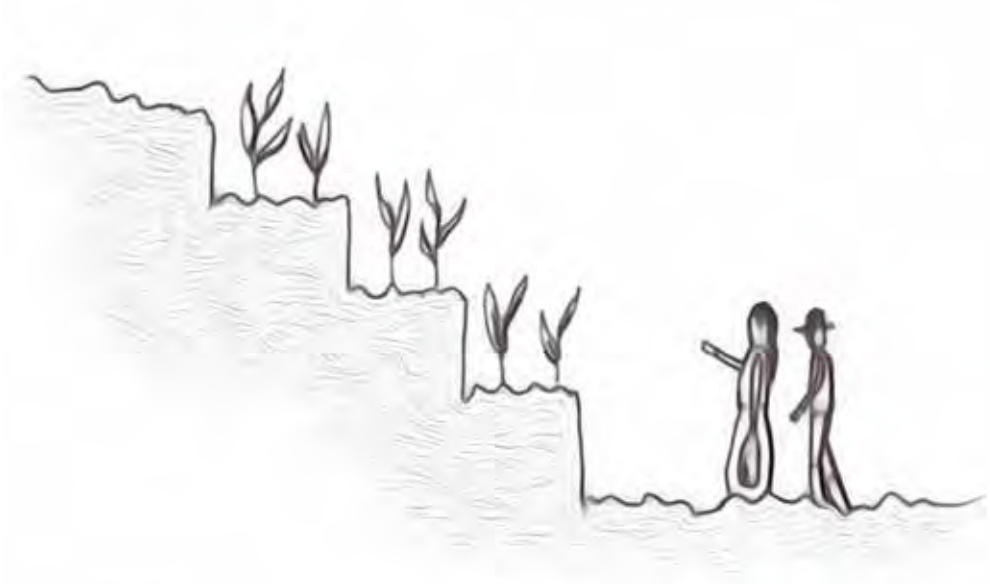


Table 9.11. Summary of threats to soil functions in Europe and Central Asia and the confidence in the assessment of their state and trend

Threat to soil function	Summary/comments	Confidence	
		Condition	Trend
Erosion	Soil erosion affects half of EU agricultural land and substantial part of agricultural lands in other parts of the region. Common Agricultural Policy measures have been positive to reduce erosion in some areas of the EU. Despite soil conservation measures, erosion of black soils under intensive farming is still seriously increasing in Central Asia	High	High
Soil organic carbon change	Improving in mineral soils due to reforestation in spite of carbon losses in agricultural soils; but still high losses in drained peatlands. Good condition in boreal soils, poor in more southern latitudes.	Medium	Medium
Urbanisation and soil sealing	Contrasting state and trends: sealed soils occupy 4.2% EU territory with declining trend, while in other parts of the region including Central Asia, soil sealing by urbanisation mostly affect suburban areas and did not experience strong changes at the national level	High	High
Soil pollution	Widespread heavy metal and biocide pollution, but improving due to less inputs. Microplastics and emerging organic compounds are being revealed as common in soils. In general better condition in forest than in agricultural soils. Radionuclide pollution is still a severe threat in some regions of Belarus and Ukraine.	Medium	Medium
Nutrient mismanagement	N, P excess is still a problem in W Europe due to over fertilizing, while nutrient deficit affects some other areas of the region. Improvement thanks to precision fertilization and increased circularity.	High	High
Salt accumulation	Locally highly affected soils in Southern Europe and widespread in Central Asia, and possible threat due to climate change in inland water scarce and coastal areas in the future. In spite of better irrigation technology preventing salinisation, climate aridification is causing salt accumulation in some soils.	Medium	Medium
Soil biodiversity change	Decreasing in many places due to intensive agriculture, but increasing due to land abandonment, following soil organic carbon change.	Low	Low
Other: Compaction	Farm machinery continues to get heavier. Intensive farming continues, but lack of objective data prevents a precise assessment.	Low	Low
Other: Waterlogging, acidification	Local degraded areas by acidification or waterlogging.	Medium	Medium
Other: Armed conflicts	Severe local soil degradation in the areas affected by armed conflicts.	Medium	Medium

Note: The assessment is based on the evidence provided in this chapter. The threats are presented in the order of most severe threat to least severe.

9.4.2 Impediments to adoption of sustainable soil management

Despite many efforts, challenges persist in achieving SSM goals in Europe and Central Asia. While some regions have succeeded in implementing effective soil management practices, others face difficulties due to factors like insufficient soil knowledge, economic constraints and conflicts. In addition, soil degradation resulting from armed conflicts poses significant environmental and agricultural challenges, highlighting the need for comprehensive soil management strategies in conflict-affected regions.

In Europe, the adoption of CAP faces several challenges and barriers due to the complexity of regulations, lack of awareness, financial constraints, limited technical support and regional disparities, to the point that farmers are still valuing the benefits regarding improved yields more than any environmental and social benefits (Fantappiè *et al.*, 2020). Addressing these challenges requires a comprehensive approach that includes improved communication, targeted support services, financial assistance, and tailored incentives to encourage farmers to adopt and benefit from CAP policies effectively.

Despite some positive developments, Türkiye's agricultural environmental performance post-2000 appears unsustainable, since support for farmers has not yet induced improvements in agricultural practices. Reasons include legislative uncertainties, inadequate support for costs, lack of environmental training and ineffective organization of agricultural producers (Yeni and Teoman, 2022).

The major threats to soil functions in Israel are soil degradation due to intensive agriculture activities, irrigation with water of lower quality, limited soil resilience under climate changes, soil loss due to urbanization and infrastructure, as well as insufficient education and low public awareness of soil health (Itkin *et al*., 2022). Intensive modern agriculture also further accelerates soil degradation rates (Eshel *et al*., 2023).

9.4.3 Conclusion and recommendations

Addressing the challenges of SSM requires capacity-building, research, cross-sectoral collaboration, and the development of appropriate policies and support systems. Despite the wealth of soil information generated in the European Union in the 2015 to 2025 period, it is not yet sufficient for a full diagnosis of the state of the soil condition that can be used to design effective policies. Soil information is essential to implement strengthened policies and to improve knowledge dissemination. Adequate financial and administrative support for farmers is also needed to promote SSM. Furthermore, the inclusion of soil degradation and SSM in international funding mechanisms and the establishment of national soil management units can contribute to enhancing soil health at all levels.

The following is a list of recommendations for SSM in the region.

Policies and regulations:

- Increase funding and financial support for SSM initiatives and programmes, particularly in less developed regions.
- Encourage cross-sectoral collaboration and partnerships between farmers, researchers, policymakers and other stakeholders to develop and implement effective soil management strategies.
- Facilitate the adoption of SSM practices by reducing bureaucracy and by developing practices that ensure regular crop yields, creating a better stimulus to farmers to adopt them.
- Prioritize aid to small family agricultural businesses to increase crop diversity, promote local food and ensure permanence in rural areas.

Land-use planning:

- Preserve soils that are best suited for biomass production, such as irrigated land or areas with black soils through direct efforts in land-use planning (biomass production is one of the most important soil ecosystem services to ensure food sovereignty).
- Soil preservation is undervalued over other soil uses (such as urban, energy and tourism).
- Develop integrated policies that address SSM alongside other environmental challenges, such as biodiversity loss, climate change, and water quality and quantity.

Soil information and monitoring:

- Enhance technological infrastructure and data collection methods to establish comprehensive soil monitoring networks, especially across Eurasia, facilitating real-time assessment of soil health indicators and trends.

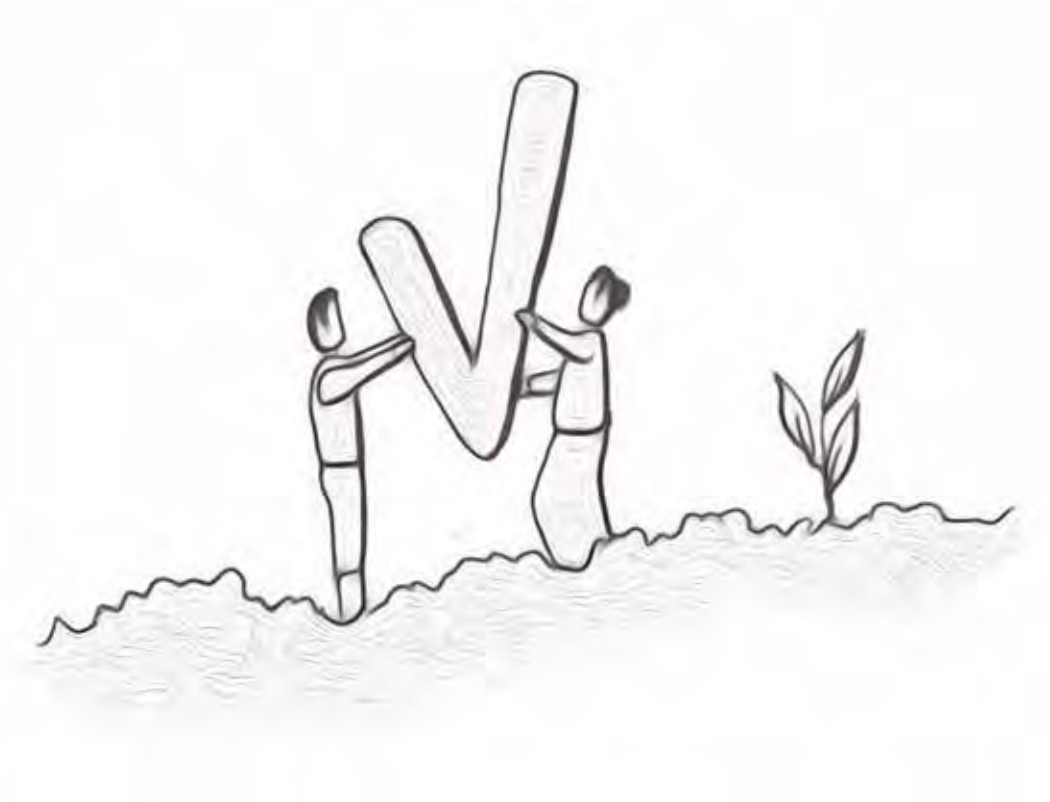
- Promote the use of technology and innovation in soil monitoring and management, including remote sensing, data analytics and precision agriculture, to enhance soil health and productivity.
- Encourage research on soil health and sustainable management practices, including the impacts of different land-use practices, crop rotations and organic amendments on soil quality.
- Incorporate soil health indicators into national and regional environmental assessments to track progress and inform policy adjustments.

Social soil awareness:

- Implement capacity-building programmes and knowledge transfer initiatives to enhance awareness and adoption of sustainable agricultural techniques by farmers, fostering resilience and productivity while mitigating soil degradation risks.
- Strengthen international cooperation on soil health and SSM, including sharing best practices and knowledge exchange.

Specific soil measures:

- Implement targeted conservation strategies and land-use policies to safeguard black soils (predominantly Chernozems), recognizing their ecological significance and contributions to agricultural productivity.
- Preserve traditional soil conservation measures, such as terraced soils in the Mediterranean, to avoid soil erosion, and to ensure efficient soil water use for plants in response to climate change, adapting new technologies to soil management.
- Implement measures to restore and conserve peatlands, including rewetting and reducing drainage when feasible in forested areas. In situations where full rewetting is not possible, changes in soil water level should be attenuated to mitigate carbon emissions, preserve biodiversity and safeguard ecosystem services for sustainable soil development and climate resilience.



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Chapter 10 | Regional assessment for Latin America and the Caribbean

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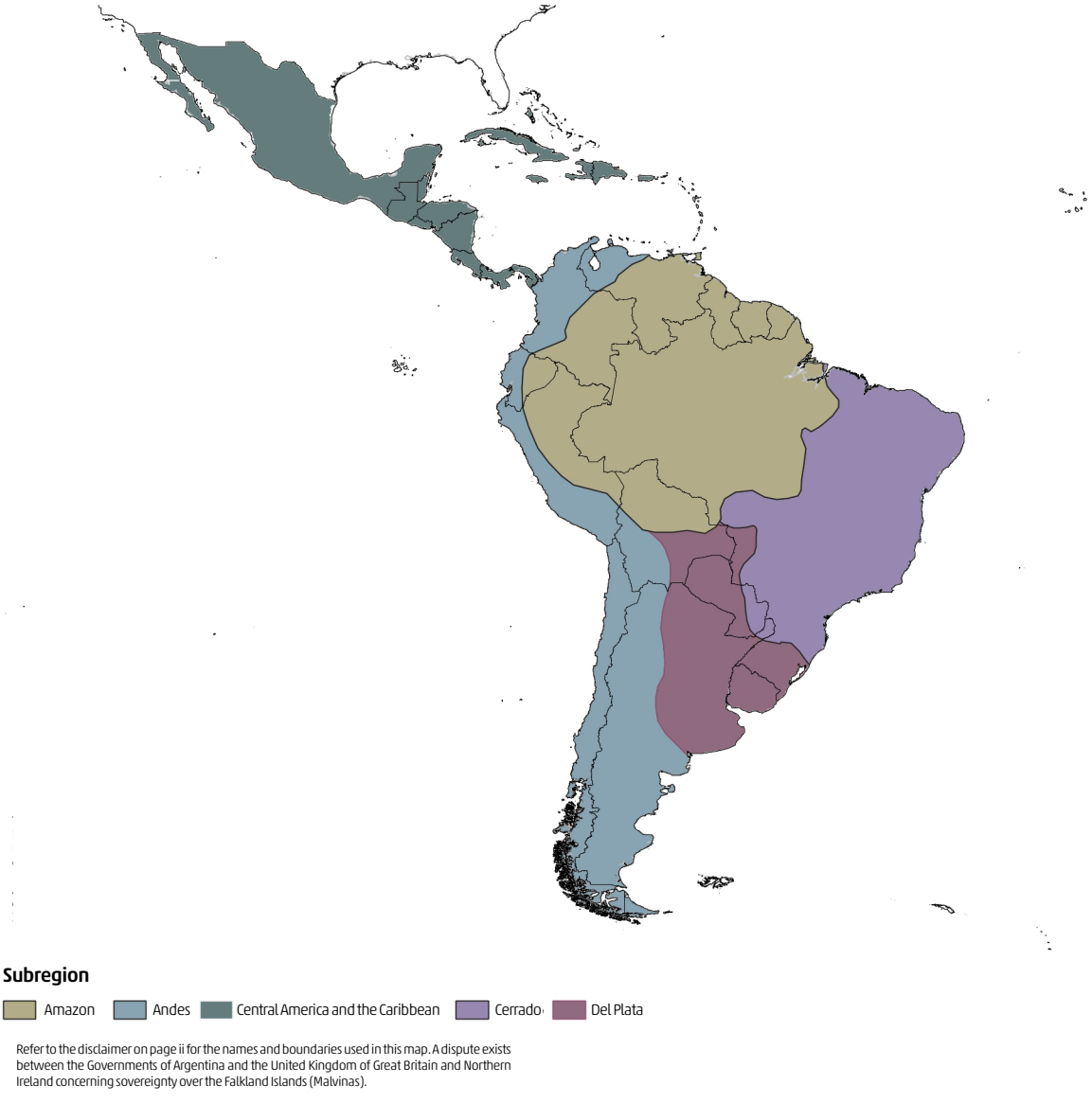
10.1 Latin America and the Caribbean regional overview

10.1.1 Characteristics of region and subregions

This chapter covers Latin America and the Caribbean (LAC), which is subdivided into the subregions of Central America and the Caribbean, Amazon Basin, Del Plata Basin, Andean countries & Patagonia, and Cerrado (Figure 10.1).

The Latin America and Caribbean (LAC) Soil Partnership comprises Argentina, the Bahamas, Belize, the Plurinational State of Bolivia, Brazil, Chile, Colombia, Costa Rica, Cuba, the Dominican Republic, Ecuador, El Salvador, Grenada, Guatemala, Guyana, Haiti, Honduras, Jamaica, Mexico, Nicaragua, Panama, Paraguay, Peru, Saint Lucia, Suriname, Uruguay, and the Bolivarian Republic of Venezuela.

Figure 10.1. Map of the Latin America and the Caribbean region showing countries and subregions



Note: The boundaries of the subregions included in this map have been established for illustrative purposes only and do not correspond to well-defined biomes or watersheds. Some subregions include multiple biomes, but are named after the predominant biome. For more details, please refer to the descriptive text for each subregion.

Source: Authors' own elaboration.

The topographic relief is dominated by the Andes mountain range, which extend along the Pacific Coast, including Argentina's Aconcagua, which rises to an elevation of about 6 961 metres above sea level (MASL). However, the relief is very low in much of the Amazon Basin and in the broad lowlands of the Gran Chaco, Pampas and Pantanal regions. Three large geomorphic areas were defined in the *Soil Atlas of Latin America and the Caribbean* (EC and FAO, 2015):

- the large shields or massifs (the most ancient and stable geologic areas);
- the Andes Mountain range (the longest in the world, extending from the Bolivarian Republic of Venezuela to Tierra del Fuego (Chile); and
- the sedimentary basins (Gardi *et al*., 2015).

The biogeographical areas of the LAC are diverse in landscapes, biomes and climates. The lowlands are predominantly tropical and subtropical, while in the highlands, the temperature patterns are more complex, as altitude influences the local temperature as much as latitude. The agroecological zones exhibit temperatures varying from tropical (lowland and highland) and subtropical, to temperate and boreal climates. Moisture regimes also vary, ranging from arid, semi-arid and dry areas, through to moist, subhumid, humid and wet areas (Fischer *et al*., 2021).

The LAC region is divided into five subregions based on geography and the relevance of the biomes in terms of agricultural production:

1. Central America, southern Mexico and the Caribbean;
2. Amazon;
3. Cerrado;
4. Del Plata Basin; and
5. Andean countries and Patagonia.



Figure 10.2. Land cover map of Latin America and the Caribbean



Refer to the disclaimer on page ii for the names and boundaries used in this map. A dispute exists between the Governments of Argentina and the United Kingdom of Great Britain and Northern Ireland concerning sovereignty over the Falkland Islands (Malvinas).

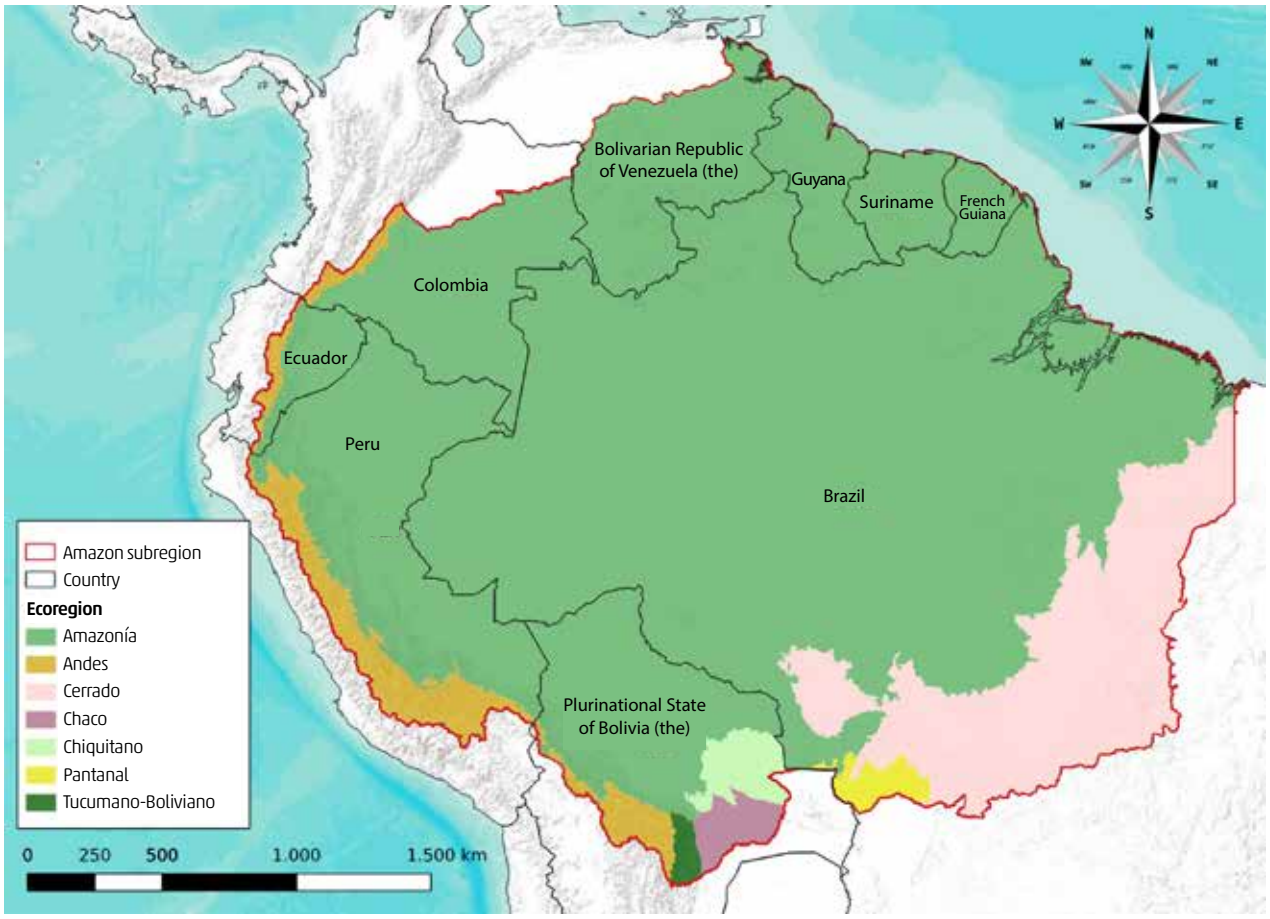
◊ *Source*: **FAO**. 2024. *GAEZ: Land and Water Resources*. [Accessed 24 October 2024] https://gaez-services.fao.org/apps/theme-1/. Licence: CC-BY-4.0.

The Central America, southern Mexico and the Caribbean subregion forms a land bridge connecting North and South America, with the Pacific Ocean to the west and the Caribbean Sea to the east. Mexico covers the largest area of this subregion. The landscape varies from the central mountain chain that dominates the interior from Mexico to Panama, to the coastal plains and the islands. Climates are essentially tropical or subtropical but vary substantially over short distances. In the subregion, cropland is concentrated on the east coast of Mexico and the west coast of Central America (Figure 10.2). At the broad scale shown in Figure 10.2, the cropland areas of Cuba

and Haiti are also evident. Tropical rainforests occupy the eastern lowlands, and evergreen forests cover the lower slopes along the Pacific coast, with forests of silver fir (*Abies* spp.) in the higher parts of the mountain range (*cordilleras*). The semi-arid and arid zones are mainly in the north and central regions (the Sonoran and Chihuahuan deserts and central highlands). The Caribbean island ecosystems have a great biodiversity and a variety of coastal and marine habitats, from rainforests, montane formations, and seasonal and dry formations to swamps and mangroves.

The Amazon subregion has been defined by the Amazon Network of Georeferenced Socio-Environmental Information (RAISG and MapBiomas, 2022) and is known as the Pan-Amazon, spanning an area of 8 470 209 km^2 across nine South American countries: the Plurinational State of Bolivia, Brazil, Colombia, Ecuador, French Guiana, Guyana, Peru, Suriname, and the Bolivarian Republic of Venezuela. Figure 10.3 shows the ecoregions present in the subregion, while Figure 10.4 shows the river basins. The Amazon subregion is dominated by tree cover, although with areas of cropland along the southern and eastern borders (see Figure 10.2).

Figure 10.3. The ecoregions of the Amazon subregion

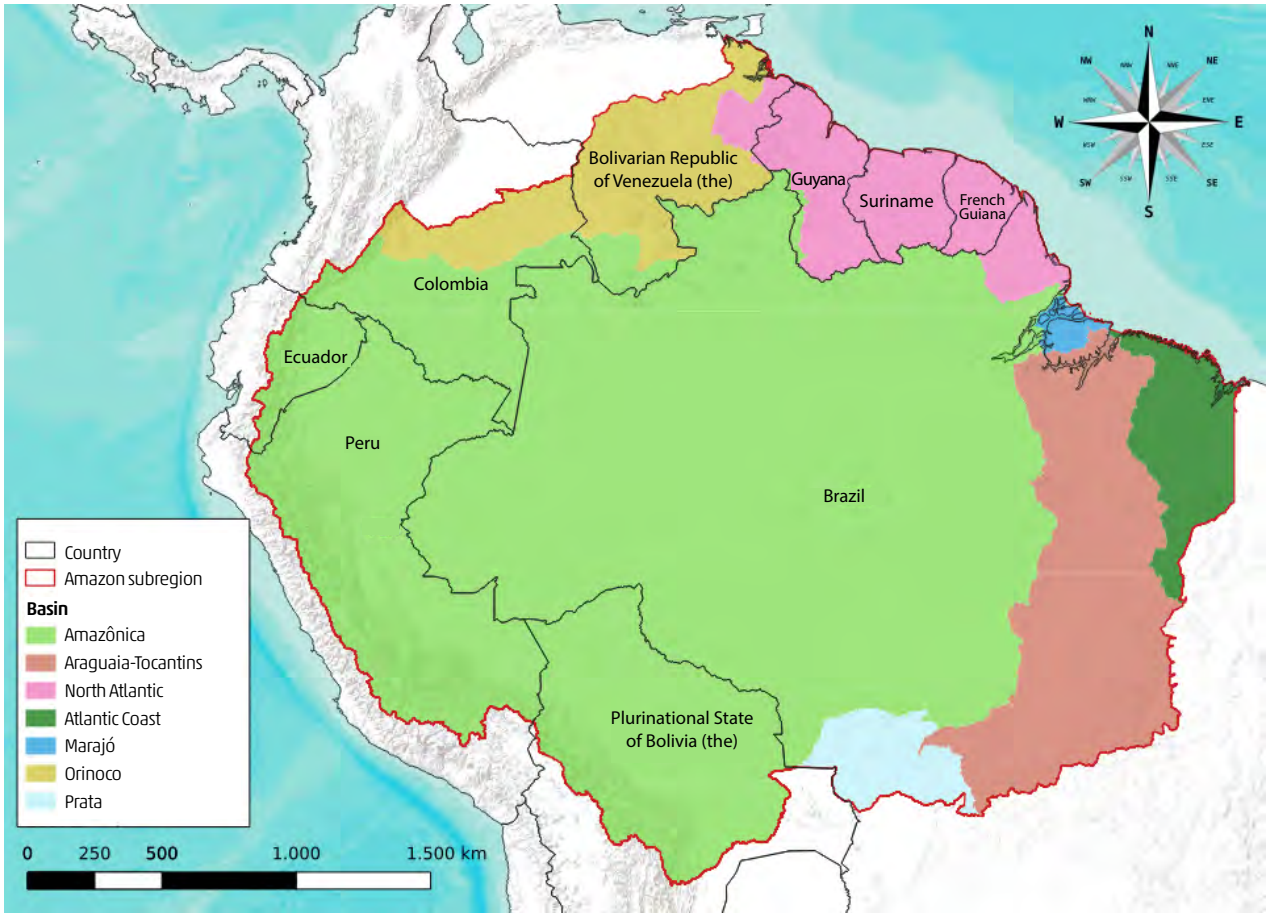


Refer to the disclaimer on page ii for the names and boundaries used in this map

Notes: BOL = Plurinational State of Bolivia], BRA = Brazil, COL = Colombia, ECU = Ecuador, GUF = French Guiana, GUY = Guyana, PER = Peru, SUR = Suriname, and VEN = Bolivarian Republic of Venezuela.

◊ *Source*: The original MapBiomas database used to create this figure can no longer be accessed. The map and accompanying data should be accessed with **MapBiomas Amazonía**. 2025. *MapBiomas Amazonía Collection 6*. [Accessed on 4 April 2025]. https://plataforma.amazonia.mapbiomas.org/cobertura?activeBaseMap=9&layersOpacity=100&activeModule=coverage&activeModuleContent=coverage%3Acoverage_main&activeYear=2023&mapPosition. Licence: CC-BY-SA.

Figure 10.4. The river basins of the Amazon subregion



Refer to the disclaimer on page ii for the names and boundaries used in this map

Notes: BOL = Plurinational State of Bolivia], BRA = Brazil, COL = Colombia, ECU = Ecuador, GUF = French Guiana, GUY = Guyana, PER = Peru, SUR = Suriname, and VEN = Bolivarian Republic of Venezuela.

The seven constituent basins are according to the definition from the Amazon Network of Georeferenced Socio-Environmental Information (RAISG).

◊ *Source*: The original MapBiomas database used to create this figure can no longer be accessed. The map and accompanying data should be accessed with **MapBiomas Amazonía.** 2025. *MapBiomas Amazonía Collection 6.* [Accessed on 4 April 2025]**.** https://plataforma.amazonia.mapbiomas.org/cobertura?activeBaseMap=9&layersOpacity=100&activeModule=coverage&activeModuleContent=coverage%3Acoverage_main&activeYear=2023&mapPosition. Licence: CC-BY-SA.

The **Amazon subregion** encompasses seven ecoregions with distinct coverages, as well as varying phenological and orographic characteristics. About 83 percent of the area is covered by the rainforests of the Amazon ecoregion, followed by the ecoregions of the Cerrado (11.3 percent), Andes (3.6 percent), Chiquitano (1.0 percent), Chaco (0.8 percent), Pantanal (0.6 percent), and Tucumano-Boliviano (0.3 percent) on the ridges. The major river basins are the Amazonas (70.2 percent) and the Araguaia/Tocantins (11.1 percent), as well as the North and Northeast Atlantic coastal basins (10.3 percent), southern parts of the Orinoco River (6.3 percent), and northern parts of the Prata River (2.1 percent). The dominant soil orders of the Amazon subregion are Ferralsols and Plinthosols (Figure 10.5).

The **Cerrado subregion** covers about 2 million km² across South America, primarily in Brazil with about 1.9 million km² (or 95 percent of the biome), and smaller areas of Paraguay and the Plurinational State of Bolivia. The Cerrado is the second-largest biome in South America and Brazil, with a wide range of vegetation (grasslands, savannahhs, and dry forests). The main soil groups are Ferralsols and Acrisols (Figure 10.4) with Gleysols adjacent to river channels. The Cerrado subregion also has one of the largest areas of contiguous cropland in the region (See Figure 10.2).

Figure 10.5. Soil map of Latin America and the Caribbean



Source: **FAO**. 2024. *GAEZ: Harmonized World Soils Database version 2.0*. [Accessed 24 October 2024] https://gaez.fao.org/pages/hwsd. Licence: CC-BY-4.0.

The **Del Plata Basin subregion** represents the soils of the Pampas biome, and is almost completely covered by agriculture (crops, pasture and forest). The Del Plata Basin biome is comprised of central northeastern Argentina, eastern Paraguay, most of Uruguay, and Brazil's southernmost state, Rio Grande do Sul. The landscape has vast plains that were originally grasslands before cultivation began and a semi-arid to humid climate (with a mean annual rainfall above 600 mm). Part of the Del Plata Basin corresponds to the Grand Chaco biome in the humid subtropical area. The Pampas in Argentina are another large area of contiguous cropland in the region (See Figure 10.2). The dominant soils of the Pampas are Phaeozems (Figure 10.5) with Acrisols to the northeast and Solonetz soils in several areas.

The **Andean countries and Patagonia subregion** is marked by the Andes mountain range. It covers an area of 2 870 000 km^2, with average elevations of 3 000 to 5 000 MASL and three mountain ranges (the Aconcagua, Nevado Ojos del Saldo and Monte Pissis) with peaks of maximum elevation at around 7 000 MASL. The Andes stretch for 8 500 km, from the western parts of the Bolivarian

Republic of Venezuela to the island of Tierra del Fuego, passing through the territories of the Bolivarian Republic of Venezuela, Colombia, Ecuador, Peru, the Plurinational State of Bolivia, Chile, and finally Argentina. The steep topography corresponds to a vertical climatic zone with various vegetation types and ecoregions. One ecoregion is the Central Andean *puna* with high elevation and montane grassland in the southern high Andes, extending from southern Peru, through the Plurinational State of Bolivia into northern Argentina (See Figure 10.3). There are also large areas of sparse vegetation or bare soil in the Atacama Desert in the north of Chile (See Figure 10.2). The subregion contains nearly unaltered blocks of habitat, representing an important area for the conservation of endemic species. Thin Leptosols and Regosols dominant in mountain areas with Calcisols in desert areas. and Andosols in southern Chile (See Figure 10.5)

The **Patagonia part of the Andean countries and Patagonia subregion** covers over 1 million km^2, shared between Argentina and Chile, in the southern tip of South America, and is naturally split by the Andes. It comprises of the southern section of the Andes, with lakes, fjords and glaciers in the west, and deserts, tablelands and steppes to the east (See Figure 10.2), associated with the Andes range, which forms an orographic barrier. Patagonia is bounded by the Atlantic Ocean to the east and the Pacific Ocean on the west, both connected by the Strait of Magellan, the Beagle Channel and the Drake Passage to the south. Thin Leptosols and Andosols dominant in western parts of this subregion and Solonetz, Luvisols and Cambisols in Argentinian areas (See Figure 10.5).

10.1.2 Status summary from the 2015 Status of the world's soil resources report

The Latin American authors of the 2015 SWSR report (FAO and ITPS, 2015) assessed that the top four threats (soil erosion, SOC change, salinization and sodification, and nutrient imbalance) were in poor condition in the region and all showed a deteriorating trend. The fifth-ranked threat, the loss of soil biodiversity, was judged to be in fair condition with a variable trend.

Overall, the authors stated that the high rate of conversion of natural ecosystems (grass–shrub–savannahhs and forest) to agricultural land was the major driver for the poor condition of many of the soil threats, with the negative effects of deforestation being most pronounced in the humid tropical region. Conversion of natural ecosystems was the major causal factor behind higher water erosion and increased land surface instability, losses of SOC and loss of soil biodiversity. The rates of water erosion and the incidence of landslides were high in the mountainous regions, especially where the vegetation had been burned or the lands overgrazed. Wind erosion was estimated to be high in the drier parts of the region.

Salinization was attributed to inadequate irrigation technology and some forms of land-use conversion. Although natural salinization and sodification were generally widespread in arid and semi-arid areas, degradation due to these processes was also occurring in humid climates due to upward salt movement caused by land clearing and overgrazing (FAO and ITPS, 2015).

Problems due to nutrient imbalances were recorded as being widespread in Latin America and the Caribbean. The largest proportion of soils in this region were found to be acidic with naturally low fertility and hence substantial inputs of N, P and K were required along with lime addition to offset the high acidity. The historical low levels of inputs were offset in some regions due to biological N fixation associated with soybean production and legume pastures.

10.2. Changes in the direct and indirect drivers of threats to soil functions

The links between the drivers and the threats to soil functions are discussed in Chapter 4 of this volume.

10.2.1 Population growth

For both the 2021–2025 and 2025–2035 periods, population growth in Latin America and the Caribbean is generally predicted to be at the global average for Central America, and below the global average for the Caribbean and South America (UN DESA, 2023). Of the most populous countries, only Cuba is projected to experience a population decrease while the Plurinational State of Plurinational State of Bolivia, Paraguay and the Bolivarian Republic of Venezuela are estimated to have population increases greater than 1 percent per year in the two time periods (Castaneira *et al.,* 2022).

10.2.2 Land-use change

In South America, several countries have experienced substantial increases in agricultural land and cropland area, with Argentina, Bolivia, Colombia and Paraguay having high increases in cropland area (Figure 10.6) (FAO, 2024a). Potapov *et al.* (2021) assessed the specific land-use conversions associated with cropland expansion, estimating that for South America over the 2000–2019 period, 61 percent of the cropland gain came from the conversion of pasture and recultivation of abandoned agricultural lands, and 39 percent from the conversion of natural vegetation or tree plantations. The majority of countries in Central America and the Caribbean had no change or only small gains in cropland in the 2017–2022 period.



Figure 10.6. Country map of changes in cropland in Latin America and the Caribbean in the 2015 to 2022 period



◊ *Source*: **FAO**. 2024. *FAOSTAT: Land Use*. [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

In the Central America and the Caribbean subregion, Barbados, Guadeloupe, Haiti, Mexico, Puerto Rico and the United States Virgin Islands have experienced substantial decreases in cropland area, as have French Guiana, Suriname and Uruguay in South America (Figure 10.6).

Forestland change

The countries of Latin America and the Caribbean experienced the second-largest net loss of forestland between 2015 and 2022 of any region, with 1 774 000 ha being lost in Central America and the Caribbean and 16 758 000 ha in South America (FAO, 2024a). Only sub-Saharan Africa (with a net loss of 26 722 000 ha) experienced higher net losses (FAOS, 2024a).

Five countries in South America experienced net losses in excess of 1 million ha in the 2015–2022 period, with Brazil having the highest net loss of any country globally at 9.7 million ha (Figure 10.7). The

2024 *State of the World's Forests* report (FAO, 2024) noted, however, that deforestation decreased by 50 percent in Brazil's legal Amazon in 2023. Two countries (Chile and Uruguay) experienced net forest gain in the period. The Falkland Islands (Malvinas) had no forestland reported during the period.

In Central America and the Caribbean, 12 countries had net losses, 7 had net gains and 11 showed no change in forestland in the 2015–2022 period (FAO, 2024a). The areal loss in Nicaragua (700 000 ha lost) was particularly high (Figure 10.7), as was the areal loss in Mexico (894 000 ha lost). Costa Rica was notable for the high areal gain achieved (a gain of 115 000 ha).

Figure 10.7. Country map of changes in forestland in Latin America and the Caribbean in the 2015 to 2022 period



Source: **FAO**. 2024. *FAOSTAT: Land Use*. [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

Permanent pastures and meadows

Overall, the change in permanent meadows and pastures has been much less in the 2015–2022 period than the change in forestland (Figure 10.8). Substantial areal losses of permanent meadows

and pastures were reported by Brazil, Chile, Columbia and Ecuador. French Guyana showed a very high gain, with lesser gains in Costa Rica, Cuba, Guatemala, Honduras and the United States Virgin Islands, with 25 countries reporting zero change.

Figure 10.8. Country map of changes in permanent meadows and pastures in Latin America and the Caribbean in the 2015 to 2022 period



Refer to the disclaimer on page ii for the names and boundaries used in this map. A dispute exists between the Governments of Argentina and the United Kingdom of Great Britain and Northern Ireland concerning sovereignty over the Falkland Islands (Malvinas).

◊ *Source*: **FAO**. 2024. *FAOSTAT: Land Use.* [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

Conservation agriculture

Central and South America has seen the widespread adoption of conservation agriculture. Kassam,

Friedrich and Derpsch (2022) estimated that by 2018/2019, approximately 83 million ha in Central and South America were under conservation agriculture and that the area under conservation agriculture had increased by 68 percent since 2008/2009. According to the estimates from 2018/2019, Brazil had 43 million ha under conservation agriculture, Argentina had 33 million ha, Paraguay had 3 million ha, the Plurinational State of Bolivia had 2 million ha, and Uruguay had 1 million ha. All are in the top 15 countries globally in terms of conservation agriculture adoption.

Agroforestry and palms

Latin America and the Caribbean has a very limited area dedicated to agroforestry, with estimates of 93 000 ha in South America, 459 000 ha in the Caribbean and 696 000 ha in Central America (FAO, 2020). Together, this represents only 2.7 percent of the global total for agroforestry. Palm production occupies 1 181 000 ha (or 10 percent of global totals), with 1 013 000 ha of palm production occurring in South America (FAO, 2020). Colombia has the highest harvested area for oil palm fruit (almost 500 000 ha) with significant areas in Brazil, Guatemala, Honduras and Mexico (FAO, 2024a)

Urbanization

On average, the countries of Latin America and the Caribbean are highly urbanized compared to the global average. In the Caribbean, in 2015, 70 percent of the population lived in urban areas, with 74 percent in Central America and 84 percent in South America (UN DESA, 2023). The growth rates in the urban population until 2035 is predicted to be the greatest in Central America (14 percent between 2015 and 2035) followed by the Caribbean (11 percent), and only 8 percent in the already highly urbanized countries of South America.

10.2.3 Nutrient management

There is greater evidence of unsustainable nutrient surpluses than nutrient mining in the countries of Latin America and the Caribbean (Figure 10.9 and Figure 10.10) (FAO, 2024b) (see Section 3.5.1 for the criteria used to identify nutrient mining and excess). The most widespread form of excessive nutrient surpluses is associated with P, with seven Caribbean countries (Antigua and Barbuda, Bahamas, Barbados, Jamaica, Puerto Rico, Saint Lucia and Trinidad and Tobago), three Central American countries (Belize, Costa Rica and Panama) and five South American countries (Brazil, Chile, Ecuador, Uruguay and the Bolivarian Republic of Venezuela) showing possibly unsustainable levels of P surpluses on soils that already have substantial P accumulations. Argentina, the Plurinational State of Bolivia and the Dominican Republic have negative P balances on soils with a history of P mining (such as negative levels of accumulated residual P) (Zou, Zhang and Davidson, 2022).

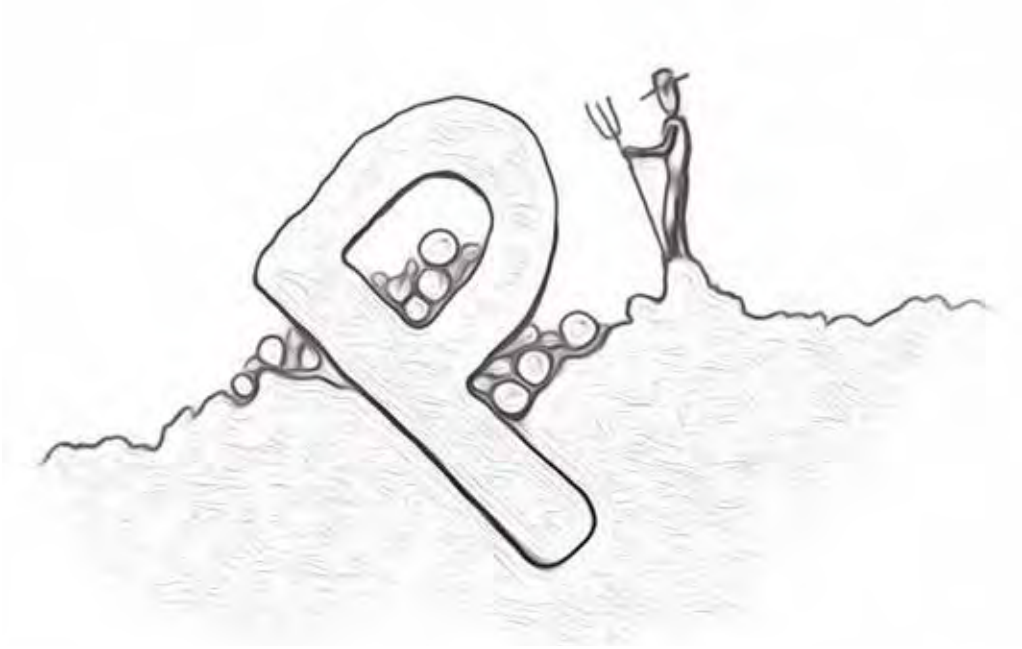


Figure 10.9. Country map of average phosphorus use in Latin America and the Caribbean in the 2017 to 2021 period



Refer to the disclaimer on page ii for the names and boundaries used in this map. A dispute exists between the Governments of Argentina and the United Kingdom of Great Britain and Northern Ireland concerning sovereignty over the Falkland Islands (Malvinas).

◊ *Source*: **FAO**. 2024. *FAOSTAT: Fertilizers by Nutrient*. [Accessed on 9 January 2024]. https://www.fao.org/faostat/en/#data/RFN. Licence: CC-BY-4.0.

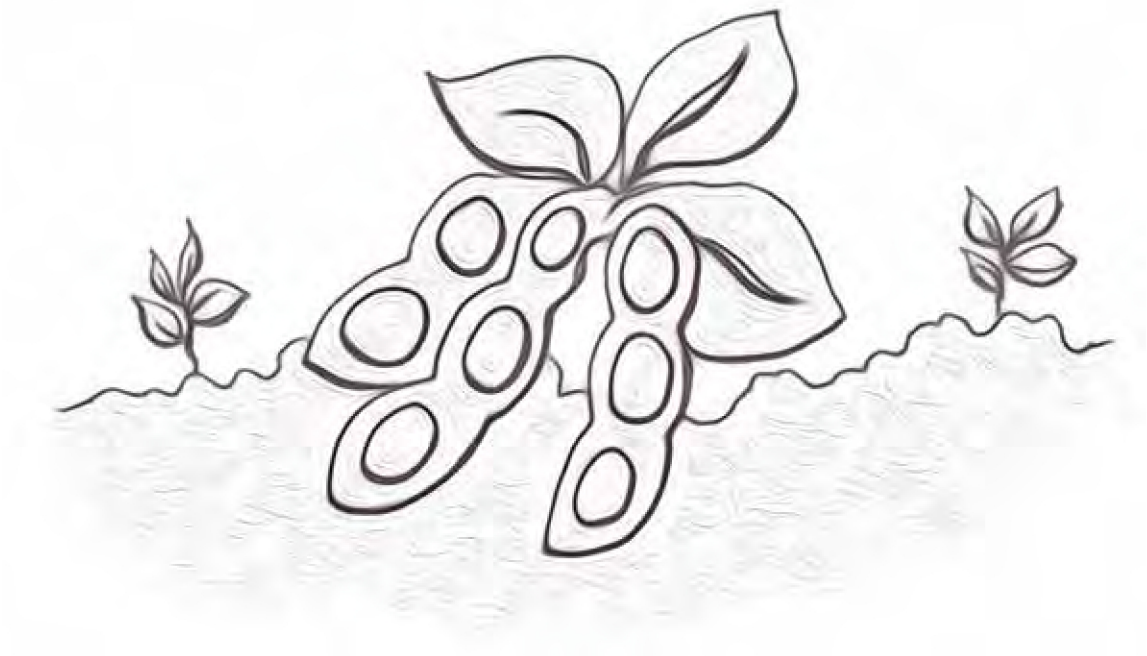


Figure 10.10. Country map of average nitrogen use in Latin America and the Caribbean in the 2017 to 2021 period



Source: **FAO**. 2024. *FAOSTAT: Fertilizers by Nutrient*. [Accessed on 9 January 2024]. https://www.fao.org/faostat/en/#data/RFN. Licence: CC-BY-4.0.

Four Caribbean countries (the Bahamas, Barbados, Puerto Rico, and Trinidad and Tobago), two Central American countries (Belize and Costa Rica) and five South American countries (Chile, Colombia, Ecuador, Suriname and Uruguay) have unsustainably high surpluses in their N budgets. Several countries (Argentina, the Plurinational State of Bolivia, Brazil, Paraguay and Uruguay) in South America have substantial N outputs that reflect the high contribution of BNF as a source of N (Table 10.1). These countries are major producers of soybeans, with harvested areas ranging from 28 percent of cropland (the Plurinational State of Bolivia) to 75 percent of cropland (Paraguay) in 2021 (FAO, 2024a).

Table 10.1. Average values for total nitrogen inputs, percentage of total nitrogen inputs from synthetic fertilizer, manure and biological nitrogen fixation, total phosphorus inputs, and percentage of phosphorus inputs from synthetic fertilizer (2017–2021)

Country	Total N inputs(kg/ha)	Synthetic fertilizer (%)	Manure (%)	Biological N fixation (%)	Total P inputs (kg/ha)	Synthetic fertilizer (%)
Antigua and Barbuda	56.3	3.7	84.9	0.0	11.4	7.8
Bahamas	178.1	52.6	39.2	4.6	27.8	43.9
Barbados	247.2	12.2	76.8	8.4	52.7	14.3
Belize	163.0	77.7	5.0	13.3	44.6	95.7
Costa Rica	241.4	67.2	28.8	2.0	27.2	49.4
Cuba	33.1	31.9	41.2	15.0	4.4	25.2
Dominica	28.9	17.6	57.7	2.5	7.1	52.3
Dominican Republic	145.1	50.1	38.3	8.5	20.1	40.5
El Salvador	94.9	62.7	22.6	8.3	10.8	58.1
Grenada	56.6	42.0	37.1	9.5	9.2	42.1
Guatemala	112.5	62.8	25.0	7.2	16.0	58.3
Haiti	39.5	10.8	43.5	29.4	6.3	35.7
Honduras	96.7	67.4	23.0	5.6	14.6	68.1
Jamaica	68.3	35.1	49.5	6.0	14.9	44.8
Mexico	89.6	65.8	22.4	6.6	13.1	62.8
Nicaragua	78.5	43.2	34.0	16.9	9.5	43.7
Panama	80.7	55.1	30.8	7.9	13.3	58.1
Puerto Rico	135.2	25.7	69.3	0.3	28.1	29.4
Saint Kitts and Nevis	33.9	7.3	62.7	11.1	5.5	13.5
Saint Lucia	66.2	38.3	51.5	0.5	14.7	42.4
Saint Vincent and the Grenadines	89.1	33.8	37.9	21.1	14.5	43.1
Trinidad and Tobago	329.2	39.5	57.0	1.6	47.6	12.8
Argentina	117.8	25.9	3.7	65.9	8.1	88.9
Bolivia (Plurinational State of)	77.1	5.4	27.1	62.2	6.4	21.8
Brazil	253.0	28.1	8.8	59.9	43.8	87.4
Chile	151.1	64.2	33.2	1.8	34.0	70.1
Colombia	178.2	57.5	33.3	5.6	33.5	61.4
Ecuador	137.7	65.6	25.2	5.1	17.5	57.5
Guyana	68.7	56.4	24.7	16.1	6.9	47.6
Paraguay	180.4	13.2	4.4	78.7	17.5	89.4
Peru	82.6	67.7	23.8	5.9	12.0	62.4
Suriname	181.5	68.9	17.8	12.7	12.3	38.5
Uruguay	176.4	41.8	9.8	43.8	29.7	88.7
Venezuela (Bolivarian Republic of)	99.9	40.7	49.8	3.1	12.1	30.8

Note: P from manure is not shown and is equal to the remainder of total P input.

Source: **FAO**. 2024. *FAOSTAT: Fertilizers by Nutrient*. [Accessed on 9 January 2024]. https://www.fao.org/faostat/en/#data/RFN. Licence: CC-BY-4.0.

Four Caribbean countries have very low N outputs that are possibly indicative of yield gaps due to insufficient supply of N. Cuba and Dominica also have low N outputs and low surpluses of N, P and K that may together produce yield gaps.

Negative K budgets (Figure 10.11) are common throughout Latin America and the Caribbean, with high negative balances in Argentina, the Plurinational State of Bolivia, the Dominican Republic, Guyana, and Saint Vincent and the Grenadines.

Figure 10.11. Country map of average potassium use in the 2017 to 2021 period



Refer to the disclaimer on page ii for the names and boundaries used in this map. A dispute exists between the Governments of Argentina and the United Kingdom of Great Britain and Northern Ireland concerning sovereignty over the Falkland Islands (Malvinas).

◊ *Source*: **FAO**. 2024. *FAOSTAT: Cropland Nutrient Balance*. [Accessed 10 October 2024] https://www.fao.org/faostat/en/#data/ESB. Licence: CC-BY-4.0.

Older, more heavily weathered soils and sandy soils in Latin America and the Caribbean may also experience deficiencies in micronutrients. Global maps prepared by Moreno-Jiménez *et al.* (2022) indicate low contents of Mn and Zn throughout tropical and subtropical South America and low contents of Cu, Fe, Co and Ni in the soils of the Cerrado subregion.

10.3 Regional status and trend of threats to soil functions for Central America, southern Mexico and the Caribbean subregion

10.3.1 Soil erosion in the Central America, southern Mexico and the Caribbean subregion

Bare soil conditions can lead to increased soil erosion by either wind or water, with large areas being detectable from satellite imagery. Countries with the most areas of bare soil (and the representative percentages) are as follows: southern Mexico (1 491 km^2 or 0.59 percent), Guatemala (131 km^2 or 0.12 percent), Cuba (84 km^2 or 0.08 percent), Nicaragua (59 km^2 or 0.05 percent), Honduras (44 km^2 or 0.04 percent), El Salvador (29 km^2 or 0.14 percent), the Dominican Republic (19 km^2 or 0.04 percent) and Costa Rica (11 km^2 or 0.02 percent) (Demattê *et al*., 2020). Croplands have the highest incidence of bare soils (766 km^2), grasslands (648 km^2) and savannahhhs (170 km^2). The main bare soil types across the region are Phaeozems (603 km^2), Vertisols (419 km^2), Luvisols (314 km^2), and Leptosols (267 km^2).

For Caribbean SIDS, a decline in vegetation cover has heightened their vulnerability to erosive forces (de Caires *et al*. 2022). Consequently, 97 percent of Haiti's forest areas have been depleted, leading to a rapid increase in erosion and thus, the loss of fertile topsoil and biodiversity (GEF, 2019). Trinidad has a higher risk of erosion due to a reduction in vegetation cover on 37 percent of the island (Melville, Sutherland and Wuddivira, 2020).

Bare soil data and land cover change data are essential to devise effective soil management strategies by identifying areas at risk of erosion, degradation or reduced productivity, enabling targeted interventions to preserve and enhance soil health. Coastal erosion and accretion occur across large areas of the subregion (for example, 13 percent of the Caribbean coast of Costa Rica [Barrantes-Castillo and Ortega-Otarola, 2023]). Coastal erosion has also become an important topic due to the possible effects of a potential sea level rise (Zhang, Douglas and Leatherman, 2004; Ashton, Donnelly and Evans, 2008). However, no updated assessments exist to account for coastal erosion in the entire area.

Water erosion

The coastal plains of Costa Rica, Cuba, El Salvador, Haiti and Mexico have the most extensive areas at risk of erosion, with soil erosion rates of over 20 t/ha annually (Borrelli *et al*., 2017). The available estimates from Borrelli *et al.* (2017) are for 2012 and most likely do not represent the higher soil erosion rates in the region occurring after the publication of SWSR 2015. In coastal areas, gullies can be formed in a short time frame (such as over a day) in response to water erosion related to heavy rains across hurricane pathways (Krasilnikov *et al.*, 2007). Inland soil erosion by water is also common across areas with young soils formed by volcanic parent materials and where there have been agricultural practices implemented in topographically complex environments (Gayer *et al*., 2019). Soil erosion rates are also associated with a rise in sea level across karstic geological landscapes and C-rich soils under mangrove ecosystems (Vázquez-Lule *et al*., 2019). Large areas in the karstic region experience intense soil erosion (Fragoso-Servón *et al*., 2020), and water-rich environments are also vulnerable to increased dissolution of their parent material in response to land-use change.

Wind erosion

Wind erosion varies with land-use, soil type and vegetation cover, and occurs in deforested areas with agricultural practices that leave soil exposed, as well as in vulnerable coastal regions.

The Caribbean's susceptibility to tropical storms and hurricanes can exacerbate wind erosion by removing the entire soil cover. In addition, accelerated winds in the area are important drivers of wind erosion, with coastal sand being transferred and deposited by the wind in continental areas, intensifying the acidification of agricultural soils during the dry period (Rivera-Solis, 2021). The rare occurrence of open savannahhh white sand ecosystems within humid, tropical rainforests has been attributed to sorting through aeolian transportation. Particle size distribution showed a unimodal pattern, with peaks at coarse silt (38 percent) and fine sand (40 percent) (Atwell *et al*., 2023).

Tillage erosion

There are no regional or national studies on tillage erosion in this subregion. Studies in Costa Rica have shown substantial tillage translocation and erosion with use of the disc plough (Tiessen *et al*., 2010) but recent and more comprehensive data is lacking.

10.3.2 Soil carbon loss in the Central America, southern Mexico and the Caribbean subregion

FAO's GSOCmap (FAO and ITPS, 2020) estimated SOC stocks in the 0–30 cm layer of mineral soils. The map showed a regional stock of SOC of 9 479 million tonnes (83 ± 26 t/ha on average). Soil organic C stocks were 109 ± 24 t/ha for Histosols, 96 ±21 t/ha for Andosols and 87 ± 13 t/ha for Nitisols, 33 ± 1 t/ha for Chernozems and 32 ± 0.01 t/ha for Durisols. Soil organic carbon stocks were 90 ± 11 t/ha for closed shrublands and 85 ± 23 t/ha for evergreen broadleaf forests. Across croplands, the SOC stocks were 59 ± 15 t/ha, with open shrublands showing values of 38 ± 5 t/ha. However, trends of changes in SOC were not detectable.

Losses in SOC were reported in Costa Rica, Curaçao and Puerto Rico, with agricultural soils losing up to 75 percent of their original SOC stocks (Otero, 2020). However, according to the GSOCseq map (FAO, 2022a), Umbrisols, Luvisols and Leptosols in the 0–30 cm layer of mineral soils had a higher SOC sequestration potential across croplands and grasslands.

Since 2015, SOC changes have been detectable in the first few centimetres of soil surface thanks to state-of-the-art earth ESMs fed with remote sensing of the environment. The NASA SMAP L4 C product (Entekhabi *et al*., 2014; Kimball, Jones and Glassy, 2014) can detect changes in time of superficial SOC by retrieving values using radar and radiometric instruments (Jones *et al.,* 2017; Ray *et al.,* 2017, 2019). Increases in SOC can be seen across large areas of southern Mexico (393 255 km^2), Nicaragua (97 038 km^2), Cuba (87 480 km^2) and Honduras (87 156 km^2), as seen by the SMAP L4 C product when tested on relatively young volcanic soils under productive ecosystems (such as temperate and tropical forests), except for Cuba (a potentially inorganic C-rich country). Soil organic carbon loss has been observed since 2015 around the Bahamas (243 km^2) in Alfisols and Arenosols. Although the SMAP L4 product has several limitations and coarse granularity, it represents a valuable benchmark for monitoring changes in the superficial soil C pool with a national-to-regional perspective. Chopin and Sierra (2019) and Vaughn *et al*. (2019) reported that changes in SOC at national and regional scales are less associated with climate and better related to land use and soil properties.

10.3.3 Soil biodiversity loss in the Central America, southern Mexico and the Caribbean subregion

Soil biodiversity plays a crucial role in nutrient cycling and affects soil respiration, soil structure (Graaff *et al*., 2015; Orgiazzi *et al*., 2016; Zhang *et al*., 2021, 2022) and soil moisture retention (Xu *et al*., 2024; Pankaj *et al*., 2024; Frey, 2024). However, soil biodiversity data in tropical and subtropical regions are scarce, or even non-existent (Guerra *et al*., 2020).

Soil respiration is an important process linked to soil organisms and is used as an indicator of organic matter decomposition. Coastal areas are a hotspot of soil respiration, with large areas having values up to 3.3 t C/ha annually (across the Yucatan Peninsula and coastal areas of the Caribbean territories), as indicated by modelling and mapping of soil respiration at the global scale (Warner *et al*., 2019). Predictions of soil respiration trends across the region show that Costa Rica, Guyana, Honduras, and Nicaragua had the largest area (a total of 7 533 km^2) experiencing significant negative trends of soil respiration between 1991 and 2018, mainly across savannahhhs, evergreen broadleaf forests and permanent wetlands (such as decreases from −3.5 to −6.1 g C/m^2 annually). In contrast, Cuba, the Dominican Republic, Haiti, Jamaica and southern Mexico showed a larger area (a total of 179 577 km^2) with significant positive trends in soil respiration between 1991 and 2018, with increases between 2.71 and 4.35 g C/m^2 annually (Warner *et al*., 2019)

10.3.4 Nutrient mismanagement in the Central America, southern Mexico and the Caribbean subregion

Digital soil maps describing the spatial variability of soil fertility indicators across the region show an average cation exchange of at least 259 millimoles (mmol)/kg and an average N content of 403 centigrams (cg)/kg (Poggio *et al*., 2021). The historical data on fertilizer consumption, broken down by nutrient and country, reveals distinct patterns (FAO, 2025a). For instance, Guatemala and Mexico, N fertilization rose consistently since the 1960s, followed by an increase in phosphate usage. Conversely, countries like El Salvador had high consumption levels of N and P fertilizers in the 1970s and again from 2005 to 2015 with lower inputs in 2020 and 2022. In contrast, Cuba had a surge in N and phosphate fertilizer usage from the 1960s until the 1990s, followed by a sudden decline in the early 1990s and continued low inputs to 2022. Haiti had low levels of N, P and K inputs throughout, from the 1960s to 2022 (FAO, 2025a).

Ludemann *et al.* (2024) investigated global NUE using data from FAOSTAT. Nitrogen use efficiency ranged from 0 to 50 percent in Costa Rica, Honduras and Mexico, P use efficiency was lowest in Belize, Costa Rica, Honduras and Panama, and K use efficiency was lowest in Belize, Costa Rica, Nicaragua and Panama. Conversely, countries such as Cuba demonstrated the highest NUE, with N use efficiency exceeding 90 percent. Similarly, Cuba, the Dominican Republic, Haiti and Jamaica had high P use efficiency, while the Dominican Republic, Cuba, El Salvador, Guatemala, Haiti, Jamaica and Mexico had superior K use efficiency. It is worth noting that low or high NUE are usually estimated as output/input ratios. Thus, low nutrient inputs usually result in high NUE but with increased soil nutrient mining and high nutrient inputs results in low NUE with mostly nutrient overuse and potential contamination.

Soil nutrient data enables informed nutrient management decisions, helping optimize fertilizer applications and organic matter additions to enhance soil fertility and crop productivity. For example, C:N ratios of at least 30:1 favour the immobilization of organic matter, while lower values (20:1) favour mineralization, with both processes affecting soil fertility (Navarro, 2003). The USDA reported a 10:1 C:N ratio as ideal for croplands, with values of 25:1 usually found in forage and legume crops with high N content (USDA, 2022). The largest areas with a C:N ratio between 10 and 30 are in Mexico (846 km^2), the Dominican Republic (54 km^2), Cuba (40 km^2) Jamaica (34 km^2), Trinidad

and Tobago (22 km^2) and Guadeloupe (2 km^2), but only 1.3 percent of the croplands area in the study region fall in this range. Alongside addressing acidification and pollution, it is essential to preserve and enhance soil biodiversity so that soil fertility is also maintained and improved, while promoting sustainable agricultural practices, and ensuring food security. Rahman *et al.* (2022) observed that decreasing and increasing soil nutrient budgets for the Caribbean were linked to increased productivity (Dominica and Grenada) or limited land availability (Barbados and Jamaica).

10.3.5 Salt accumulation in the Central America, southern Mexico and the Caribbean subregion

While some countries of the subregion benefit from soils with a high agricultural potential, such as Cuba, the Dominican Republic, Honduras, Mexico and Nicaragua, the soil fertility across large areas is currently affected by some degree of increased soil salinity and sodicity. It is also worth noting that soil salinity levels have regional variations. Coastal areas, especially in the Caribbean, tend to experience higher soil salinity due to the proximity of seawater, with soil salinity capable of ranging from 3 to 8 dS/m or even higher in some cases (González-Pedraza, Chiquillo Barrios and Escalante, 2022). In the humid tropical Caribbean, the soils are typically non-saline, as abundant rainfall continually flushes out excessive soluble salts (Atwell and Wuddivira, 2019). However, coastal areas face elevated soil salinity due to the intrusion of saltwater as sea levels rise (Stubbs *et al*., 2023). In a study conducted in Trinidad using electromagnetic induction, Atwell *et al*. (2016) found that the salinity of soil adjacent to brackish wetlands displayed EC levels reaching up to 8 dS/m.

Human activities such as improper irrigation and deforestation can also increase soil salinity. Thus, managing soil salinity is crucial for sustainable agriculture, as high salinity can negatively impact soil health and crop yield. It is essential to implement strategies to mitigate these effects and maintain productive soils in the subregion. The recent development of the GSASmap (Omuto *et al.*, 2021) reveals large areas of the region affected by soil salinity, although there is still coarse granularity due to an at-present lack of soil salinity data and information provided by Caribbean countries that would be able to generate country-specific soil salinity reports.

Since 2015, studies on the spatial variability of soil salinity have increased, and remote sensing and machine learning have facilitated the spatial characterization of soil salinity across differing scales. Such efforts to map soil salinity worldwide have made digital assessments of soil salinity available that are useful benchmarks for improving the quality of estimates and enabling soil salinity monitoring systems (Ivushkin *et al.*, 2019). According to those authors, the countries with the largest land areas at risk of increased soil salinity are southern Mexico (389 873 km^2), Cuba (74 253 km^2), Honduras (57 880 km^2), and Nicaragua (46 963 km^2) with direct implications to the soil fertility and yields of a wide diversity of production systems. While certain regions boast considerable agricultural potential, marked by positive trends in moisture (Zhang *et al.,* 2022b) and SOC within their croplands (Kimball *et al.,* 2022), such as Cuba (0.74 percent and 100.4 g/m^2 annually), the Dominican Republic (1.40 percent and 98.3 g/m^2 annually), Honduras (0.62 percent and 59.4 g/m^2 annually), Mexico (0.51 percent and 66.4 g/m^2 annually), and Nicaragua (0.67 percent and 37.0 g/m^2 annually), they concurrently face salinity issues exceeding 2 dS/m. These results confirm the urgent need to reverse the expansion of soil salinity and implement long-term monitoring systems.

10.3.6 Soil moisture change in the Central America, southern Mexico and the Caribbean subregion

Soil moisture is the most important variable affecting the soil-forming environment, structure, stability and erosion, and its variation regulates biological activity and C sequestration rates in soils. Radar-based remote sensing can be useful for quantifying soil moisture content across large geographical areas. The Central America, southern Mexico and the Caribbean subregion has experienced a soil moisture decline since the early 1990s (Guevara, Taufer and Vargas, 2021). Using remotely sensed soil data from Zhang *et al*. (2022), important hotspots of superficial (0–10 cm soil depth) soil moisture decline were observed across southern Guatemala, southern Haiti, Honduras, Jamaica, Nicaragua, Panama, and the Yucatan Peninsula (Castelblanco Rivera and Guevara, 2024). The results showed that 338 750 km^2 of land had negative soil moisture trends between 2015 and 2022, with the main changes occurring across Durisols, Acrisols, Leptosols, and Nitisols. Superficial soil moisture increased across large areas with flat to gentle terrain slopes across Cuba (35 000 km^2), Honduras (49 375 km^2), the gulf plains of Mexico (69 375 km^2) and Nicaragua (23 125 km^2), probably as a response to increased flooding events. Flooding as an extreme event has become progressively evident in the Caribbean due to a higher number of intense rainfall and storm surges from hurricanes (Fontes de Meira and Phillips, 2019). Changes in soil moisture conditions at the regional scale have been historically impacted by land-use intensification, sea level rise, and extreme climate events. However, they are mainly based on multiple available products that may require further verification for detailed interpretation. There is a pressing need for improved quality, quantity and accessibility to soil data and information across the region.



10.4 Regional status and trend of threats to soil functions for the Amazon subregion

10.4.1 Introduction

Threats to soil functions in the Amazon subregion primarily stem from diminishing natural vegetation cover and land-use and land-cover (LULC) changes and transitions. The MapBiomas Network monitors LULC changes and transitions in the Amazon since 2017, creating annual maps from 1985 to the present using Landsat imagery with a 30 m spatial resolution.

Between 1985 and 2015, the Amazon Basin showed a 9 percent reduction in its forest vegetation, followed by an additional 1 percent decrease between 2015 and 2021 (Figure 10.7) (MapBiomas Amazonía, 2022). This resulted in a cumulative loss of approximately 9 million ha during the latter period, with 90 percent comprising of forest formations (MapBiomas Amazonía, 2022). Notably, woodland savannahh formations witnessed the most substantial relative losses in the Amazon, with a 26 percent reduction (13 million ha) from 1985 to 2021. Non-forest vegetation had similar relative losses, with the area of wetlands decreasing by 12 percent between 2015 and 2021 (about 2 million ha).

The primary drivers of deforestation in the Amazon are farming and silviculture. Between 1985 and 2021, the area for these purposes increased by 74 million ha (MapBiomas Amazonía, 2022). Since 2015, however, oil palm cultivation, silviculture and agriculture have shown increased annual rates of land area expansion. As a result, the total area used for agriculture increased by 10 million ha between 2015 and 2021, reaching a total of 123 million ha.

Pastureland, however, remains the primary usage of deforested areas, consistently being the dominant land use in the Amazon, covering 60 to 70 percent of the agricultural area throughout the entire 1985–2021 period (MapBiomas Amazonía, 2022). For instance, the area of pastures (84 million ha) has surpassed that of non-forest vegetation (73 million ha) in the 2011 to 2021 period, as well as the combined areas of savannahhs, mangroves and flooded forests (74 million ha). This dominance reflects the main trend of LULC changes and transitions in the subregion.

The most recent data from MapBiomas Amazonía (2022) reveals that between 1985 and 2015, natural vegetation coverage decreased by 8.4 percent (approximately 65 million ha), with an additional 1.4 percent decrease (approximately 10 million ha) between 2015 and 2021 (83 percent of the loss between 2015 and 2021 is from forests). The expansion of pastureland remains the primary reason for deforested areas, followed by agriculture, silviculture and oil palm cultivation (Table 10.2 and Figure 10.12) (IDEAM, 2023; MapBiomas Amazonía, 2022).

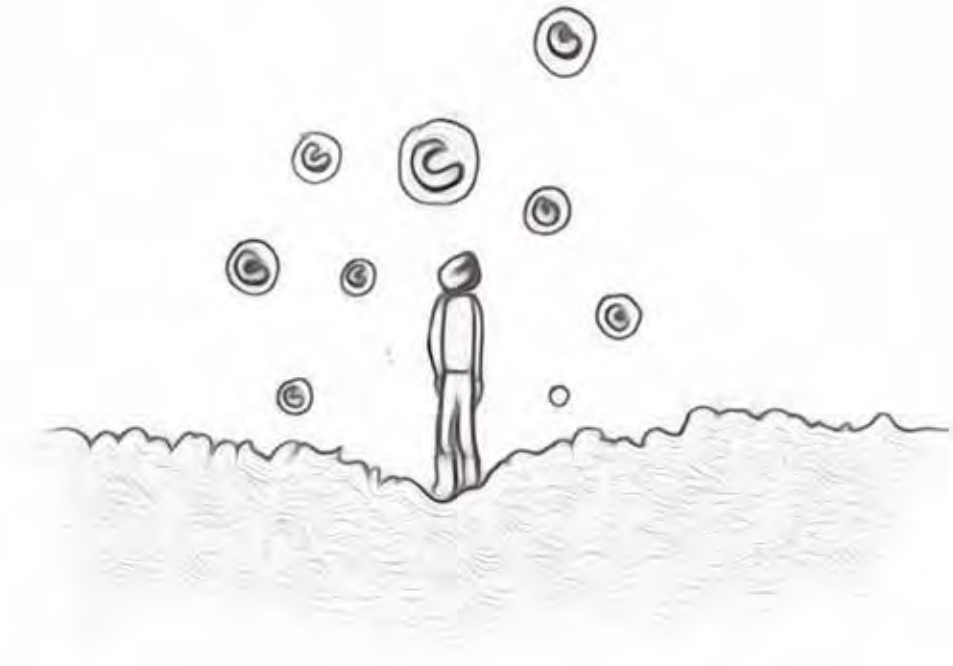


Table 10.2. Area occupied by natural land cover and anthropogenic land use in the Amazon Basin in 1985, 2015 and 2021, and their percentage change in the 1985–2015 and 2015–2021 periods

LULC class	Area in 1985 (ha)	Area in 2015 (ha)	Area in 2021 (ha)	Change in area (1985–2015) (%)	Change in area (2015–2021) (%)
1. Natural forest					
1.1. Forest formation	609 677 457	559 889 832	551 673 646	−8	−1
1.2. Savannahh formation	50 066 302	37 934 419	37 090 212	−24	−2
1.3. Mangrove	909 786	933 548	916 747	3	−2
1.4. Flooded forest	37 302 970	36 244 707	36 233 281	−3	0
Total (1)	697 956 515	635 002 506	625 913 886	−9	−1
2. Non-forest natural formation					
2.1. Wetland	16 114 604	16 921 229	14 836 440	5	−12
2.2. Grassland	53 955 981	50 970 137	52 296 910	−6	3
2.3. Rocky outcrop	302 526	311 374	308 370	3	−1
2.4. Other non-forest formation	5 935 032	5 793 712	5 690 606	−2	−2
Total (2)	76 308 143	73 996 452	73 132 326	−3	−1
3. Farming and silviculture					
3.1. Pasture	29 172 018	80 288 982	83 743 701	175	4
3.2. Agriculture	3 407 264	19 024 778	22 608 651	458	19
3.3. Silviculture	20 463	595 512	736 299	2 810	24
3.4. Oil palm	35 552	136 392	205 562	284	51
3.5. Mosaic of uses	16 383 463	13 852 514	16 080 712	−15	16
Total (3)	49 018 760	113 898 178	123 374 925	132	8
4. Non-vegetated area					
4.1. Urban infrastructure	368 370	832 193	929 613	126	12
4.2. Mining	47 315	433 293	571 174	816	32
4.3. Other non-vegetated area	3 313 308	3 099 032	3 305 187	−6	7
Total (4)	3 728 993	4 364 518	4 805 974	17	10
5. Water					
5.1. River, lake or ocean	16 540 913	16 367 203	16 407 955	−1	0
5.2. Glacier	172 629	100 922	93 267	−42	−8
Total (5)	16 713 542	16 468 125	16 501 222	−1	0
6. Non observed					
Total (6)	12 425	8 600	10 044	−31	17

◊ *Source:* **MapBiomas Amazonía.** 2022. *Colección 4.0 de la Serie Anual de Mapas de Cobertura y Uso del Suelo de la Amazonía [Collection 4.0 of the Annual Series of Cover and Land Use Maps of the Amazon].* [Accessed on 2 November 2023]. https://amazonia.mapbiomas.org/map/coleccion-4/. Licence: CC-BY-SA.

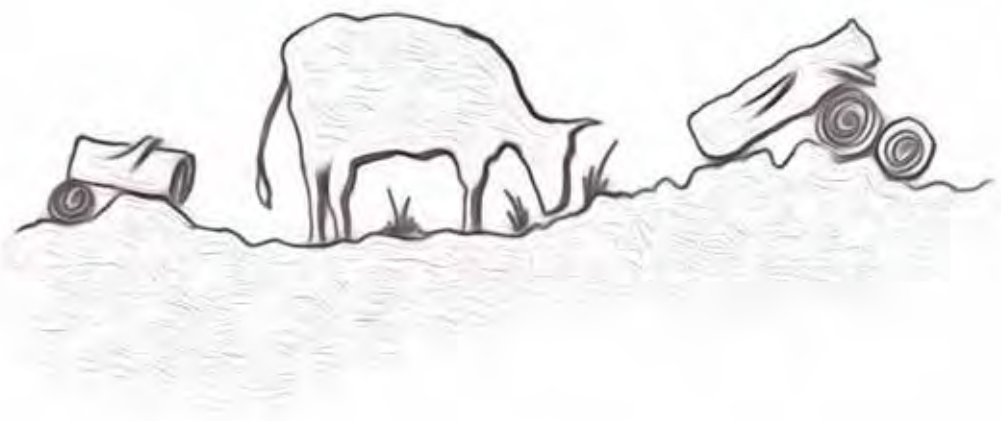
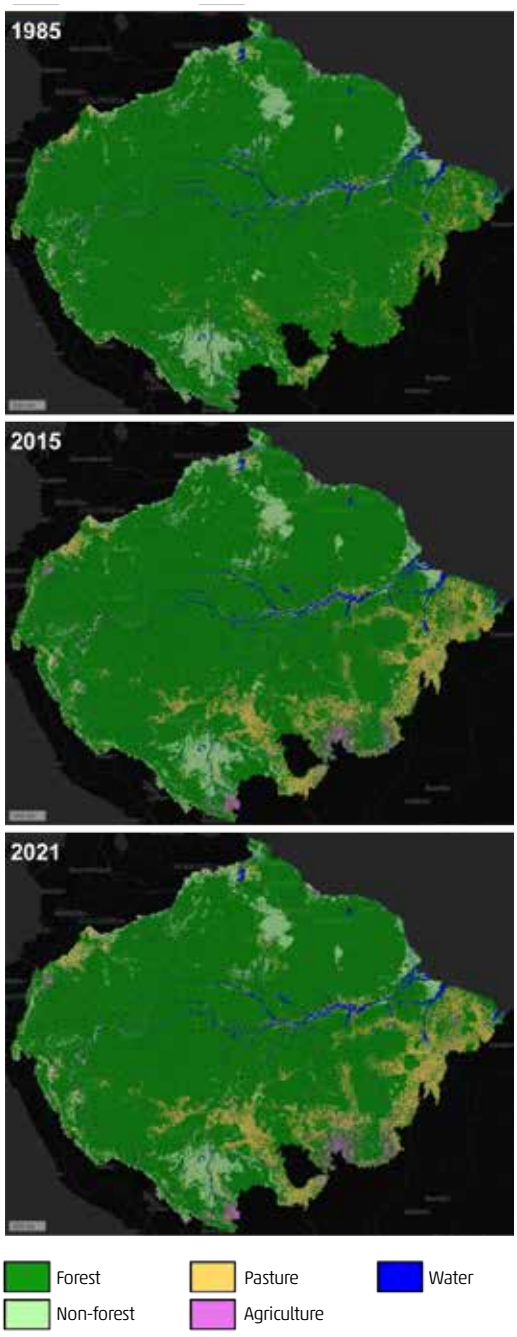


Figure 10.12. Land-use and land-cover changes in the Amazon subregion in 1985, 2015 and 2021



◊ *Source*: The original MapBiomas database used to create this figure can no longer be accessed. The map and accompanying data should be accessed with **MapBiomas Amazonía**. 2025. *MapBiomas Amazonía Collection 6*. [Accessed on 4 April 2025]. https://plataforma.amazonia.mapbiomas.org/cobertura?activeBaseMap=9&layersOpacity=100&activeModule=coverage&activeModuleContent=coverage%3Acoverage_main&activeYear=2023&mapPosition. Licence: CC-BY-SA.

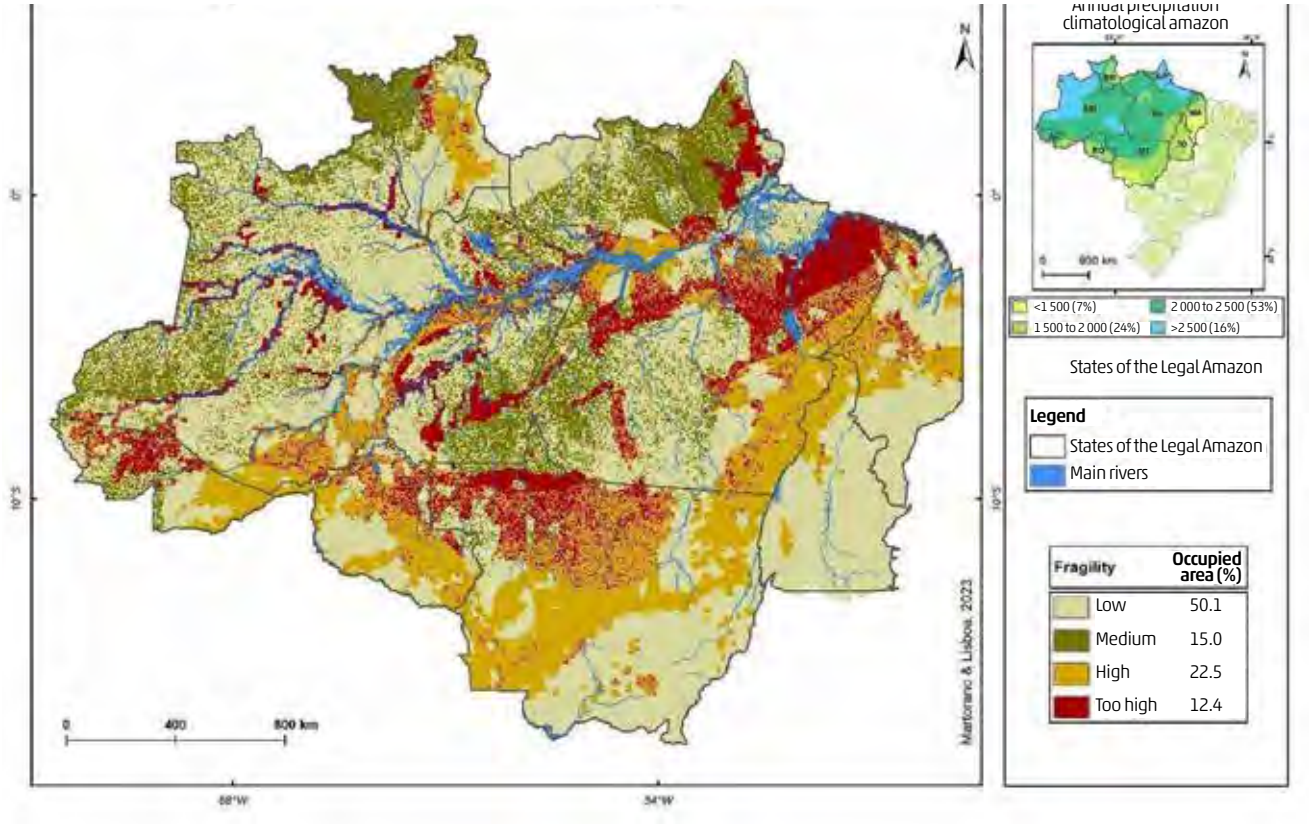
10.4.2 Soil erosion in the Amazon subregion

Water erosion

Rainfall is the primary cause of soil erosion in the Amazon subregion, a tropical region known for high and intense precipitation. Recent climate change has increased rainfall, thus intensifying the occurrence of extreme rainfall events and raising the risks of soil erosion (CIAT and CORMACARENA, 2017; Gallo, 2022; Schiesari *et al.*, 2020). This has also increased the occurrence of severe droughts, such as the one in 2023 that reduced water levels in major rivers such as the Amazonas and Negro (Wagner *et al.*, 2024). Approximately 35 percent of anthropogenic areas in the Brazilian Amazon

face the serious risk of soil loss due to the erosive potential of high intensity rainfall (Figure 10.13) (Martorano *et al.*, 2017). Large areas of the Amazon lowlands are comprised of highly erodible soils due to the high silt contents (Godoi *et al.*, 2021).

Figure 10.13. Map of land fragility due to water erosion in anthropogenic areas in the Amazon



Refer to the disclaimer on page ii for the names and boundaries used in this map.

◊ *Source*: adapted from **Martorano, L.G., Lisboa, L.S., Villa, P.M. & de Moraes, J.R. da S.C.** 2017. Fragilidade das terras pelo processo erosivo das chuvas em áreas antrópicas e declivosas na Amazônia Legal [Fragility of land due to the erosive process of rain in anthropogenic and steep areas in the Legal Amazon]. Conference presentation at Amazônia e seus solos: peculiaridades e potencialidades [Amazon and its soils: peculiarities and potentialities]. Brazilian Soil Science Congress, 30 July to 1 August 2017. Viçosa, Brazil, Brazilian Soil Science Society (SBCS). http://www.alice.cnptia.embrapa.br/handle/doc/1080857

Recent decades have witnessed the removal of native vegetation, periodic fires, inadequate land management, colonization, and livestock farming, contributing to soil fragility and accentuating erosion processes (Bravo-Medina *et al.*, 2017; Landholm *et al.*, 2019; Rodríguez *et al.*, 2021; Bravo-Medina *et al.*, 2023; Gallo, 2022; Kuhn *et al.*, 2023; Rodríguez *et al.*, 2021; Schiesari *et al.*, 2020). The annual deforestation rate of about 1.5 million ha between 2015 and 2021 is lower than the previous three decades (2.1 million ha per year between 1985 and 2015), but is still high (See Table 10.2 and Figure 10.12) (IDEAM, 2023; MapBiomas Amazonia, 2025).

Conservation practices like no-till annual cropping systems generally reduce soil erosion. However, the consistently high air temperature and humidity throughout the year hasten the decomposition of plant residues (Gallo, 2022). Integrated crop–livestock–forestry systems and other agroforestry arrangements are notable solutions by maintaining year-round plant growth, ensuring permanent soil cover (Bravo-Medina *et al.*, 2023) and providing protection against extreme rainfall events.

Wind erosion

Wind erosion is a concern in the north Amazon, particularly in the Orinoco River basin. The Orinoco low-level jet stream generates high wind speeds over Colombia and the Bolivarian Republic of Venezuela, with average velocities exceeding 8 m/s, peaking at 17 m/s (Jiménez-Sánchez *et al.*, 2020). Wind erosion is prevalent during tillage in the dry season, suspending fine soil particles in

the air due to ploughing and carrying them with the wind. Ongoing deforestation for agricultural use throughout the region (See Table 10.2 and Figure 10.7) (MapBiomas Amazonía, 2025), coupled with climate change, is amplifying climatic dynamics, and wind erosion may become a concern in other parts of the Amazon in the near future (CIAT and CORMACARENA, 2017; Feng *et al.*, 2023).

Tillage erosion

Many Amazonian rivers originate in the Andes, a mountainous landscape where contour cultivation and slow-forming terraces are commonly used as soil conservation practices. The construction of terraces is relatively simple, developed slowly through tillage translocation which requires low levels of labour and materials (Dercon *et al.*, 2007, 2003). However, this method causes a net downslope soil movement, leading to gradients in morphological, physical, chemical, biological, and hydrological soil properties over time, thinning the soil on upper slopes and reducing overall production (Caulfield *et al.*, 2020; Dercon *et al.*, 2003).

Hillslope mass movement

In the Amazon Basin subregion, the Andes landscape with the highest elevations and steepest slopes covering 303 797 km^2, is also the most susceptible to hillslope mass movement (Villacorta, Fidel and Zavala Carrión, 2012). Deforestation and ongoing land-use changes have persisted after 2015 (MapBiomas Amazonía, 2022), while the rise in Au prices has led to increased demand and mining activities (WRI, 2020). Mining – using both highly or minimally mechanized technology – increases the susceptibility of zones for mass movements (Velásquez Ramírez *et al.*, 2021).

In the Amazon lowlands, mass movements are triggered by river erosion, a phenomenon known as "*terras caídas*" or "fallen soil". Lateral erosion of riverbanks is linked with undermining, landslides, slips and collapses along the riverbanks formed in the alluvial deposits of Amazonian rivers (Bandeira *et al.*, 2018). The increased use of waterways to transport crops such as soybean from fields in the region and the considerable growth of urban infrastructure, especially near rivers (a 12 percent increase from 2015 to 2021) (MapBiomas Amazonía, 2022), is likely heightening the frequency of mass movements. Moreover, climate change is elevating the risk of landslides, due to increasingly extreme rainfall and droughts (Kuhn *et al.*, 2023).

It is therefore essential to establish and enforce monitoring and inspection policies in vulnerable areas to prevent additional soil loss and protect human lives, as well as control mining activities.

10.4.3 Soil carbon loss in the Amazon subregion

Mineral soils

The soils in the Amazon Basin subregion have low nutrient levels, heavily relying on organic matter for sustained plant nutrition through nutrient cycling (IDEAM, 2023; IGAC, 2015; Davidson *et al.*, 2012). Deforestation and land clearing reduce SOC, as high temperatures and humidity accelerate the decomposition of organic matter, disrupting nutrient cycling and impacting soil biodiversity (CIAT and CORMACARENA, 2017). Soil organic carbon loss is further influenced by water erosion and the edge effects resulting from ongoing deforestation and the spatial fragmentation of natural vegetation (Barros and Fearnside, 2016; Riquetti *et al.*, 2023a).

Soil organic carbon loss persists as a threat in the subregion. As an example, the Brazilian Amazon had 20.2 billion tonnes of organic C in the top 0–30 cm of soil in 1985 (MapBiomas, 2023a). Due to intense LULC changes since 1990, organic C storage decreased to 19.9 billion tonnes by 2015 and has continued to decline, reaching 19.8 billion tonnes in 2021 (MapBiomas, 2023a). Land-use and land-cover changes have been similarly profound across the entire subregion, indicating substantial SOC losses between 1990 to 2020. Conversion from forests to pastures can initially

increase C content in the topsoil (0.27 t/ha annually) but decreases in deeper layers (Rittl, Oliveira and Cerri, 2017). However, silvopastoral and agroforestry systems have the potential to reverse losses by enhancing SOC storage, compared to pastures, natural regeneration and secondary forest in deforested Amazon soils (Bermeo *et al.*, 2022; Olaya-Montes *et al.*, 2021; Rodríguez *et al.*, 2021; Suárez *et al.*, 2021).

Organic soils including wetland soils

Wetlands within non-forest natural formations in the Amazon experienced a significant 12 percent reduction between 2015 and 2021, corresponding to a loss of about 2 million ha (See Table 10.2 and Figure 10.12) (MapBiomas Amazonía, 2025). Wetlands were converted to pastures, agriculture or a mosaic of uses at a rate of 60 000 ha per year (MapBiomas Amazonía, 2022). However, estimating LULC changes in wetlands and flooded forests is a challenge due to seasonal water level fluctuations and complex vegetation responses to radiation.

The Andes glaciers, the water source of most Amazonian rivers, showed a significant decline of 42 percent from 1985 to 2015, followed by an additional 8 percent reduction (See Table 10.2) with annual loss rates varying from 2 400 ha in 2015 to 1 300 ha in 2021. Areas once occupied by glaciers are now mapped as either rock outcrops, other non-vegetated areas, or as river, lake or ocean categories. However, over time, the exposed rocky areas may evolve into grasslands, thus recovering some of the SOC lost (MapBiomas Amazonía, 2022; Velásquez Ramírez *et al.*, 2021).

10.4.4 Soil biodiversity change

The soils of the Amazon host a diverse and dense fauna, noted for its high taxonomic and functional richness, making it among the most diverse fauna globally (Lavelle *et al.*, 2021). Nonetheless, deforestation and fires remain a significant threat to soil organisms (Heijink *et al.*, 2023; MapBiomas Amazonía, 2022). Deforestation diminishes SOM and moisture, while elevating soil temperatures and altering edaphic conditions, endangering some soil organisms (Franco *et al.*, 2019; Moreno, Rueda-Uribe and Andrade-Pérez, 2018).

The establishment of intensive agriculture and pastures on cleared lands leads to a decrease in species richness and diversity within edaphic macroinvertebrate communities (Braga *et al.*, 2013; Decaëns *et al.*, 2018; Duran-Bautista *et al.*, 2020; Lavelle *et al.*, 2021; Navarrete *et al.*, 2016; Rodríguez-León *et al.*, 2021). Soil microfauna communities respond differently to LULC changes compared to macroinvertebrate communities, potentially impacting the ecological roles they play (Carvalho *et al.*, 2016; Peña-Venegas, Sterling and Andrade-Ramírez, 2021). Generally, microorganisms involved in root symbiosis exhibit more resilience (Lima *et al.*, 2009; Silva *et al.*, 2018), altering the composition of the communities without a notable impact on species richness and abundance (Leal, Siqueira and Stürmer, 2013). Some bacterial groups gradually recover after land-use intensification, approaching levels similar to those found in undisturbed forests (Pedrinho *et al.*, 2023).

Research on the natural regeneration of forests and reforestation in the Amazon subregion shows a recovery of over 90 percent in biological communities within a decade, linked to the restoration of soil functionality (Rodríguez-León *et al.*, 2021). To reduce the impacts of grasslands it is necessary to introduce low grazing intensity practices in cattle farms (Navarrete *et al.*, 2016), and to move toward agrosilvopastoral systems.

The current legislation imposes frail policies on the conservation of biodiversity, and the quality of forest remnants should be better considered (Decaëns *et al.*, 2018). It is also crucial to acknowledge the significant role of Indigenous Peoples in preserving forest and soil biodiversity. The utilization of edible mushrooms by the Yanomami tribes, and their involvement in ecotourism, exemplifies the

vital contribution of these communities to the conservation and sustainable management of soil biodiversity in the Amazon (Oliveira and Estrela, 2022; Sanuma *et al.*, 2016; da Cruz Pinto, de Mattos Vieira and Ishikawa, 2019).

In addition, Indigenous Peoples employ territorial management and ecological practices, rooted in their food and knowledge systems, that successfully maintain biodiversity, combat land degradation and ensure long-term food security. While scientific acknowledgement of these methods is growing, effective policy frameworks are still required to protect these practices and integrate their sustainable, scalable lessons into broader environmental strategies (FAO, Alliance of Bioversity International and CIAT, 2021).

10.4.5 Nutrient mismanagement in the Amazon subregion

The low nutrient content and high Al saturation are key limiting factors for monocropping agriculture in the Amazon (Bravo-Medina *et al.*, 2017, 2023; IGAC, 2015; Olaya-Montes *et al.*, 2021). Phosphorus is the most limiting nutrient and is primarily supplied to plants through the mineralization of organic matter under natural conditions (Moreira *et al.*, 2013). After deforestation, SOM rapidly depletes due to burning during land preparation and swift decomposition, decreasing the availability of both P and N (Cerri and Frade Junior, 2021; Hölscher *et al.*, 1996). There is low P in 56 percent of Amazonian soils, but its application via chemical fertilizers is inefficient due to rapid leaching by the heavy rainfalls of the Amazon (da Silva Cravo, Smyth and Bastos, 1987). In the medium term, intensive agriculture and pasture (cattle) increase soil acidity compared to silvopastoral systems and natural forest (Bermeo *et al.*, 2022; Bravo-Medina *et al.*, 2017). The lack of fertilization usage in the Peruvian Amazon and the continue prevalence of the slash and burn systems are depleting soil fertility with a rate of deforestation of 150 000 ha per year (La Barrera Noa, 2021).

Agriculture in the Amazon continues to expand (Table 10.2 and Figure 10.12) (MapBiomas Amazonía, 2025). However, much of this expansion lacks proper nutrient management due to limited soil analysis and the absence of recommendations tailored for their specific soils and climate. The use of liming and mineral fertilizers in recently cleared areas remains limited due to high prices in local markets, primarily due to transportation costs. Over the decades, these practices have resulted in soil nutrient depletion and more acidification, leading to reduced productivity and soil degradation, particularly in pastureland. In 2022, 15 percent of the pasture area in the Brazilian Amazon had “low vigour”, indicating possible soil nutrient degradation. The 2022 figure shows an increase from 2019, when the proportion of low-vigour pastureland was at its lowest (12 percent) since 2000, and 22 percent of pasturelands were rated as “degraded” (MapBiomas Brasil, 2023).

The establishment of agroforestry systems and the careful management of pastures in the Amazon contribute to improved soil fertility (Bermeo *et al.*, 2022; Olaya-Montes *et al.*, 2021; Suárez *et al.*, 2021). Nitrogen deficiencies can be addressed by utilizing naturally occurring legumes as cover crops or green manure. To increase P content, strategies that incorporate organic matter as biochar are recommended, similar to those used in the formation of the Amazonian Anthrosol (terra preta or dark earth) (de Castro *et al.*, 2022). Organic fertilization using “*cupuaçu*” fruit husks (*Theobroma grandiflorum*), rich in K, is highly effective when combined with liming for nutrient replenishment (Silva *et al.*, 2022). Additionally, the use of liming to regulate acidity and increase the availability of exchangeable cations could recover degraded areas in the Amazon (da Silva Cravo, Smyth and Brasil, 2012). Enhancing nutrient management in the Amazon also involves collaborative research initiatives with local farmers and Indigenous Peoples to blend scientific and traditional expertise in agroecology and organic production (Alfaia *et al.*, 2009; Pinho *et al.*, 2023).

10.4.6 Soil pollution in the Amazon subregion

The increase in gold prices has prompted a surge in demand and mining, along with government support. For instance, in 2018, over 188 mining titles were registered in the Colombian Amazon (WRI, 2020). This has led to a rapid expansion of mining operations in the Amazon, marking an 816 percent increase from 1985 to 2015, followed by a 32 percent rise from 2015 to 2021, resulting in an annual increase from 13 000 to 23 000 ha per year (Table 10.2 and Figure 10.12) (MapBiomas Amazonía, 2025). Many of the new mining operations are located within protected areas and conservation units, as well as Indigenous Peoples' lands and territories, without their Free, Prior and Informed Consent (*Office of the United Nations High Commissioner for Human Rights (OHCHR, 2021), June 2). Brazil: UN experts deplore attacks by illegal miners on indigenous peoples; alarmed by mercury levels. United Nations*) with a significant increase in the number of illegal airstrips (MapBiomas Brasil, 2023). Gold mining practices (many of which are illegal) persist in the Amazon, causing soil and water pollution (Velásquez Ramírez *et al.*, 2021). Studies confirm the contamination of river-dwelling people by Hg, a substance historically used due to its simplicity and cost-effectiveness in gold extraction. In a study involving 462 adults from eight riverine communities and one urban area, higher Hg exposure was observed among the riverine population (90 percent) compared to urban (57.1 percent). Alarmingly, 75.6 percent of the participants exhibited Hg concentrations exceeding the safe limit of 10 µg/L (Meneses *et al.*, 2022).

Diverse strategies can be employed to reverse the current trend in mining practices observed in the Amazon (MapBiomas Brasil, 2023). First, it is essential to remove illegally installed miners from Indigenous Peoples' lands and territories and ensure the respect to the right of Indigenous Peoples to a Free, Prior and Informed Consent (FPIC). Additionally, it is crucial to control the sale of metallic Hg for mining purposes, enhance the ability to trace the Au production chain, geolocate heavy machinery and monitor real-time signs of illegal mining expansion, along with the permanent protection of Indigenous Peoples rights over their lands, territories and resources, and conservation units.

10.4.7 Urbanization and soil sealing in the Amazon subregion

The Amazon has had a significant growth in population since the early 1970s (Perz, Aramburú and Bremner, 2005). The population of the Brazilian Amazon rose from 8.2 million in 1972 to 28.4 million in 2021 (Projeto Amazônia 2030, 2023). Population growth has driven the growth of urban infrastructure, which increased by 126 percent from 1985 to 2015 and by another 12 percent from 2015 to 2021 (Table 10.2). The increasing urban infrastructure means that previous land uses such as pasture and agriculture are being substituted for urban uses, leading to soil sealing and the loss of agricultural fields near urban centres (MapBiomas Amazonía, 2022). Urban expansion contributes to increased costs of food production and transport and affects the water cycle in the region, potentially exacerbating issues with water recharge and land movement near rivers (Kuhn *et al.*, 2023).



10.5 Regional status and trend of threats to soil functions for the Cerrado subregion

10.5.1 Soil erosion in the Cerrado subregion

There is extensive scientific support for conservation agriculture and no-till farming as best practices for controlling soil erosion in the Cerrado (Kassam *et al.*, 2015). Recently, systems like agroforestry and crop–livestock–forest integration (CLFI) have emerged in the Brazilian Cerrado as alternatives toward an agriculture that is sustainable (Anghinoni *et al.*, 2021). Despite the increase in research and government programmes toward reducing soil losses and other threats (Gurgel and Laurenzana, 2016; Polidoro *et al.*, 2021), it is still necessary to strengthen regulatory agencies to establish and monitor erosion prevention measures.

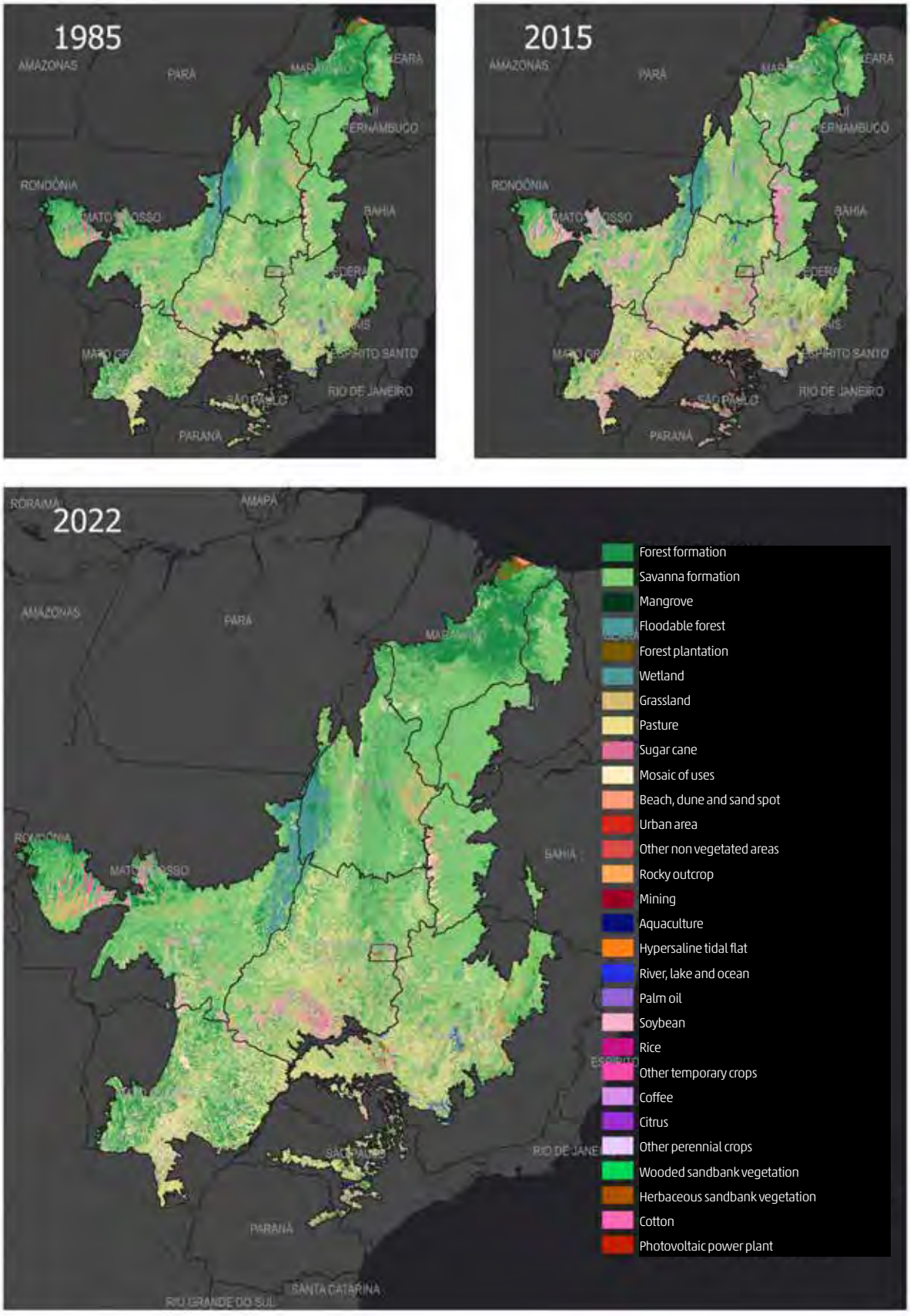
Water erosion

The annual rainfall in the Cerrado ranges from 1 000 mm, bordering the "Caatinga" biome in the semi-arid region of Brazil, to 1 800 mm, near the Amazon rainforest transition. Rainfall erosivity is high in the northern part of the Amazon biome in soils with high clay contents (Oliveira, Wendland and Nearing, 2013; Teixeira *et al.*, 2022). The mean annual rainfall is 1 444 mm annually, with an annual runoff of 83 mm, and an annual soil loss of 8.9 t/ha (Fonseca, Uagoda and Chaves, 2022). Higher rates of runoff and losses occur in soils without permanent cover (such as found in conventional tillage and bare soil). Soil loss can exceed 200 t/ha annually, associated with land-use changes, such as the conversion of natural vegetation into cropping or pastures, without conservation practices (Oliveira, Wendland and Nearing, 2013). Data from 20 sets of runoff erosion plots, covering 5 states and 10 locations in the Brazilian Cerrado, were compiled by Fonseca *et al.* (2021) and correlated with environmental and management factors and the corresponding soil loss tolerances. The authors found mean annual soil loss rates of 8.9 t/ha and that soil loss was significantly higher in soils with impermeable subsurface horizons and in land uses without permanent cover.

Batista *et al.* (2023) showed that centre-pivot irrigation could lead to soil erosion – mainly due to excessive watering rates in periods of low rainfall – or to the formation of circular pivot channels that could then lead to the incision of gullies. Records from 2010 to 2019 indicate an increasing trend in the number of natural disasters related to ravines and gullies (Kuhn *et al.*, 2023), coinciding with urban growth and land-use changes related to deforestation in the Cerrado, over recent decades. The rise in water erosion-related disasters is also tied to the expanding agricultural frontier, and accelerated water erosion. The incidence of ravines and gullies correlates with the release of substantial sediment loads, contributing to river silting and altered water dynamics.

MapBiomas data (MapBiomas, 2023) revealed that between 1985 and 2022, the Cerrado's forest cover decreased significantly, and the rate of forest area loss slowed from 850 000 ha annually (1985–2015), to 570 000 ha annually (2015–2022) (Table 10.3). Similarly, non-forest natural formations diminished, albeit at a slower rate. Farming areas expanded less rapidly, and pasturelands decreased in the latter period. Thus, while the deforestation rate has slowed, the continued conversion of land for agriculture could increase soil erosion threats and it is therefore crucial to address the connection between land management practices and water erosion to mitigate these soil threats in the Cerrado. The land-use dynamics in the Cerrado between 1985 and 2020 show a shift towards agricultural and pasture coverage (Figure 10.14). From 1985 to 2000, 15 percent of the region had at least one change in land use or cover. This value dropped to 5 percent from 2015 to 2020 (Table 10.3), suggesting that the land use stabilized. The reduction in stable forested areas points to ongoing deforestation and conversion into agrosystems, and requires policies for sustainable development and the conservation of the Cerrado biome.

Figure 10.14. Land-use and land-cover in the Brazilian Cerrado in 1985 (top left), 2015 (top right) and 2022 (bottom)



Refer to the disclaimer on page ii for the names and boundaries used in this map.

◊ *Source:* **MapBiomas.** 2023. *MapBiomas Land Cover and Use. Collection 8 of the Annual Land Cover and Land Use Maps of Brazil (1985-2022).* [Accessed on 3 November 2023]. https://data.mapbiomas.org/dataset.xhtml?persistentId=doi:10.58053/MapBiomas/VJIJCL. Licence: CC-BY-SA.

Table 10.3. Area occupied by natural land covers and anthropic land uses in the Cerrado ecoregion in 1985, 2015, and 2022, and their percentage change in 1985–2015 and 2015–2022 periods)

LULC class	Area in 1985 (ha)	Area in 2015 (ha)	Area in 2021 (ha)	Change in area (1985–2015) (%)	Change in area (2015–2022) (%)
1. Natural forest					
1.1. Forest formation	40 576 323	34 453 554	33 899 330	−0.2	0.0
1.2. Savannahh formation	79 772 962	59 536 650	55 531 431	−0.3	−0.1
1.3. Mangrove	62 978	67 948	68 232	0.1	0.0
1.4. Flooded forest	7 015	5 864	4 833	−0.2	−0.2
Total (class 1)	120 419 278	94 064 017	89 503 827	−0.2	0.0
2. Non-forest natural formation					
2.1. Wetland	6 785 717	6 095 414	6 233 634	−0.1	0.0
2.2. Grassland	11 321 869	8 760 070	8 819 175	−0.2	0.0
2.3. Rocky outcrop	569 130	568 995	569 081	0.0	0.0
Total (class 2)	18 684 380	15 432 161	15 629 352	−0.2	0.0
3. Farming and silviculture					
3.1. Pasture	34 253 917	55 389 527	52 326 826	0.6	−0.1
3.2. Agriculture	4 319 658	23 876 672	26 727 938	4.5	0.1
3.2.1. Temporary crop	3 981 117	22 478 237	25 208 546	4.6	0.1
3.2.1.1. Soybean	1 230 106	15 080 266	19 226 773	11.3	0.3
3.2.1.2. Sugar cane	258 117	2 899 838	3 046 503	10.2	0.1
3.2.1.3. Rice	5 421	68 102	93 949	11.6	0.4
3.2.1.4. Cotton (beta)		127 968	253 595		
3.2.1.5. Other temporary crops	2 487 474	4 302 062	2 587 726	0.7	−0.4
3.2.2. Perennial crop	170 347	617 768	679 383	2.6	0.1
3.2.2.1. Coffee	105 660	344 179	360 447	2.3	0.0
3.2.2.2. Citrus	49 572	217 680	228 591	3.4	0.1
3.2.2.4. Other perennial crops	15 115	55 909	90 345	2.7	0.6
3.3. Silviculture	546 999	2 817 315	3 225 857	4.2	0.1
3.5. Mosaic of uses	28 625 417	16 107 282	19 553 304	−0.4	0.2
Total (class 3)	67 745 992	98 190 795	101 833 925	0.4	0.0

Note: The land-cover changes in terms of natural vegetation (forest and non-forest areas) from 1985 to 2022 indicates a decrease in forest area from 120.4 million ha in 1985 to 89.5 million ha in 2022. Non-forest areas also declined but at a slower rate. The relative loss in forest area from 1985 to 2015 was 24 percent, with an annual loss rate of 0.66 percent. The annual rate of loss accelerated to 3.1 percent between 2015 and 2022.

◊ *Source:* **MapBiomas.** 2023. *MapBiomas Land Cover and Use. Collection 8 of the Annual Land Cover and Land Use Maps of Brazil (1985-2022).* [Accessed on 3 November 2023]. https://data.mapbiomas.org/dataset.xhtml?persistentId=doi:10.58053/MapBiomas/VJIJCL. Licence: CC-BY-SA.

Wind erosion

Wind erosion is prominent on sandy soils, due to their loose and unconsolidated structure. The decline in annual rainfall and the weakening of the South Atlantic convergence zone further exacerbates the threat (Campos and Chaves, 2020), albeit mainly to rainfed agriculture, and highlights the relationship between climatic shifts and soil erosion. The erosion-altered water dynamics contribute to a river's sedimentation, especially in low-elevation areas with shrubs and grasses, and in wetlands or "*veredas*". A combination of decreased rainfall and high evapotranspiration leads to plant stress, and a reduction in vegetative cover and topsoil protection, further degrading the soil.

Wind erosion is most intense when fields are being prepared for cultivation or in degraded pastures with low soil cover (Cordeiro *et al.*, 2020). Climatic and environmental conditions, combined

with changes in land use, have been linked to the increase in wildfires in the Brazilian Cerrado, particularly in the states of Maranhão and Tocantins. To prevent wind erosion, conservation practices are necessary, such as the maintenance of vegetative cover and windbreaks, the minimization of soil disturbance, and strategies to combat and prevent wildfires, as well as understanding the relationship between land cover, climate change and wind erosion.

Tillage erosion

Conventional farming systems, with intensive tillage and bare soil after cultivation, increase tillage erosion, deplete soil fertility and reduce crop yields and productivity (Polidoro *et al*., 2021). Tillage erosion also leads to the silting and eutrophication of rivers, lakes, and reservoirs, increasing floods and intensifying droughts (Kuhn *et al*., 2023). Tillage erosion disrupts the soil structure, displacing nutrients and releasing C. Under specific conditions, such as waterlogging or anaerobic states, eroded soil particles can also generate methane emissions (Lal, 2004), due to the soil acting as a C reservoir for plant material and microorganisms (Singh *et al*., 2023).

The rates of tillage erosion are minimal under no-till farming and uptake of the technique in the Cerrado has been very high. Five of the seven states in Brazil with the highest area of no-till or conservation agriculture relative to the area of annual cropping are found in the Cerrado region, with 87 percent in the Mato Grasso, 81.9 percent in Parańa, 70.7 percent in Goías, 70 percent in Distrito Federal and 69.3 percent in Mato Grosso do Sul (Fuentes-Llanillo *et al*., 2021).

10.5.2 Loss of soil organic carbon in the Cerrado subregion

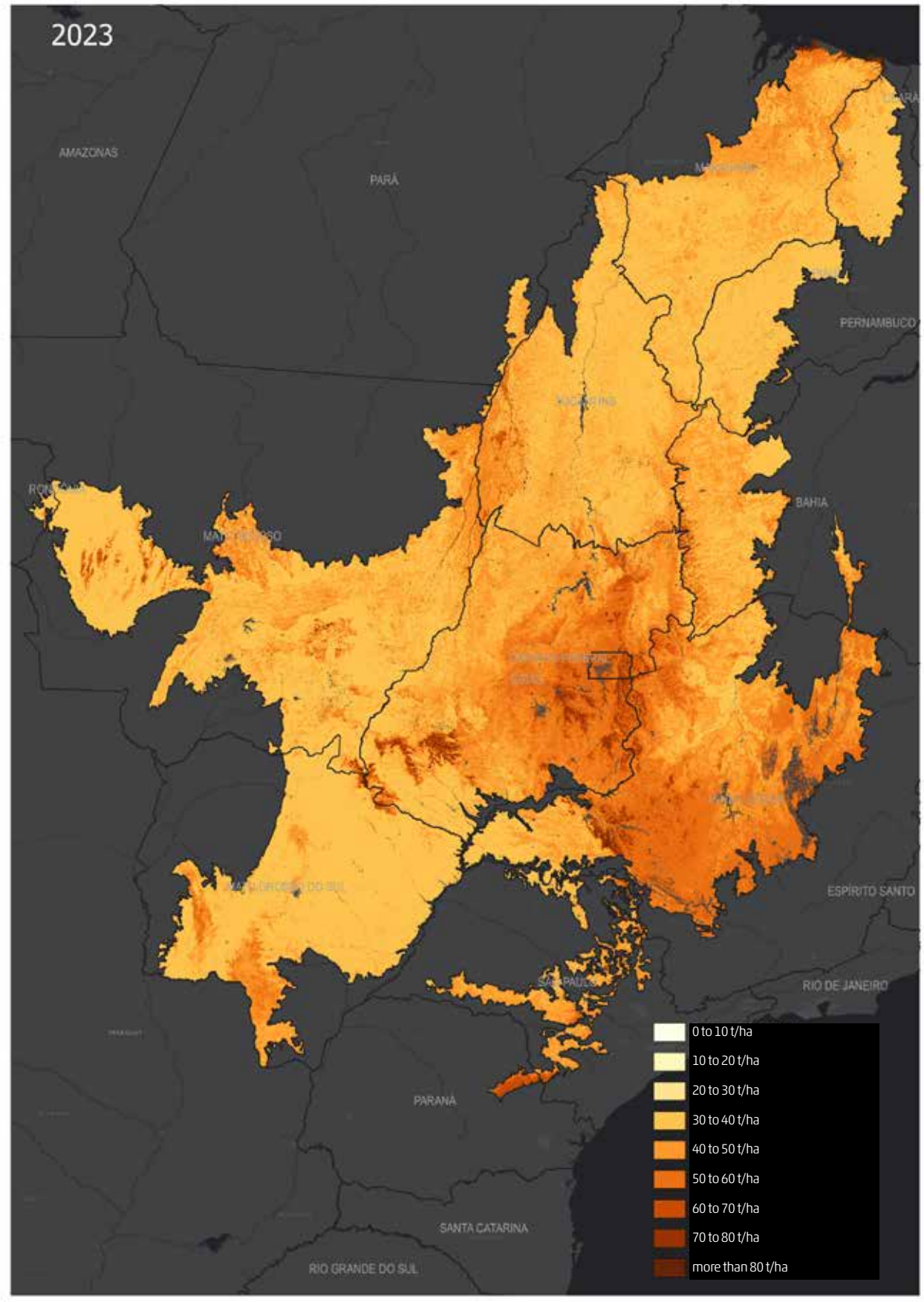
Mineral soils

It is estimated that 50 percent of the natural vegetated area of the Brazilian Cerrado has been converted to agricultural cropping (Beuchle *et al*., 2015; Dias, Moschini, and Trevisan, 2017; de Almeida Rodrigues *et al*., 2022), increasing potential losses of SOC. Cerrado soils contain about 34 percent of Brazil's soil C stocks (71.3 billion tonnes of C), storing 24 billion tonnes of C in the first 100 cm of soil (Gomes *et al*., 2019). According to Castellano, Santos and Menegário (2022), the C stock in different locations, soil classes, and under native vegetation and agriculture in the Cerrado was 13.5 ± 6.7 billion tonnes for the 0–30 cm layer, and 30.5 ± 18.9 billion tonnes for the 0–100 cm layer.

In 2015, pastureland spanned 53 million ha (26.6 percent) of the Cerrado (Parente *et al*., 2019), with 15 million ha (7.8 percent of the biome) showing low vegetative vigour, indicating severe degradation, while 16 million ha (7.9 percent) had high vegetative vigour, with no signs of degradation. Recent trends indicate a reduction in total pastureland area to 50 million ha (25.1 percent of the biome) (Parente *et al*., 2019). Encouragingly, the condition of the pastureland has improved from 2015 to 2022, with the highly degraded area of pastureland reducing to 26.9 percent of the biome, and the proportion of pastureland in good condition increasing to at least 33.4 percent (MapBiomas Brazil, 2022).

According to MapBiomas (2022), between 2015 and 2021, the total Cerrado SOC stocks appeared to be stable, shifting slightly from 8.2 billion tonnes to 8.1 billion tonnes in the topsoil (0–30 cm) (Figure 10.15), with nearly half of the SOC stocks in pastures and annual crops. The average SOC stock per unit area has remained stable in the natural areas of the Cerrado. A slight increase of SOC in agricultural areas – from 42 to 43 t/ha – was attributed to the improved vigour of pastures (Figure 10.16), due to better management practices (Parente *et al*., 2019). The ecological effects of degraded pastures extend beyond C storage, as they also affect biodiversity and increase soil erosion. The restoration of degraded pasture areas is crucial for ecological balance and agriculture and livestock productivity.

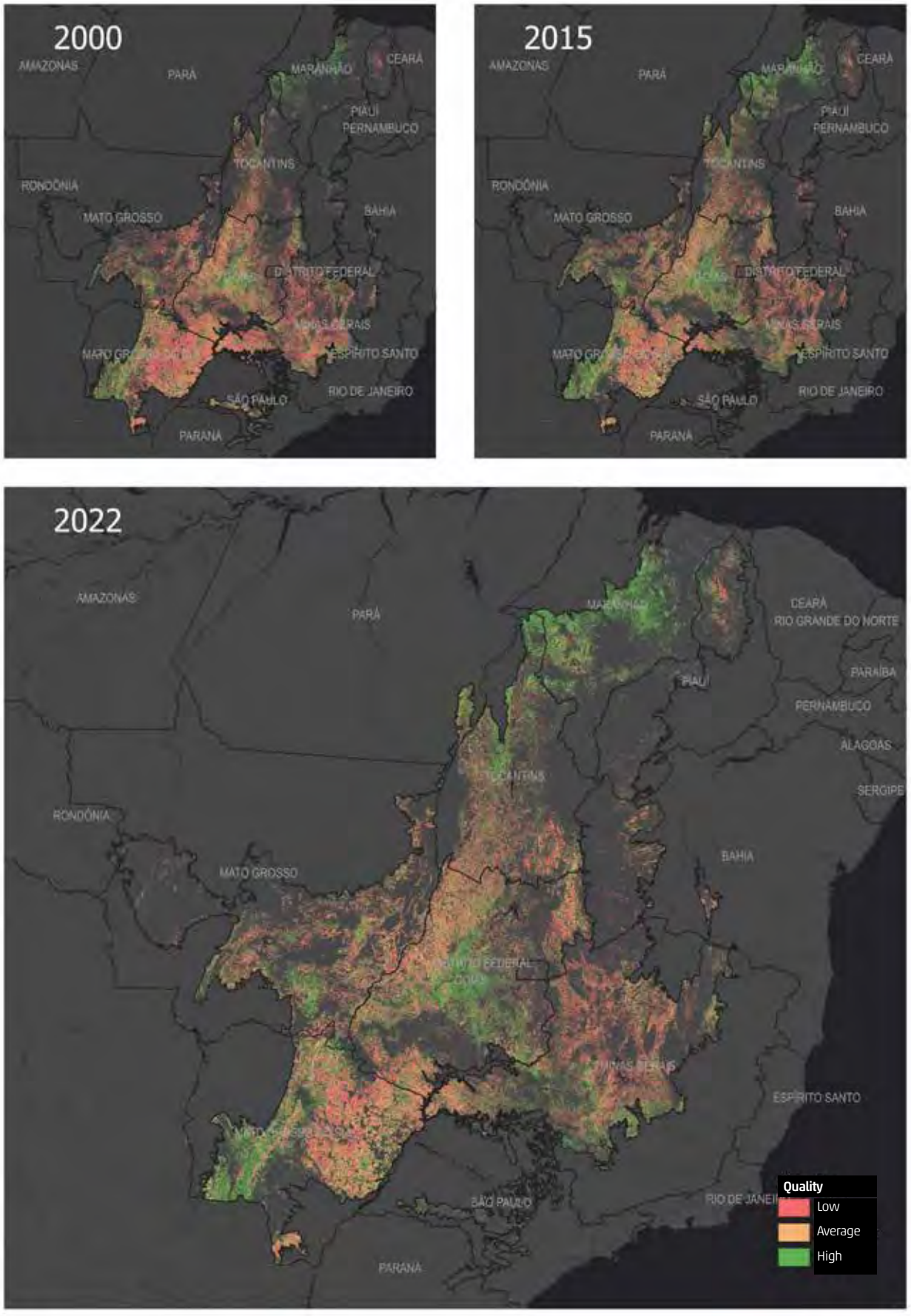
Figure 10.15. Soil organic carbon stock maps of the topsoil (0–30 cm) of the Brazilian Cerrado in 2021



Refer to the disclaimer on page ii for the names and boundaries used in this map.

◊ *Source:* **MapBiomas.** 2023. *Annual mapping of soil organic carbon stock in Brazil 1985-2021 (beta collection). Map collection.* [Accessed on 6 November 2023]. https://data.mapbiomas.org/citation?persistentId=doi:10.58053/MapBiomas/DHAYLZ. Licence: CC-BY-SA.

Figure 10.16. Pasture vigour within the Brazilian Cerrado, comparing 2000 (top left), 2015 (top right), and 2022 (bottom)



Refer to the disclaimer on page ii for the names and boundaries used in this map.

◊ *Source:* **MapBiomas**. 2023. *MapBiomas Land Cover and Use. Collection 8 of the Annual Land Cover and Land Use Maps of Brazil (1985-2022).* [Accessed on 3 November 2023]. https://data.mapbiomas.org/dataset.xhtml?persistentId=doi:10.58053/MapBiomas/VJIJCL. Licence: CC-BY-SA.

Organic soils including wetland soils

The Cerrado's wetlands ecosystem, called *veredas*, mainly consist of Gleysols, Histosols and Fluvisols. It is estimated that more than 300 000 ha of the *veredas* have soils with histic (organic) horizons (Canto *et al*., 2020). Organic C stocks in Brazilian Histosols average 204 t/ha up to 1 m depth (Valladares *et al*., 2016). Santos *et al*. (2023) reported C stocks of 523 ± 56 t/ha to 576 ± 68 t/ha, and C losses of 10 percent (53 t/ha), attributed to the erosion and decomposition of SOM following the drainage of two Histosols studied. In addition to the hydrological importance for aquifer recharge, controlling floods and improving quality of water, the *veredas* are hotspots of enormous diversity of fauna and flora, as well as storing organic C, regulating biogeochemical cycles and ultimately helping to stabilize the climate by preventing emissions of GHGs to the atmosphere (Araújo *et al*., 2023; Horák-Terra *et al*., 2022).

Soil inorganic carbon

In the Cerrado, studies on soil inorganic C are scarce. However, fire is a common factor influencing the dynamics of SOM, increasing the mineralization of more labile SOM fractions and forming black carbon (inorganic or pyrogenic C) (Schmidt and Noack, 2000). Black carbon is rapidly incorporated into the soil, and due to its chemical composition and morphology, is not easily distinguishable from organic C from biological processes (Skjemstad and Graetz, 2003). Looking at Ferralsols from Serra do Espinhaço (Brazil), Schellkenes *et al*. (2023) found that soils with thick A horizons and high C content had more C originating from trees, with a greater amount of charcoal and higher soil macrofauna activity. The contribution of charcoal, with relative persistence against degradation and faster downward movement through bioturbation, led to higher C content in the surface horizons.

10.5.3 Soil biodiversity loss in the Cerrado subregion

The Cerrado is one of the world's 34 biodiversity hotspots (Mittermeier *et al*., 2004), with over 4 800 vertebrate animals and plants living only in this biome (Sawyer *et al*., 2017). Invertebrates have a high endemism, including 40 percent of global termite species (Constantino, 2015). Another emblematic species, the giant armadillo (*Priodontes maximus*) has major impacts on soil functions and is red listed as vulnerable (Brazil, 7 June 2022). Nearly 1 200 species in the Cerrado are endangered (IBGE, 2023), and the agricultural expansion in the region has been projected to drive as many as 480 plant species to extinction by 2050, negatively affecting ecosystems and service provisioning (Strassburg *et al*., 2017).

The Cerrado's soil biodiversity is a source of potentially useful species for farming, particularly of microorganisms like bacteria, archaea and fungi, but land-use conversion leads to major changes in their communities (Castro *et al.,* 2008; Souza, Mendes and Reis-Junior, 2016). Native N^2-fixing microbes allow for large-scale soybean production in Brazil, including the Cerrado, reducing N fertilizer costs by up to USD 15.2 billion annually (Telles, Nogueira and Hungria, 2023). Many soil invertebrates inhabit Cerrado soils and are threatened by agricultural intensification (Pompermeier *et al*., 2020). Soil macrofauna studies in the Brazilian Cerrado using standard methods (ISO, 2011) showed that termites represented 52 to 64 percent of the total population, except in agriculture lands (20 percent of the total), where they decreased. Annual cropping reduced overall macrofauna populations and group taxa richness by 72 percent and 33 percent, respectively, while pastureland and CLFI systems had higher group diversity and more active populations.

The transition to agriculture significantly diminishes the habitat complexity required by a diverse array of soil organisms. The simplification of the ecosystem is likely to exacerbate the loss of endemic and specialized species, which could impair ecological resilience and the capacity for natural regeneration. The LULC changes in the Cerrado (Table 10.4) since 2015 (MapBiomas, 2023b) has implications in the decline of soil biodiversity.

Table 10.4. Number of land-use and land-cover changes in the Cerrado subregion over 1985–2000 and 2015–2021 periods

Number of changes (LULC)	1985–2000 (ha)	2015–2020 (ha)	1985–2020 (ha)
0	165 851 296	189 581 130	143 999 033
1	24 103 919	8 204 076	33 496 654
2	6 967 751	660 966	12 924 476
3	1 283 136	9 147	4 811 035
4	224 514	89	2 162 542
5	20 295	3	697 353
>6	4 500	0	364 319

◊ *Source:* **MapBiomas.** 2023. *MapBiomas Land Cover and Use. Collection 8 of the Annual Land Cover and Land Use Maps of Brazil (1985-2022).* [Accessed on 3 November 2023]. https://data.mapbiomas.org/dataset.xhtml?persistentId=doi:10.58053/MapBiomas/VJIJCL. Licence: CC-BY-SA.

Fire events are increasing in frequency, duration, and severity, affecting soil biodiversity in the Cerrado (Nagel *et al.*, 2023). The prevalence of fire has shifted from natural to human-induced occurrences (Silva *et al.*, 2021), often set to clear land for agriculture or to control weeds in pastures. Fires occur mainly during the dry season and are now more widespread and recurrent, critically endangering soil biodiversity and the balance of the ecosystem. The natural Cerrado vegetation originally covered 51 million ha and is now reduced to only 28 million ha, while the role of natural or human-induced fires have increased significantly (MapBiomas, 2022). The sharp change from just 11 million ha of farming land in 1985 to 79 million ha in 2022 (MapBiomas, 2022) underlines a profound transformation in land utilization. The recurrent blazes, regardless of source, eradicate the soil's cover, leading to erosion, and disrupt the delicate network of life underground, thereby curtailing both the diversity and the functional capacity of the soil's biotic community.

A pioneering initiative was adopted in Brazil in 2020, involving the inclusion of soil enzymes in routine soil analysis to assess soil health in agricultural lands, It is known as "soil bioanalysis" or SoilBio and by November 2022, a database containing 15 788 soil sample results had been collected (de Carvalho Mendes *et al.*, 2024).

10.5.4 Nutrient mismanagement in the Cerrado subregion

Nutrient mining

Overall, the balance of nutrient consumption in Brazil decreased from the 2013–2016 period to the 2017–2020 period (da Cunha *et al.*, 2023). Fertilizer use efficiency rates in the 2017–2020 period were 51.2 percent for N, 48.4 percent for P_2O_5 and 73.6 percent for K_2O (Cunha *et al.*, 2023)). The use of N-fertilizers has decreased along the years, and, except in the case of soybeans, where the N is mainly from BNF, there is an urgency to improve the dissemination and adoption of good practices for efficient use of fertilizers. Conversely, secondary macronutrients (such as Ca and Mg) and micronutrients are being applied in quantities that meet the export demands of crops (Cunha *et al.*, 2023).

Agricultural management of nutrients in the Cerrado has evolved greatly in terms of technology. The combination of crop cultivars better adapted to the soils, the adaptation of machinery and the efficient fertilizers and various bioinputs, has increased the efficiency of nutrients management. Since the start of the twenty-first century, there has been a wide-scale adoption of the no-till practices in the Cerrado, but due to crop intensification (where two crops are grown per season) and drought conditions, the results in terms of nutrient balance are still below the optimal, with SOC and N often being diagnosed as below adequate levels in agricultural soils. More studies are needed, especially for improving the crop rotation using legumes and gramineous plants (Anghinoni *et al.*, 2019; Anghinoni *et al.*, 2021; Cunha *et al.*, 2023).

Acidification

Soils in the Cerrado are highly weathered and naturally acidic, requiring liming to increase crop productivity. However, the successive fertilizer applications to replenish losses by leaching and the absorption by the crops over time (and depending on the dose applied), can increase soil acidification, especially in the case of N fertilizers (Francioli *et al*., 2016). The application of products with slow or controlled release technology has increased significantly in the Cerrado soils with better results compared to conventional fertilizers. But this is a new field of research, requiring investment for development and testing the technology as well as regulations.

10.5.5 Salt accumulation in the Cerrado subregion

Saline and sodic soils are mainly observed in the northeast of Brazil, in some irrigated perimeters and coastal areas. Soils with a higher salt content occur naturally in the Pantanal biome, which borders the Cerrado. However, as water quality is threatened by the ongoing expansion of irrigated crops in the Cerrado and urban expansion, (Bichsel *et al*., 2016), this represents a potential future risk of salt accumulation.

Irrigation and drainage

The total irrigated area in the Brazilian Cerrado exceeds 1.3 million ha, which corresponds to about 5 percent of the total area of the biome, In 2022, 93 percent of the area was under centre-pivot irrigation, with the remaining 7 percent being irrigated by flooding (MapBiomas, 2023). The area under centre-pivot irrigation surged from 38 000 ha in 1985 to 864 000 ha in 2015 and to 1 million ha by 2022. Soybeans under irrigation expanded from 2 000 ha in 1985 to 891 000 ha in 2022. Similar rates of growth were observed in sugar cane and other temporary crops. Cotton, not present in the 1985 data, reached 8 000 ha by 2022. Perennial crops under irrigation also grew, especially with coffee and citrus. Inundation or surface irrigation by flooding for rice expanded from 5 000 ha in 1985 to 94 000 ha by 2022 (MapBiomas, 2022).

Alteration of hydrological conditions

According to de Almeida Rodrigues *et al*. (2022), land-use changes in the Cerrado from forest vegetation to areas cleared for cropland or pastureland increased the subregion's land surface temperature (LST) by about 3.5 °C; while converting native grasslands increased average LST by 0.9 °C in cropland and by 0.6 °C in pastureland, In addition, changes in land use led to a 10 percent reduction in the water recycled into the atmosphere and a 0.9 °C increase in soil temperature throughout the biome, compared to the historic baseline under native vegetation. In relation to the hydrological cycle, Nóbrega *et al*. (2017) analysed the impact of land-use change on edaphic attributes and surface water flow in watersheds under the Cerrado *sensu stricto* and extensive pasture. The conversion to pasture has led to degradation, lower evapotranspiration, higher water flow and a higher runoff coefficient. These changes may increase salt accumulation in biomes such as the Pantanal.

10.5.6 Soil pollution in the Cerrado subregion

The Cerrado has one of the world's largest agricultural outputs, with high pesticide use in monocrops (principally cotton and cereals). Glyphosate, 2,4-D, and atrazine are the key products (Ribeiro *et al*., 2023), as well as mancozeb and imidacloprid. Although there are no comprehensive studies about the total area of the Cerrado affected and the pesticide residues in soils, Teodoro *et al*. (2023) gave an estimated average consumption of pesticide per crop, along with the risks associated with

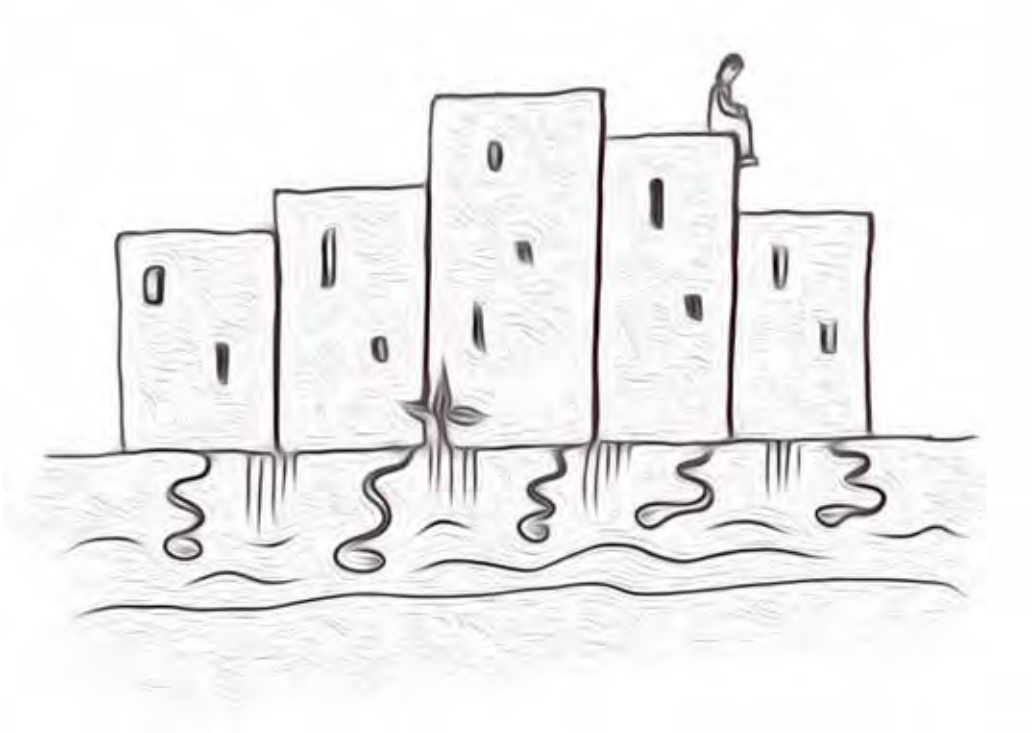
glyphosate and atrazine contamination in the state of Goiás, in the Brazilian Cerrado, where soybean is the predominant crop (78.04 percent of the cultivated area), followed by maize and sugar cane. The usage of pesticides was estimated as: soybean (17.7 litres [L] per hectare), maize (7.4 L/ha) and sugar cane (4.8 L/ha).

Urgent government action and policies are needed to regulate agrochemical use, as well as the monitoring of residues in soil and water, and the evaluation of risks for humans and animals. The contamination of aquifers and natural waterways also requires long-term studies and investments in controlling soil erosion (among other factors that increase this threat).

10.5.7 Other threats in the Cerrado subregion

Studies on urban expansion and soil sealing are scarce, but the Cerrado population was greater than 25 million in 2022, and urban areas have increased by 9 percent from 752 000 ha in 2015, to 820 000 ha in 2022 (Mapbiomas, 2023).

The expansion of agriculture and the fragility of the Cerrado inevitably leads to the increasing degradation of the biome and soils, with habitat fragmentation and biodiversity reduction, invasion by exotic species, aquifer pollution, an increase in fires and GHG emissions, rising temperatures and longer periods of drought, imbalances in C and nutrient cycles and, possibly, changes in regional climate (de Almeida Rodrigues *et al.*, 2022). The reduced number of conservation units and the depletion of natural resources, soil, water, and biodiversity, require urgent measures from the government, investments in research and policies to protect areas still under natural vegetation.



10.6 Regional status and trend of threats to soil functions for the Del Plata Basin subregion

10.6.1 Soil erosion in the Del Plata subregion

Soil erosion in the Del Plata Basin subregion is mostly caused by water in steep humid landscapes and by wind under semi-arid conditions with sandy soils. The specific drivers of soil erosion in the Del Plata Basin subregion are deforestation (mainly in the Chaco biome), the replacement of grasslands or pasturelands by annual crops, overgrazing, monocropping and planted forests. In Uruguay, a system called conservation forest plantation – which covers only one-third of the total surface and in contour – has proven to be effective in reducing erosion risks (Denis Lepiane and García Préchac, 1997).

As agriculture expands over the Pampa natural grasslands, the erosion trend could increase (Borrelli *et al*., 2017). Management options to reduce or eliminate the threat of soil erosion are those as recommended in conservation agriculture, such as adoption of no-till, the planting of cover crops, the construction of terraces on steeper slopes and preserving the grasslands of the most vulnerable soils.

The main barrier to effective soil erosion control and the adoption of conservation management options is the financing required by farmers to construct terraces for contour cultivation. There is also a lack of regulations that favour soil conservation and law enforcement. Many institutions in the countries of this subregion have low quality research data, leading to a knowledge gap, and long-term and large, watershed-level plot studies on soil erosion are needed. Any studies also need to be supported by government policies that include financial assistance for universities and institutions, and that are able to fund an increase in research personnel.

Water erosion

More than 1 million km^2 in Argentina is estimated to be affected by water erosion (Casas and Albarracin, 2015) and it is also the main threat in the other countries of the region. This scenario did not change much since the 2015 SWSR report (FAO and ITPS, 2015). Gaitán *et al*. (2019) estimated a cumulative loss of USD 10.06 billion due to decreases in soybean, wheat and maize yields by soil erosion in Argentina. Since 2015, trends in the water erosion threat show a decrease or a stabilization in the areas due to the prevalence of no-till. However, despite the generalized adoption of no-till farming in the subregion, some crop rotations are based on a single and low residue crop as soybean, with a low production of phytomass that does not favour ground cover protection against erosion (Rubio *et al*. 2019a, 2019b). At least in Argentina, the proportion of land managed by soybean monoculture is diminishing in the last years. Notable exceptions are areas under no-till farming in southern Brazil and in the southeastern Buenos Aires province, where thicker layers of crop residues are left to cover the soil.

More frequent extreme weather events would worsen water erosion threats, given changes in rainfall patterns. Based on RUSLE predictions, an increase in soil erosivity (the R-factor) has been projected for eastern Argentina and southern Brazil, with a decrease in the R-factor for southeastern Brazil. Both the increased rainfall intensity and increased drought may be harmful and indicate increased soil erosion vulnerability (Riquetti *et al*., 2020b). A significant increase in water erosion nationwide has been predicted in Argentina, affecting the whole Pampas region in the Del Plata Basin subregion (Casas and Albarracín, 2015; Casas, Godagnone and de la Fuente, 2022a). Huge variations in the predicted soil erosion loss (1.6 to 2 020 t/ha annually) have been estimated, due to different rainfall intensities, soil cover and soil relief (Gaitán *et al*., 2017).

Wind erosion

In the semi-arid sector of the Del Plata Basin subregion, soils are formed on loessic materials and are highly susceptible to wind erosion. Wind erosion was largely promoted by land-use changes, replacing grasslands and pastureland by annual crops like soybean and maize. An increase of wind erosion processes was registered in these semi-arid areas, when not covered by crop residues (Casas and Albarracín, 2015; Casas, Godagnone and de la Fuente, 2022b). In the sandy soils of the Brazilian Pampa, the "arenization" process is a consequence of the high mobility of soils and sediments subjected to wind and water erosion and promoted by agriculture and overgrazing (Suertegaray, 1987; Rovedder and Eltz, 2008; Rovedder, Eltz and Schenato, 2019). Despite being a well-recognized threat, the rate of wind erosion has accelerated since 2015, mainly due to the expansion of soybean farms on natural grasslands. It is therefore imperative to implement appropriate restoration techniques tailored to the specific needs of eroded areas.

Tillage and root crop harvesting

No-till farming has been extensively adopted by the farmers in the Del Plata Basin subregion (Derpsch and Friedrich, 2009; Rubio *et al*., 2019b). However, despite the predictions about the healthier soils using such practices (Calegari *et al*., 2020), soil erosion still persists (Casas and Albarracin, 2015; Casas, Godagnone and de la Fuente, 2022a). This is due to the ineffectiveness of the protection offered by low phytomass associated with non-diverse crop rotations, with almost continuous production of soybean crops (Novelli, Caviglia and Piñeiro, 2017). Complementary conservation practices are necessary to control runoff and the associated processes, from soil erosion and nutrient and pesticide mobilization to water resources (Deuschle *et al*., 2019). This should be combined with more diverse and intensified crop rotations and integrated crop–livestock systems (de Moraes *et al*., 2019; Rubio *et al*., 2019b). The area under tuber and root crop cultivation is not high, with the main crops being potato, groundnut and some vegetables (Kuhwald *et al*., 2022).

Mass movement

The Del Plata Basin subregion is mainly covered by large plains with some sloped areas (such as the Tandilia and Ventania hill systems in the southern Buenos Aires province). Except in the steep areas where high soil erodibility has been predicted using RUSLE (Gaitán *et al*., 2017), mass movements are not expected to be significant. However, in Argentina, the province of Entre Ríos shows the greatest impact of mass movements, with only 25 percent of its area not susceptible to erosion, constituting a major issue (Panigatti, 2010).

10.6.2 Loss of soil carbon in the Del Plata subregion

Mineral soils

The Del Plata Basin subregion has a predominance of black soils (Chernozems), considered to be some of the best soils in the world for crop production (FAO, 2022b), because of their high organic matter content and natural fertility. As already reported in the 2015 SWSR report, continuous cropping has increased SOC mineralization and reduced its content, causing lower productivity and affecting ecological services (FAO and ITPS, 2015; FAO, 2022a). The SOC content of the agricultural soils of the Argentine Pampa decreased between 2011 and 2019 (Sainz-Rozas *et al*., 2019), as well as in the agricultural soils of Uruguay due to the change of land use to annual crops (García-Prechac *et al*., 2022) with consequences for CO_2 emissions (Baldassini *et al*., 2023). Likewise, the Brazilian Pampa is under environmental stress, given that 22 percent of the area is cultivated with soybean, and the area under silviculture increased by 17 times between 1985 and 2022, with a further 97 800 ha being converted from natural grasslands to agriculture between 2000 and 2019 (MapBiomas Project, 2022).

Fortunately, the replacement of grasslands by annual crops – the main cause of a reduction in SOC – has decreased since 2015 in some areas, due to the adoption of more sustainable practices, such as cover cropping, more diverse crop rotations and precision agriculture (Nicolas *et al*., 2022; Satorre and Andrade, 2021; Wyngaard *et al*., 2022).

The Pampa grasslands and pastures are highly recommended for climate mitigation through SOC sequestration (Bai and Cotrufo, 2022; Poeplau, 2021). Studies show that the inclusion of gramineous and leguminous cover crops could increase SOC and its labile fractions content across the Pampas (Duval *et al*., 2016; Frasier, Quiroga and Noellemeyer, 2016; Frasier *et al*., 2016; Álvarez, Berhongaray and Giménez, 2021; Beltrán *et al*., 2018; Landriscini *et al*., 2020; Viglizzo *et al*., 2019, 2020). The inclusion of grasses as cover crops has also increased SOC stock by 3.1 t/ha in the Pampa's coarse-textured soils (Beltrán *et al*., 2021). In Uruguay, areas of Molisols and Vertisols under crop–pasture rotations with no-till have proved to be the best combination of soil usage and management to keep SOC stocks very close to the original content, when compared to pastures in long-term experiments (García-Préchac *et al.,* 2022).

Organic soils including wetland soils

The Del Plata Basin is a connected freshwater wetland system that includes areas in Argentina, the Plurinational State of Bolivia, Brazil, Paraguay, and Uruguay, and under diverse climatic zones. There are 29 Ramsar sites, which comprise almost 85 000 km^2 and 18 biosphere reserves, covering almost 361 000 km^2 (CIC, 2017). One-third of the freshwater marshes in the lower basin have been lost due to cover replacement, with pasture expansion accounting for 70 percent of losses and forestry for 18 percent (Sica *et al*., 2016). In these ecosystems, according to Puig, Olguín-Salinas and Borús, 2016, concentrations of *Escherichia coli*, Cd, Pb, Fe, Mn, and ammonium (NH_4^+) exceeded guideline levels under a severe drought, and a dispersal of cyanobacteria appeared under a high-flow pulse in an La Niña year, increasing the risks for ecosystem and humans. Other threats are foreseen due to the expansion of the Paraguay-Paraná hydroway for commercial trade, and the planned installation of new dams in the Paraná basin (Botana and Fernández, 2017; Baigún and Minotti, 2021).

10.6.3 Soil biodiversity loss in the Del Plata subregion

In the Del Plata Basin subregion, the greatest impacts on soil biodiversity comes from changes in land use from grasslands, pasturelands and savannahhhs into growing annual crops. These changes first occurred in the early twentieth century but were intensified in the beginning of the twenty-first century (Baeza and Paruelo, 2020). Between 2000 and 2020, at least 2.4 million ha of grassland (9 percent of the remaining grassland area in 2001) were converted (Baeza *et al*., 2022). The losses are concentrated in Brazil and Uruguay, with the majority of losses due to the expansion of forestry (Baeza *et al*., 2022). In a recent publication, Paruelo *et al.* (2024) assessed the environmental performance of Uruguayan farms larger than 5 ha, using seven synoptic indicators derived from remote sensing.

Besides the loss of biological corridors for the regional fauna caused by agricultural expansion, studies on soil biodiversity are less encompassing and based on localized data. In two farms of the Entre Rios province, El Mutjar *et al.* (2021) observed that a low grazing intensity in temperate woodlands increased bacterial biodiversity while decreasing fungal community structure (associated with higher vegetation cover.) In the Brazilian Pampa, the conversion from natural grassland to agropastoral systems was found to maintain populations of native earthworms and possibly enhance some ecosystem services (Gillanders *et al*., 2023). It was found that a greater biodiversity was recorded under natural forest and grasslands (Góes *et al*., 2021), while the biological community was affected when natural grasslands were converted into monocultures or eucalyptus plantations (Stroher *et al*., 2023). López Zieher, Vivanco and Yahdjian (2020) found that

bacterial communities in grassland soils were associated with changes in plant communities after agricultural abandonment. However, results from the Biospas project showed that the adoption of good agricultural practices in productive no-till fields increased leaf litter and soil invertebrate abundance and modified faunal composition (Bedano *et al.,* 2016). The inclusion of gramineous and leguminous cover crops improved and stabilized soil microbial activity in the western Pampa region (Frasier, Quiroga and Noellemeyer, 2016; Frasier *et al.,* 2016).

Another factor is the impact of herbicides in the no-till farming systems of Pampean soils, where glyphosate is used. Studying three different soils, Hernández-Guijarro *et al*. (2018) observed that the application of glyphosate appeared to disrupt the bacterial association network. In the humid Pampean region, gene microbial community analysis showed a stronger microbial seasonal dynamic under conservation agriculture compared to monocropping (Fernandez-Gnecco *et al*., 2021).

10.6.4 Nutrient mismanagement in the Del Plata subregion

Nutrient mining

In the 2015 SWSR report, nutrient imbalances were reported in agriculture and livestock production in the region, due to low dosages of fertilizers and the low adoption of BNF (FAO and ITPS, 2015). In Argentina, annual average deficits of 22.4 kg/ha for N, 6.9 kg/ha for P and 2.1 kg/ha for S still exist (de Astarloa and Pengue, 2018; Koritschoner *et al.,* 2023). In Brazil, the nutrient deficit was also high for N, P and S, with the exception of soybean crops which provided BNF. A deficit was also reported for exchangeable cations (Ca, K and Mg) in Pampean soils, where grain production increased without adequate fertilizer replacement (Sainz-Rozas *et al*., 2019; Larrea *et al*., 2023). At current rates of exchangeable cation depletion, deficiencies could become evident in the medium and short term.

Acidification

The soils of Del Plata Basin subregion were generally considered to have a positive surface cation and there was concern that modern agricultural practices could acidify some soils with low carbonate content (Álvarez *et al*., 2020). However, a recent study by Larrea *et al*. (2023) on Pampas Mollisols did not find any acidification due to decreases in exchangeable cation contents in unfertilized crops. However, Iturri and Buschiazzo (2016) observed light acidification associated with N fertilization in Pampean Hapludolls.

10.6.5 Salt accumulation in the Del Plata subregion

Irrigation and drainage

A study on the supplementary irrigation of crops in the Argentine Pampa showed that most water used had high sodium bicarbonate concentrations, which increased pH by 9 percent, EC by 34 percent and ESP by 177 percent (Álvarez, Rimski-Korsakov and Torres-Duggan, 2020). High pH caused by irrigation with sodic waters is usually found in urban and peri-urban horticulture soils around Buenos Aires (Paladino *et al*., 2020) and other cities.

Alteration of hydrological conditions and coastal waterlogging

The 2015 SWSR report (FAO and ITPS, 2015) suggested that topsoils in large plains with saline groundwater and sodic soils (such as Solonetz) may be salinized by soluble salts rising through capillary action, promoted by land clearing and overgrazing. These drivers of topsoil salinization and groundwater continue to be present. Coastal waterlogging is a natural feature of tidal marshes

in the Samborombón bay coast, at the mouth of the Rio de La Plata in Argentina, a Ramsar site (Government of Argentina, 2024). It is likely that salinization will increase if sea levels rise due to climatic changes.

10.6.6 Soil pollution in the Del Plata subregion

In the Del Plata Basin subregion, medium and large cities coexist with farms using extensive agricultural production and the high use of pesticides, increasing the risk of soil pollution and adverse effects on the local population. Between 2011 and 2013, the National Institute of Agricultural Technology (INTA) analysed pesticides in 1 357 samples of surface and groundwater in the western and southeastern Pampas, jointly monitoring with the Regional Consortium of Agricultural Experimentation (CREA) (Vázquez-Amábile *et al*., 2018). The levels of agrochemicals were mostly below the limit of quantification (LOQ) and far below the critical limits. Their presence in the samples was linked to rainfall events, with AMPA, the major metabolite of glyphosate, only detected in surface waters due to soil erosion after extreme rainfall in 2012. Regarding nitrates, the average level in water courses was 5 mg/L, while the concentration in groundwater varied with precipitation and landscape position, in many cases exceeding 10 mg/L (Vázquez-Amabile *et al*., 2018).

In agricultural soils the main source of non-point pollution are herbicides, which are mainly used for weed control in no-till systems, as illustrated by many studies from 2017 to 2021 (Andrade *et al*., 2017; Primost *et al*., 2017; Okada *et al*., 2018; Lupi *et al*., 2019; Pérez-Bidegain *et al*., 2021). Biodegradation is the major process responsible for the degradation of glyphosate (Fernandez-Gnecco *et al*., 2021; Hernández-Guijarro *et al*., 2018; Bento *et al*., 2019), with rates being determined by the soil properties in the loess soils of the Pampa. As for the breakdown of AMPA, the rate is lower than expected, which may have a higher impact on soils.

Brazil has very high pesticide use, with an average of 12.6 kg/ha of total pesticide application compared to a global average of 2.4 kg/ha (FAO, 2025b) As noted by Niemeyer, Chelinho and Sousa (2017), the predominant herbicides were glyphosate (62 percent), 2,4-D (15 percent), atrazine (7 percent), paraquat (5 percent), and diuron (2 percent). Soil and water contamination with glyphosate and AMPA were studied by Fernandes *et al.* (2018, 2023). According to these authors, herbicide residues still persist in biofilms within the river sediments, which should also be sampled in addition to water to monitor contamination.

Argentina and Brazil also have no restrictions on protecting aquatic life from the use of atrazine, acetochlor, imidacloprid and glyphosate in crops (Pérez *et al*., 2021).

Soil pollution with heavy metals like Cr, Pb, Cd and Ni and groundwater pollution with nitrate and hyper-fertilization with P are of concern for human health, mainly in the Matanza-Riachuelo basin, near Buenos Aires (Argentina), which is also highly polluted with irregular industrial discharges (Mendoza *et al*., 2015; Cittadino *et al.,* 2020; Paladino *et al*., 2020; Ceballos *et al.,* 2021; Magni, Castro and Rendina, 2021). This issue is not a new phenomenon, but its status is largely dependent on the strength of controls put in place by the Government of Argentina.

10.6.7 Urbanization and soil sealing in the Del Plata subregion

The Del Plata Basin subregion is characterized by its low rural population (Taboada *et al*., 2021), but it has some of the most populated megacities of Latin America, such as Buenos Aires (Argentina), Porto Alegre (Brazil) and Montevideo (Uruguay). The three cities are surrounded by crop belts, which are increasingly encroached upon by new unorganized urbanizations. Soil sealing around these cities is an increasing threat.

10.7 Regional status and trend of threats to soil functions for the Andean countries and Patagonia subregion

10.7.1 Soil erosion in the Andean countries and Patagonia subregion

Water erosion

All the lands located along the Andes mountain range are susceptible to landslides, soil erosion and flooding in the valleys, due to their steep topography, the high spatial and temporal variability of precipitation (e.g. El Niño or La Niña events), and the heterogeneity of the lithological composition and diversity of soils. A literature review by Molina *et al.* (2021), including over 137 case studies in 6 countries of the subregion (Argentina, the Plurinational State of Bolivia, Chile, Colombia Ecuador, Peru, and the Bolivarian Republic of Venezuela) showed that more than 84 percent reported erosion rates above 70 t/km^2 annually and more than 65 percent reported sediment production above 250 t/km^2 annually. The acceleration of erosion above natural rates is due to human activities such as deforestation, overgrazing, trail construction and the abandonment of agricultural land. The analysis of 118 case studies (Vanacker *et al.*, 2022) showed that the conservation of natural vegetation in the Andes (moorlands, *punas* [high, treeless plateaus] and forests) was effective in controlling erosion.

A modelling study for estimating annual average soil losses in continental South America using RUSLE, estimated the average soil loss for South America overall of 3.8 t/ha annually, whereas Chile had average annual rates of >20 t/ha and Colombia had average annual rates of 8 t/ha (Riquetti *et al.*, 2022). Regarding sediment transport rate, the Guyanas had the lowest values (<1 t/ha annually), while the highest soil losses were in the Colombian Andes Cordillera and the central south region of Chile. (Riquetti *et al.*, 2022, 2023a).

Detailed information on the proportion of soil area in Colombia degraded by erosion was reported by IDEAM and UDCA (2015), with information by hydrographic area for the continental and the insular territories of the country, according to Colombia's Regional Environmental Authority (REA) and department subdivisions. About 40 percent of those soils (45 million ha) were affected by some degree of erosion, varying from severe to moderate. The main factors were deforestation, poor soil management and different agricultural activities. Livestock farming had the most degraded areas by erosion, as livestock farms occupy about 77 percent of the territory. In a more recent work by González Betancourt *et al.* (2021), about 21 percent of the country's soils presented light erosion, 17 percent moderate, 3 percent severe, and 59 percent did not present erosion. It is necessary to have a more comprehensive and detailed assessment of the area affected by erosion to ensure that sustainable management practices are adopted to decrease this threat.

In Patagonia, different productive systems have been established, such as mining (coal, gas and oil), forestry (mostly in the northern area of Patagonia) and agriculture and livestock production, with livestock production being of great relevance for the entire Chilean-Argentinean extension of Patagonia. Each system has contributed to changes in the Patagonian soil, reflected in the change in plant biodiversity (Radic-Schilling *et al.*, 2021) and the increase in bare soil, consequently increasing the levels of anthropogenic erosion. According to CIREN (2010a, 2010b), 576 000 ha in the Aysén region and 746 000 ha in Magallanes had very severe erosion rates, with approximately 253 000 km^2 of the Andean and Patagonia regions of Argentina having high rates of water erosion (>10 t/ha annually), mainly in the provinces of Neuquén and Santa Cruz (Gaitán *et al.*, 2017). In

one study site in the ecozone, annual erosion rates were calculated using ^{137}Cs redistribution over approximately 50 years, averaging 22 t/ha in grazed lands and 33 t/ha in plantation and rangeland slopes (La Manna *et al*., 2019).

Wind erosion

Wind erosion is a serious hazard in arid and semi-arid land in Argentina, the Plurinational State of Bolivia, Chile and Peru. These areas are also threatened by desertification, which is intensified by droughts especially in the desert and cold steppes of the Andean highlands and in the Atacama Desert. The overgrazing of sheep and goats is another factor. A study in the Coquimbo region at Las Cardas Range Station (CEALC) (Patón, 2021), showed that unappropriated goat grazing management was one of the main factors of desertification in northern Chile.

The perennial drought and salt-tolerant shrub *Atriplex nummularia* (a type of saltbush) thrives in arid and semi-arid areas, and is used for various purposes, such as fodder for livestock. The plants protect the soil as a green belt and the adequate handling of animals in this area is extremely relevant to protecting soils against wind erosion and fighting desertification.

In Patagonia, soils developed from volcanic ash are highly erodible when they remain bare. The erosion processes involve both the loss of light sand particles, and the removal of microaggregates rich in organic matter and silt fractions, favouring the loss of soil C (La Manna *et al*., 2021). Wind-eroded soils occupy 65 473 km^2 of the Andean and Patagonian region of Argentina, mainly in the arid parts of Patagonia, Cuyo (the central western region) and in the northwestern region (Casas, Godagnone and de la Fuente, 2022). There is evidence showing a decrease in wind speed and hence erosivity from 1990 to 2020 (Merino and Gassmann, 2021), and no significant changes from 2001 to 2010, according to globally-modelled data (Yang *et al*., 2022). The most important contributors to wind erosion are the overgrazing of grassland, forest wildfires in Patagonia, the use of conventional tillage in semi-arid pampa and land-use change in northwestern Argentina.

10.7.2 Soil carbon loss in the Andean countries and Patagonia subregion

Mineral soils

The soils in Colombia have a low C storage capacity, with 72 percent of the country's soils having less than 1.5 percent of C in the first 30 cm of soil depth. The majority of the soils with higher C are in the Amazon biome (IGAC, 2015).

According to data provided by the Forest and Carbon Monitoring System of the Institute of Hydrology, Meteorology and Environmental Studies (IDEAM), in Colombia, 158 893 ha was deforested in 2019, 171 685 ha was deforested in 2020, 174 103 ha was deforested in 2021, and 123 517 ha was deforested in 2022 (IDEAM *et al*., 2021; MADS, 2022). The main causes of deforestation are from slash-and-burn agriculture, combined with poor extensive livestock practices, the production of illicit crops, illicit mining and illegal logging, and the expansion of the agricultural frontier into non-permitted areas (MESD, 2022).

Colombia stores a total of 6.2 billion tonnes of SOC across the five natural regions, with the Andean natural region being the highest (83.8 t SOC/ha), and the Orinoquia natural region being the lowest (35.3 t SOC/ha). Detailed data is available in the work carried out by FAO, the World Soil Alliance (AMS) and the Agustín Codazzi Geographic Institute (IGAC) (Bolívar *et al*., 2021).

The SOC sequestration potential of agricultural soils in Colombia was estimated by Araujo *et al*. (2021) for a 20-year period (2020–2040), under four scenarios:

1. business-as-usual (BAU);
2. sustainable soil management measures with organic matter additions of 5 percent (SSM1);
3. 10 percent organic matter (SSM2); and
4. 20 percent organic matter (SSM3).

The average SOC sequestration rate varied from −0.022 tonnes of C/ha annually (BAU) to 0.225 tonnes of C/ha annually (SSM3). Regarding the relative SOC sequestration rate for each scenario, it varied from 0.062 tonnes of C/ha annually (SSM1) to 0.247 tonnes of C/ha annually (SSM3) (Araujo *et al.*, 2021). These estimates indicate that in any SSM scenario the increase in SOC was positive.

In the Bolivarian Republic of Venezuela, the total and average relative annual sequestration rates of SOC amount to 4 755 million tonnes C and 0.195 million tonnes C, respectively, placing the country in thirty-first place out of the 119 countries that participated in the training sessions and construction of the national maps of SOC sequestration potential (FAO, 2022a). In the coastal area of Peru, under the tropical and subtropical dry forest (commonly found in mountainous landscapes), medium fertility soils such as Luvisols and Cambisols favour the growing of such crops as maize, bean, potato, sugar cane, fruits and coffee. The main threat to this biome is deforestation, causing the loss of SOC in bare soils and an increase in soil temperature.

The soils of Chilean-Argentinian Patagonia are associated with the activity of glaciation, mainly in the steppe zone (Casanova *et al.*, 2013; Pisano, 1977) and recent volcanic activity in the Andes zone (INIA, 2014). Both situations have resulted in the formation of Mollisols, Chernozems and Kastanozems in the steppe zone (Borrelli and Oliva, 2001) and Andosols and Espodosols in the mountain ranges zone (INIA, 2014; Panigatti, 2010). In addition, glacial activity has generated valleys with soils including Histosols, Luvisols and Gleysols (Filipová, Hédl and Covacevich, 2010; Panigatti, 2010; Valle, Ivelic-Sáez and Radic, 2021). Each of these soils has a specific and relevant function within the Patagonian ecosystem, of which C sequestration and freshwater storage are most relevant. The expansion of productive systems, such as agriculture and livestock production, have caused changes in the soils, with decreasing C content (Peri *et al.*, 2018; Pfeiffer *et al.*, 2020). However, land-use change towards afforestation in the Andean countries and Patagonia subregion of Argentina did not show significant changes in SOC (Peri *et al.*, 2022), and is probably associated with site-dependent responses, conditioned by multiple factors (Buduba, Defossé and Irisarri, 2017; La Manna *et al.*, 2021; Guo *et al.*, 2022).

Organic soils including wetland soils

The Chilean Soil Organic Carbon (CHLSOC) database, a recent inventory and a multi-institutional collaborative effort, has helped to build the largest SOC database to date in Chile, comprising of 13 612 data points collected from numerous sources (Pfeiffer *et al.*, 2020). The SOC values vary from near 0 percent to 83.3 percent, which reflects the variety of Chilean ecosystems. Peat bogs comprise an extensive area located on the Pacific coast of Chilean Patagonia (the zone from 44° to 55° south) and have a high ecological and scientific value, as well as providing ecosystem services at the national level (Pfeiffer *et al.*, 2020).

The CHLSOC database (Pfeiffer *et al.*, 2020) estimates that Chilean Patagonian landscapes store 570 tonnes of C/ha, mainly in peatlands (Perez-Quezada *et al.*, 2023) and black soils (Ivelic-Sáez *et al.*, 2022). In Argentina it has been determined that the C stock of the Patagonian steppe and forests varies between 1.3 and 515 tonnes C/ha (Fröjd *et al.*, 2022; Gaitán *et al.*, 2023; Peri *et al.*, 2018). In Colombia, the *páramos*, located in elevations higher than 3 000 MASL, are an emblematic ecosystem, with around 50 percent of all the *páramos* in the world, supplying water to approximately

70 percent of the population (Sarmiento *et al*., 2013). In Argentina, Chile and Patagonia, the great expanses of wetland watersheds such as peatlands, *vegas*, *mallines* and lagoons are critical in terms of the balance of SOC and soil biodiversity (Iturraspe, 2010; Ivelic-Sáez *et al*., 2021; Urciuolo and Iturraspe, 2022). In both environments, the reduction of wetland soils have large implications for the degradation of the ecosystems.

10.7.3 Soil biodiversity loss in the Andean countries and Patagonia subregion

The Andean countries and Patagonia subregion hotspots represent 20 percent of planetary biodiversity and endemic species, many in communities of Indigenous Peoples, natural grasslands and moorlands. Among the main drivers of threats to soil biodiversity are forest fires, which lead to changes of physical, chemical and biological soil properties. In Chile, "megafires", particularly in the summer of 2017 and 2023, have had a highly negative impact. In the 1984–2023 period, fires affected an area of 3.1 million ha in Chile (with 1.5 million ha being affected by fires between 2015 and 2022) (Francke-Campaña, 2019, 2021a, 2021b).

Unique species are found in the montane grasslands and shrublands in the high-altitude mountains. This habitat type includes the *puna* and the *páramos* of the northern Andes, which are the most extensive examples of ecosystems in the subregion (Orgiazzi *et al*., 2016). In Peru, this habitat is found above 3 000 MASL on shallow Regosols and Leptosols, usually rich in organic matter, and Solonchaks in the driest areas. In some parts of the country, soil biodiversity is being negatively affected by excessive exposure to fertilizers and agrochemicals through the intensive farming of traditional crops such as potato, carrot (*Daucus carota*) and quinoa (*Chenopodium quinoa*). Soil biodiversity is threatened by the loss of plant biodiversity, the introduction of invasive plants and threats such as tillage, erosion and soil pollution. Likewise, some recent activities such as lithium (Li) mining raise serious concerns about the future of the *puna* wetlands where organic soils have developed (Izquierdo *et al*., 2018, Chiappero, Vaieretti and Izquierdo, 2021) due to the intensive water use and protracted duration of the mining (Vera *et al*., 2023).

10.7.4 Nutrient mismanagement in the Andean countries and Patagonia subregion

No comprehensive literature review is available that covers the countries in the Andean countries and Patagonia subregion. Data presented in Table 10.5 indicate that Chile, Colombia and Ecuador have N input levels indicative of N surplus. Chile and Ecuador also have P input levels at potentially unsustainable levels, coupled with large P accumulation in their soils.

Chile has established soil protection laws and programmes related to nutrient mismanagement. The Soil Framework Law project, led by the Chilean Soil Science Society, the Geological Society of Chile, and the NGO, Sustainable Soil (and sponsored by the Senate Commission of Agriculture), was approved in the Senate in January 2022 (Salazar *et al*., 2022).

10.7.5 Salt accumulation in the Andean countries and Patagonia subregion

At the country level, Brazil, Chile, Colombia and Peru were estimated to have the highest rates of annual increase in the total area of soils with a soil electrical conductivity (ECe) ≥4 dS/m. Additionally, Francke-Campaña (2021c) reported that by the end of the twenty-first century, the drylands of South America would become salinization hotspots.

According to the results of the zoning map of soil degradation due to salinization in Colombia (IDEAM, CAR and UDCA, 2017; IDEAM, 2019), the continental and insular areas have 12.3 percent of land with some degree of salinization. Of this salt-affected land, 7.9 percent is from the continental area, with 0.2 percent having a severe level of salinization, 1.3 percent having a very severe level of salinization and 5.1 percent with a moderate level of salinization. This process occurs naturally but is increased due to anthropogenic activities.

In the coastal area of Peru there is a desert and xeric shrubland biome where Calcisols, Gypsisols, Arenosols, Regosols and Solonchaks predominate. The area is characterized by its low precipitation, high evaporation, windy conditions and warm tropical climate, (FAO, 2015). Soils in this region are often affected by natural salinization, but land-use changes and overgrazing have also caused secondary salinization, potentially affecting its agricultural productivity.

A great threat to Patagonian soils is the constant danger of salt accumulation due to the combination of intrinsic characteristics of the country's geomorphology, the negative water balances (with a greater evapotranspiration than rainfall) and the increase in bare soils as a result of production systems. In Argentina, natural salt accumulation is common in the western arid and semi-arid regions. In these areas, those processes are triggered in non-irrigated areas by land-use changes when perennial, deep-rooted native vegetation is increasingly being replaced by annual crops, or by excessive irrigation and deficient drainage in irrigated lands (Guida-Johnson, Abraham and Cony, 2017).

10.7.6 Soil pollution in the Andean countries and Patagonia subregion

Peru is one of the 14 Latin American countries that has As water contamination, attributed to smelting activities in mining areas (FAO, 2015). Other anthropogenic sources of soil contamination in Peru, include the intensive use of fertilizers and pesticides, industrial activity, poor management of solid and liquid waste, and inadequate land occupation (Diaz, 2016). The neighbouring countries of the Plurinational State of Bolivia and Chile also report As contamination of soils.

Non-essential metals derived from soil and other sources are present in considerable quantities in various agricultural products in Ecuador (Romero-Estévez *et al.,* 2023). The concentrations of As, Cr, Cu, and Ni in farmland soils – influenced by mining activity – exceed the Ecuadorian quality guidelines for agricultural soils, and concentrations of As, Cd and Zn in local crops are higher than the maximum permissible levels set by FAO (Romero-Crespo *et al.,* 2023).

Soil pollution in the Argentinian Andes and Patagonia regions is also mainly related to mining activities and petroleum extraction (Fernández *et al*., 2022). Soils affected by mining waste usually have elevated levels of heavy metals or other toxic elements that exceed the threshold level of national environmental regulations (Kirschbaum *et al*., 2012; Villegas and Turel, 2022; Do Campo, Valenzuela and Ferro, 2020; Heredia, 2021; Bermudez *et al*., 2010; Marinho *et al*., 2017; Vicente and Valenzuela, 2019). In the Patagonian region, occasional spills of oil and its by-products have also caused contamination with PAHs, modifying the physical, chemical and biological properties of soils (Barquín, Ríos and Nudelman, 2011; Madueño *et al*., 2011).

10.7.7. Other threats in the Andean countries and Patagonia subregion

Soil degradation and desertification are a multicausal and combined process that affect much of the land in the Andean countries and Patagonia subregion. The soil map depicting arid, very dry, dry and dry subhumid lands of Colombia with soils susceptible to degradation due to desertification indicate that they occupy 26.7 percent of the territory (IDEAM, 2019). As the deserts of Peru and other arid regions have the lowest SOC reserves, their conservation is essential due to the fragility of their ecosystems.



10.8 Trends in sustainable soil management for the Latin America and Caribbean region

The information presented in this chapter enables the assessment of the state and trend of threats to soil functions through a management-based approach. Instead of directly measuring or modelling the physical severity of each soil threat, as was done in the 2015 report (FAO and ITPS, 2015), the current report assesses the level of adoption of sustainable soil management (SSM) practices known to mitigate each soil threat.

Overall, the state of SSM to address the threats to soil function is assessed as being in poor or fair condition across the five subregions of the LAC subregion (Figure 10.17 and Table 10.5). The trend is dominated by deteriorating tendencies or variable conditions in most subregions. Generally, the Cerrado and the Andean countries and Patagonia subregions have marginally better-assessed conditions in trends than the other regions.

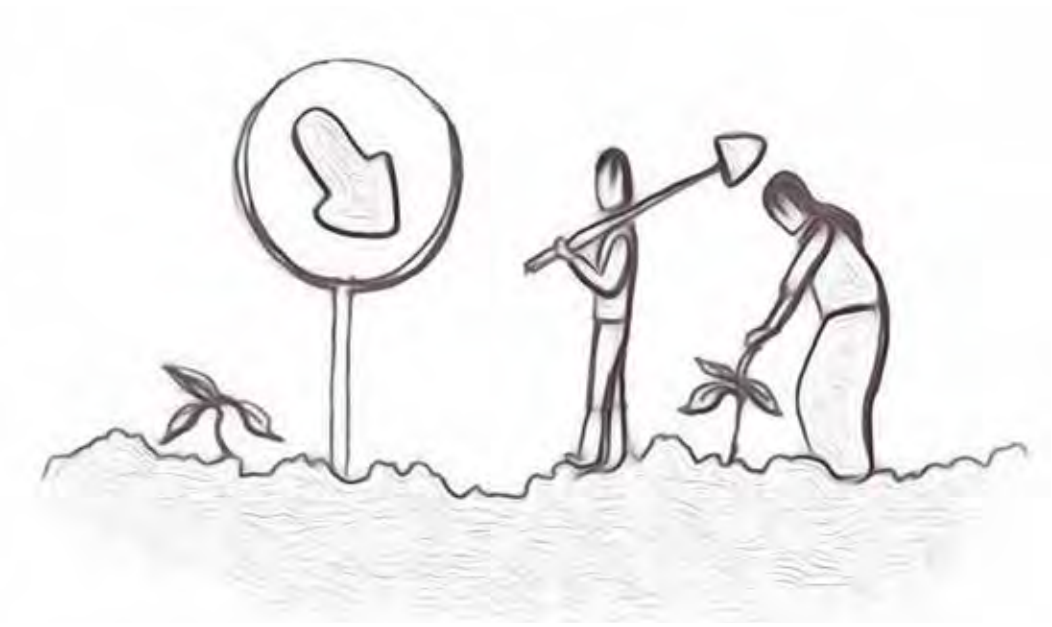
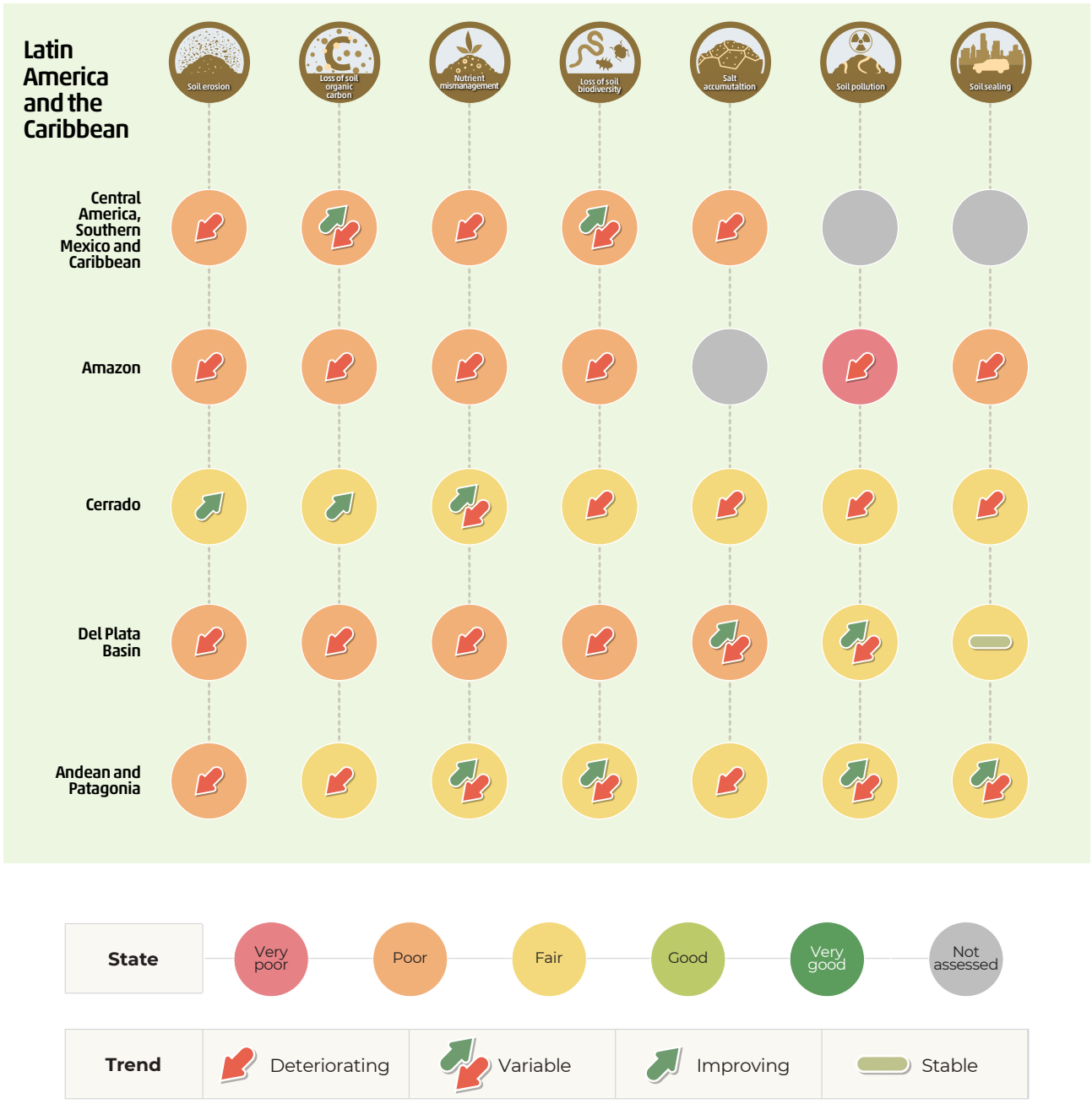


Figure 10.17. Assessment of state and trend of threats to soil functions in Latin America and the Caribbean through a management-based approach



Note: For further details, see Table 10.5.

Table 10.5. Summary of threats to soil functions in Latin America and the Caribbean and the confidence in the assessment of their state and trend

Subregions	Threat to soil functions	Confidence	
		State	Trend
Central America, southern Mexico and the Caribbean	**Soil erosion**: Soil erosion is driven by bare soil conditions, water erosion, and wind erosion in dry climates and sandy soils. Coastal erosion, exacerbated by sea level rise, is also a concern.	Low	Medium
	Loss of SOC: Soil organic carbon stocks are variable with soil types, as well as in croplands and open shrublands. Losses of SOC reflect the complex interplay between human activities and natural processes.	Low	Medium
	Loss of soil biodiversity: Data in tropical and subtropical regions is limited. Modelling in coastal areas indicates high values of respiration, especially across the Yucatan Peninsula and coastal Caribbean territories and savannahhs and evergreen broadleaf forests. experiencing negative trends.	Low	Medium
	Nutrient mismanagement: Spatial variability in soil fertility indicators across the subregion, with notable trends in fertilizer consumption.	Low	Medium
	Salt accumulation: Coastal areas, particularly in the Caribbean, face elevated soil salinity due to seawater intrusion, exacerbated by rising sea levels. Human activities such as improper irrigation and deforestation further contribute to soil salinity.	Low	Medium
	Soil moisture: Soil moisture has declined across the subregion since the early 1990s. Increases are noted in flat terrain prone to flooding. Extreme events like hurricanes exacerbate these trends.	Low	Medium
	Soil pollution	Not assessed	Not assessed
	Urbanization and soil sealing	Not assessed	Not assessed
Amazon Basin	**Loss of soil biodiversity**: Pasture and agriculture expansion continue at the expense of native vegetation, posing threats to soil biodiversity. Deforestation and fires lead to soil degradation, including decreased SOC, moisture and increased temperatures. Over-exploited areas face severe biodiversity loss.	High	High
	Loss of SOC: Ongoing deforestation and conversion to pasture/agriculture accelerate SOC loss. Overexploited areas suffer severe SOC depletion.	High	High
	Soil erosion: Deforestation exposes soils to erosion and mass movements are common in mountainous landscapes (gravity and rainfall) and lowlands (river erosion). The high erosive potential of rainfall may increase with climate change and the wind erosion become widespread.	High	Medium
	Nutrient mismanagement: Conversion to agriculture and pasture speeds up nutrient leaching and SOC losses, with depletion of nutrients and more acidification. The agricultural expansion, in general, lacks in soil analysis and recommendation of fertilizers tailored for this environment.	Medium	Medium
	Soil pollution: Mining operations in the Amazon (including numerous illegal operations) are rapidly expanding, fuelled by rising gold prices, even within protected areas. This leads to soil and water contamination with Hg.	Medium	Medium
	Urbanization and soil sealing: Lands once covered by natural forest or used for agriculture and other purposes are occupied by urbanization. Soil sealing, mainly in the large cities near the deltas, affects the natural cycles of the fluvial systems in the Amazon.	High	High

Subregions	Threat to soil functions	Confidence	
		State	Trend
Cerrado	**Soil erosion**: Deforestation, overgrazing and low adoption of agriculture conservation practices in some areas are the main factors. Adoption of conservation practices (no-till, agroforestry, integration of crops, livestock, and forestry), play a significant role in mitigating erosive processes, with improvements in some areas of the Cerrado.	High	High
	Loss of SOC: Carbon stocks are experiencing stabilization in some areas with agriculture. The data on lands with conversion of natural vegetation to agriculture in the last decade indicates some progress in SSM.	High	High
	Loss of soil biodiversity: Biodiversity is significantly impacted by agricultural expansion. Over-exploitation, deforestation and fires, mainly in the pasture lands and expansion areas, affect negatively soil biodiversity.	High	High
	Nutrient mismanagement: Nutrient mismanagement occurs in some croplands due to excessive usage of fertilizers and a cumulative nutrient overload over time. However, there are improvements in nutrient management, through the use of crop varieties better suited to Cerrado soil and climate.	Medium	Medium
	Soil pollution: Soil pollution principally comes from agrochemicals used in the monocropping farms. One of the significant challenges is the pollution of aquifers.	Low	Medium
	Salt accumulation: Higher risks exist in the drier areas, and lands irrigated with central pivot without proper management of water.	Low	Medium
	Urbanization and soil sealing: Urbanization replaces land once covered by natural forest or used for agriculture and other purposes, as populations increase.	High	High
Del Plata Basin	**Soil erosion**: The erosion of soil due to water occurs in steep humid landscapes, while wind erosion occurs in semi-arid conditions and sandy soils. The trend is deteriorating as a function of local factors, mainly due to low soil cover and lack of structures to control and manage runoff and proper practices to manage wind incidence.	High	High
	Nutrient mismanagement: Deficits persist for macronutrients (N, P and S) and new deficits are observed in bases (K, Ca and Mg) and micronutrients (Zn). Insufficient fertilization is generalized, and some evidence on soil pH decrease trend.	High	High
	Loss of SOC: There are losses or no improvement of SOC contents, mainly due to monocropping, overly simple rotations and unsustainable grazing systems.	High	High
	Loss of soil biodiversity: Monocropping or extremely simple rotations lead to an increasing decline of soil biodiversity.	Medium	Medium
	Salt accumulation: This primarily occurs in large plains with high and saline groundwater and sodic soils. Upward soluble salt rises are promoted by land clearing and overgrazing. In some areas, the use of improper water (high sodium bicarbonate content) for uncontrolled irrigation is leading to sodification.	High	High
	Soil pollution: Soil pollution principally occurs as non-point pollution by residues or by-products of herbicides.	Medium	Medium
	Urbanization and soil sealing: They are increasing around the large cities.	Medium	Medium

Subregions	Threat to soil functions	Confidence	
		State	Trend
Andean countries and Patagonia	**Soil erosion**: Erosion by water occurs in intensive agricultural and fruit lands. There are fires in forest soils in hilly topography with a reactivation of erosion processes in most of the Andean countries. Wind erosion has increased in dry climates, in western Argentina, other Andean countries and Patagonia, intensified by livestock overgrazing, deforestation and increased cropping	High	High
	Loss of SOC: Losses occur due to factors such as megafires and increase in soil temperature due to climate change, lower crop productivity, erosion and non-adoption of conservation practices, and the expansion of agriculture and livestock production in arid and semi-arid environments. Soil organic carbon has also declined in organic soils and wetlands.	High	High
	Salt accumulation: Major factors are the lower availability and quality of irrigation water, lower salt leaching and the concentration of surface salts due to capillarity in the central-northern area of Chile, the coastal areas of Colombia and Patagonia.	High	High
	Loss of soil biodiversity: This is principally due to changes in land use and soil cover, natural forest megafires, water use and pollution by mining activities that affect wetlands.	Low	Medium
	Nutrient mismanagement: Excessive use of fertilizers and manure, and cumulative nutrient overload has occurred through time. This is coupled with losses of surface SOC in the lands with high intensity usage and fires. In some places, there is insufficient fertilization to overcome nutrient export and soil fertility is declining. Soil acidification is limited to some soils, mainly those originated from volcanic materials.	Medium	High
	Soil pollution: This is due to mining, oil extraction, industrial processes, livestock production and agricultural activities, and show increasing trends towards the accumulation of inorganic and organic contaminants in soil and water.	Low	Medium
	Urbanization and soil sealing: They occur primarily in some valleys and flood plains near the major cities and capitals, advancing on agricultural fertile soils and on peri-urban forest soils.	Medium	Medium

Note: **The threats are presented in the order of most severe threat to least severe for each subregion.**

Source: Authors' own elaboration.

The LAC region presents a high diversity of landscapes, biomes, agriculture and populations. As a result, the condition and trend in soil management of threats to SSM varies intensively**.**

Soil erosion is the main threat in all subregions, except for the Amazon Basin, where it is the third most important threat after soil biodiversity and SOC change. In contrast, urbanization and soil sealing are placed last in all subregions or not even mentioned as a threat. This is explained by the fact that the subregions of LAC encompass large territories and thus are less densely populated compared to other world regions. While soil pollution is rated in various positions, some pollution threats are particularly serious in terms of human health, such as those produced from mining and oil extraction operations, especially in the Amazon Basin and some Andean countries.

Soil organic carbon change is the second or third most important threat in the LAC subregions. The main causes of this increasing threat are fires, the non-adoption of conservation practices, the expansion of agriculture (monocropping) and livestock production (some in unsuitable lands), and the substitution of the natural vegetation (mainly in wetlands, which are natural hotspots for SOC stock). As pointed out in Section 10.3.2 concerning Central America, southern Mexico and the Caribbean subregion, SOC losses reflect the complex interplay between human activities and natural processes.

Although many soils in the LAC region have a naturally low nutrient content (due to the dominant tropical climate), some subregions report nutrient mismanagement, due to the excessive use of fertilizers. In other areas, there is an inadequate addition of nutrients and intense agriculture in more

fertile lands, such as the black soils of the Del Plata Basin subregion. Nutrient mismanagement is also related to lack of proper recommendations tailored for specific biomes such as the Amazon Basin subregion and lesser usage of soil analyses. The Central America, southern Mexico and the Caribbean subregion shows a variable condition, with notable trends in fertilizer consumption.

Although soil biodiversity is an important threat in all subregions, correlated with other factors and directly affected by deforestation and fires, the quality of the information on this threat is very limited and mostly indirectly evaluated, with trends showing contrasting patterns.

Additionally, salt accumulation, fuelled by improper irrigation and drainage practices, human activities and climate change, pose risks to soil health and crop yields, necessitating urgent mitigation strategies. Soil moisture decline, influenced by land-use changes and extreme weather events, adds to the complexities of SSM. These threats are particularly relevant in southern Mexico, the Caribbean and Patagonia, and in all coastal areas.

Examples of promising trends are the increase of no-till systems (with cover crops and residue conservation), integrated crop–livestock–forestry systems, improved management of pastures and low-intensity grazing, the adoption of CSA, development of more efficient fertilizers and integration of various bioinputs, wide adoption of BNF in soybean production, and the valuing of traditional knowledge, such as biochar.

Impediments to the adoption of SSM across all the LAC subregions are the lack of data quality, quantity and accessibility, along with insufficient monitoring of the implementation of SSM protocols and practices. Only by promoting SSM, targeting interventions and improving legislation, will it be possible to mitigate threats to soil health, preserve ecosystem resilience and ensure long-term agricultural sustainability.

A strategic roadmap for LAC toward the governance of soil and water can only be achieved by integrated and unified approaches. Comprehensive national and regional soil data are essential, together with contributions of experts from the local research institutions and universities. This should be part of national and regional development policies and seen as a priority in government investments.



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Chapter 11 | Regional assessment for the Near East and North Africa (NENA)

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11.1 Regional overview

11.1.1 Characteristics of the region

The Near East and North Africa (NENA) region is one of the most land and water-constrained regions of the world and is predicted to become hotter and drier in the future due to climate change. The region has four challenges related to the dominant climate: aridity, recurrent drought, desertification and salinization, with salinization being partly human-induced (Choukr-Allah *et al.*, 2023). Over 43 percent of the NENA region is desert, while dry and semi-dry areas constitute a further 40 percent of the area. An annual precipitation of ≥300 mm is estimated to occur over about 17 percent of the NENA area and is the focus of agricultural activity and food production (AOAD, 2007).

The countries of the Near East are Bahrain, the Islamic Republic of Iran, Iraq, Jordan, Kuwait, Lebanon, Oman, Palestine, Qatar, Saudi Arabia, the Syrian Arab Republic, the United Arab Emirates and Yemen. The countries of North Africa are Algeria, Egypt, Libya, Mauritania, Morocco, the Sudan and Tunisia (Figure 11.1).

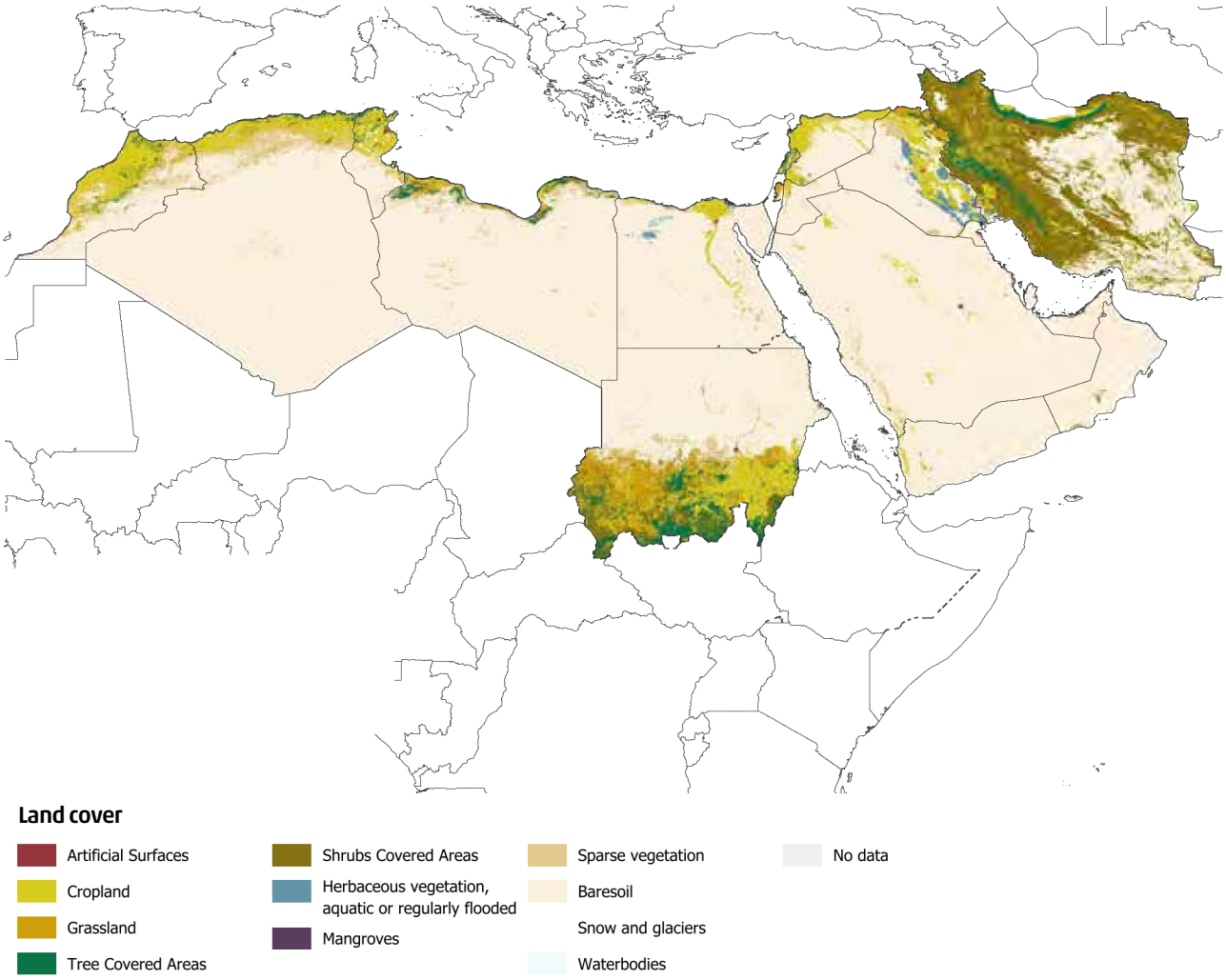
Figure 11.1. Location and extent of the Near East and North Africa region



◊ Source: Author's own elaboration based on FAO Global Soil Partnership Regional Partnerships. https://www.fao.org/global-soil-partnership/regional-partnerships/en/

About one-third of the NENA region is considered to be agricultural land (cropland and pastures), while only 5 percent is arable (OECD and FAO, 2018) (Figure 11.2). The rest of the land is either urban or desert. Due to the dry climate, about 40 percent of the cropped area in the region requires irrigation. Only 4 percent of land in the region has soil of high or good suitability for rainfed cereal cultivation, with about 40 percent of marginal land and about 55 percent unsuitable for cereal production (OECD and FAO, 2018).

Figure 11.2. Land cover in the Near East and North Africa region



Refer to the disclaimer on page ii for the names and boundaries used in this map. Final boundary between the Republic of Sudan and the Republic of South Sudan has not yet been determined. Final status of the Abyei area is not yet determined.

Source: **FAO**. 2024. GAEZ: Land and Water Resources. [Accessed 24 October 2024] https://gaez-services.fao.org/apps/theme-1/. License: CC-BY-4.0.

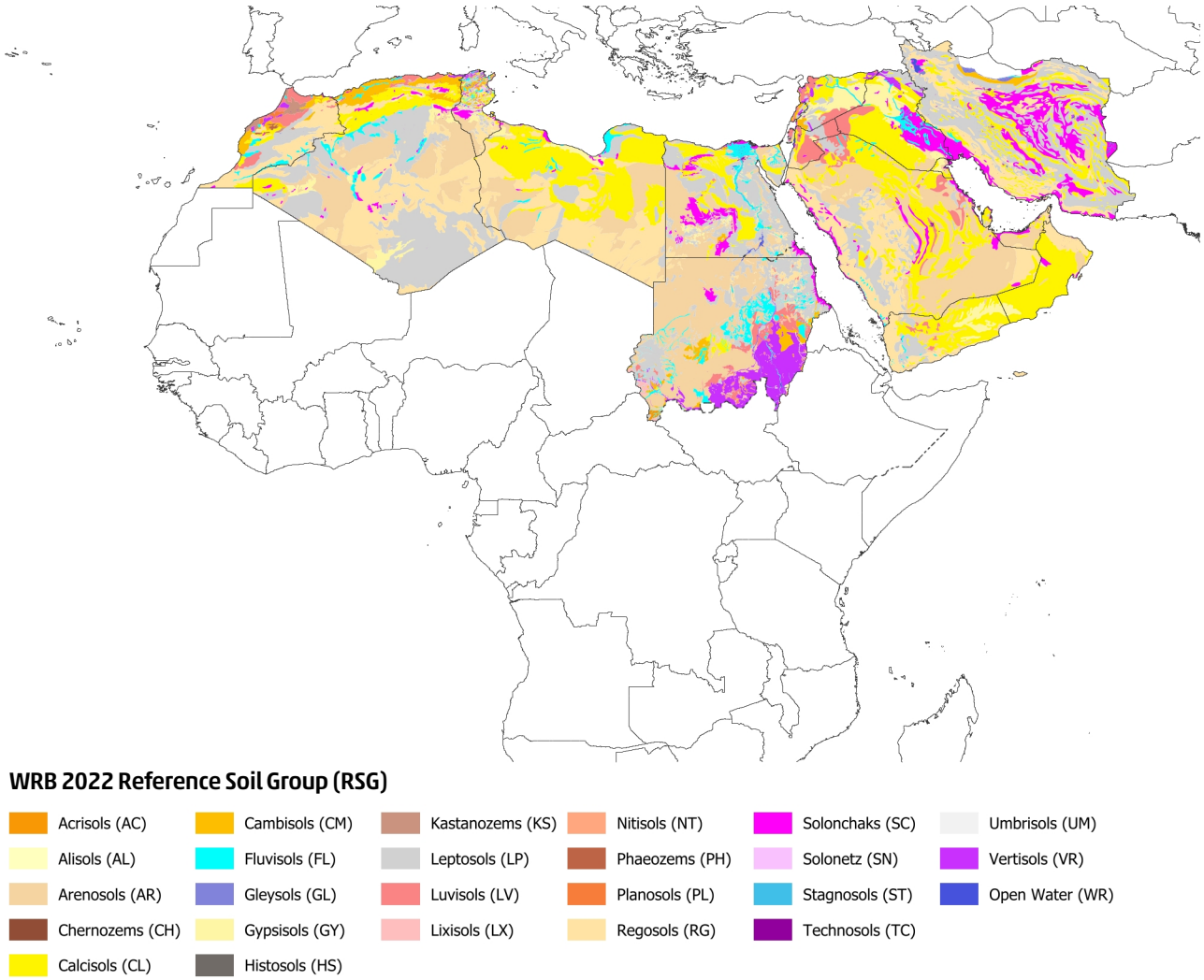
Much of the arable land in the NENA area (whether irrigated or rainfed agricultural land, forage land or forestland) suffers severely from desertification and degradation of various types (OECD and FAO, 2018).

The fertility limitations of soils in the NENA region are the result of low natural soil nutrient content, low SOM content, dry weather, high salinity with limited fresh water for irrigation, and nutrients not being replenished after being removed by crops (OECD and FAO, 2018).

Most soils in the NENA area are generated from calcareous rocks, which have a coarse texture and are composed of unweathered materials with low fertility, and so do not meet crop requirements

for mineral nutrients. The major soil orders in the NENA region include Leptosols, Arenosols and Calcisols (Figure 11.3). Despite the limited inherent fertility of these soils, farmers all over the NENA region continue to farm with minimal, if any, fertilization in most cases. Most soils are low in SOM due to high rates of decomposition in hot weather and limited organic inputs from plants. The lack of SOM contributes to degraded soil physical and fertility characteristics, and consequently, some degraded lands have been removed from the farming system (AOAD, 2007).

Figure 11.3. Soils of the Near East and North Africa region



Refer to the disclaimer on page ii for the names and boundaries used in this map. Final boundary between the Republic of Sudan and the Republic of South Sudan has not yet been determined. Final status of the Abyei area is not yet determined.

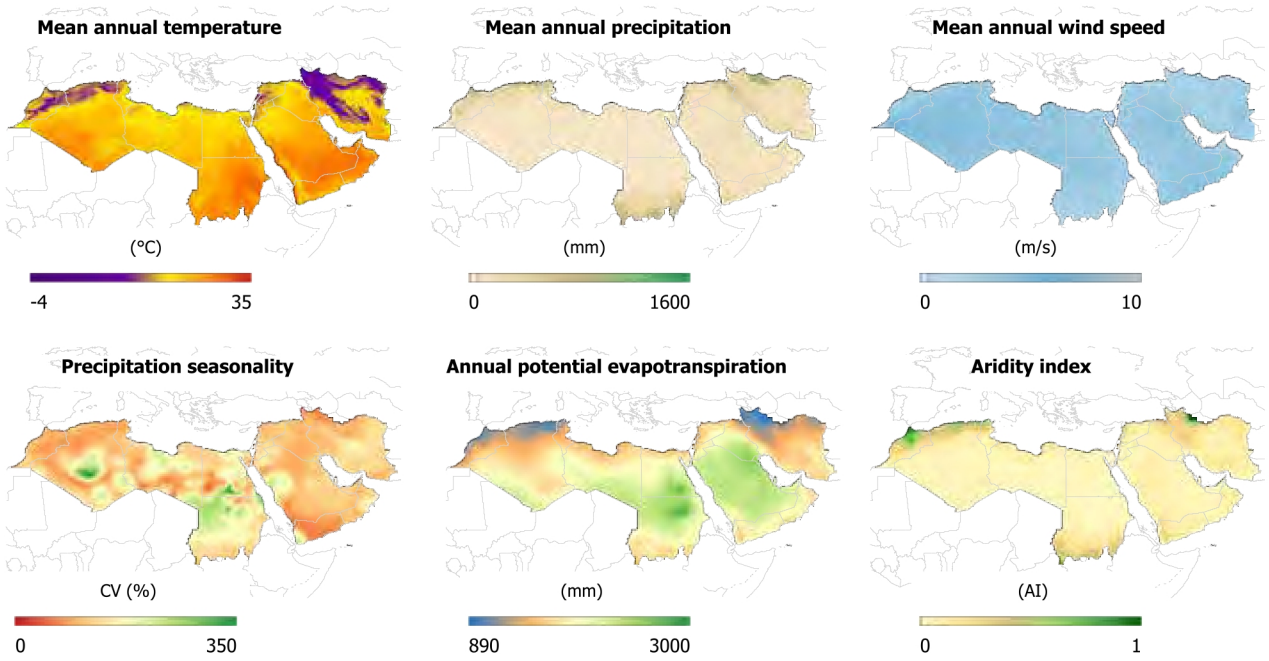
Source: **FAO**. 2024. *GAEZ: Harmonized World Soils Database version 2.0*. [Accessed 24 October 2024] https://gaez.fao.org/pages/hwsd. License: CC-BY-4.0.

The degradation of soil fertility is a major limiting factor for food security in the NENA region that continues to threaten sustainable land management, influencing all aspects of rural societal development. However, the intensification of agricultural production is the key for both food and the financial needs of farming societies.

Agriculture has always been a strategic sector for the socioeconomic development of the NENA countries. The agricultural sector has undergone many agricultural and rural development programmes and structural reforms to enable each country in the region to achieve food security and contribute to economic growth.

There is climatic variation in the NENA region (Figure 11.4). Annual precipitation is generally less than 500 mm, except in some western Levant and Maghreb coastal areas where it ranges from 1 000 to 1 300 mm. Thermal dynamics also vary, with the mean annual temperature in Gulf countries exceeding 30 °C, while coastal North Africa and Levant have milder climates (20–25 °C). The NENA region experiences elevated annual potential evapotranspiration (PET) levels, often surpassing 1 500 mm in many areas and reaching 2 000 mm in Gulf countries (Figure 11.4). Paradoxically, this abundance of PET, combined with the region's characteristic seasonality of rainfall, contributes to aridity. Infrequent but intense rainfall events fail to provide sufficient moisture to offset the substantial evaporation and transpiration rates, resulting in water deficits and extended dry periods.

Figure 11.4. Spatial distribution of mean annual temperature, mean annual precipitation and mean annual wind speed (top row) and precipitation seasonality, annual potential evapotranspiration and aridity index (bottom row) for the 1960–2022 period



Refer to the disclaimer on page ii for the names and boundaries used in this map. Final boundary between the Republic of Sudan and the Republic of South Sudan has not yet been determined. Final status of the Abyei area is not yet determined.

Note: Tmean = mean annual temperature, Precip = mean annual precipitation, WSpd = mean annual wind speed, Precip_Seas = precipitation seasonality), PET = annual potential evapotranspiration, and AI = aridity index.

◊ *Source:* **Harris, I., Osborn, T.J., Jones, P. & Lister, D.** 2020. Version 4 of the CRU TS monthly high-resolution gridded multivariate climate dataset. *Scientific Data*, 7(1): 109. https://doi.org/10.1038/s41597-020-0453-3

11.1.2 Status summary from 2015 Status of the world's soil resources report

The 2015 SWSR report from the NENA region stated that three of the top five threats (soil erosion, contamination and soil sealing from urban expansion) were in very poor condition and showed a deteriorating trend (FAO and ITPS, 2015). The report's NENA region assessment was the most negative for any region. The other two threats in the top five (salinization and sodification and SOC change) were judged to be in poor condition, again with a deteriorating trend.

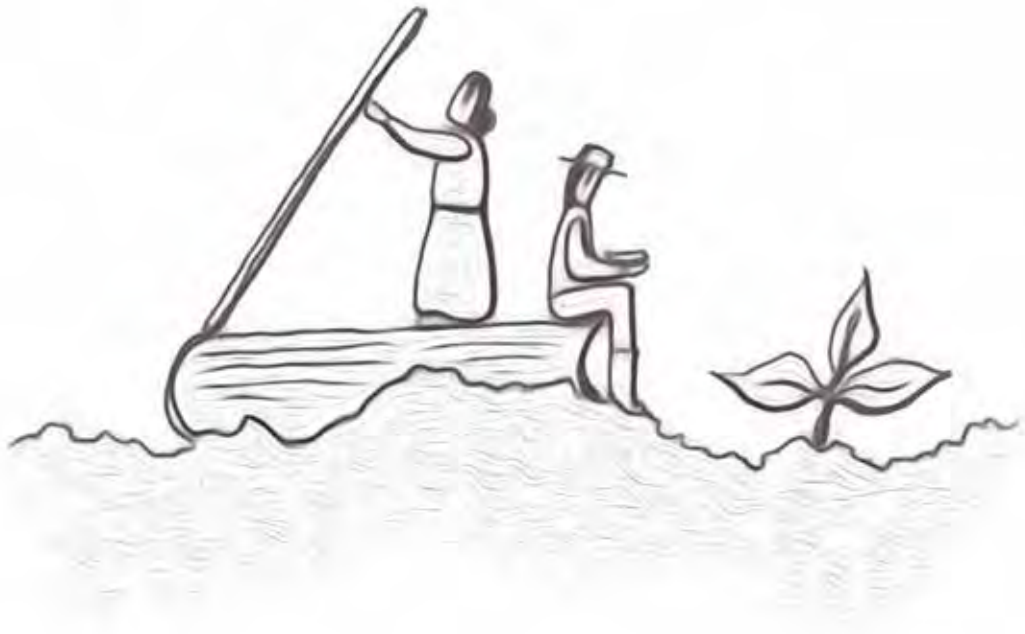
Erosion was judged to be the most severe threat, based on the widespread wind erosion and resulting dust storms that occurred throughout the region. However, attempting to stabilize sand in the source areas was difficult and very expensive. Water erosion was limited in its extent and it was felt that suitable control measures were available to limit its effects.

Salinization and sodification make up the second most severe threat. Salinization was and remains a widespread problem due to the generally high temperatures, inappropriate irrigation techniques and seawater intrusion in the coastal areas of the region. Although suitable control measures were available, socioeconomic conditions limited the implementation of these measures in some countries. Sodification was less common but is locally important in countries such as the Islamic Republic of Iran (Mesgaran *et al.*, 2017).

Although SOC levels were found to be generally low throughout the region, the high temperatures caused very high turnover of any C added to the soils and limited the possibility of significant C additions.

Contamination and soil sealing due to urban expansion were largely attributed to the growth of urban population throughout the regions. Significant soil contamination was also associated with oil production and refining in some areas.

The report judged that the information available for the loss of soil biodiversity and nutrient imbalance was limited and hence there was low confidence in the assessment of these threats.



11.2 Changes in the direct and indirect drivers of soil functions

11.2.1 Indirect drivers: population growth

The most important indirect driver of threats to sustainable soil management (SSM) is population growth. Population growth in the NENA region is predicted to be higher than the global average through to 2035 (UN DESA, 2023). Population growth will be slower from 2021 to 2025 than in the 2015 to 2021 period but still increasing. The population in the NENA region is projected to grow from 521 million in 2021 to 635 million by 2035 and 771 million by 2050 (UN DESA, 2022). The highest rates of population growth by 2025 will be in Iraq, Palestine, the Syrian Arab Republic, the Sudan, and Yemen.

11.2.2 Direct drivers: land conversion

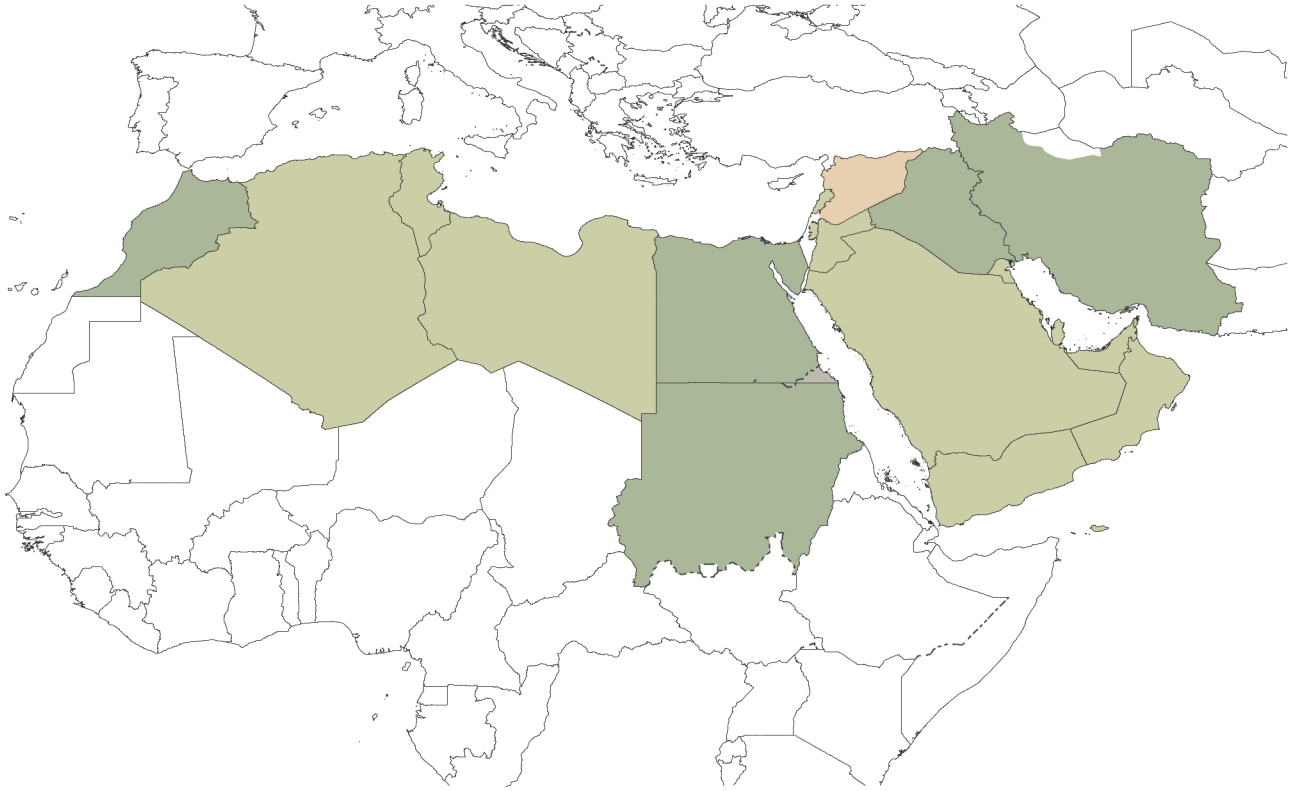
Conversion of land from natural vegetation to agricultural uses is a direct driver of soil change. In most settings, land-use conversions cause the rates of harmful soil threats to increase.

Agricultural and forestland change

Countries in the NENA region have generally experienced relatively small shifts in agricultural land use in the 2015–2022 period. The largest increases in cropland in the NENA region in the 2015–2022 period have occurred in the Sudan (a gain of 0.91 million ha), Egypt (a gain of 0.27 million ha) and Morocco (a gain of 0.24 million ha) (FAO, 2024a) (Figure 11.5). The only country with significant areal loss of cropland between 2015 and 2022 is the Syrian Arab Republic (a loss of 0.29 million ha representing a 5 percent decrease of the 2015 cropland area). Cropland area in Palestine also decreased by 26.2 percent between 2015 and 2022 (FAO, 2024a).



Figure 11.5. Country map of change in cropland area from 2015 to 2022



Source: **FAO**. 2024. *FAOSTAT: Land Use*. [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

Land-use conversion from forest to other land uses has the greatest negative impact on soil functions in the NENA region. The area of forestland is small with only the Islamic Republic of Iran (10.8 million ha), and the Sudan (19.2 million ha) having significant areas of forestland in 2015 (Figure 11.2) (FAO, 2024a). Forestland remained unchanged in the Islamic Republic of Iran between 2015 and 2022 but the Sudan has experienced significant deforestation (a loss of 1.2 million ha representing a 6.2 percent decrease in forest area between 2015 and 2020) (Figure 11.6) (FAO, 2024a).

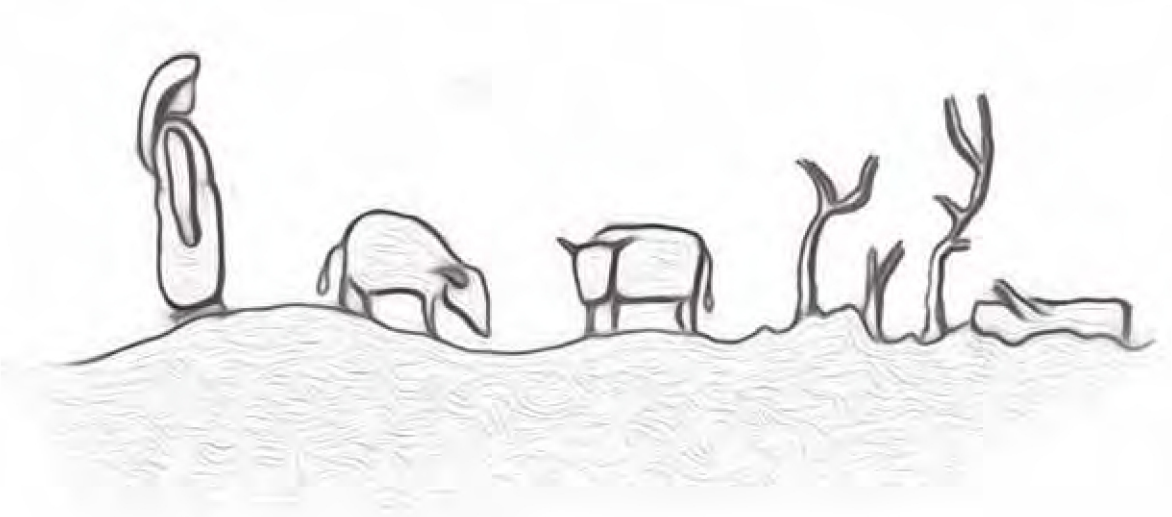


Figure 11.6. Country map of change in forestland area from 2015 to 2022

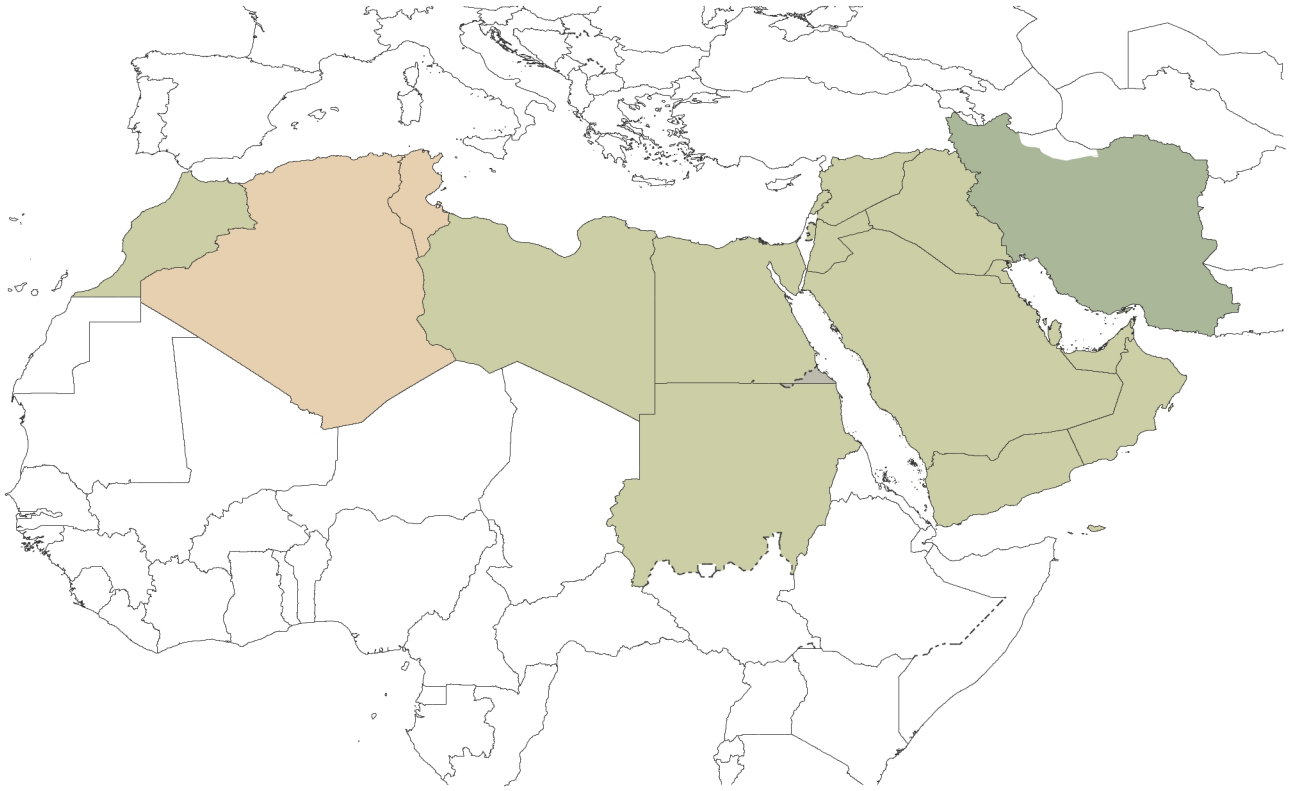


Source: **FAO**. 2024. *FAOSTAT: Land Use*. [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

The reported changes in pastures and permanent meadows in NENA between 2015 to 2022 are relatively minor (Figure 11.7) (FAO, 2024a) with only Tunisia (with a –3.4 percent loss of area) reporting a change greater than ±1 percent.



Figure 11.7. Country map of change in area of pastures and permanent meadows from 2015 to 2022



Source: **FAO**. 2024. *FAOSTAT: Land Use*. [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

Irrigation extent and change

There have been increases in the land equipped for irrigation throughout most countries in the NENA region (FAO, 2024a). Irrigation is important in many countries in the region, with six countries having over 20 percent of agricultural land under irrigation and Egypt having the highest percentage overall (98.3 percent of its agricultural land equipped for irrigation) (FAO, 2024a). Increases in irrigated land between 2015 and 2022 have been highest in Egypt (an increase of 0.23 million ha), Saudi Arabia (0.18 million ha) and Morocco (0.16 million ha) (FAO, 2024a). Palestine has the highest percentage increase (18.8 percent increase in land equipped for irrigation). Increases in irrigation have a major impact on the allocation of water resources in countries and (if improperly installed), can lead to increases in salinization and waterlogging.

Conservation agriculture

Adoption of conservation agriculture in the NENA region has been limited. Kassam, Friedrich and Derpsch (2022), estimated that the area under conservation agriculture in NENA in 2018–2019 was 387 200 ha, or approximately 0.4 percent of the total cropland in the region. The highest adoption by area has been in the Islamic Republic of Iran (estimated at 300 000 ha) followed by the Syrian Arab Republic (30 000 ha), Tunisia (14 000 ha), Morocco (13 000 ha), Iraq (12 000 ha) and the Sudan (10 000 ha). Jordan and Lebanon only have small areas that have adopted conservation agriculture.

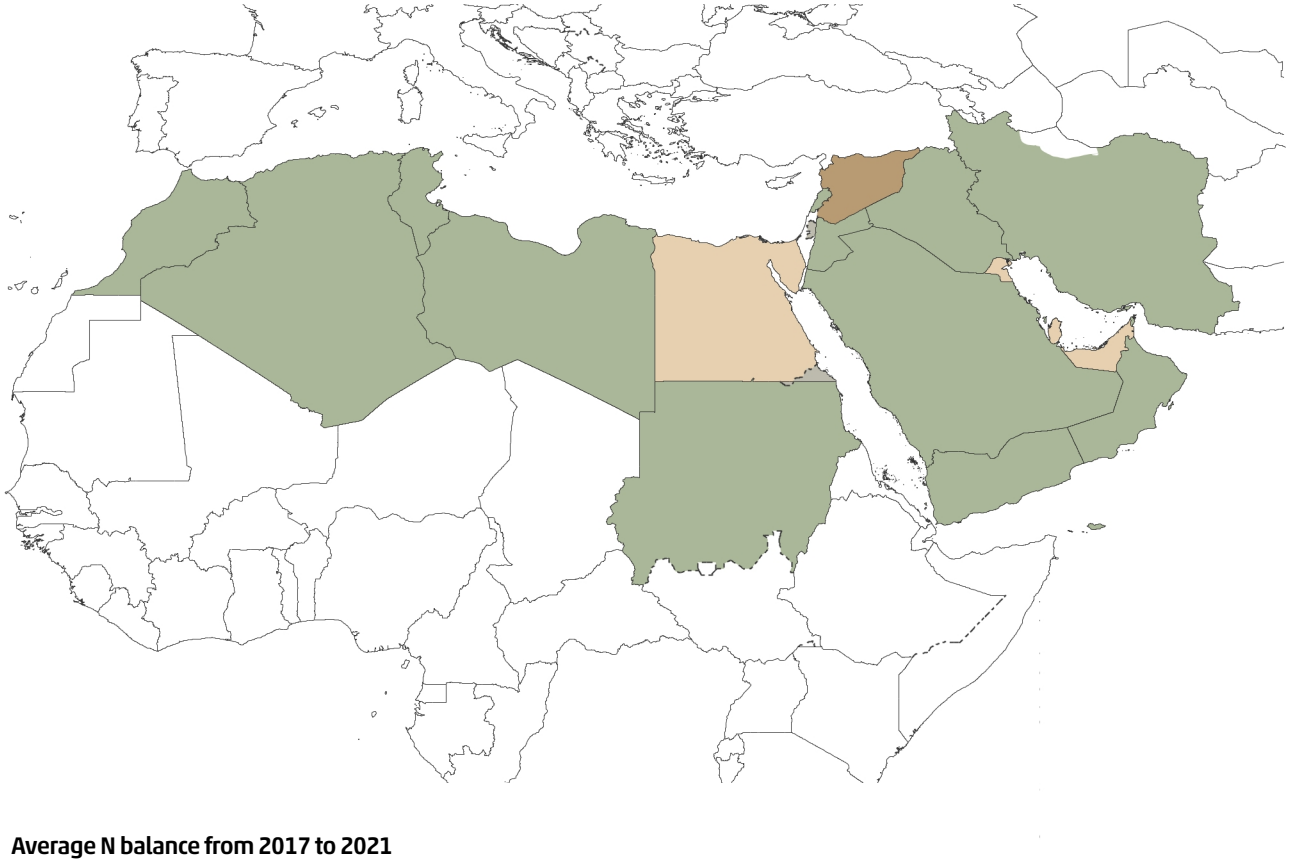
Urbanization

Most countries in the region are highly urbanized (UN DESA, 2018) with 15 of the 19 countries having a percentage of urban population greater than the global mean (53.9 percent urban population on average in 2015) (UN DESA, 2018). Moreover, the urban population in many of these countries is projected to grow substantially by 2050, especially those of North Africa. Urban growth rates in cities of the region are almost all in the 1 to 3 percent range.

Nutrient mismanagement

Four countries in NENA show evidence of excess N balances based on average N use data for the 2017 to 2021 period (Figure 11.8) (FAO, 2024b). Only the Syrian Arab Republic has a negative N balance for that period.

Figure 11.8. Country map of average values (2017 to 2021) for nitrogen use

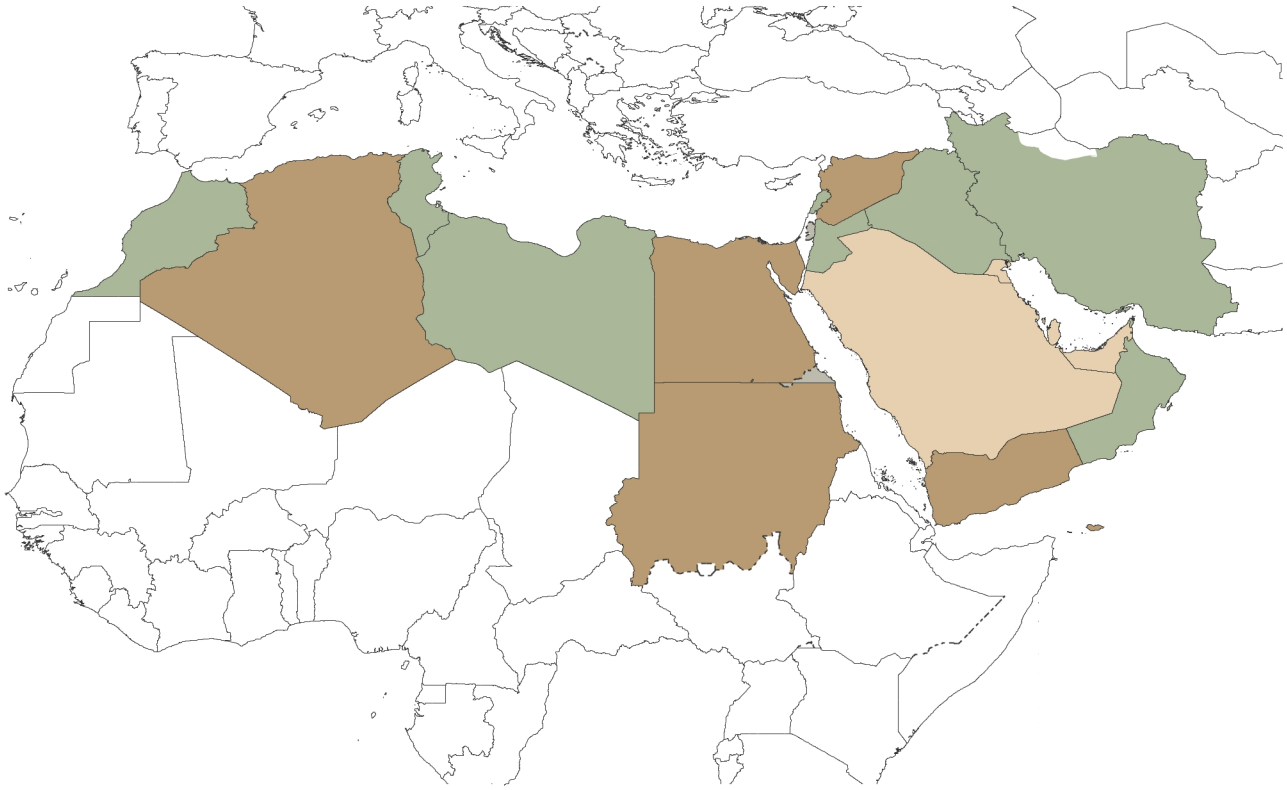


◊ *Sources:* **FAO**. 2024. *FAOSTAT: Fertilizers by Nutrient*. [Accessed on 9 January 2024]. https://www.fao.org/faostat/en/#data/RFN. Licence: CC-BY-4.0.

◊ **Zou, T., Zhang, X. & Davidson, E.A**. 2022. Global trends of cropland phosphorus use and sustainability challenges. *Nature*, 611(7934): 81–87. https://doi.org/10.1038/s41586-022-05220-z

Five of the Gulf States (Bahrain, Kuwait, Qatar, Saudi Arabia and the United Arab Emirates) show evidence of average excess P balances in the 2017 to 2021 period (Figure 11.9) (FAO, 2024b). Evidence of P mining is more widespread, with the Syrian Arab Republic, Yemen, Algeria, Egypt and the Sudan all showing negative P balances (FAO, 2024b) and long-term depletion of residual soil P (Zou, Zhang and Davidson, 2022).

Figure 11.9. map of average values (2017 to 2021) for phosphorus use



Refer to the disclaimer on page ii for the names and boundaries used in this map. Final boundary between the Republic of Sudan and the Republic of South Sudan has not yet been determined. Final status of the Abyei area is not yet determined.

◊ *Sources:* **FAO**. 2024. *FAOSTAT: Fertilizers by Nutrient*. [Accessed on 9 January 2024]. https://www.fao.org/faostat/en/#data/RFN. Licence: CC-BY-4.0.

◊ **Zou, T., Zhang, X. & Davidson, E.A**. 2022. Global trends of cropland phosphorus use and sustainability challenges. *Nature*, 611(7934): 81–87. https://doi.org/10.1038/s41586-022-05220-z

Small average negative K balances are common in many countries of NENA for the 2017–2021 period (Figure 11.10) (FAO, 2024b). Only Lebanon has a more substantial K deficit (an average annual deficit of 19.7 kg K/ha) (FAO, 2024b).

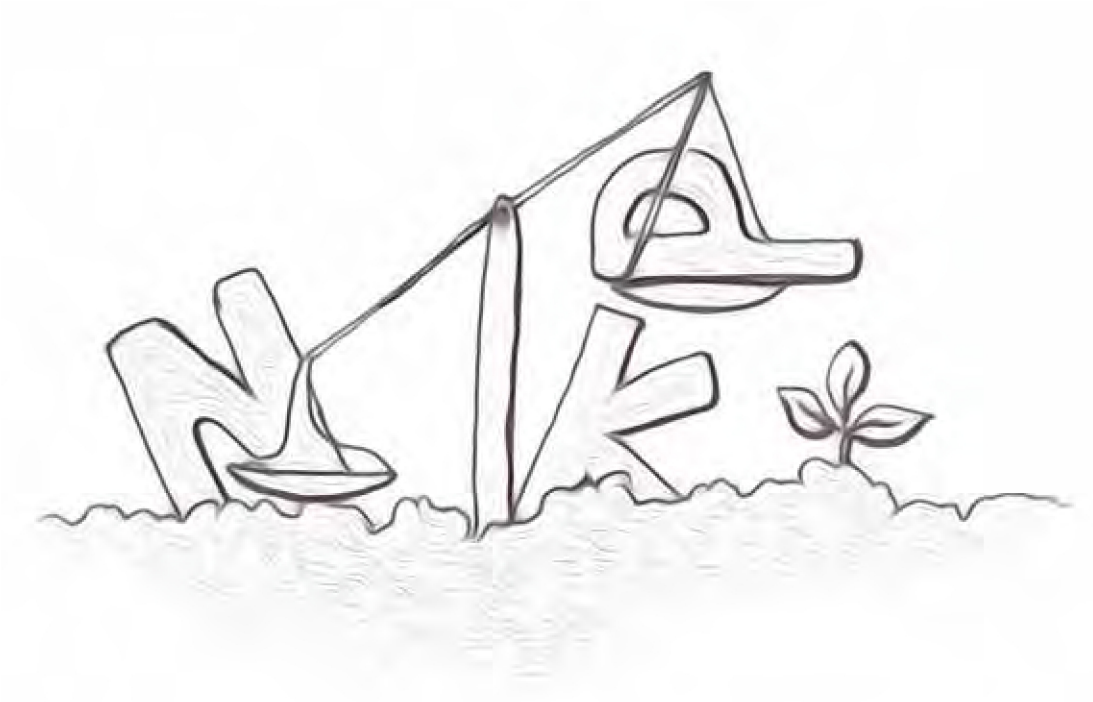
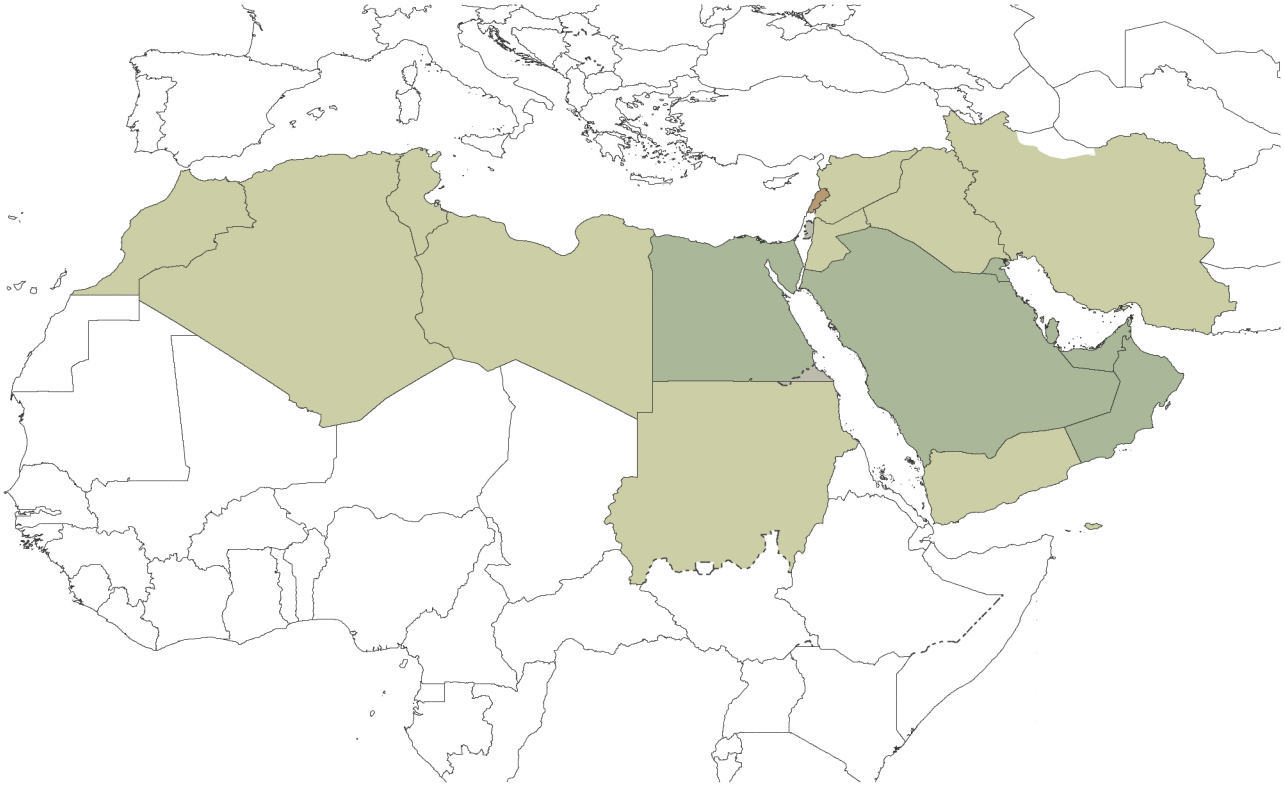


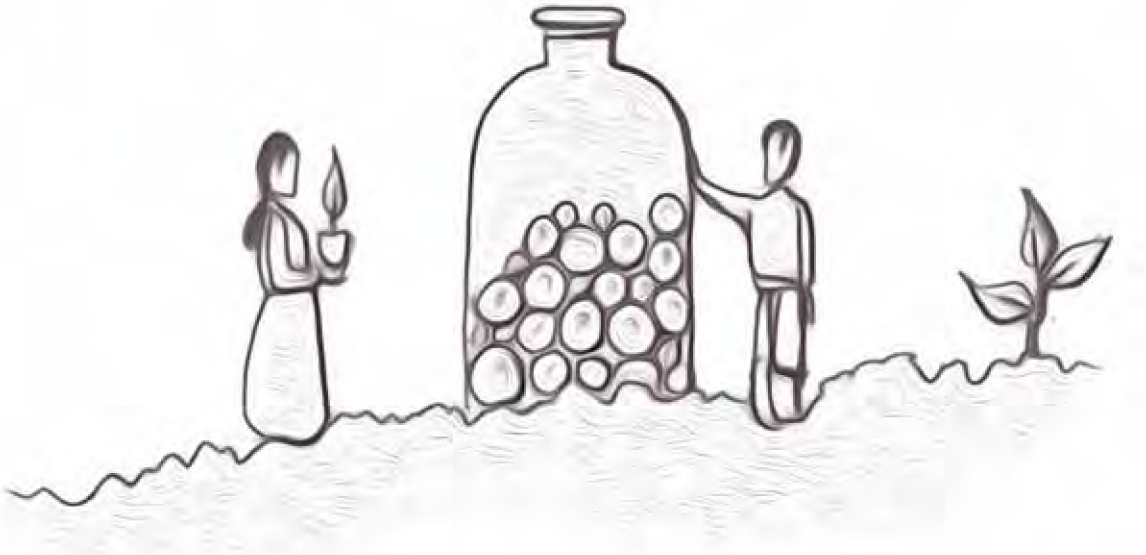
Figure 11.10. Country map of average values (2017 to 2021) for potassium use



Refer to the disclaimer on page ii for the names and boundaries used in this map. Final boundary between the Republic of Sudan and the Republic of South Sudan has not yet been determined. Final status of the Abyei area is not yet determined.

◊ *Sources:* **FAO**. 2024. *FAOSTAT: Fertilizers by Nutrient*. [Accessed on 9 January 2024]. https://www.fao.org/faostat/en/#data/RFN. Licence: CC-BY-4.0.

◊ **Zou, T., Zhang, X. & Davidson, E.A**. 2022. Global trends of cropland phosphorus use and sustainability challenges. *Nature*, 611(7934): 81–87. https://doi.org/10.1038/s41586-022-05220-z



11.3 Status and trend of threats to soil functions in the Near East and North Africa (NENA) region

11.3.1 Soil erosion in the Near East and North Africa region

Water erosion

Soil erosion is a global issue, leading to the loss of surface soil, degradation of the agricultural landscape, the loss of soil carbon and nutrients, a decrease in soil moisture-holding capacity, direct damage to crops, a decline in surface water quality due to suspended solids, and the siltation of hydro-agricultural infrastructure. Water erosion can provide particle size selectivity, where the finer particles are moved down slopes while the coarse particles are redeposited near their erosion sites. Thus, sorting and redistribution of eroded materials can have both agronomic and environmental repercussions.

When the quantity of sediment lost by erosion exceeds the rate of natural soil regeneration, the soil is gradually degraded and plant productivity declines (Borelli *et al.*, 2018). Estimates of the average tolerable soil loss by water erosion range from 1 to 10 t/ha annually, depending on the economic and environmental issues of a region (see Section 3.2.2 for more information). An annual value of 10 t/ha is used by several countries as a threshold for tolerable soil loss by water erosion (DiStefano and Ferro, 2016).

Borelli *et al.* (2018) use an annual threshold of 20 t/ha to identify hotspots of soil erosion above which implementation of soil conservation measures are considered critical.

The countries of the Near East have a risk of high rates of water erosion. Although no regional studies of water erosion rates are available, more localized studies throughout the region have identified hotspots of erosion by water (Husein, Kalkha, Bäumier, 2019).

Soils in the Syrian Arab Republic, especially in the coastal region, have a high susceptibility to water soil erosion due to lithology, high rainfall intensities, rough relief, topsoil fertility, degraded vegetation and soil, and accelerated human activity (Mohammed *et al.,* 2020; Abdo, 2021a). About 18 percent of the agricultural land is prone to soil loss hazard that exceeds 100 t/ha annually, mainly in western, mountainous areas (ACSAD 2007; Husein, Kalkha and Bäumier, 2019). For example, water erosion in the Wadi-Qandeel river basin threatens farming, food production, and thus secure sustainability. The intensive pattern of rainfall, high runoff peaks, rugged terrain, steep slopes, shallow soil profiles, and sparse vegetation are the major physical criteria that promote water erosion, especially in the areas surrounding rivers and waterbodies. An annual rate of soil loss of 93.02 t/ha was recorded, with a spatial annual average reaching 58.22 t/ha (Abdo, 2021).

In the Al-Abrash River basin in the Tartous Governorate in the west of the Syrian Arab Republic, the amounts of soil loss were classified by Jouhra (2022) into three categories of low, moderate and high. About 12 percent of the river basin had moderate soil erosion and 6 percent had high soil erosion. Both the moderate and high soil loss categories would be considered hotspots of erosion (i.e. >20 t/ha annually).

The Eastern Mediterranean region, due to its harsh climate, is known for its sensitivity to erosion, particularly in mountainous areas (Traboulsi, 2012). The areas more prone to soil erosion and degradation are those with steep slopes, poor vegetation, and barren agricultural land throughout the winter season.

Several studies of water erosion hazards have consistently shown high rates of water erosion in Lebanon. FAO (1977) indicated that the annual erosion rates in the Lebanese mountains reached

50–70 t/ha. Bou Kheir, Cerdan and Abdallah (2006) estimated that 35 percent of central Lebanon soil was affected by higher rates. El Hage Hassan *et al.* (2015) indicated that 36 percent of the Aaqoura mountainous area (northeast Lebanon) had a high water erosion rate and Charbel and El Hage Hassan (2017) suggested that 49 percent of the Bkassine forest (southern Lebanon) was also subject to a high rates of water erosion. The southern Lebanon region is one of the regions most exposed to water erosion. The problem has not been managed, although it threatens soil quality and sustainable agricultural production.

Application of the Revised Universal Soil Loss Equation (RUSLE) model for estimating soil erosion in the Gaza Strip in Palestine using a geographic information system shows annual soil erosion values of between 0.11 and 504 t/ha with a mean of 4.7 t/ha (Almughari, 2021). The areas at risk of high soil erosion rates account for 10 percent of the Gaza Strip and are concentrated in Gaza City and the valleys (Gaza Valley, Beit Hanoun Valley, and Salqa Valley) and in the coastal area of the cliffs.

Jordan has serious soil erosion. This is by no means a new problem but one that has intensified as human population pressures on the land have increased (Beaumont and Atkinson, 1969). FAO, UNEP and UNESCO (1979) and Battikhi and Arabiat (1983) reported that part of the highlands of Jordan was classified within annual soil loss categories of 50–200 t/ha and >200 t/ha. The results of the RUSLE application on the Asreh watershed shows that the mean annual soil loss rate is about 7.8 t/ha (Al Ayyash, 2021). About 73 percent of the area has been classified as having a low annual soil loss rate (<10 t/ha), 14 percent as moderate (10–20 t/ha) and only 13 percent of the area classified as having a high soil loss rate (>20 t/ha)

In Iraq, the delineation of soil erosion zones is important to protect soil and manage fertile land in a sustainable manner. For example, a quantitative assessment of annual soil erosion by water in the northern part of Kirkuk Governorate in the north of Iraq was investigated through the integration of remote sensing, Geographical Information System (GIS) software and the RULSE soil erosion model (Al-Abadi, Ghalib and Al-Qurnawi, 2016). The estimated annual soil losses ranged from 0 to 245 t/ha with an average of 2 t/ha. The spatial pattern of classified soil erosion rate indicated that the areas at moderate to high risk were in the northeast with a small area in the east, while the minimal to low soil erosion risk zones covered the remaining areas. In the Alibag catchment, in the Iraqi Kurdistan region, annual soil loss values range between 0 and 16.6 t/ha (Keya, 2018).

To compare the amounts of soil eroded in the Euphrates River Watershed, Khassaf and Al Rammahi (2018) used the RUSLE model, remote sensing and GIS over the years 2013 to 2017. Overall, 99.7 percent of the watershed area had slight soil erosion loss (i.e. 0 to 2.5 t/ha annually) while 0.17 percent of the watershed had soil erosion of the slight to moderate type (i.e. 2.5 to 5 t/ha annually). Moderate to extremely high rates of soil erosion loss (50–100 t/ha annually) were found only in a very small area (0.006 percent) the riverbed of the Euphrates (Khassaf and Al Rammahi, 2018).

The impact of soil erosion is more serious in regions that have sparse vegetation and experience intense rainfall, such as the Islamic Republic of Iran. Water erosion is the main cause of land degradation in the southwest of the country and leads to the impoverishment of fertile soils and farmland (Sadeghi, 2011). The application of integrated RUSLE, GIS and remote sensing has been used in a few studies for estimating soil loss in the south of the Islamic Republic of Iran (e.g. Ostovari *et al*., 2021).

In the southwest of the Islamic Republic of Iran, the annual soil loss (24.6 t/ha), varied from zero in the flat areas up to 154 t/ha in the hilly areas. Erosion classes of very high, severe, and very severe covered 25 percent (6 983 ha) of the site. The length and slope (LS) factor was identified as the most influential link to soil erosion (Ostovari *et al*., 2021).

Soil erosion is a cause of soil degradation in Saudi Arabia and is influenced by the country's topography, soil type, and climate. The soil erosion risk was reclassified into five erosion risk classes and showed that 19.3 percent (37 740 ha) of the Wadi Yalamlam was under very severe risk (Bahrawi *et al*., 2016).

In North Africa, the soil erosion impact has been intensified by global warning and increased drought, especially in North African countries, such as Algeria and Tunisia, where an arid climate is dominant. Soil erosion by water has caused serious regional soil loss. For example, a study in the Mellegue catchment (a large cross-border basin shared between Algeria and Tunisia) showed that the catchment has undergone significant soil water erosion (Weslati and Serbaji, 2023). The average annual erosion risk also increased from 1.58 t/ha in 2002 to 1.78 t/ha in 2020, leading to an increase in total estimated soil loss of 54 000 t/ha in 2020 compared to around 25 500 t/ha in 2002. Maps of erosion risk show that highly eroded areas are more frequent downstream of the basin (Table 11.1).

Table 11.1. Comparative soil loss rates in the Mellegue watershed (Algeria and Tunisia) between 2002 and 2020

Class (t/ha annually)	Erosion rate	2002						2020					
		Tunisia		Algeria		Total		Tunisia		Algeria		Total	
		(km²)	(%)	(km²)	(%)	(km²)	(%)	(km²)	(%)	(km²)	(%)	(km²)	(%)
Missing data	–	70	1.6	87	1.4	157	1.5	72	1.6	80	1.3	152	1.4
0–7	Very weak	4 173	94.9	5 837	95.0	10 010	94.9	4 036	91.7	5 633	91.7	9 669	91.7
7–15	Weak	96	2.2	127	2.1	223	2.1	162	3.7	239	3.9	400	3.8
15–30	Moderate	32	0.7	58	0.9	90	0.9	74	1.7	108	1.8	181	1.7
30–60	High	15	0.3	16	0.3	31	0.3	34	0.8	51	0.8	85	0.8
>60	Very high	12	0.3	21	0.3	33	0.3	22	0.5	35	0.6	57	0.5
Total	–	4 399	100	6 145	100	10 545	100	4 399	100	6 145	100	10 545	100

◊ *Source:* **Weslati, O. & Serbaji, M.-M.** 2023. Spatial assessment of soil erosion by water using RUSLE model, remote sensing and GIS: a case study of Mellegue Watershed, Algeria–Tunisia. *Environmental Monitoring and Assessment*, 196(1): 14. https://doi.org/10.1007/s10661-023-12163-z

The northwest of Algeria is prone to soil water erosion with a very high annual soil loss (Achite and Ouillon, 2007). The wadis (dryland rivers) also deliver sediment into reservoirs at very high rates.

Tunisia has a serious risk of soil water erosion, with 6.4 percent of Tunisia affected by a very high annual soil loss rate, estimated at >30 t/ha, and a further 4.2 percent with high mean annual soil losses (between 20 and 30 t/ha) (Weslati and Serbaji, 2023). The most eroded areas were identified in the southwestern, central and western parts of Tunisia.

In Morocco, the annual sediment yield is between 100 and 4 620 t/h in 38 catchments (Vanmaercke *et al.*, 2014). Water erosion is intense in the High Atlas mountain range, while the Anti-Atlas has only a low to medium erosion risk (Tairi and Elmouden, 2019).

In Egypt, the RUSLE model integrated with GIS has been used to estimate soil loss in Wadi Umm Ashtan and Wadi Umm El-Rakham basins located in the northwest coast of the country (Shoman, 2019). The highest value of predicted annual soil erosion was 563 t/ha for Wadi Umm Ashtan and 212 t/ha for Wadi Umm El-Rakham. Soil erosion of Wadi Umm Ashtan and Wadi Umm El-Rakham basins were classified into six classes: slight (about 99 percent of both areas), moderate (about 0.5 percent), and high (about 0.1 percent), with very small areas of very high, severe and very severe erosion classes (Shoman, 2019) (Table 11.2).

Table 11.2. Soil loss rates and distribution in Wadi Umm Ashtan and Wadi Umm El-Rakham

Annual soil loss rate (t/ha)	Soil erosion hazard class	Wadi Umm Ashtan				Wadi Umm El-Rakham			
		Area		Annual soil loss		Area		Annual soil loss	
		ha	%	tonnes	%	ha	%	tonnes	%
0–7	Slight	10 522	99.4	36 827	95.0	8 308	99.3	29 078	97.0
7–15	Moderate	40	0.4	440	1.0	47	0.6	517	2.0
15–25	High	10	0.1	200	1.0	8	0.1	160	1.0
25–45	Very high	7	0.1	245	1.0	4	0.0	140	0.0
45–60	Severe	2	0.0	105	0.0	1	0.0	53	0.0
>60	Very severe	3	0.0	934	2.0	1	0.0	136	0.0
Total		10 584	100	38 751	100	8 369	100	30 083	100

Source: **Shoman, M.** 2019. Estimation of soil erosion hazard in some wadis of the north western coast Egypt using RUSLE model and GIS. *International Journal of Advanced Research (IJAR)*, 7(5): 290–302.

Wind erosion

Wind erosion is a complex geomorphic process affected by numerous variables (Wenqian *et al.*, 2023). Around 28 percent of the world's degraded land area is impacted by wind-induced soil erosion (Středová *et al.*, 2021). In the NENA region, wind erosion is of environmental concern due to the arid and semi-arid climates, limited vegetation, and vast desert landscapes (Ostovari *et al.*, 2022). Wind erosion causes adverse effects on fragile ecosystems, agriculture and human livelihoods in the region (Awadh, 2023), leading to soil degradation and negative impacts on plant growth and ecosystem health (Zhang, 2021). Recent studies (Gholami *et al.*, 2020; Rashki, Middleton and Goudie, 2021; Zittis *et al.*, 2022) have indicated that rising temperatures and shifting precipitation patterns are worsening wind erosion, resulting in the loss of fertile topsoil, reduced agricultural yields, and increased airborne dust particles that endanger local populations' health (Tian *et al.*, 2021).

While specific data on the exact extent of wind-eroded land may vary and evolve over time, it is well-documented that vast stretches of the NENA region's arid and semi-arid landscapes are vulnerable to wind erosion. Some of the most affected countries include Egypt, the Islamic Republic of Iran, Libya, Morocco, Saudi Arabia, and Tunisia (Zittis *et al.*, 2022).

Primary drivers of wind erosion include prevalent arid and semi-arid conditions marked by low rainfall and high temperatures (Figure 11.4). Peak wind speeds, often exceeding 6 m/s at a height of 2 m occur throughout the region and exceptionally high wind speeds, surpassing 8 m/s, have been documented in western Mauritania and southern Saudi Arabia. The heightened atmospheric dynamism underscores the necessity for robust soil conservation and land management strategies.

This arid environment creates ideal conditions for wind erosion, as the absence of vegetation cover and loose, dry soils render the region susceptible to soil erosion driven by strong winds. Consequently, the combination of high PET, seasonal rainfall patterns, and arid conditions serves as a significant driver of wind erosion in the NENA region.

Climatic erosivity (CE) describes the climate's capacity to generate conditions conducive to wind erosion (Skidmore 1986; Středová *et al.* 2021). Climatic erosivity accounts for factors such as soil surface dryness and the occurrence of erosive wind events. Lou, Wang and Cai, (2019) and Yue *et al.*, (2019) examined spatial and temporal fluctuations in CE as an indicator for assessing the potential scale at which climate change may impact the risk of wind erosion. The main determinants of CE include wind speed and precipitation patterns (Fenta *et al.*, 2020).

The mean annual CE-factor has substantial spatial variability within each country (Figure 11.11). The spatial patterns of climate indicators observed between 2015 and 2022 varies between the NENA countries. Climatic erosivity exhibits considerable intensity, with CE-factor values consistently

surpassing 100 across all countries, primarily concentrated in the Gulf and North Africa (the regions suffering from serious wind erosion and desertification problems). In specific areas, such as Algeria, Egypt, Mauritania, Tunisia, and the United Arab Emirates, the CE factors exceed 300 due to high wind speeds, signifying an elevated susceptibility to wind erosion. The Levant area has CE values below 100, which are the lowest values observed (Figure 11.11). Climatic erosivity is positively correlated with variables such as wind speed and mean annual temperature, and inversely correlated with precipitation. At the specific site level, wind speed emerges as the most strongly correlated variable, which suggests that wind speed has exerted the most substantial influence on the climatic erosivity. The CE was calculated using the equation established by FAO in 1979 which assesses the climate's capacity to generate conditions conducive to wind erosion (Lou, Wang and Cai, 2019; Yue *et al*., 2019):

$$CE = \frac{1}{100}\sum_{i=1}^{i=12} {U_i}^3 (\frac{PET_i - P_i}{PET_i}) d_i$$

where CE is climate erosivity, U_i U_i denotes the mean wind speed measured at 2 m height, PET_i PET_i is potential evapotranspiration (mm), P_i P_i is the precipitation (mm), (*i* refers to the month), and d_i d_i refers to the number of days in the month.



Figure 11.11. Temporal distribution for (a) mean annual climatic erosivity values for each year, (b) mean annual climatic erosivity values and (c) mean annual climatic erosivity values for the Near East and North Africa and adjacent countries over the 2015–2022 period



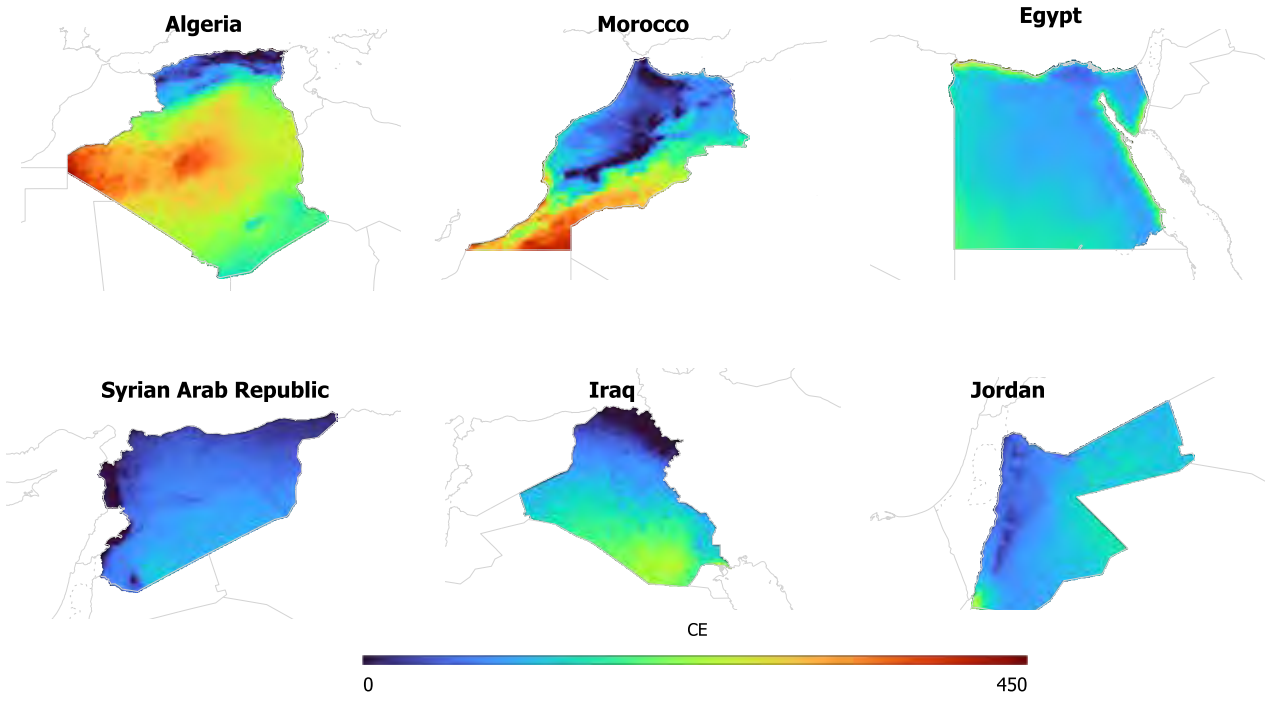
Notes: In (b), the black horizontal line represents the median. The upper and lower vertical lines indicate the 95th and 5th percentile values, respectively. The red dots indicate outliers.

Country abbreviations: AE = United Arab Emirates , BH = Bahrain, DJ = Dijbouti, DZ = Algeria, EG = Egypt, IL = Islamic Republic of Iran, IQ = Iraq, Jo = Jordan, KW = Kuwait, LB = Lebanon, LY = Libya, MA = Mauritania, MR = Morocco, OM = Oman, PS = Palestine, QA = Qatar, SA = Saudi Arabia, SD = Sudan, SO = South Sudan, SY = Syrian Arab Republic, TN = Tunisia, and YE = Yemen.

◊ *Source:* **Harris, I., Osborn, T.J., Jones, P. & Lister, D.** 2020. Version 4 of the CRU TS monthly high-resolution gridded multivariate climate dataset. *Scientific Data*, 7(1): 109. https://doi.org/10.1038/s41597-020-0453-3

From 2015 to 2022, around 74 percent of the NENA region was classified as having a high to very high risk of wind erosion. The areas most prone to wind erosion were in Algeria, Mauritania, Morocco, and Somalia (Figure 11.11 and Figure 11.12). Conversely, the coastal regions situated within the Levant had a low risk due to their abundant vegetative cover, predominantly consisting of forests and perennial grasslands, along with heightened winter and spring precipitation. The Gulf countries, specifically Bahrain, Kuwait and Qatar were susceptible to wind erosion. The mean values of CE fluctuated within a range of 228 in 2019 and 262 in 2018, with a peak during 2017 and 2018 (Figure 11.11). The highest CE values were in Algeria, Mauritania, Morocco, and the Sudan, indicating an elevated susceptibility to wind erosion.

Figure 11.12. Spatial distribution of mean annual values of climatic erosivity in selected countries over the 2015–2022 period



Refer to the disclaimer on page ii for the names and boundaries used in this map

Note: DZ = Algeria, EG = Egypt, IQ = Iraq, JOR = Jordan, LB = Lebanon, MA = Morocco, QA = Qatar, SYR = Syrian Arab Republic, and AE = United Arab Emirates.

◊ *Source:* **Harris, I., Osborn, T.J., Jones, P. & Lister, D.** 2020. Version 4 of the CRU TS monthly high-resolution gridded multivariate climate dataset. *Scientific Data*, 7(1): 109. https://doi.org/10.1038/s41597-020-0453-3

Local authorities facing the challenge of wind erosion in vulnerable regions should prioritize several key actions (UNCCD, 2022). Encouraging sustainable land management practices – including afforestation and reforestation – can help stabilize soils and reduce the risk of erosion. Implementing efficient water management strategies to enhance soil moisture levels is important, while public awareness campaigns on the importance of sustainable land use and vegetation cover could help build community support. Early warning systems to predict and respond to wind erosion events also need to be established. Lastly, promoting and enforcing land-use regulations to prevent further degradation and adopting innovative technologies for erosion control are crucial steps in mitigating the impacts of wind erosion.

Consequently, mitigation policies must integrate Indigenous Peoples-including Indigenous nomadic pastoralists – whose traditional ecosystem management effectively curtails soil erosion. Their traditional mobility and water-harvesting practices protect fragile soil structures and shallow

water lenses, offering proven, community-led adaptation strategies against wind erosion across arid ecosystems. Their tailored land-use practices maintain soil moisture and safeguard vulnerable water lenses from severe climatic degradation (UNCCD, 2024).

The NENA region's susceptibility to wind erosion is shaped by the combination of high aridity, limited precipitation and the arid and semi-arid conditions that prevail. Climate challenges present an environmental threat to numerous NENA countries, with Algeria, Bahrain, Kuwait, Mauritania, Morocco, the Sudan, and Qatar particularly affected (See Figure 11.2). The adverse impact of wind erosion on soil quality, agricultural productivity and the overall environmental equilibrium in these nations cannot be overstated. The climatic diversity across the region, from high aridity to coastal regions with sporadic higher precipitation, underscores the necessity for tailored mitigation strategies. Thus, taking proactive measures to combat wind erosion is pivotal in ensuring the sustainable development of the NENA nations amid the evolving challenges of a changing climate.

11.3.2 Soil carbon loss in the Near East and North Africa region

Soil organic matter is made from living and dead plants, animals and microbes within and on the soil surface and is an indicator of soil fertility. Soil organic matter makes soils more productive by providing nutrients for soil microbes and bonding soil particles for a stable structure. Soil organic carbon is the major component of soil organic matter.

Soil organic carbon content

Near East and North Africa countries are mainly semi-arid or arid with low levels of soil organic matter. Darwish, Atallah and Fadel (2018) estimated that although the NENA region covered 18 percent of the Earth's surface, it only had 4.1 percent of the global SOC stock. Most countries in the NENA region have moderate to relatively low total stocks of SOC (Figure 11.13) due to the dominance of soil types associated with dry conditions (such as Xerosols, Arenosols, and Lithosols) (Darwish and Fadel, 2017). Mean topsoil SOC contents (i.e. to 30 cm depth) are estimated to range between 20 to 30 tonnes SOC/ha in most of the NENA countries but are lower than 20 tonnes SOC/ha in Bahrain, Kuwait, Oman, Saudi Arabia, the United Arab Emirates and Yemen, and higher than 30 tonnes SOC/ha in Comoros, Gaza, Lebanon, and the Sudan (Darwish and Fadel, 2017). Overall, the mean SOC stock in the topsoil (0 to 30 cm) of the NENA countries is estimated at 37 tonnes SOC/ha and at 78 tonnes SOC/ha for the profile to 1 m depth (Darwish and Fadel, 2017).

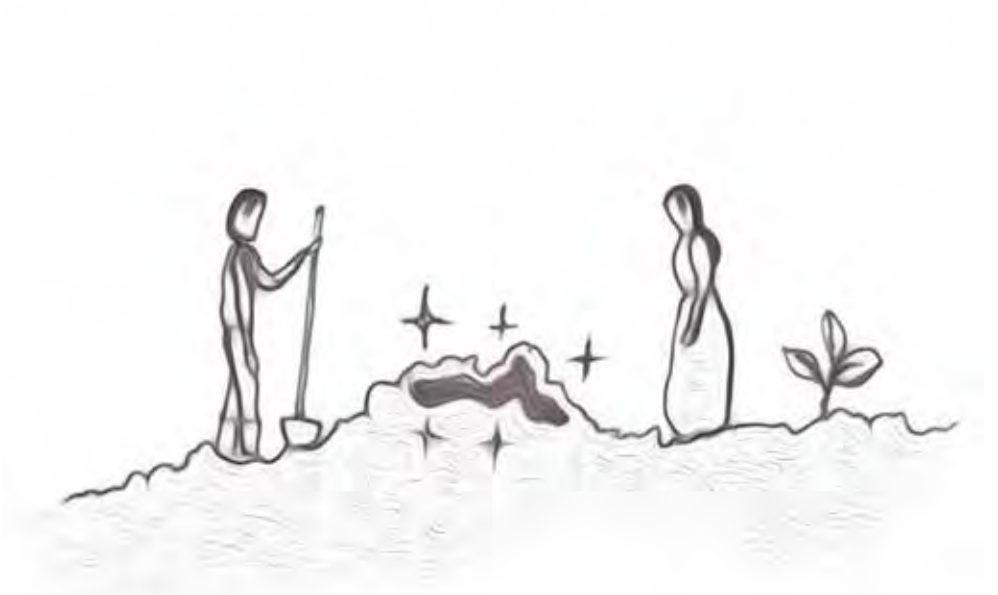


Figure 11.13. Spatial view of soil organic carbon stock in the topsoil (0–30 cm) across the countries of the Near East and North Africa region



Refer to the disclaimer on page ii for the names and boundaries used in this map.

◊ *Source:* **FAO**. 2024. *Regional action plan for sustainable soil management in the Near East and North Africa region*. Rome. https://doi.org/10.4060/cd2814en

Country-specific studies of SOC contents show a considerable range of SOC levels. For example, Husein, Kalkha and Bäumier (2019) reported that the SOC content in the northeast of the Syrian Arab Republic in the topsoil of the Al Balikh plain ranged between 0.6 and 8.7 g C/kg of soil, whereas SOC in the south of the Syrian Arab Republic ranged between 2.1 and 7g C/kg of soil (Mohammed *et al.*, 2020). However, Abdulla (2012) pointed out that only 2 percent of total area of Syrian soils were considered to be rich in organic matter. Baraket *et al.* (2021) studied the spatial variability of SOC in semi-desert areas of North Africa (Tunisia) and found that SOC did not exceed 2 g C/kg of soil. Soil organic carbon in Lebanese soils ranged between 1.5 and 14 g C/kg of soil. In the northern Nile Delta of Egypt, SOC content ranged between 2 and 12 g C/kg of soil (Arshad *et al.*, 2020). In the mangrove stands along the Egyptian Red Sea coast, 20 g C/kg of soil (at 0–10 cm depth) has been reported (Eid and Shaltout, 2015). In Morocco, in the Sidi Bennour District on the Doukkala Plain, high SOM levels were concentrated in Vertisols, and low SOM levels were observed in immature soils, with an overall mean of 1.4 percent of SOM (Bouasria *et al.* 2021)

Decline of soil organic carbon content

Unlike soils in humid regions, dryland soils have higher losses of SOC due to soil degradation (Farage *et al.*, 2007). Dryland soils are especially vulnerable to land degradation and desertification and associated SOC losses, as SOC storage decreases with increasing temperature and decreasing soil water content.

The warm, arid, climate conditions limit plant growth and contribute to the rapid biodegradation of soil organic material, leading to soils with a tendency to have naturally low SOM contents (Darwish and Fadel, 2017). This also suggests that less livestock manure is being returned to the soil than at some time in the past. Livestock overgrazing presumably also reduces vegetation cover and thus limits the return of OM to the soil.

Thus, weather variability, drought and depleted vegetation are dominant causes for SOM decline (Zdruli and Zucca, 2023). Under Mediterranean conditions, the succession of dry summers and wet winters increases the emission of CO_2 by promoting the decomposition of SOC (Jarvis *et al.* 2007).

Soil erosion and urban expansion have led to an irreversible loss of soil cover, SOC and other nutrients (Darwish and Fadel, 2017). Darwish and Fadel (2017) also estimated that 25 billion tonnes of OC

had been irreversibly lost due to water erosion and another 53.6 billion tonnes lost due to urban expansion.

Ploughing and land-use change from native forest to crops are considered to cause a rapid loss of SOC (Darwish and Fadel, 2017). In Algeria, Boubehziz *et al.* (2020) focused on SOC depletion in the Tiffech watershed (in the northeast of Algeria), using geostatistics and GIS software. The maps produced showed a good conservation of SOC stocks in areas with dense vegetation. Conversely, poor SOC contents were estimated where land degradation factors had taken place, such as the decline of forests and subsequent erosion. A study conducted by Bel-Lahbib *et al.* (2022) in the Doukkala plains (one of the largest irrigated areas in Morocco) showed that the integration of new technologies in agriculture had negative impacts on soil quality, especially SOM loss.

Forest fires and natural disasters cause the deterioration of vegetation cover and thus a decline in soil chemical properties, especially SOC content. Van der Werf *et al.* (2010) and Guenon *et al.* (2011) reported that fires eliminated large quantities of organic matter from ecosystems through vegetation burning and the oxidation of soil organic matter. Two years after the fire in western Algeria, SOM was significantly lower in the burnt soil compared to a control zone (Allam *et al.*, 2020). Following a forest fire in the Al Qardaha district, Latakia Governerate, in the Syrian Arab Republic, the SOC content ranged between 8.8 and 45.5 g C/kg in the control sites, and between 6.6 and 21.8 g C/kg at burned sites (Barakat *et al.*, 2022).

Overgrazing of forests also causes a decline in the organic resources that reach the soil, so laws were passed by the Syrian government prohibiting grazing in forest areas (Syrian Arab Republic, 2018).

Increases in soil organic carbon content

Given the low SOC stores and limited productivity of soils of the NENA region, the possibility of significant SOC sequestration is small (Darwish, Atallah and Fadel, 2018). Practices such as the introduction of legumes as part of a cereal–legume rotation, the application of N fertilizers, and the adoption of no-till has been shown in site-specific studies to cause a notable increase of SOC (Darwish, Atallah and Fadel, 2018). The effects of crop rotations on SOC are related to the amounts of above and below ground biomass produced and retained in the system (Darwish, Atallah and Fadel, 2018). Both appropriate soil management and land abandonment may enhance the soil carbon pool (Boukhdoud *et al.* 2016).

In some cases, the observed increases in SOC have been substantial. For example, Arshad *et al.* (2020) demonstrated that changing land use in the northern Nile Delta of Egypt increased SOC content. Soil organic carbon was measured at 3.77 g C/kg in virgin soil, increasing to 5.22 g C/kg in fish farms, and to 6.54 g C/kg in croplands. An evaluation of the effects of human activity over the last decades on SOC, using satellite imagery and GIS in the Nile Delta area of Egypt also showed that land use and human activities affected SOC content, and that changing from agricultural activities to fishponds increased SOC, due to organic additions used as nutrients for fish (Mohamed *et al.*, 2019).

Some national studies have investigated the effects of management change on SOC contents. For example, a study to predict SOC changes between 2000 and 2030 for Jordan, using the Global Environment Facility Soil Organic Carbon (GEFSOC) modelling system, suggested that SOC could increase in heavily-used areas under irrigation but was likely to decrease in grazed rangelands that cover most of Jordan, giving an overall decrease in total SOC over time (Al-Adamat *et al.*, 2007). In Morocco, no-till farming increased SOC compared to conventional tillage (Aboutayeb, El-Yousfi and El-Gharras, 2020).

Land-use policies must address soil conservation and carbon sequestration to maintain and improve soil health and sustain food production.

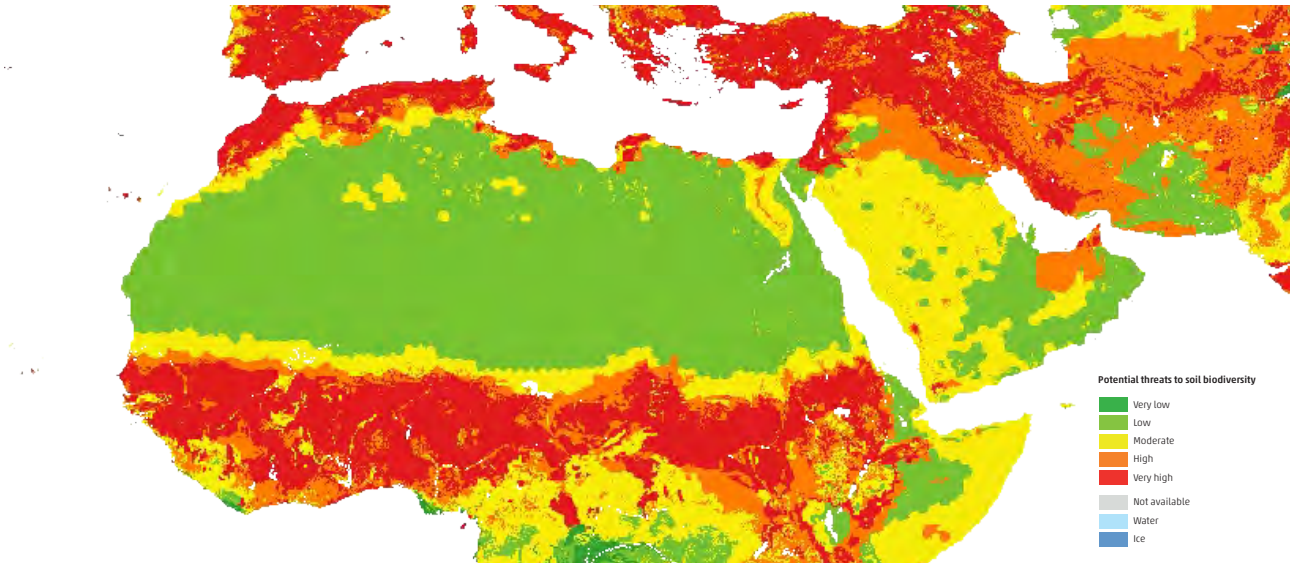
11.3.3 Soil biodiversity loss in the Near East and North Africa region

The Near East and North Africa region is the home to one-tenth of the globally recorded plant species (Zdrulli and Zucca, 2023). The NENA region has many diverse ecosystems, including mountains, forests, drylands, oases, and coastal and marine. The Mediterranean coastline areas of the region are categorized as biodiversity hotspots, as many plant and animal species are extinct or endangered (Myers *et al.*, 2000). Climate change presents a large and increasingly severe factor endangering the status of the ecosystems in the form of droughts, desertification and wildfires. For example, in 2023, Algeria, Lebanon, Morocco, Tunisia, and the Syrian Arab Republic had huge fires due to drought and high temperatures.

The NENA region is hot, dry and saline in many areas, which is reflected in the low concentration of organisms, compared to the global level. Both the amount of soil microbial biomass C and the richness of soil fungi are very low throughout the region (Orgiazzi *et al.*, 2016). The number of species, abundance and biomass of earthworms were at the low end of the scale in the coastal areas of the NENA region, and there were insufficient studies to assess earthworms in the interiors of both North Africa and the Near East (Phillips *et al.* 2020).

The potential threats to soil biodiversity are classed as very high in some areas of North Africa, specifically the coastal areas from Casablanca (Morocco) to Tunisia and around urban areas such as Tripoli and Benghazi (Libya) and the Nile Delta (Egypt) (Orgiazzi *et al.*, 2016) (Figure 11.14). In the Near East, the threats to soil biodiversity are classed as very high along the Mediterranean coastline of Lebanon and the Syrian Arab Republic, as well as large areas of Iraq and the west of the Islamic Republic of Iran (Figure 11.14). Such areas are judged to be at a very high risk of desertification (Orgiazzi *et al.,* 2016) which seems to be the main threat to soil biodiversity. Generally, the desert areas of the NENA region are judged to have a low or a very low risk of soil biodiversity decline, as the impacts of various drivers of degradation considered by Orgiazzi *et al.* (2016) are minimal.

Figure 11.14. Potential threats to soil biodiversity for the Near East and North Africa region



◊ *Source:* **Orgiazzi, A., Bardgett, R.D., Barrios, E., Behan-Pelletier, V., Briones, M.J.I., Chotte, J-L., De Deyn, G.B.** *et al.* 2016. *Global Soil Biodiversity Atlas.* Luxembourg, Publications Office of the European Union.

Mitigating the loss of soil biodiversity in the NENA drylands requires stabilizing the land rights of traditional stewards. Indigenous Peoples and pastoralists can only protect and restore soil food webs in the long term if policies and investments safeguard their customary resource rights (UNCCD, 2025).

Soil biodiversity is both directly and indirectly affected by land cover (Figure 11.3). The land cover classes in the NENA region are dominated by bare land, which comprises nearly 80 percent of the area, which lowers overall biodiversity (although the desert region does host many valuable and rare species of fauna and flora).

Several countries reported that changes in land and water use and management, invasive plant species, climate change and pollution, natural disasters, wildfires, earthquakes, and heavy storms are among the main threats to plant, animal, forest, and aquatic genetic resources (FAO, 2019) (Table 11.3). In many of the region's countries, war and armed conflicts have also seriously affected biodiversity.

Table 11.3. Drivers of change reported to affect biological and genetic resources in the Near East and North Africa of relevance to soil biodiversity

Drivers of change	Countries reporting	Examples or description
Changes in land and water use and management	Jordan, Lebanon, the Sudan, the United Arab Emirates, and Yemen	Oil drilling, gold mining and unsustainable water management practices have led to a reduction in arable land area, crop production and productivity. Increased mechanization and the expansion of traditional rainfed agriculture have led to a decrease in rangeland area. Human population growth and the expansion of the agriculture sector have increased demand for water.
Changing economic, sociopolitical and cultural factors	Iraq, Jordan, the Sudan, the United Arab Emirates, and Yemen	Open market policies and changing food consumption patterns are a threat to the conservation and use of the region's biodiversity for food and agriculture.
Overexploitation and overharvesting	Iraq, Jordan, Oman, the Sudan, the United Arab Emirates, and Yemen	Overgrazing has resulted in land degradation, soil erosion and the loss of grass species.
Climate change and natural disasters	Egypt, Jordan, Lebanon, Oman, the Sudan, the United Arab Emirates, and Yemen	Changing climatic conditions (predicted to include rising temperatures, greater fluctuation in rainfall patterns and longer droughts), will contribute to increasing the frequency and severity of natural disasters such as floods and fires.
Population growth and urbanization	Jordan, Oman and the United Arab Emirates	Human population growth and urbanization put pressure on the availability of arable land and on sanitation infrastructures. When the latter are outdated or overloaded, waste is discharged directly into the environment and finds its way into surface or groundwater.
Pollution	Jordan and the United Arab Emirates	Increased use of pesticides is negatively affecting biodiversity in and around production systems.

◊ *Source:* **FAO.** 2019. *Near East and North Africa Regional Synthesis for The State of the World's Biodiversity for Food and Agriculture.* Rome. https://openknowledge.fao.org/server/api/core/bitstreams/3f353815-8ba1-4c23-87c7-55dc6271e0c1/content

The NENA region includes diverse ecosystems and some of the most important biodiversity areas in the world, as well as being the originating sites of many domesticated species. The region has 33 biological reserves in 12 countries that are internationally recognized as being rich in biodiversity, and cultural, social and economic values and 153 wetlands protected under the Ramsar Convention on Wetlands (UNESCO, 2021). All countries in the NENA region are parties to the Convention on Biological Diversity (CBD, 1992) but coordination within and between countries, involving policies and legislation, as well as national surveys and records are minimal.

There is a lack of information available on soil biodiversity in the NENA region. Several studies on the global distribution of soil organisms, such as Phillips *et al.* (2020) for earthworms, or Puissant *et al.* (2021) for nematodes largely exclude the region, due to the lack of studies available. The sections for NENA in the 2020 *State of Knowledge of Soil Biodiversity* are short and concluded that, "Most countries in this region reported that despite the relevance of soil biodiversity to the country, very little has been done to enhance it due to a lack of resources." (FAO *et al*., 2020, p. 407).

11.3.4 Nutrient mismanagement in the Near East and North Africa region

Overview

Nutrient mismanagement is associated with both the over- and undersupply of nutrients to crops. The oversupply of nutrients may contribute to environmental issues in air and the eutrophication of water, while nutrient undersupply is one cause of yield gaps between potential yields and actual yields of agricultural crops.

A widely used measure to assess the over- and undersupply of nutrients is the nutrient balance, which compares nutrient inputs from all major sources with nutrient uptake by the crop. Data from FAOSTAT can be used to calculate nutrient balances for the major macronutrients (N, P and K) (FAO, 2024b) and provides a comparative measure across all NENA countries. A substantial oversupply of N and P (high positive balances) indicates a high risk of air and water quality degradation, while an oversupply of K is not associated with environmental issues. The undersupply of N, P or K indicates that insufficient nutrients are supplied to achieve optimum plant growth and that nutrients are being withdrawn from soil organic or mineral stores.

A second measure of potential mismanagement is the accumulated residual amount of P in soils (Zou, Zhang and Davidson, 2022). The oversupply of P through time causes it to accumulate in the soils, where it can be transported to waterbodies and cause degradation of water quality. The undersupply of P causes a decrease in soil P stores and is an example of nutrient mining.

Bahrain and Kuwait have a substantial oversupply of N, P and K and the United Arab Emirates has an oversupply of N and P (Figure 11.8 and Figure 11.9). All three countries also have high residual P levels in their cropland soils. In all three countries, the area of cropland is small (5 000 ha in Bahrain, 14 000 ha in Kuwait and 90 000 ha in the United Arab Republic). Egypt has a high oversupply of N but a small undersupply of P and a large undersupply of K, indicating very unbalanced nutrient inputs.

The undersupply of nutrients is a serious issue in the Syrian Arab Republic, where all three macronutrients have negative nutrient balances (Figure 11.8, Figure 11.9 and Figure 11.10). Yemen also has low positive N balances and inputs and undersupply of P and K, possibly due to both countries having experienced political instability and military conflicts since 2015.

Oman, Qatar and Saudi Arabia have high positive N balances (i.e. between 50 and 85 kg N/ha (FAO, 2024b) but balances for the other two nutrients are close to zero. Algeria, the Islamic Republic of Iran, Iraq, Morocco and the Sudan show mining of soil P reserves in the past but current balances of N, P and K are close to zero or positive. Jordan, Lebanon, Libya, Morocco, the Sudan and Tunisia have N, P and K balances close to zero, but levels of inputs overall are low in Libya, Morocco, the Sudan and Tunisia.

The reasons for the widely differing levels of nutrient supply are often country-specific and are explored in the following section.

Gulf states

In Saudi Arabia, wheat, barley, fruits, vegetables, and fodders are the most common agricultural products, with agricultural sustainability depending on socioeconomic and climatic conditions (Tirth *et al.*, 2021).

With a mean annual average rainfall of around 100 mm (Hussain, Alquwaizany and Al-Zarah, 2010), agriculture in the Arabian Gulf continues to use fast-depleting ground water resources for irrigation to meet the ever-increasing demand for food and fodder. Water stress, especially under arid conditions, is one of the key factors limiting food production (Hanson, Bali and Sanden, 2008; Al Naeem, 2008). Achieving higher yields is important for profitable agricultural production, which in turn removes large quantities of nutrients from the soil. Hence an adequate supply of nutrients is essential for obtaining profitable yields and to maintain high cereal grain quality (Bernardi *et al.*, 2013). The yield of cereals has been reported to increase significantly after P and K fertilizer application (Berg *et al.*, 2007). There has been an increase in the land area cultivated for cereals in Saudia Arabia, which in turn has led to a significant increase in the use of mineral fertilizers. In Saudi Arabia, in 2020, 87 percent of the P inputs and 84 percent of the N inputs were from fertilizer (FAO, 2024a), which is higher than the average for other NENA countries.

Deficiencies of micronutrients are a problem in crop production on coarse-textured (sandy) calcareous soils, common in the agricultural areas of arid regions (Prasad, Bashour and Al-Shanghitti, 1984). From several thousand samples analysed from Saudi Arabia, it was found that soils generally had a coarse to medium texture, low organic matter, a pH between 7.5 and 8.5, and were deficient in Fe and Zn (Prasad, Bashour and Al-Shanghitti, 1984)**.**

Soils in Kuwait, Oman and the United Arab Emirates are also mostly sandy and lack sufficient organic matter and fertility to sustain productivity. The sandy soils lack soil structure to support holding of water and nutrients to sustain vegetation growth (Alshankiti and Gili, 2016).

Iraq

There is a crucial need for nutrient application in Iraq, which should be considered through adopting the principles of fertilizer applications and integrating different kinds of fertilizers from different sources to ensure a fair nutrient supply as required.

The shortage of good quality fertilizers, especially macronutrients, is affecting crop production. Due to availability issues, while the Iraqi Ministry of Agriculture is able to subsidize farmers with a certain amount of chemical fertilizers (either from the Ministry of Industry or from importation), most farmers mainly depend on local markets to meet their fertilizer demands. Urea for N, and triple superphosphate for P are produced locally but in insufficient quantities, especially in recent years and diammonium phosphate and K fertilizers must be imported. However, the private sector is active in chemical fertilizer investment.

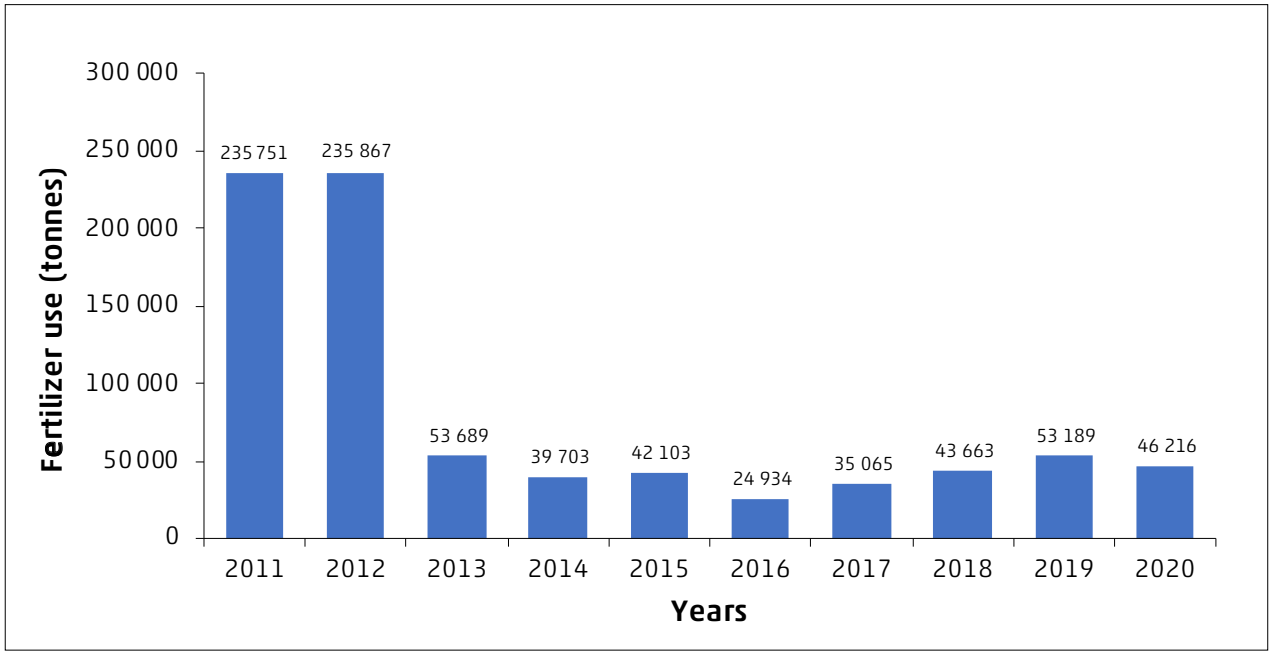
Arid and semi-arid lands, especially alkaline soils with high content of soil carbonates – which constitute over 90 percent of arable land in Iraq – have a deficiency in micronutrients. Because of this, the Iraqi Ministry of Industry prepared and distributed a foliar feed containing Fe, Zn, Mn and Cu to farmers to apply to wheat crops with good results (Salman and Ali, 2016).

Syrian Arab Republic

The unrest in many of the NENA countries has impacted on the availability of mineral fertilizers for farmers due to restricted fertilizer flow in the international market, and the increase in prices. Libya, the Sudan, the Syrian Arab Republic and Yemen, and are typical examples, with very low usage of N, P, and K fertilizers (14.7 kg/ha for Libya, 3.9 kg/ha for the Syrian Arab Republic, 7.1 kg/ha for the

Sudan and 10.6 kg/ha for Yemen) (Figure 11.15). While the Syrian Arab Republic was more than self-sufficient in wheat production and food security prior to the civil war, the country needed to import extra cereal grain for human and animal feed during the period of the war. For example, in 2007 the Syrian Arab Republic imported 44 569 tonnes of wheat and exported 999 151 tonnes; imports rose to 340 391 tonnes in 2012 (the year after the civil war began) and exports fell to 3 tonnes; and by 2023 imports were 71 297 tonnes and exports were 48 142 tonnes (FAO, 2025). Fertilizer use in the Syrian Arab Republic dropped from 235 751 tonnes in 2011 to 46 216 tonnes in 2020 (Figure 11.15), which has affected land productivity and the quality of food produced.

Figure 11.15. Fertilizers used over the 2011–2020 period in the Syrian Arab Republic



◊ *Source:* Authors' own elaboration based on **MOAAR (Ministry of Agriculture and Agrarian Reform).** 2020. Statistical Report (2020). In: *MOAAR*. Damascus. [Cited 18 June 2024]. https://moaar.gov.sy/category/statistical-group-2022

While many countries in the NENA region have attempted to map their country's soil fertility to establish SSM practices and highlight the factors causing soil fertility degradation, most maps are restricted to small areas of each country and no regional maps are available. For example, the General Commission for Scientific Agricultural Research (GCSAR) in the Syrian Arab Republic produced a map of the most important soil fertility indicators on the outskirts of Damascus (GCSAR, 2023), showing that most fertility problems in the area were related to a very high $CaCO_3$ content in the soil. Such a map highlights the necessity and importance of preparing soil fertility maps at the national level as an effective tool and the first step in promoting the optimal use of fertilizers. A great amount of work is required by all NENA countries to map their soil fertility traits on a national scale.

The Islamic Republic of Iran

Only 12 percent of the land in the Islamic Republic of Iran is suitable for cultivation and deficiencies in one or more nutrients are common (Karimian and Tehrani, 2018). The challenges facing soil fertility and plant nutrition in the Islamic Republic of Iran include low SOC and nutrient content, imbalanced plant nutrition, low nutrient efficiency, ineffective soil fertility systems environmental stresses, non-efficient fertilizer application, and inadequate knowledge transfer to users.

Given the conditions, recommending appropriate fertilizers is a multifaceted and complex task but is a prerequisite to the integration of soil fertility and plant nutrition management. It requires the

development of a framework that incorporates all the factors involved into a single management package that lays out the correct application of an optimal assortment of chemical, organic, and biological sources of nutrients, that are suitable to the country's environmental and local conditions (Moshiri *et al.*, 2022).

An analysis (OC, available P and soil pH) of the soil fertility status of land growing wheat in the Shadkam plain of Fars province, was carried out by Azadi, Zareian and Shakeri (2023), from 210 topsoil samples. It was determined that 4 130 ha (10.8 percent of the area) had a very high fertility class, 7 615 ha (20 percent) had a high fertility class, 23 942 ha (62.8 percent) had a moderate fertility class, and 2 452 ha, (6.4 percent) had a low fertility class. The mapping was accurate for predicting soil fertility in this study area.

Agricultural lands in the northwest of the Islamic Republic of Iran have a high pH, low SOM, low total N, and low available P, with micronutrient deficiencies being the main limiting factors for soil fertility of agricultural lands (Aqdam *et al.*, 2023). Therefore, to improve soil fertility in the area, there is a need to improve the condition of macro- and micronutrients, especially total N.

Mesgaran *et al.* (2017) evaluated the capacity of the Islamic Republic of Iran's land for sustainable crop production based on soil properties, topography, and climate conditions, and using high-resolution spatial data sets (Figure 11.16). The Islamic Republic of Iran's land suitability for cropping was classified as very good (0.4 percent or 0.6 million ha), good (2.2 percent or 3.6 million ha), medium (7.9 percent or 12.8 million ha), poor (11.4 percent or 18.5 million ha), very poor (6.3 percent or 10.2 million ha), unsuitable (60 percent or 97.4 million ha), and excluded areas (11.9 percent or 19.3 million ha). In addition to the limitations caused by low precipitation, low SOC, steep slopes, and high soil sodium (Na) content were the predominant soil and terrain factors limiting the agricultural land suitability (Figure 11.16). About 50 percent of the country's existing croplands are located in low-quality lands, representing- an unsustainable practice. There is little possibility for cropland expansion to increase production but the redistribution of cropland to more suitable areas may improve sustainability and reduce pressure on water resources and ecosystems.

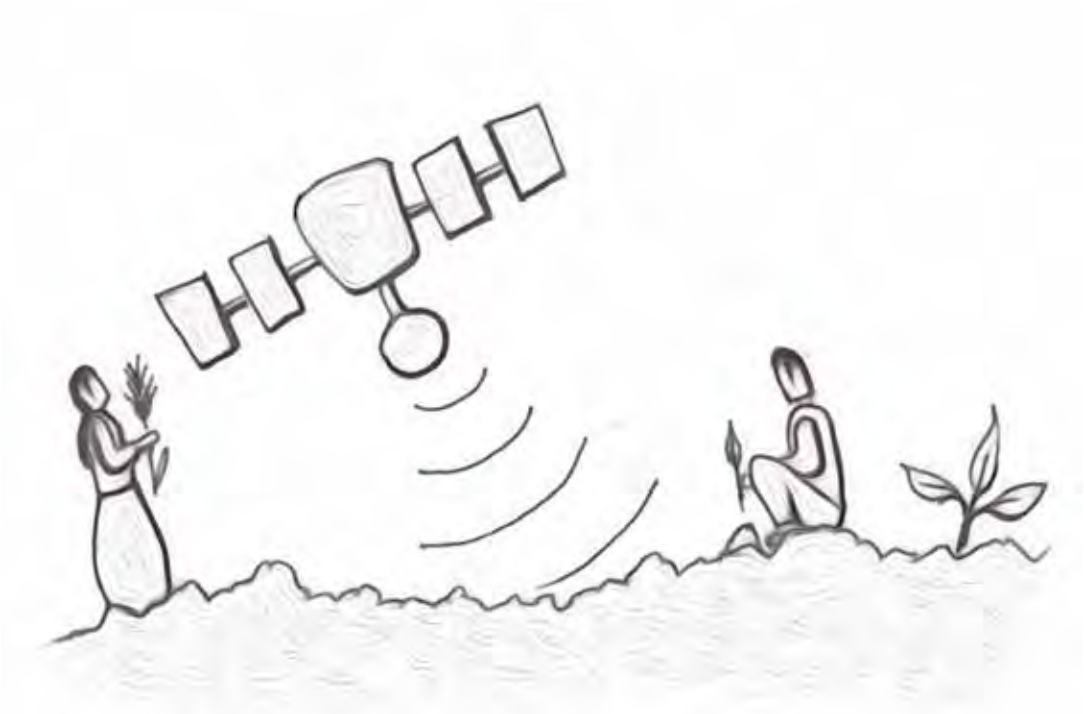
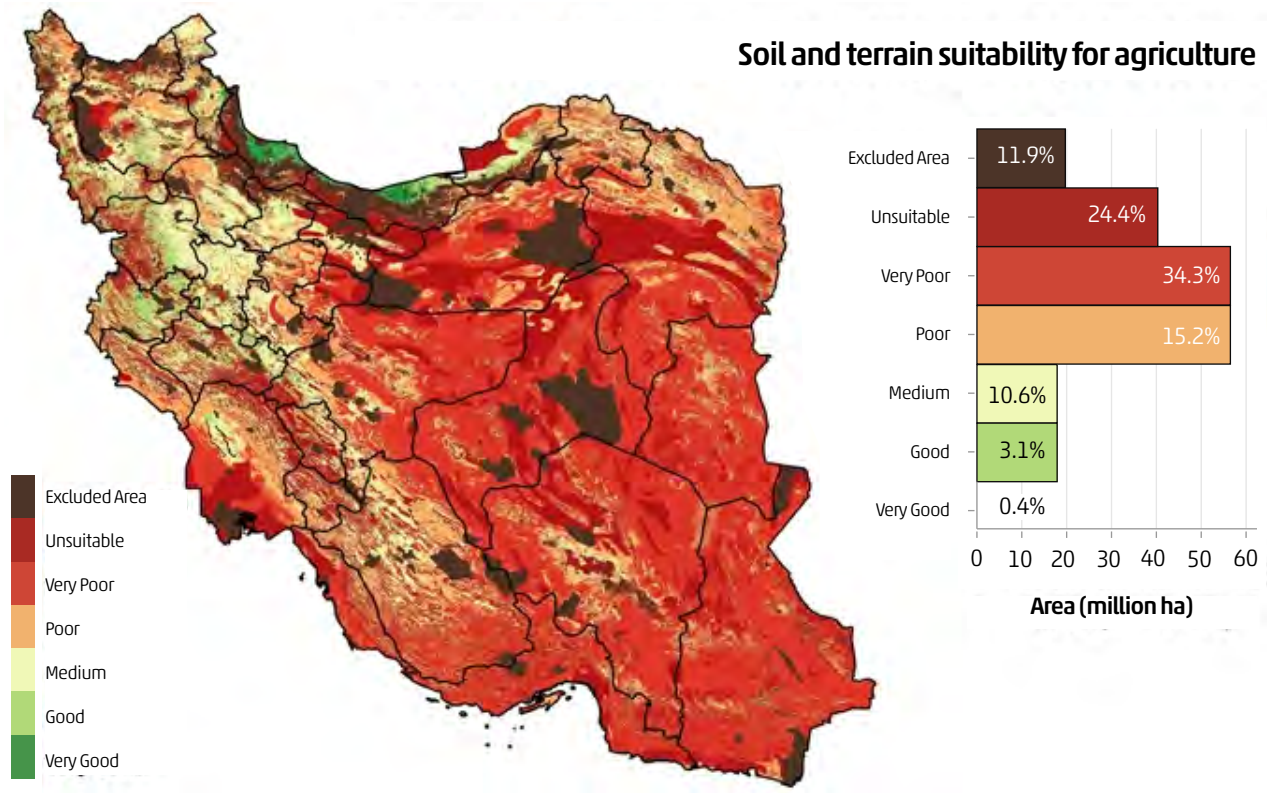
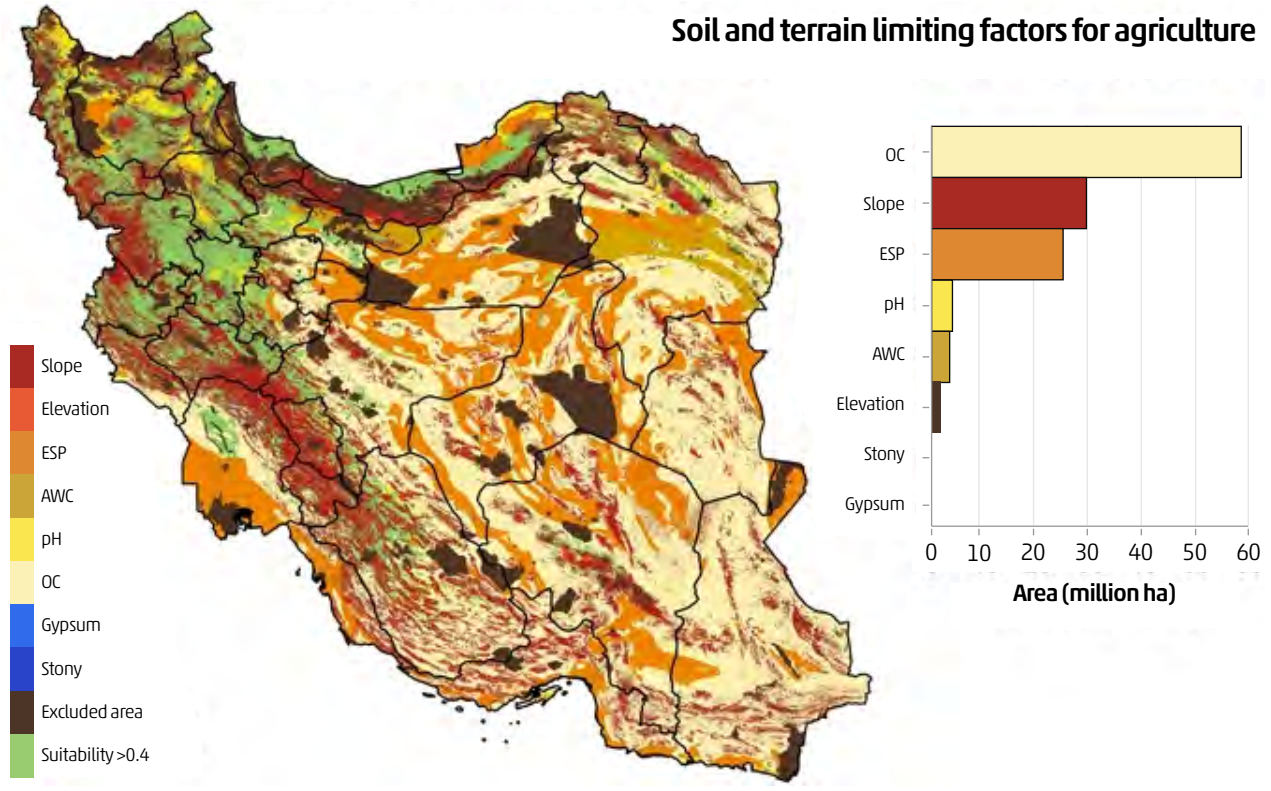


Figure 11.16. Land suitability for agriculture and soil limiting factors





Refer to the disclaimer on page ii for the names and boundaries used in this map

◊ *Source:* **Mesgaran, M.B., Madani, K., Hashemi, H. & Azadi, P.** 2017. Iran's Land Suitability for Agriculture. *Scientific Reports*, 7(1): 7670. https://doi.org/10.1038/s41598-017-08066-y

North Africa

Fertilizer use in the North Africa region varies between countries. Among the seven countries involved in the North Africa International Plant Nutrition Institute (IPNI) programme (Gharous and Boulal, 2016), Egypt was the biggest consumer of fertilizers (an average of 500 kg/ha). The fertilizer use in other countries only covered between 30 and 50 percent of the real need (an average of 50 kg/ha), with the quantities used also generally well below the recommended rates. Thus, fertilizer inputs need to be increased to reverse the current trends of low crop productivity and land fertility degradation, and contribute to the goal of the respective national agricultural strategies.

Soil and soil fertility research have played a role in the improvement of fertilizer use. Initial efforts focused on the identification of nutrient constraints in the field, mainly N and P (Gharous and Boulal, 2016). A major effort involved the establishment of a soil test calibration programme network for North Africa and West Asia in 1985 involving the International Center for Agricultural Research in the Dry Areas (ICARDA), and the National Institute for Agricultural Research (INRA), to provide a rational basis for fertilizer use in the field and to stimulate awareness of the potential value of soil testing as a basis for fertilizer recommendations (Gharous and Boulal**,** 2016). In the context of climate change, cereals represent potential crops for increasing crop diversity, agricultural productivity, and farmers' income in semi-arid regions (Taaime *et al.*, 2023).

The area of arable land in Morocco is estimated at 8.4 million ha, which represents approximately 20 percent of the total area of the country, with more than 80 percent being rainfed agriculture. Seventy percent of farms are smallholdings (less than 5 ha) and frequently in fragmented parcels (El Gharous and Boulal, 2016).

In Morocco the recorded yields between 1990 and 2022 of four crops (maize, barley, sorghum and wheat) have been highly variable (Epule *et al.*, 2023). Wheat had the highest yield followed by barley, maize and sorghum. In Morocco fertilizers and nutrients are generally dominated by N fertilizers, followed by phosphate and K.

Evaluation of the effects of applied fertilizer nutrients in Morocco showed a linear relationship between maize yields and fertilizers (Epule *et al.*, 2020). The study showed that only a 0.14 percent increase in yield was related to N fertilizer, while a 7 percent increase was due to phosphate, and a 4.1 percent increase was due to P. Phosphate fertilizers tend to explain more of the changes in maize yield (R^2 of 7 percent), a statistic that is relatively higher than those recorded for N and potash. A similar trend was observed for barley, wheat and sorghum cereals, in which phosphate fertilizers outperformed N and K fertilizers (Epule *et al.*, 2020).

Recommendations for nutrient management

The following three steps are suggested to achieve an integrated management system:

1. The participation of all stakeholders is needed in the design, prioritization, implementation, and monitoring processes.
2. The integrated use of chemical, organic and biofertilizers need to be well adapted to the nutritional requirements of the plant varieties grown in the established cropping system in terms of the amounts, timing and method of application (with due consideration of environmental stresses).
3. Local knowledge and know-how of the integrated management system needs to be disseminated with the help of local extensionists and volunteer farmers from a pilot farm for the replication and achievement of the expected outcomes in other farming sites.

11.3.5 Salt accumulation in the Near East and North Africa region

Soil salinization is a global issue, particularly in regions with arid and semi-arid climates. Over 1 381 million ha of soils are affected by salinity and sodicity worldwide, of which about 67 percent are affected by human activities (FAO, 2024c). In irrigated systems, farming practices are the main contributors to soil salinity and sodicity increases, with about 14 km^2 of the world's irrigated lands being lost daily due to salinization (Singh, 2022). The extent of salinity development in terrestrial environments varies primarily due to differences in rainfall, temperature, and soil drainage capacity.

Soil salinity may be defined as the presence of dissolved salts in the soil to the extent that plant growth and performance are impacted. Due to plant physiological characteristics and salt tolerance, the impact of soil salinity and sodicity can be different for different plants (Saadat *et al.*, 2018). In other words, in one place and for a certain plant, a certain level of salinity can be limiting, while the same level of salinity is not limiting for another plant or in another place. In the NENA region, soil salinity threatens the sustainability of irrigated agriculture and water management and impacts food production. In 2020, according to the United Nations Environment Programme (UNEP), the annual economic losses caused by salinity in the NENA region were estimated to be around USD 1 billion. This translates to a range between USD 1 600 and USD 2 750 per ha of affected lands (UNEP, 2020).

Estimates of the area of saline soils within NENA counties vary widely, due in part to different thresholds of electrical conductivity (EC) used to identify saline soils (Table 11.4). The most consistent estimate for the region is from the global map of salinity from Hassani, Azapagic and Shokri (2020) using machine learning techniques and a comprehensive set of soil, climate, topographic and remote sensing data. The percentage of saline soils were estimated from 1980 to 2018, along with the mean values of the areas with salinity greater than 4 dS/m (a common threshold for saline soils).

Table 11.4. Estimates of area of salt-affected soils in Near East and North Africa countries

Country	References
Bahrain	
0.44 million ha have EC values ≥4 dS/m.	Hassani, Azapagic and Shokri (2020)
Iran (Islamic Republic of)	
88.3 million ha have EC values ≥4 dS/m	Hassani, Azapagic and Shokri (2020)
Iraq	
30.5 million ha have EC values ≥4 dS/m.	Hassani, Azapagic and Shokri (2020)
Jordan	
4.2 million ha have EC values ≥4 dS/m.	Hassani, Azapagic and Shokri (2020)
Kuwait	
209 000 ha are affected by salinity.	Burezq, Shaid and Baron (2021)
1.5 million ha have EC values ≥4 ds/m.	Hassani, Azapagic and Shokri (2020)
Lebanon	
0.038 million ha have EC values ≥4 ds/m.	Hassani, Azapagic and Shokri (2020)
Oman	
10.5 million ha have EC values ≥4 ds/m.	Hassani, Azapagic and Shokri (2020)

Country	References
Qatar	
1 million ha have EC values ≥4 ds/m.	Hassani, Azapagic and Shokri (2020)
Saudi Arabia	
68.2 million ha have EC values ≥4 ds/m.	Hassani, Azapagic and Shokri (2020)
Syrian Arab Republic	
11.7 million ha have EC values ≥4 ds/m.	Hassani, Azapagic and Shokri (2020)
United Arab Emirates	
3 million ha have EC values ≥4 ds/m.	Hassani, Azapagic and Shokri (2020)
Yemen	
6.5 million ha have EC values ≥4 ds/m.	Hassani, Azapagic and Shokri (2020)
Algeria	
63.9 million ha have EC values ≥4 ds/m.	Hassani, Azapagic and Shokri (2020)
Egypt	
Saline, saline sodic and sodic soils have a strong presence in the Nile Delta and represent an average of 37% of the total cultivated soils. The north Delta contains the highest area of saline and saline-sodic soils, reaching 46%.	Mohamed (2016)
6.7 million ha have EC values ≥4 ds/m.	Hassani, Azapagic and Shokri (2020)
About 33% of cultivated land is already salinized. Almost 30–40% of irrigated farmlands are saline. 60% of the northern cultivated land and 20% of the middle and southern Delta regions are saline. In the Nile Valley (Upper Egypt), saline soils account for about 25% of the cultivated areas.	Choukr-Allah (2021)
Libya	
21.5 million ha have EC values ≥4 ds/m.	Hassani, Azapagic and Shokri (2020)
Morocco	
85.2 million ha have EC values ≥4 ds/m.	Hassani, Azapagic and Shokri (2020)
Approximately 30% of the irrigated area is saline and average yield losses may be as high as 50%. It has been reported that more than 37 000 ha of irrigated land is already affected by salinization.	Choukr-Allah (2021)
Sudan	
237 million ha have EC values ≥4 ds/m.	Hassani, Azapagic and Shokri (2020)
Tunisia	
2.1 million ha have EC values ≥4 ds/m.	Hassani, Azapagic and Shokri (2020)
Approximately 1.5 million ha, or roughly 10% of the country's area is affected by different degrees of salinity and about 25% of water resources have a salt concentration exceeding 3 g/L. As a result, about 50% of the total irrigated area is considered to be at high risk of salinization.	Choukr-Allah (2021)

◊ *Sources*: **Choukr-Allah, R.** 2021. Use and Management of Saline Water for Irrigation in the Near East and North Africa (NENA) Region. In: K. Negacz, P. Vellinga, E. Barratt-Lennard, R. Choukr-Allah & T. Elzenga, eds. *Future of Sustainable Agriculture in Saline Environments*, pp. 51–70. Abingdon, UK, Taylor & Francis Group.

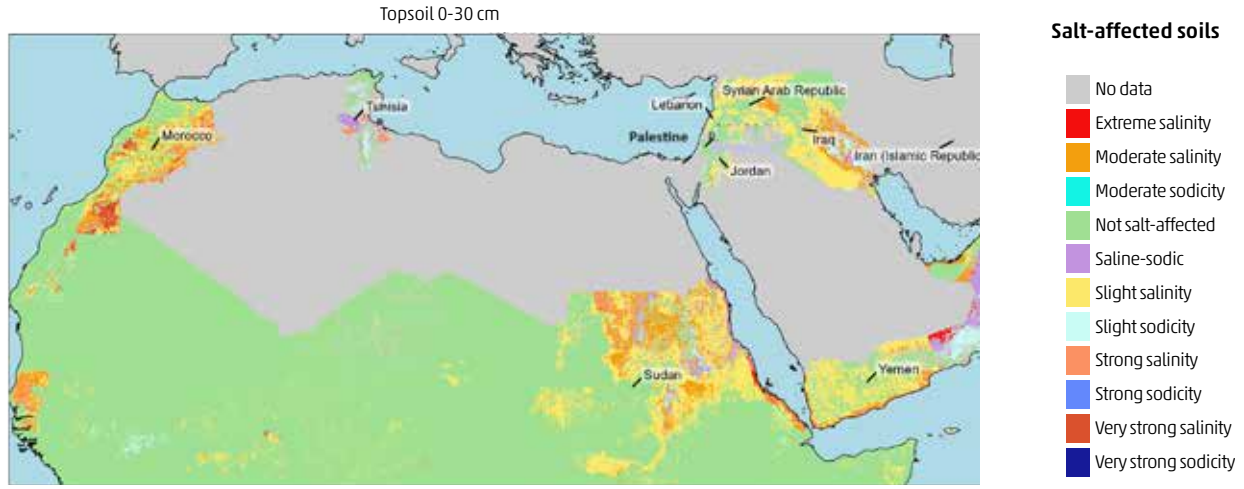
◊ **Burezq, H., Shahid, S.A. & Baron, H.J.** 2021. Salts in the terrestrial environment of Kuwait and proposed management. Conference presentation at the Global symposium on salt-affected soils. Halt soil salinization, boost soil productivity, 20–22 Oct 2021. Rome, FAO. https://www.fao.org/fileadmin/user_upload/GSP/GSAS21/day3/PS3_03_Burezq.pdf

◊ **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 (PNAS)*, 117(52): 33017–33027. https://doi.org/10.1073/pnas.2013771117

◊ **Mohamed, N.N.** 2016. Management of salt-affected soils in the Nile Delta. In: A.M. Negm, ed. *The Nile Delta*. The Handbook of Environmental Chemistry, 55, pp: 265–295. Cham, Switzerland, Springer.

Among the Near East countries, only Lebanon, with only 0.4 percent of its land mass occupied by saline soils, is largely free of salinity issues (Hassani, Azapagic and Shokri, 2020) (Figure 11.17 and Figure 11.18). Yemen and Palestine have 10 to 20 percent of their land area occupied by saline soils, followed by 20 to 30 percent of Saudi Arabia, 30 to 40 percent of Oman and the United Arab Emirates, 40 to 50 percent of the Islamic Republic of Iran, Jordan, and the Syrian Arab Republic, and 50 to 60 percent of Bahrain and Iraq. Kuwait (with 82 percent of its land area occupied by saline soils) and Qatar (with 87 percent) have the highest area of saline soils.

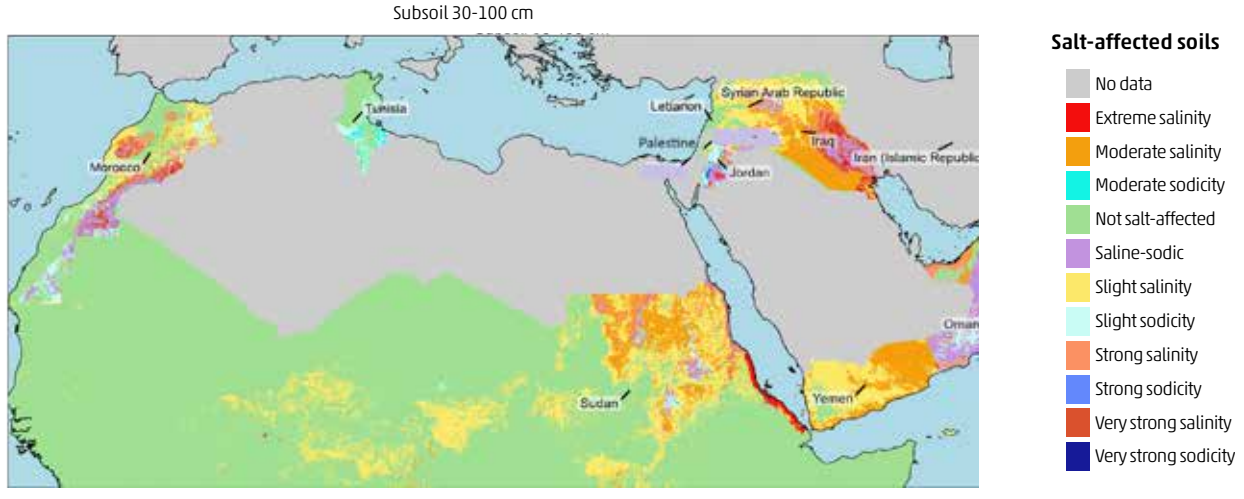
Figure 11.17. Salt-affected soils in the Near East and North Africa region at 0–30 cm depth (topsoil)



Refer to the disclaimer on page ii for the names and boundaries used in this map.

Source: **FAO**. 2024. *Regional action plan for sustainable soil management in the Near East and North Africa region*. Rome. https://doi.org/10.4060/cd2814en

Figure 11.18. Salt-affected soils in the Near East and North Africa region at 30–100 cm depth (subsoil)



Refer to the disclaimer on page ii for the names and boundaries used in this map.

Source: **FAO**. 2024. *Regional action plan for sustainable soil management in the Near East and North Africa region*. Rome. https://doi.org/10.4060/cd2814en

The countries of North Africa have lower estimates of saline land (Table 11.4) Figure 11.17 and Figure 11.18). Algeria has the highest percentage of its land area occupied by saline soils (26.3 percent) followed by Morocco, Tunisia, and Libya (between 10 and 20 percent) and Egypt and the Sudan (less than 10 percent).

Hassani, Azapagic and Shokri (2020) also estimated the change in saline area across the NENA region between 1980 and 2018. Yemen had the highest annual increases in saline area of 1.2 percent. The rate of increase was also high for the Islamic Republic of Iran (0.49 percent), Egypt and Palestine

(both 0.55 percent), and Saudi Arabia and the Syrian Arab Republic (both 0.39 percent). Increases in Iraq and Jordan were between 11 and 15 percent, while close to zero or non-significant for Algeria, Bahrain, Kuwait, Lebanon, Libya, Oman, and Qatar. Annual decreases in the area of saline land were estimated for Tunisia (−0.26 percent), Morocco (−0.39 percent) and the Sudan (−0.98 percent).

Estimates of the impact of salinity on agricultural and irrigated land are important for the sustainability of agricultural production. Estimates of the percentage of agricultural land affected by salinity are high in many countries, ranging from 30 to 35 percent in the Islamic Republic of Iran and the Nile Delta soils in Egypt, to 46 percent in Jordan and 80 percent in the United Arab Emirates. The percentage of irrigated land in North Africa that is salt-affected is also estimated to be high to very high, with 20 to 30 percent in Algeria and Morocco, 30 to 40 percent in the Syrian Arab Republic and Egypt and up to 50 percent in Tunisia.

The map of soil resources and potential of the Islamic Republic of Iran (Banai, 2001), indicated that the areas with soils affected by salinity covered 55.6 million ha, mostly located in the central plateau of Khuzestan province, and the southern plains. Of the country's agricultural lands, 6.8 million ha had soils with varying degrees of salinity, of which about 4.3 million ha were derived from a parent rock of andesite . About 1.1 million ha (27.5 percent) had a salinity of 4 to 16 dS/m, limiting the yield of most crops. More than 2.4 million ha (57 percent) had a salinity of 16 to 32 dS/m, preventing all but a few salt-resistant varieties from being grown. More than 660 000 ha (15.5 percent) had a salinity of more than 32 dS/m, and were considered uncultivable lands with no economic justification for their reclamation (Banai, 2001). More recently Mesgaran *et al*. (2017) (Figure 11.16) estimated that of the total cropping area in the Republic of Iran (about 24.6 million ha), 8 percent had a land suitability class of very poor and 17 percent was unsuitable. The combined effects of high sodicity (shown as ESP in Figure 11.16) and salinity were a major cause of the poor ratings.

So far, no general study has been undertaken on the geographical spread of secondary (anthropogenic) salinity and the contribution of geological and natural salinity in the Islamic Republic of Iran. The available data is also scarce about the amount of reduction in the performance of native agricultural products of the country due to salinity and the area of abandoned agricultural land due to salinity and desertification (Moameni, 2010).

According to Banai (2001), about 44.5 million ha of Iranian agricultural lands are at risk of salinization due to the country's arid and semi-arid climate and improper agricultural and horticultural management.

According to recent studies and investigations, the area of agricultural land in the Islamic Republic of Iran is about 25.1 million ha (Saadat *et al*., 2023). Of this, about 10.5 million ha face various restrictions, mainly salinity (EC >2 dS/m). About 8.2 million ha (33 percent) of agricultural lands have a salinity of 2 to 16 dS/m, which limits the yield of most agricultural products. More than 1.3 million ha (5.5 percent) have a salinity of 16 to 32 dS/m, and 0.9 million ha (3.7 percent) have a salinity more than 32 dS/m.

11.3.6 Soil pollution in the Near East and North Africa region

Assessment of soil pollution in agricultural land

Research on soil pollution in the NENA region has primarily focused on metal and metal-like elements (Table 11.5) (Rodriguez Eugenio, McLaughlin and Pennock, 2018).

There are no regional studies of soil pollution in agricultural soils in the NENA region. However, studies of subregions of countries or specific sites have demonstrated hotspots of pollution. In some cases, the causes of the hotspots are related to the high salts content in parent rocks but in others are attributable to anthropogenic activities. Common sources of pollution in agricultural soils are the excessive application of conventional and organic fertilizers, pesticides, animal manure and sewage sludge, proximity to urban areas or industrial zones, and contamination from military activities.

Table 11.5. List of chemical elements associated with soil pollution and their main effects on health

Element	Symbol	Main effects
Manganese	Mn	Readily taken up by plants but low risk to humans
Zinc	Zn	Readily taken up by plants but low risk to humans
Copper	Cu	Readily taken up by plants but low risk to humans
Nickel	Ni	Readily taken up by plants but low risk to humans
Cadmium	Cd	Significant risks to human and animal health, and most toxic element (with lead) for humans
Lead	Pb	Significant risks to human and animal health and most toxic element (with Cd) for humans
Chromium	Cr	Low uptake risk by plants
Vanadium	V	Associated with petroleum processing and use
Selenium	Se	High risk for food chain contamination and poses risks to human and animal health
Cobalt	Co	High risk for food chain contamination and poses risks to human and animal health
Zirconium	Zr	Low uptake risk by plants
Rubidium	Rb	Low human health risk
Arsenic	As	High risk for food chain contamination and poses risks to human and animal health
Titanium	Ti	Low uptake risk by plants
Beryllium	Be	Low human health risk
Tungsten	W	Limited environmental impact
Mercury	Hg	Toxic to animals and humans

◊ *Source:* **Rodríguez-Eugenio, N., McLaughlin, M. & Pennock, D.** 2018. *Soil Pollution: a hidden reality.* Rome, FAO. https://openknowledge.fao.org/server/api/core/bitstreams/3f7e6959-db0b-44d3-971e-109bcfe78195/content

Studies from the Islamic Republic of Iran have shown widely differing levels of soil pollution by trace elements (Table 11.5). The total values and distribution of Fe, Mn, Zn, Cu, and Ni in relation to lithology, topography and geopedological processes showed a direct dependency on the parent rock (i.e. geogenic origin) with the enrichment of Fe, Cu and Ni in the middle soil horizons and Zn and Mn concentration within the surface horizons (Rezapour, Golmohammad and Ramezanpour, 2014). The background levels of trace elements in the calcareous soils of the Hamedan province are at tolerable levels (Beygi and Jalali, 2017). However, the testing of 41 soil samples from Tehran city showed heavy contamination with Cd (Fazeli *et al.*, 2018). A clear anthropogenic origin of trace elements was shown, notably Cd, Pb and Zn, likely resulting from the excessive application of conventional and organic fertilizers, pesticides, animal manure and sewage sludge.

An analysis of 53 soil samples from the agricultural soils of Kermanshah province showed greater concentrations of Zn (81 percent), Cu (98 percent), Ni (100 percent), Cr (98 percent) and Mn (79 percent) than the reported world background values, with high enrichment factors and high geoaccumulation index for Ni and Cu in agricultural soils, indicating considerable pollution (Ahmadi, Karami and Afyuni, 2019). Similar results from the assessment of the spatial distribution and pollution level of trace elements in agricultural soils of Aghili plain, Khuzestan province showed low to moderate degree of contamination with a probable ecological risk factor (unitless) for the biological communities exceeding 65, resulting from the excessive application of fertilizers, herbicides and pesticides (Ahmadi, Karami and Afyuni, 2019).

A review of 39 studies that covered 46 agricultural sites across 17 provinces in the west and north of the Islamic Republic of Iran showed a moderate and high integrated pollution index of 45.5 percent for Cd, 16.7 percent for Pb and 13 percent for Cu (Jalali, Antoniadis and Nafaji, 2021). A more comprehensive study undertaken in 2022 collected 247 Iranian studies and papers and analysed 21 400 soil samples for trace element contamination (except for Pb). It showed a low pollution index (<1) for agricultural and urban soils, with moderate contamination by Cr and severe contamination by As and Ni for the soils of industrial zones (Shahbazi, Fathi-Gerdelidani and Marzi, 2022). The levels

of Zn, Cu, Co, Mn, Se, and Mo in the coastal soils of southwest parts of the country were in the low-risk contamination range (Hamid *et al.*, 2022). The concentration and spatial distribution of toxic elements (As, Cd, Cu, Ni, Pb and Zn) in surface soils of the northwest of the country showed similar distribution for Cd, Cu, Pb, Ni, and Zn, indicating the same source. With respect to the degree of contamination, the soil samples from the north, northwest and northeast of the Islamic Republic of Iran were classified as having a very strong contamination index (Mohammadi *et al.*, 2023).

In Iraq, the assessment of soil quality in the 1970s and 1980s from 21 000 soil samples representing 70 percent of the country were similar to reported world range values, except some Cr, Ni and V outliers (Al-Bassam and Yousif, 2014). Subsequent studies in Iraq of specific locations showed higher levels of contamination. The environmental assessment of the sediments of Al-Hammar marsh in southern Iraq showed minimal contamination by Zn, V, uranium (U), Cr, As, Fe and Mn and high contamination by S, Mg, chlorine (Cl), Ca, P, bromine (Br), Se, Mo, Ni, Co, Cu and strontium (Sr) (Abdullah and Al-Tawash, 2015). An environmental assessment of agricultural soils of North Baiji City, Iraq, showed a moderate enrichment of Zn, silver (Ag), Co, Cd, Sb, Sr and Cr, and a significant enrichment of As and Ni (Abed, Altawash and Ali, 2015).

Contamination of agricultural soils has also been noted in other NENA countries. Studies of the spatial distribution of trace elements in the Quaternary cultivated soils of Lebanon indicated Cr and Ni accumulation in the soil that exceeded 100 mg/kg, with the geoaccumulation index showing moderate to strong contamination with Pb and moderate hazard from Ni, Cu, Cd and Cr (Darwish, Hamze and Nsouli, 2021). Accumulation and pollution load indices in a densely cultivated area of Al-Qassim province in Saudi Arabia, indicated extremely high Cd, significant Pb and moderate Zn enrichment, and a high ecological risk associated with moderate contamination by Pb with a minimal enrichment by other trace metals (Al-Wabel *et al.*, 2017).

In some locations, contamination in agricultural soils is associated with proximity to river systems. In Iraq, the sediments of the Euphrates River were found to be polluted by Ni (≤199 mg/km) and Sr (≤543 mg/km), moderately polluted by Cu (≤47.9 mg/km), Co (≤22.8 mg/km), Cr (≤111 mg/km) and Se (≤1.1 mg/km) with permissible levels of As (≤10.2 mg/km), Cd (≤0.29 mg/km), Pb (≤14.8 mg/km) and Zn (≤91 mg/km) (Issa and Qanbar, 2016). The degree of contamination by trace elements and pollution level index in agricultural soils located in the El Tarf region in Algeria showed an increasing trend with the direction of the Bounamoussa River flow, with a high and unsafe level of Fe and Pb (Zaoui and Benselhoub, 2020). The assessment of the quality of arable soils in the Hama floodplain, in the northwest of the Syrian Arab Republic showed a strong and very strong geoaccumulation index values for Cd, Pb and Cu in different soil layers reaching 4.22 (very strong accumulation), with a concentration of Cu and Cd of 127 mg/kg and 9.8 mg/kg respectively, with potential health and environmental risks (Husein, Kalkha and Bäumier, 2019). High levels of V, Ni, Cr, Zn, Cr and Pb in agricultural soils from El-Gharbia Governorate (Shokr *et al.*, 2016) and Quessna district, in the southwest of the Nile Delta in Egypt were also reported (Khalifa and Gad, 2018).

Irrigation waters can also be a source of pollution. The distribution and migration of trace elements by irrigation with treated wastewater showed slight enrichment of Cr in the upper soil layer, with declined stock of Zn and Cu in the calcareous soil of Elhajeb-Sfax, Tunisia, against the accumulation of elements in the Fluvisol subsoils, promoted by the type of applied irrigation technique (Belaid *et al.*, 2019).

Another source of contamination comes from the proximity to urban areas and industrial zones. A recent determination of trace element concentrations in the soil of the Al-Qadisiyah Governorate in the south-central area of Iraq showed heavy pollution from Cd and Ni in the various layers of the soils, which was attributed to proximity to a major drain. The high levels of Cd, Ni, Pb, and Zn were also attributed to industries with fossil fuel combustion, and other anthropogenic sources including agricultural nutrients, soil conditioners and sludge, which caused a baseline level of contamination with a partial decline in soil quality (Al-Khuzaie and Abdul Maulud, 2022).

Soil contamination from conflicts

The conflicts in the NENA region since 2015 have also left a legacy of soil pollution. The concentration of 15 trace elements in the urban soils of Al-Anbar province, Iraq, showed the presence of unusual toxic metals (Rb, Sr and Zr) and high levels of As, Cd, Cr, Cu, Ni, Pb, Rb, Sr, V and Zr that are usually only found in trace amounts in the natural environment (the higher levels are probably explained by military actions) (Al-Sudani *et al.*, 2021). Topsoil sampling and analysis from 66 sites located in marginal agricultural land or peri-urban settings from the northwest of the Syrian Arab Republic showed concentrations of Ni of 133.30 ± 72.12 mg/kg and Cr of 122.14 ± 52.25 mg/kg, which exceed the norms for agricultural lands (Alhasan *et al.*, 2023). The content of Cd, As and Co in Sarmada, North Aleppo in the Syrian Arab Republic exceeded European thresholds, explained by the impact of intense conflict as well as the effects of informal settlement, unregulated industry and irrigation using untreated wastewater.

A soil remediation project in northern and southern Kuwait started in 2021 within the framework of the Kuwait Environmental Remediation Program (KERP) to treat the formed oil lakes after the first Gulf War (Al-Qallaf, Al-Baroud and Al-Kandari, 2020; Al-Mebayedh, Niu and Lin, 2023). This soil remediation programme was put in place by the UN to treat the consequences of a substantial oil spill and the related environmental disaster. The programme goals will be to clean and remediate 26 million m^3 of contaminated soil from a surface area of 114 km^2, with consequent revegetation of the affected area (Al-Qallaf, Al-Baroud and Al-Kandari, 2020).

Soil contamination from industrial activities and traffic

Industrial activities and traffic generate several sources of contamination, including air, water and soil pollution.

Near East

There are many examples of localized soil pollution related to industrial activity. Mapping of soil pollution in the vicinity of the Bama Pb-Zn mining area and the Esfahan and Mobarakeh iron and steel plants of the Islamic Republic of Iran determined the geogenic origin of Ni, Co, Cr, Fe and Al and the anthropogenic origin of Pb, Zn and Cd (Esmaeili *et al.*, 2014). Assessing the impact of land use on soil contamination in Golestan province, in the Islamic Republic of Iran showed higher trace element accumulation in soil surrounding industrial plants than in soils through agricultural and natural land uses. There were higher soil enrichment factors for Se and As, although no serious effects on the environment were recorded (Ghorbani, Moghads and Kashi, 2018). Assessment of soil pollution levels, geoaccumulation and the ecological risk of trace elements in agricultural soils around the Nakhlak Pb-Zn mine, confirmed moderate to heavy contamination with Pb and Zn and the absence to moderate contamination with Cd and Cu (Baghaie and Aghili, 2019). Enrichment factors indicated moderate contamination with Mn, Ni, and Cu, significant contamination with Cd and Zn, and high enrichment with Pb providing a moderate level of potential ecological risk.

Since the start of the twenty-first century, soil contamination has been increasing in Iraq, with trace elements reported in sediments and soils (e.g. Cr in soils reaching 570 mg/kg) (Muslim, Al-Mayyahi and Al-Ghasham, 2022). A 2013 screening of a large number of trace elements in 27 soil samples from 6 sites in Erbil city, Iraq, showed soils to be within the normal values by international standards, despite some permissible levels being exceeded, with Co, Mn and Pb reaching critical levels in industrial zones (Mohammed, Kettanah, and Smai, 2013). Mine clearance and disposal through controlled explosions in the Halgurd-Sakran National Park caused moderate to high soil enrichment with Ni, Cr, Mn, Co and Cu through large parts of Iraq (Hamad, Balzter and Kolo, 2019). Contamination attributed to urban activities has impacted the agricultural soils of Baghdad, with

Zn at 133 mg/kg, Co at 31 mg/kg and Pb at 154 mg/kg, and an oil refinery in Kurkuk, Iraq raising the level of soil Cd to 12.6 mg/kg.

The release of phosphogypsum waste during transport and incorrect storage of raw materials contaminated soil over a 2 km area in Lebanon's coastal zone, surrounding phosphate plants. Localized contamination of radionuclides of the uranium decay chain (^{238}U, ^{235}U, ^{232}Th, ^{226}Ra, ^{210}Po, ^{210}Pb, ^{40}K and ^{137}Cs) were measured (El Samad *et al.*, 2014).

Assessments of the hazards of trace elements associated with the use of phosphate fertilizers produced in the Eastern Mediterranean showed an accumulation of Zn, Pb, Cd, Cu, Sb, Ag, Pd, Nb, and Mo that linearly or strongly correlated with the level of P_2O_5 in the fertilizers used (Azzi *et al.*, 2017). Exogenous Pb radionuclides accumulated on the soil surface in areas surrounding a phosphate production plant in north Lebanon (Saba *et al.*, 2019). Assessing the radioactivity of some Lebanese soils showed an accumulation of ^{238}U (a mean of 37 Bq/kg), ^{232}Th (24 Bq/kg), ^{226}Ra (46 Bq/kg), ^{40}K (206 Bq/kg), and ^{137}Cs (23 Bq/kg) denoting a permissible level for all elements except ^{226}Ra which was higher than the world average (El Samad, Baydoun and Abdallah, 2018).

Studies on the origin and toxicity level of Fe, Mn, Zn, Cu, Ni, Cd, Cr, Co, and Pb in soil samples of the coast of Saudi Arabia affected by industrial activities, indicated the anthropic origin of Zn, Cu, Cr, and Pb but with the lowest adverse biological effects, despite a heavy to moderate risk of contamination by Cr (Almasoud, Usman and Al-Farraj, 2015). Trace element characterization and spatial distributions in 68 surface and subsurface soil samples from 34 locations around the Al-Jalamid phosphate mine, Saudi Arabia, showed levels of U and Th close to the world average. It also demonstrated moderate contamination by As, hafnium (Hf) and Sb with a contamination factor >3 indicating an increasing pollution load index in the soil surface and a permissible load in the subsoil (El-Taher *et al.*, 2016).

Analysis of trace elements in sand samples and sediments near a disposal site for reject brine from the Alkhobar desalination plant, Saudi Arabia, reveled higher levels of Fe, Mn, Cr, Cu, As, Sr, and Zr than was reported by other studies on sediments and soils of the Arabian Gulf, with the highest contamination factor for As (Alshahri, 2017). Trace metal accumulation (Cd, Mo, europium [Eu], Hf, terbium [Tb], and ytterbium [Yb]) in the soils of Ras Tanura city, close to the oldest and largest oil refineries located in eastern Saudi Arabia, exceeded the permissible level and the world average, causing pollution, notably by Cd, with a mean value of 39.9 mg/kg, 113 times greater than the background level (Alshahri and El-Taher, 2018).

In the industrial city of Adra, in the Syrian Arab Republic, a cement company and the surrounding area had concentrations of Pb and Cd that exceeded the permissible limits of 214.73 mg/kg for Pb and 1.55 mg/kg for Cd in subsoil, but allowable values in the rest of the area (Al-Thalji and Nazem, 2022).

In the United Arab Emirates, decreasing concentrations of trace metals (Cd, Zn, Pb, and Mn) were reported near agricultural lands and forests with increases in the areas surrounding industries and gas stations, derived from fossil fuel and traffic, with Cd and Hg posing ecological risks (Al-Taani, Nazzal and Howari, 2019). Evaluation of soil pollution by industrial activities in Omdurman city, in the Sudan indicated a low content of Ni, Zn, Co, As and Pb with concentrations exceeding background values for Cr, Cu, As and Cd (Ali and Ateeg, 2015).

North Africa

Near the Cairo-Alexandria agricultural highway, high levels of Pb, Ni, Co and Cd were reported in surface soils (0–15 cm) and subsurface soils (15–40 cm) in industrial plants, with trace element concentrations, particularly Pb, exceeding the permissible limits (Hashim *et al.*, 2017). A principal component analysis of the contamination factor index showed that the soils of the Gerga district, Egypt, had been affected by industry, urban and agricultural activities with zero to moderate

contamination with trace elements (Co, Ni and Pb) of anthropogenic origin (Said, Salman and Elnazer, 2019).

Analyses of soil samples collected from Abu-Kammash (Libya) revealed pollution by Ti, Be, W, P, and V due to exposure to petrochemical wastewater, which can accumulate in the soil–food chain (Banana *et al.*, 2019). Checking the Libyan soils for trace element contamination using the geoaccumulation index, pollution load and overall metal pollution indices revealed a range from unpolluted to extremely and severely contaminated soils with Ni, Cr, Cu and Pb, caused by industry and traffic from surrounding cities notably in Benghazi, Al-Marj, Tripoli, Misurata and Khomes (Aishah and Elssaidi, 2019).

The surface sediments of the central coastal area of Gabes Gulf, Tunisia had increasing levels of trace elements, notably Zn, Cd, Cr, Pb, Cu and Hg, partly derived from the Tunisian Chemical Group (TCG) producing phosphogypsum and other industrial wastes (El Zrelli *et al.*, 2015).

Population exposure to contamination

Near East

Some work has been undertaken assessing risks to human health from a variety of food products across the region. Although still within the permissible limits, the concentration of Hg and Se was significantly higher in cow milk coming from traditional farms in the Islamic Republic of Iran, whereas a higher content of Ni was detected in milk coming from industrial farms in the country (Arianejad *et al.*, 2015). Testing 96 surface soil samples from Ahvaz, the capital of Khuzestan province, in the Islamic Republic of Iran, showed considerable contamination by Pb, Zn, Cu, Cd, As, and Cr, with the highest enrichment factor for Pb and Cu, and a high potential ecological risk from Pb but with neglected carcinogenic risks for children and adults and higher exposure of children (Ghanavati, Nazarpour and De Vivo, 2019). Similarly, a safe health limit of the possible exposure risks to Pb and Cd by the consumption of raw and pasteurized milk using the target hazard quotient (THQ) for adults and children was recently reported in the east of the Islamic Republic of Iran (Madani-Tonekaboni *et al.*, 2019) and in Tabriz (Beikzadeh *et al.*, 2019). The low content of Pb and Cd in eggs suggested low risk from the consumption of eggs by the average Iranian (Hoseini *et al.*, 2023).

Health risk from trace element bioavailability in Kermanshah Province in the west of the Islamic Republic of Iran was tested using 121 agricultural soil samples and revealed the highest trace element enrichment and ecological risk was in an area considered as an agricultural hub that exported food crops, with the highest bioavailability of Cd, followed by Cu and As (Rastegari *et al.*, 2021). Although a higher concentration of Cd, As and Pb was detected in mares' milk in Yazd, the content of Pb, Cd and As in the milk was proven to be safe and lower than the permissible limit (Alipour *et al.*, 2023). Mapping of a large area for trace element contamination and potential link with stomach cancer was undertaken by overlying soil pollution maps and patients' distribution in the country, and suggested that the concentrations of As and Fe could be considered as stomach cancer risk factors (Mohammadnezhad *et al.*, 2023).

The concentrations and bioavailability of As, Co, Cr, Cu, Mn, Ni, Pb, V, and Zn in 38 agricultural land and playground soils from an urban area of Baghdad, Iraq, showed lower cancer risk through inhalation and ingestion than the threshold level of concern for children and adults but with a 97 percent chance for children and 90 percent for adults to have hazardous indices from the cumulative effect of these elements (Hamad *et al.*, 2014).

The content of As, Zn, Ni and Pb was analysed in some soils of Alexandria and Al-Musayyib cities in the Babylon Governorate, Iraq, along with alfalfa plant samples. While the single pollution index of Ni was the highest in soils (1.29–1.32), the ecological risk index for soil As was dominant (12.30–14.10)

with the highest Zn enrichment factor for leaves (1.76–1.82) and the largest biological concentration factor of 0.32–0.36 (Wotaify, 2019). Testing 174 soils and 26 vegetable and fruit samples from 15 areas in and around Sulaymaniyah and Halabja cities (the Kurdistan region of Iraq) for potentially toxic trace elements, showed agricultural soil contamination near waste disposal sites with a level of accumulation in vegetables of Ni, As and Cd that posed health risks to children (Hawrami *et al.*, 2020).

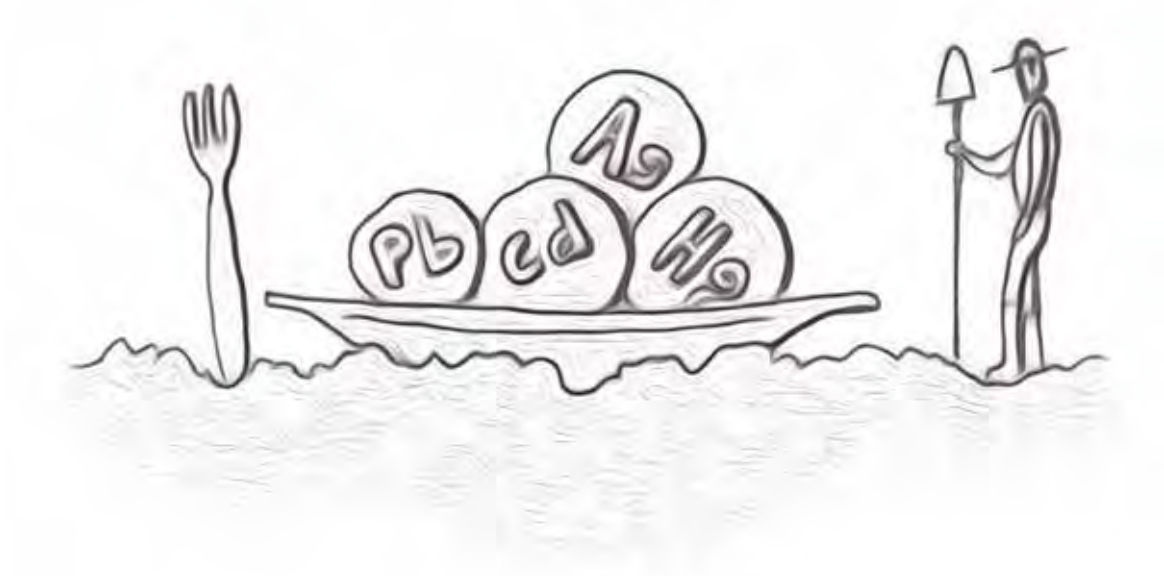
Population exposure to Pb, Zn, Cu, Cr, Mn, Ni and Fe in a Jordanian industrial zone showed an increased risk of exposure with high probability of cancer in children and adults (Al-Shaweesh *et al.*, 2019). An analysis of 950 samples from different Lebanese water, soil and food sources showed a permissible level of ^{137}Cs in milk with higher ^{40}K activity in food samples exceeding values reported from non-Mediterranean countries, along with the detection of ^{131}iodine (I) in grass samples (El Samad *et al.*, 2017).

Population toxicological exposure in Lebanon to trace elements from the consumption of pitta showed a high exposure of local residents to As, Pb and Co – including six to nine-year-old children – that exceeded the toxicological reference values (Lebbos *et al.*, 2018). The survey showed high levels of Ni (1 292 µg/kg), As (400 µg/kg) and Pb (260 µg/kg) in white bread and showed an excess of the tolerable daily intake for the Lebanese population (Lebbos *et al.*, 2019). Beside the large emission of NOx in the atmosphere, the Beirut marine port explosion in 2021 of 2 750 tonnes of Nitropril™ resulted in the complete disappearance of 9.5 tonnes of Class I insecticides with the resulting potential human and environmental risks (Hajj *et al.*, 2021).

Analysing water, soil and anuran livers from Al-Hayr, Riyadh, Saudi Arabia showed there to be good environmental quality with no risk from pollution and bioaccumulation of trace elements in the soil–food chain (Hasayen *et al.*, 2017).

North Africa

While values of trace elements in the soils of the Nile Delta of Egypt showed low contamination levels of Ni, Zn and As (Badawy *et al.*, 2017), the ecological risk of trace elements in El Sadat City, pointed to a high As enrichment factor (5–10) (Badawy *et al.*, 2016). Trace element pollution (Pb, Cd, As and Hg) of soil and water derived from industry and municipal wastes in northern Egypt was linked to the frequency of liver cancer, supported by the high level of these metals in the blood of observed patients (Elwakil *et al.*, 2017). A high concentration of Pb, Cd and Al was detected in milk and dairy products from the city of Alexandria, Egypt (Mesgaran *et al.,* 2019). A recent assessment of trace element dietary intake by milk and processed products in Mansoua, Egypt showed the maximum limit of Pb and Cd to be exceeded by between 44 percent and 68 percent of dairy products. However, there were no potential health risks from the consumption of such dairy products (Elafifi *et al.*, 2023).



Conclusions on soil pollution

A common trend in recent studies assessing soil contamination in the NENA region is the reported increased concentration of Ni and Cr, mainly attributed to a geogenic origin, pointing to more frequent soil contamination with Pb, Cd, As, Zn and Mn caused by anthropogenic activities. Unusual toxic trace elements such as Rb, Sr and Zr were reported due to armed conflicts. In addition, new trends in the assessment of soil contamination and the implications have been observed in the NENA region:

- More studies are focusing on the assessment of soil contamination at local level, applying more dense soil sampling and analyses using more sophisticated equipment and tools including remote sensing.
- Additional indices are extensively used related to geogenic accumulation, bioaccumulation, pollution load index and contamination.
- Public exposure to dietary intake of trace elements continues to attract researchers, focusing on food safety and human exposure from edible food derived from urban, peri-urban and coastal farming systems, and the aquatic environment.
- More studies are focusing on urban soil pollution hazards and the health conditions of peri-urban agricultural soils.
- While attention to the risks from industrial wastes remains a major focus, new worldwide attempts are being made to develop the use of industrial waste as compost to decontaminate polluted soils, improve agricultural production in wheat-based rotation systems and improve above and below ground ecosystem services (Pradhan *et al.*, 2023).
- There is additional attention being paid to addressing population exposure to radioactivity levels.



11.4 Trends in sustainable soil management

11.4.1 Summary of state of and trend in sustainable soil management

The information presented in this chapter was used by the ITPS members of this region to assess the state and trend of threats to soil functions through a management-based approach. Instead of directly measuring or modelling the physical severity of each soil threat, as was done in the 2015 SWSR report (FAO and ITPS, 2015), the current report assesses the level of adoption of SSM practices known to mitigate each soil threat.

The state reflects the effectiveness of management practices in place as of 2015 in preventing or reducing a specific soil threat. At the regional or subregional scale, changes in management occur at decadal or longer time scales and hence the state as of 2015 is relevant for the 2015–2024 assessment period.

The trend indicates the direction of change in the adoption of SSM practices between 2015 and the finalization of this report (late 2024). In short, trend reflects the how SSM adoption has progressed over time.

Grassland, sparse vegetation cover and rainfed agriculture each make up about 5 percent of the NENA region, while irrigated crops make up less than 1 percent of the total area. Thus, the NENA region is becoming increasingly dependent on food imports, due to demographic pressure, rapid urbanization, water scarcity and climate change (FAO and ITPS, 2015).

Soil erosion by water and wind remains the main threat to soil and the largest cause of degradation of agricultural lands in the NENA area, with erosion also causing the loss of fertility (Figure 11.19 and Table 11.6). Water erosion in NENA countries is more evident in mountainous regions than in the coastal regions. As previously mentioned, rates of soil erosion are high, often exceeding 100 t/ha annually in many western mountainous areas of Algeria, the Islamic Republic of Iran, Lebanon, Morocco, the Syrian Arab Republic, and Tunisia. In Egypt, erosion losses are even higher in some specific areas, with values ranging between 200 and 550 t/ha annually.

Wind erosion is an environmental concern associated with arid and semi-arid climates, limited vegetation, and the vast desert landscapes of the NENA region. Wind erosion results in the loss of fertile topsoil, reduced agricultural yields and increased airborne dust particles that endanger local population health. The susceptibility of land to wind erosion can be assessed using CE, which is a measure of factors such as surface soil dryness and the occurrence of erosion wind events. The highest CE values were documented in Algeria, Mauritania, Morocco and the Sudan, indicating an elevated susceptibility to wind erosion in these countries.

Nutrient mismanagement issues differ among NENA countries (Figure 11.19). In countries with small agricultural land areas like Kuwait, Qatar and the United Arab Emirates, there is an overload on fertilizer application with a high positive annual N balance (i.e. between 50 and 85 kg N/ha), and P is accumulated in the soil. Egypt has a very high oversupply of N but a small undersupply of P and a large undersupply of K, indicating very unbalanced nutrient inputs. Deficiencies of micronutrients are a serious problem in crop production on coarse-textured, calcareous soils of arid regions. Calcareous soils represent a large portion of all the agricultural soils in the region and are commonly deficient in Fe and Zn.

Figure 11.19. Assessment of state and trend of threats to soil functions in the Near East and North Africa through a management-based approach



Note. See Table 11.8 for details.

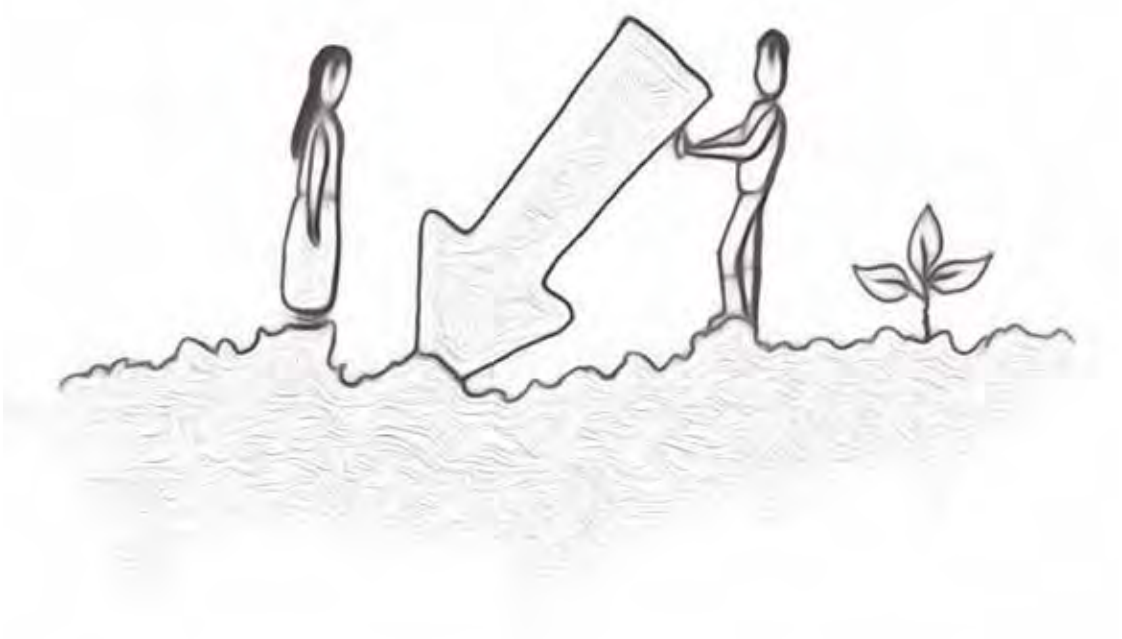


Table 11.6. Summary of threats to soil functions in the Near East and North Africa and the confidence in the assessment of their state and trend

Threat	Summary	Confidence	
		State	Trend
Soil erosion	Water erosion is severe in the mountains of coastal regions throughout the region, in inland wadis and in croplands. Estimates of the land area with annual soil loss greater than 10 t/ha are >15 percent for the Islamic Republic of Iran, Jordan, Lebanon, Saudi Arabia, the Syrian Arab Republic, and Tunisia. Wind erosion is high in approximately 70 percent of the NENA region (especially southern areas of North Africa) with low risk only in the coastal regions.	Medium	Medium
Nutrient mismanagement	There is excess nutrient application of N and P in Kuwait, Qatar and the United Arab Emirates, and N in Egypt. There is significant nutrient mining due (in part) to conflicts in the Sudan, the Syrian Arab Republic and Yemen.	High	High
Soil organic carbon loss	There are losses due to wind and water erosion and substantial expansion of urban lands, with decomposition of SOC being high under Mediterranean climates. There is very limited adoption of reduced tillage systems.	Medium	Medium
Soil biodiversity loss	Soil biodiversity threats are classed as very high in some areas of North Africa, specifically the coastal areas from Casablanca to Benghazi and the Nile Delta. Desertification is a major threat to soil biodiversity. In the Near East, the threats to soil biodiversity are classed as very high along the Mediterranean coastline of Lebanon and the Syrian Arab Republic, as well as large areas of Iraq and the western parts of the Islamic Republic of Iran.	High	High
Salt accumulation	There is a high proportion of salinized land in NENA. The loss of vegetation cover is linked to changes in salinity in global modelling. There are increases in irrigated land since 2015 in most NENA countries (especially the Near East) which increase risks due to improper water use.	Medium	Medium
Soil pollution	Hotspots of pollution are increasingly found resulting from either agricultural mismanagement practices, armed conflicts, oil refineries and cement companies. In conflict areas, unusual toxic metals (Rb, Sr and Zr) and high levels of other heavy metals are more evident.	Medium	Medium
Urbanization and soil sealing	The region is already highly urbanized and high continuing growth in urban population puts surrounding land at risk.	Medium	Medium

Note. The threats are presented in the order of most severe threat (erosion) to least severe (urbanization and soil sealing).

The unrest experienced in many of the NENA countries has had a significant effect on the availability of mineral fertilizers for farmers. This was the result of restricted fertilizer flow in the international market, as well as increased prices. Libya, the Sudan, the Syrian Arab Republic and Yemen are typical examples of low fertilizer usage (N, P and K) per hectare of cropland. The Syrian Arab Republic was more than self-sufficient in wheat production and food security prior to the start of the civil war in 2011, but now is known to import extra cereal grain for human and animal feed. The use of fertilizers in the Syrian Arab Republic in 2020 (46 216 tonnes) was about one-fifth of the levels in 2011 (235 751 tonnes) (FAO, 2024b).

NENA countries are in semi-arid and arid areas with a dominance of dryland soils that have low levels of organic matter and poor soil quality. Although the NENA region covers 18 percent of the Earth's surface, it is estimated to have only 4.1 percent of the global SOC stock. Most of the countries of the NENA region have moderate to relatively low total C stocks due to the dominance of soil types associated with dry conditions, such as Xerosols, Arenosols and Lithosols.

There has been a continuous decline in the SOC in agricultural land since 2015, due to the decline in livestock and open animal husbandry, and the resulting decline in manure inputs to the soil. Weather variability, drought and depleting vegetation are dominant causes for SOM decline. Under Mediterranean conditions, the succession of dry summers and wet winters increases the emission of CO_2 by promoting the decomposition of SOC. Soil erosion and urban expansion have also led to irreversible loss of soil cover, SOC and other nutrients.

Most countries in the NENA region reported that despite the relevance of soil biodiversity to the country, very little has been done to enhance it, due to a lack of resources. In the global map of potential threats to soil biodiversity, threats to soil biodiversity are classed as very high in some areas of North Africa, specifically the coastal areas from Casablanca (Morocco) to Tunisia and around urban areas such as Tripoli and Benghazi (Libya) and the Nile Delta in Egypt. In the Near East, the threats to soil biodiversity are classed as very high along the Mediterranean coastline of Lebanon and the Syrian Arab Republic as well as large areas of Iraq and the west of the Islamic Republic of Iran. Many of threatened areas are judged to be at very high risk of desertification, and this seems to be the main threat to soil biodiversity. Generally, the desert areas of NENA are judged to be of a low or very low risk to soil biodiversity due to minimal human impact.

Salt accumulation is still a major threat to agriculture in the NENA region. Estimates of the impact of salt accumulation on agricultural and irrigated land are important for the sustainability of agricultural production. Estimates of the percentage of agricultural land affected by salinity are high in many countries, ranging from 30 to 35 percent in the Islamic Republic of Iran and the Nile Delta soils in Egypt, to 46 percent of cultivated land in Jordan and 80 percent in the United Arab Emirates. The percentage of irrigated land in North Africa that is salt-affected is also estimated to be high to very high, with 20 to 30 percent in Algeria and Morocco, 30 to 40 percent in the Syrian Arab Republic and Egypt and up to 50 percent in Tunisia. According to figures from UNEP (2020) cited previously, the economic losses caused by salinity alone in the region are estimated to be around USD 1 billion per year. This translates to a range of USD 1 600 to USD 2 750 per ha of affected lands.

There are no regional studies of soil pollution in agricultural soils in the NENA region. Studies of subregions of countries or specific sites have demonstrated hotspots of pollution. Contaminated sites are mainly the result of human activity that include the excessive use of fertilizers and sewage sludge, irrigation with treated wastewater, armed conflicts, exposure to petrochemical wastewater, industrial wastes and discharges, and the disposal of rejected brine from water desalinization plants.

11.4.2 Priorities for sustainable soil management

The priorities for SSM are as follows:

- The NENA region is subject to soil erosion from water or wind, which poses a major threat to agricultural land. It is necessary to increase land cover in rural areas by planting, according to specific ecological and socio-economic contexts, trees, shrubs, grasses and deep-rooting perennial plants that could limit the effects of a dry climate or water runoff.
- Soil organic matter levels and soil biodiversity could be improved by increasing the input of organic C from different sources such as animal manure, farmyard composting and organic by-products of agricultural industries.
- Improving the utilization of organic matter sources requires the education of farmers on how to compost all organic farm residue.
- Farmers need to be encouraged to follow an appropriate sustainable nutrient management regime using organic and inorganic fertilization based on soil monitoring data. There should be a full fertility map for each NENA country with different layers that include all parameters that define soil fertility.

- Fertilizers should be made available to farmers at reasonable prices, especially in areas of unrest.
- Soil contamination with heavy metals is not a widespread problem in the NENA region but is a serious problem in hotspots. Monitoring concentrations of heavy metals should be a priority.
- Soil salinity is a persistent problem, and actions should be taken to limit the spread of salinity through the implementation of integrated management systems.



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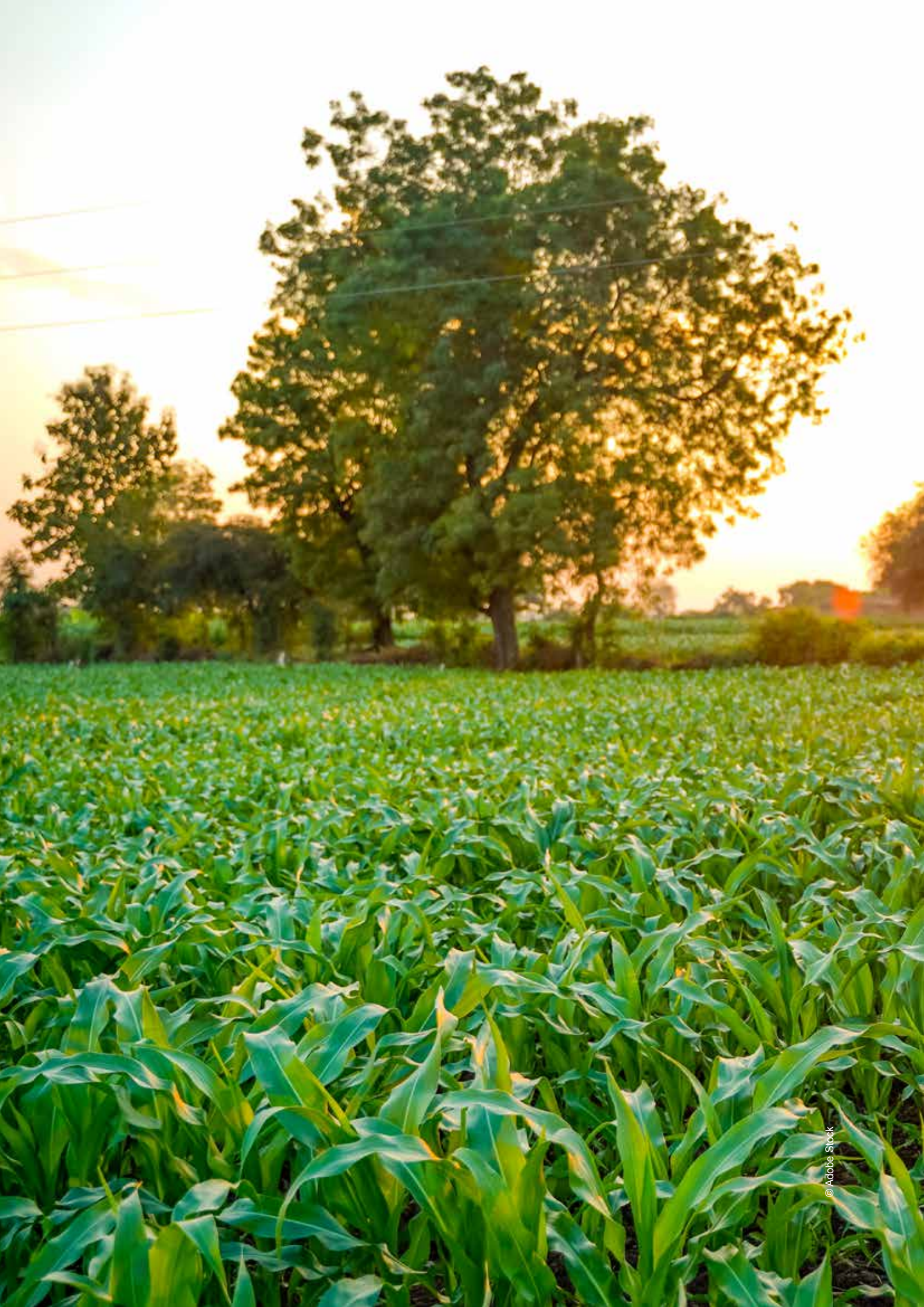
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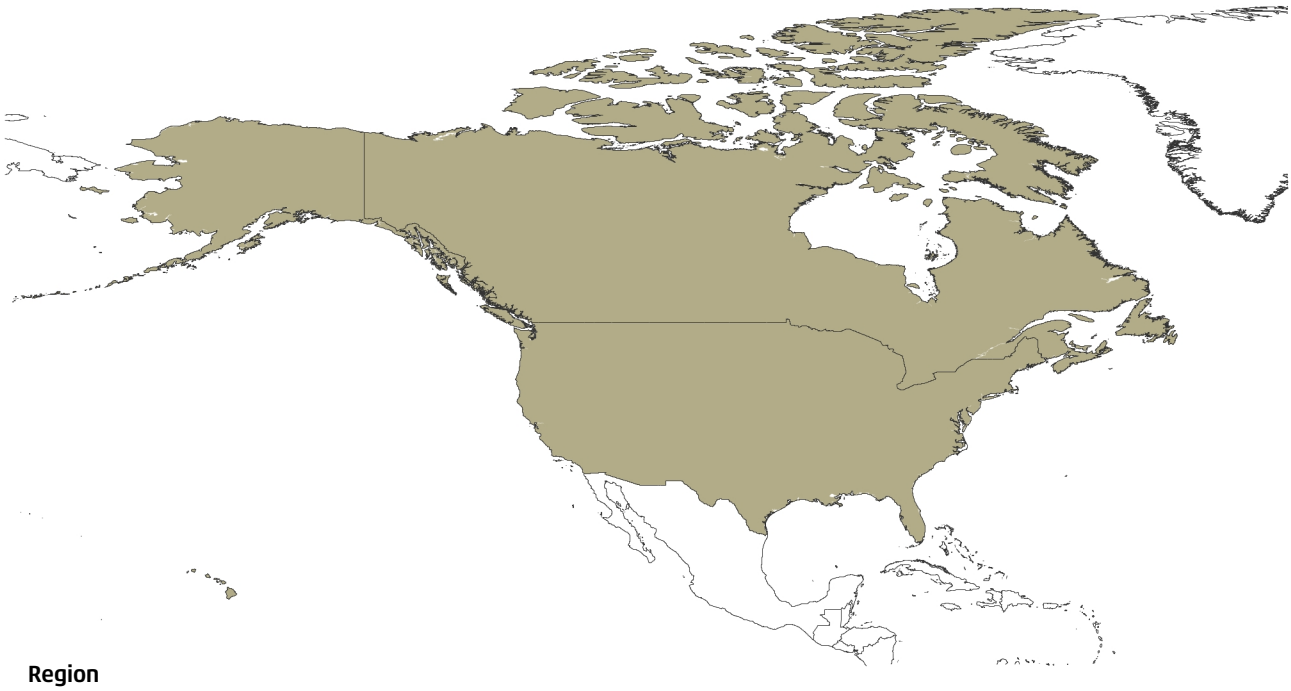
Chapter 12 | Regional assessment for North America region

12.1 Regional overview

12.1.1 Characteristics of the region

The North America region comprises Canada and the United States of America (Figure 12.1). For some soil threats this chapter also includes information from northern Mexico although Mexico is formally included in the Latin America and the Caribbean regions. The northern Mexico information was included given the similarity of this region with the southwestern area of the United States.

Figure 12.1. Map of the North America region

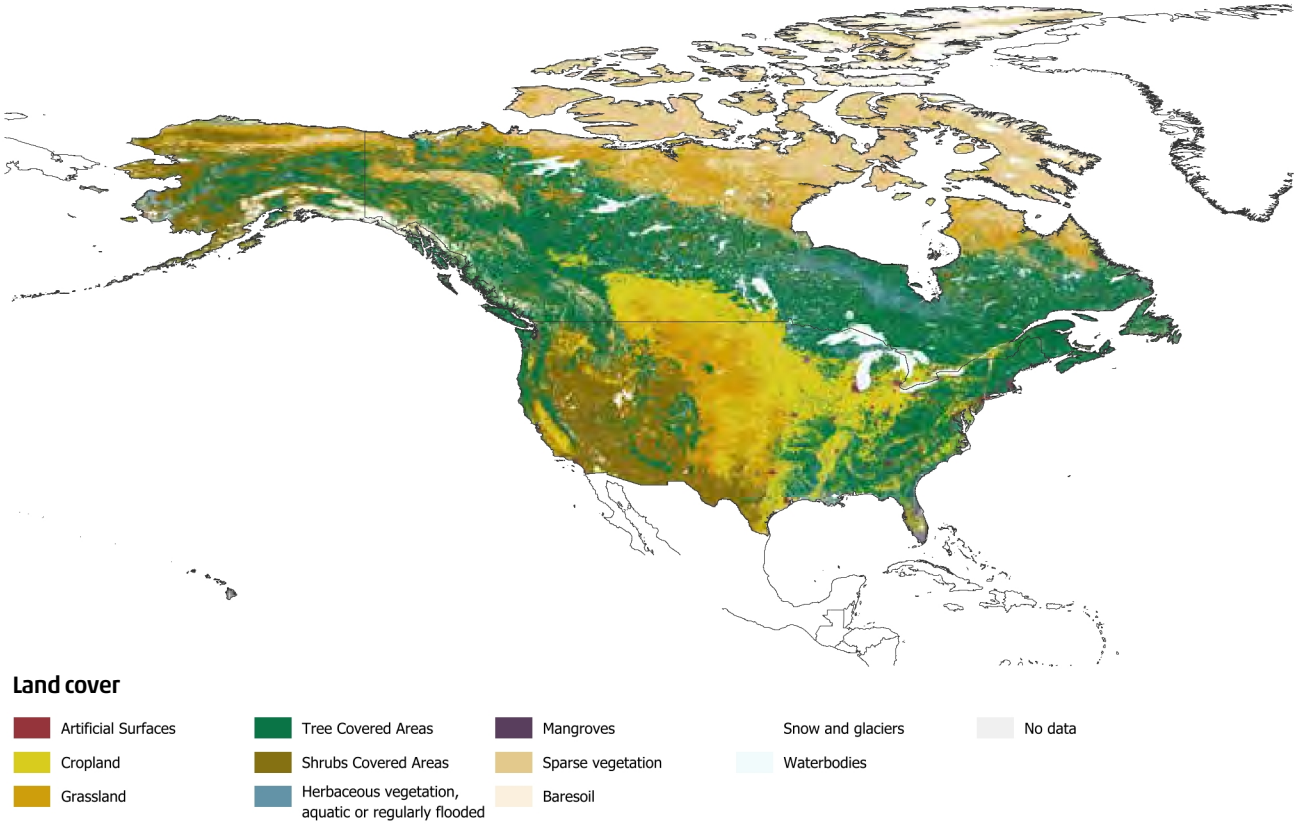


Source: Adapted from **UN Geospatial.** 2020. Map Geodata [Shapefile]. New York, USA, UN.

All five of FAO's global ecological zones (Fischer *et al.*, 2021), from tropical to polar, occur in the region together with an enormous diversity in soil and soil-forming factors, including climate. Average annual rainfall ranges from less than 10 to more than 1 000 mm while average annual temperature ranges from less than 0 °C to more than 25 °C. Soils in this region provide critical ecosystem services both locally and globally, with the central temperate plains being key to global food production, while boreal forests and arctic permafrost storing immense amounts of C.

The largest area of cropland in North America is in the central part of the region from the Gulf of Mexico to the boundary with the forested region in west central Canada (Figure 12.2). Other large areas of contiguous cropland occur south of the Great Lakes, along the Mississippi valley, in central California and along the eastern coast of the United States. The western cropland area in the United States grades to shrub-covered semi-arid lands and this cover extends into northern Mexico (Figure 10.1). The vast boreal forest stretches from the east coast of Canada through to the west coast and southern Alaska and is bordered by shrub-covered tundra and bare soil to the north (Figure 12.2). Much of the eastern United States is also tree-covered.

Figure 12.2. Land cover Map of North America



Refer to the disclaimer on page ii for the names and boundaries used in this map.

Source: **FAO.** 2024. GAEZ: Land and Water Resources. [Accessed 24 October 2024] https://gaez-services.fao.org/apps/theme-1/. Licence: CC BY 4.0.

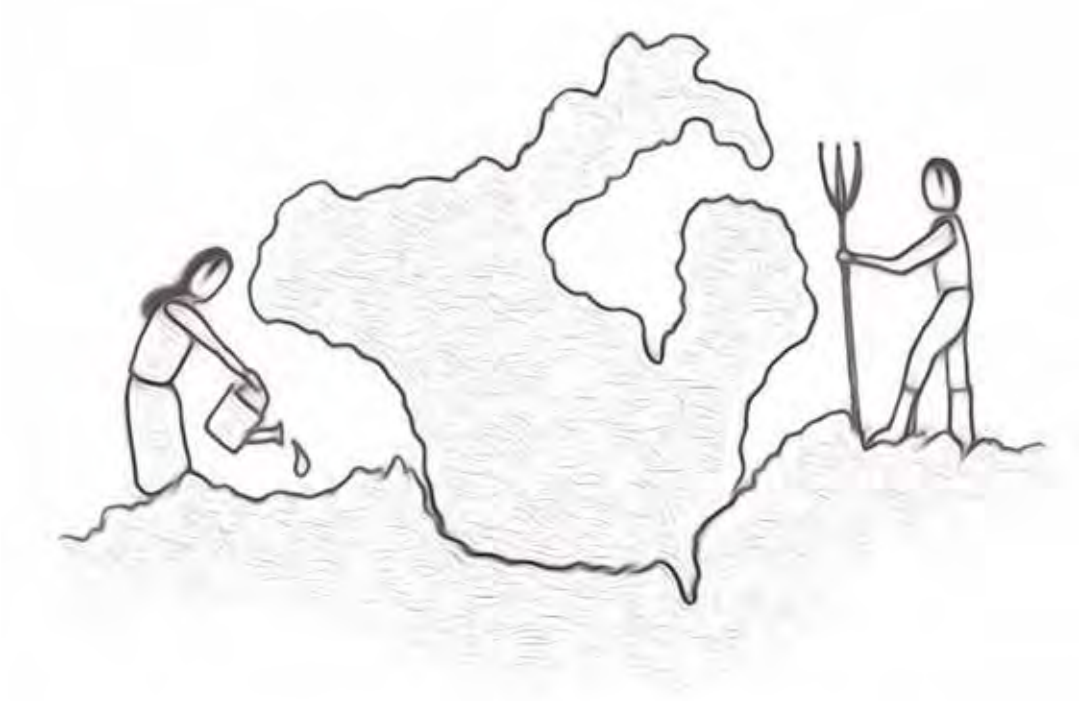
12.1.2 Summary of status and trend from the 2015 Status of the world's soil resources report

The North America regional assessment of condition and trend in the 2015 SWSR report (FAO and ITPS, 2015) was relatively positive, with six of the threats being assessed to be in fair condition, two in good condition and only one in poor condition. The majority of threats were either improving or stable.The only threat rated as poor in condition and deteriorating in trend was nutrient imbalance, the second-most severe threat. The authors stated that the excess application of fertilizers in many areas was causing degradation of surface water quality and producing increased emissions of N^2O to the atmosphere. In part, the deterioration in water quality was linked to sediment and nutrient runoff from cropland (judged to be the greatest threat). The linked issues of erosion and nutrient impacts on water quality were most severe in the northern Midwestern United States, the Mississippi River basin and agricultural regions of central Canada. However, widespread adoption of reduced and no-till in the Great Plains had substantially lowered the erosion risk in this region.

The adoption of reduced and no-till had also led to stabilization or increases in SOC in some regions and was responsible for the improving trend overall in SOC levels. However, the authors expressed concern about the effects of global warming on C storage levels in permafrost and the organic soils in northern Canada and Alaska in the United States.

Both soil acidification and contamination were judged to be in good condition and with an improving trend. In both cases, the improving trend was due to legislation passed in the previous decades that had limited further spread of the threats.

The authors also judged that both the loss of soil biodiversity and sealing and urbanization (due to urban expansion) were difficult to assess due to the lack of information.



12.2 Changes in the drivers of threats to soil functions

12.2.1 Overall population and urban population growth

While both Canada and the United States were highly urbanized by 2015, projected increases in urbanization by 2035 are substantially lower than global averages (Table 12.1). Population growth in Canada is projected to be higher than in the United States by 2035 (Table 12.2).

Table 12.1. Rate of change in urbanization for North America

Country	Urban population (2015) (%)	Change in urban population (2015–2025) (%)	Change in urban population (2015–2035) (%)	Change in urban population (2015–2050) (%)
Canada	81.3	1.0	3.2	7.4
United States	81.7	2.5	5.3	9.2
Global averages	71.2	3.3	7.8	15.4

◊ *Source:* **UN DESA.** 2018. *World Urbanization Prospects: The 2018 Revision.* New York, USA. [Cited 20 February 2024]. https://population.un.org/wup/assets/WUP2018-Report.pdf

Table 12.2. Population in 2021 and projections for population in 2025 and 2035 and percentage change per year for countries of North America

Country	Population 2021	Change in population (2015–2021) (%) (2015–2025) (%)	Population (2025)	Change in population (2021–2025) (%) (2015–2050) (%)	Population (2035)	Change in population (2025–2035) (%)
Canada	38 155 000	1.1	39 431 000	0.8	42 503 000	0.8
United States	336 998 000	0.6	343 603 000	0.5	360 016 000	0.5
Global averages		1.1		0.9		0.8

◊ *Source:* **UN DESA.** 2023. *World Population Prospects 2022: Summary of Results.* New York, USA. https://www.un.org/development/desa/pd/sites/www.un.org.development.desa.pd/files/wpp2022_summary_of_results.pdf

12.2.2 Land-use change

The United States only had small net changes in agricultural land use in the 2015–2022 period (Table 12.3). Canada experienced a substantial decline of 4.9 percent in permanent meadows and pastures (Table 12.4) with Canada and the United States having very similar trajectories in terms of forestland (Table 12.5), showing very minor net declines in forestland area.

Table 12.3. Land area, agricultural land area, cropland land area and change in agricultural land and cropland between 2015 and 2022

Country	Land area (ha)	Agricultural land area (%)	Change in agricultural land area (2015–2022) (%)	Cropland area (%)	Change in cropland area (2015–2022) (%)
Canada	878 870 000[1]	6.5	–1.6	4.4	0.1
United States	914 742 000	45.1	–1.1	16.9	–2.9

[1] While FAO (2024) is used for the land area of Canada in this table, Canada's land area is estimated by the Government of Canada as 909 350 000 ha.

◊ *Sources:* **FAO.** 2024. *FAOSTAT: Land Use.* [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.
Statistics Canada. 2011. Geography. In: *Statistics Canada.* Ottawa. [Cited 7 March 2023]. https://www150.statcan.gc.ca/n1/pub/11-402-x/2011000/chap/geo/geo-eng.htm

Table 12.4. Land area, forestland area and areal change between 2015 and 2022

Country	Land area (ha)	Forest land area (%)	Change in forestland area (2015–2022) (%)	Areal change in forestland area (2015–2022) (ha)
Canada	878 870 000	39.5	−0.1	−262 000
United States	914 742 000	33.9	−0.1	−300 000

◊ *Source:* **FAO.** 2024. *FAOSTAT: Land Use.* [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

Table 12.5. Land area, permanent meadows and pastures land area for 2022 and change in permanent meadows and pastures between 2015 and 2022

Country	Land area (ha)	Permanent meadows and pastures area (2022) (%)	Change in permanent meadows and pastures area (2015–2022) (%)
Canada	878 870 000	2.1	-4.9
United States	914 742 000	28.2	0.0

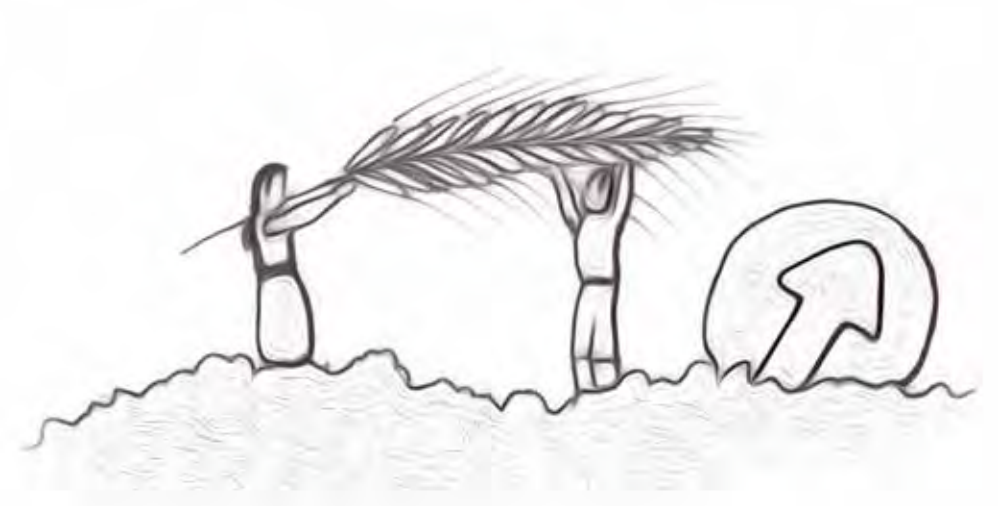
◊ *Source:* **FAO.** 2024. *FAOSTAT: Land Use.* [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

The estimated loss of wetlands since 1700 has been substantially higher in the United States than in Canada (Table 12.6), in part reflecting the much larger area of agricultural land in the United States relative to Canada (Table 12.3).

Table 12.6 Natural wetland changes between 1700 and 2020

Country	Wetland area (1700) (km^2)	Wetland area (2020) (km2)	Wetland loss (km^2)	Loss (%)	Percentage of global wetland losses (%)
Canada	2 176 000	2 093 000	83 000	3.8	2.18
United States	1 517 000	924 000	593 000	39.1	15.6

◊ *Source:* **Fluet-Chouinard, E., Stocker, B.D., Zhang, Z., Malhotra, A., Melton, J.R., Poulter, B., Kaplan, J.O.** *et al.* 2023. Extensive global wetland loss over the past three centuries. *Nature*, 614(7947): 281–286. https://doi.org/10.1038/s41586-022-05572-6



12.2.3 Nutrient management

At the national level, the estimated average values of N, P and K inputs and outputs in both Canada and the United States are approximately in balance (Table 12.7). The United States has a small negative balance in P, but agricultural soils (on average) have substantial residual P stores that have accumulated through time. As discussed in Section 12.3.2, there are major regional variations in nutrient use in large countries like Canada and the United States, and hotspots of nutrient imbalances and losses remain significant contributors to GHG emissions and water pollution.

Table 12.7. Average values (2017 to 2021) for nutrient use in North America

Country	N outputs[1] (kg/ha)	N nutrient balance[2] (kg/ha)	P nutrient balance[3] (kg/ha)	Accumulated P residual[4] (kg/ha)	K nutrient budget[5] (kg/ha)
Canada	60.6	35.8	1.6	13.0	1.0
United States	85.5	35.4	−0.3	237.0	3.5

[1] A very low N surplus (<20 kg/ha annually) may be associated with nutrient mining.

[2] Negative values for N nutrient balance indicate the mining of N from soil organic matter. Nitrogen surpluses greater than 80 indicate a high risk of N pollution.

[3] Negative values for P nutrient balance indicate the mining of P from soil.

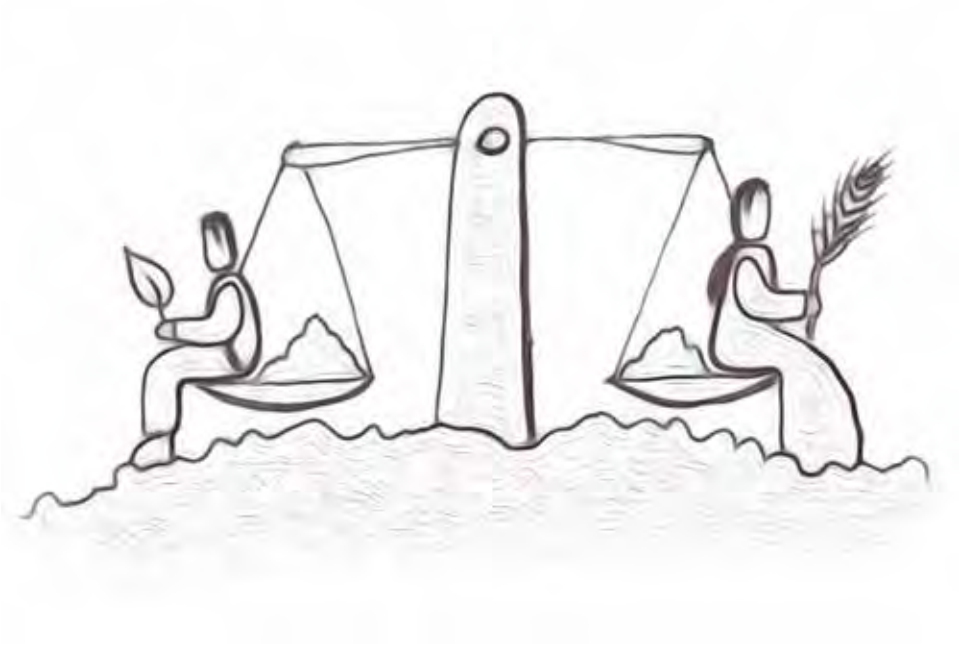
[4] Negative values for accumulated P residual indicate depletion of soil P reservoir since 1961. Positive values indicate a surplus of P additions.

[5] Negative values for K indicate the mining of K from soil.

◊ *Sources:* **EU Nitrogen Expert Panel.** 2015. *Nitrogen Use Efficiency (NUE) - an indicator for the utilization of nitrogen in agriculture and food systems.* Wageningen, Kingdom of the Netherlands, Wageningen University. https://research.wur.nl/en/publications/nitrogen-use-efficiency-nue-an-indicator-for-the-utilisation-of-n

◊ **FAO.** 2024. *FAOSTAT: Fertilizers by Nutrient. [Accessed on 9 January 2024]. https://www.fao.org/faostat/en/#data/RFN. Licence: CC BY 4.0.*

◊ **Zou, T., Zhang, X. & Davidson, E.A.** 2022. Global trends of cropland phosphorus use and sustainability challenges. Nature, 611(7934): 81–87. https://doi.org/10.1038/s41586-022-05220-z



12.3 Regional status and trend of threats to soil functions for North America region

12.3.1 Soil erosion in the North America region

Soil erosion is a major threat to soil function in North American croplands and is driven primarily by tillage, water and wind (Lobb *et al*., 2023). Soil erosion caused by crop production is the focus of this report, but soil erosion associated with land use and climate change also threatens forest and rangeland soils. For example, landslides in the mountainous landscapes of North America erode soil, particularly where agroforestry practices include clear-cutting (Montgomery *et al*., 2000). Moreover, the increase in the frequency and severity of wildfires over the last several decades (Abatzoglou and Williams, 2016) has led to increases in post-fire soil erosion (e.g. Larsen *et al*., 2009).

In Canada, trends in cropland soil erosion are assessed using the Soil Erosion Risk Indicator (SoilERI), which was initiated by the Government of Canada in 1993 as part of a nationwide assessment of a suite of agri-environmental indices (Lobb *et al*., 2023). Estimates of soil erosion risk were made in five-year increments from 1981 to 2021 using separate indices of erosion risk due to water, wind, and tillage, and their combined soil losses. Water erosion was predicted using a model based on the USLE (Wischmeier and Smith, 1978) and RUSLE2 (USDA, 2008). Wind erosion was predicted using a modified version of the Wind Erosion Equation (Woodruff and Siddoway, 1965), and tillage erosion was evaluated using a tillage erosion model (Lobb *et al*., 2023).

The measures of soil erosion risk in Canada have generally decreased since 1981, with most improvement occurring in the 1990s. However, there is evidence that the risk of soil erosion has been increasing in recent years with changes in tillage systems. The SoilERI estimated that in 1981, only 35.6 percent of cropland was eroding at a “very low” or “acceptable” annual rate of <6 t/ha. However, by 2011, largely due to adoption of no-till cropping, 77 percent of cropland was estimated to be eroding by water, wind and tillage processes at an annual rate of <6 t/ha, with a slight decline to 76 percent by 2016. By 2016, there was a risk of annual erosion at rates of >6 t/ha due to water erosion on 11.6 percent of cropland, wind erosion on 2.7 percent, and tillage erosion on 11.6 percent of cropland (Lobb *et al*., 2023). The areas in Canada with the highest risk of erosion include southern Ontario and parts of the Maritime provinces, where water and tillage erosion are the predominant risks (Figure 12.3), particularly where root vegetables are grown using intensive tillage. Some of the Prairie provinces are also subject to low-to-moderate erosion risk, primarily due to wind and tillage erosion (Figure 12.3). Nationwide, the farmland area classified to be at moderate to very high risk of erosion declined from ~30 percent in 1981 to ~15 percent in 2016 (Lobb *et al*., 2023). The goal of Canada is to have no land in higher risk classes (Lobb *et al*., 2023).

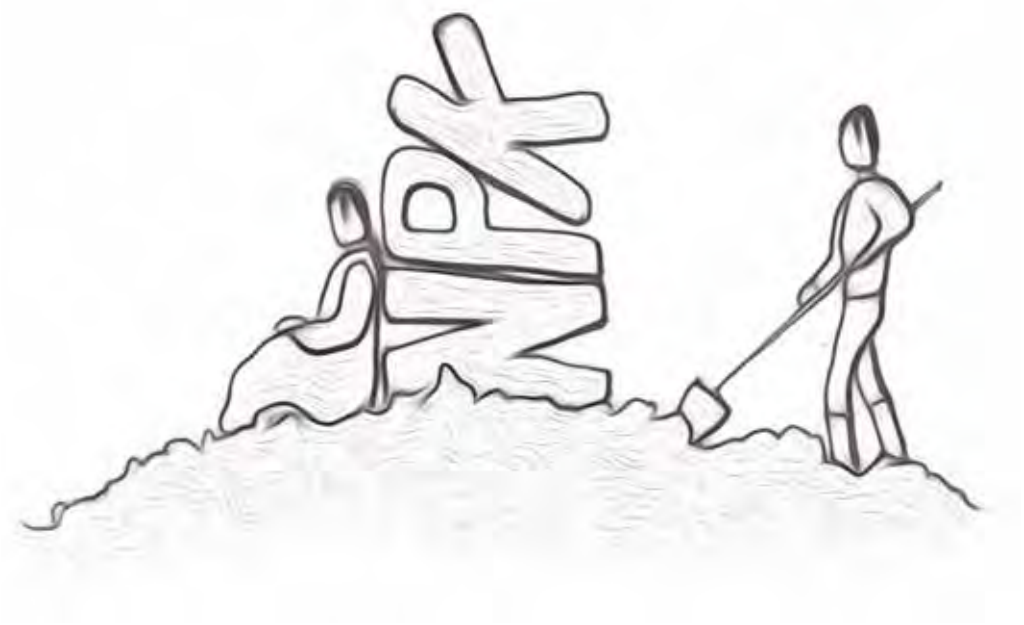
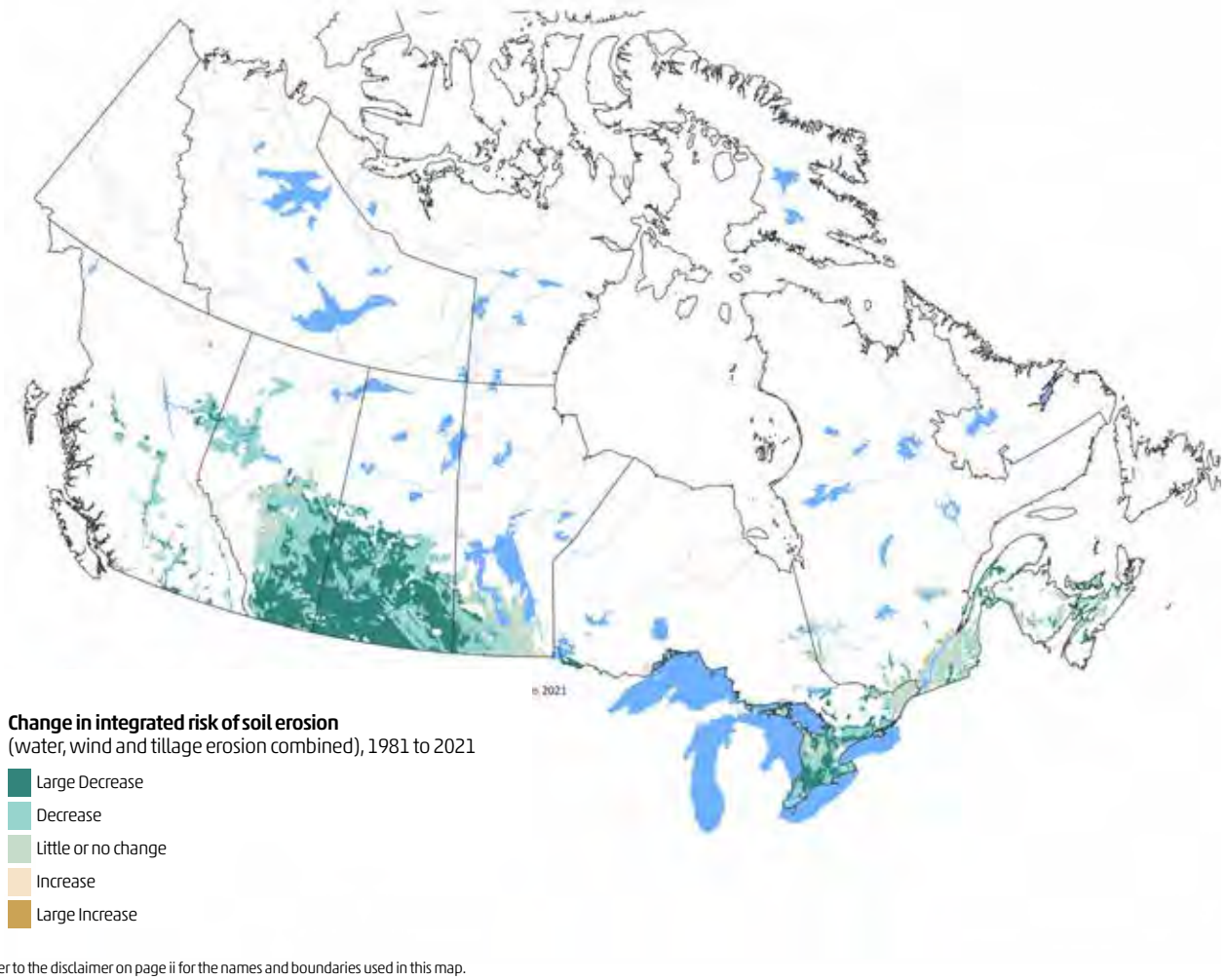


Figure 12.3. Map of the change in erosion risk due to the combined influence of water, wind and tillage erosion in Canada from 1981 to 2016



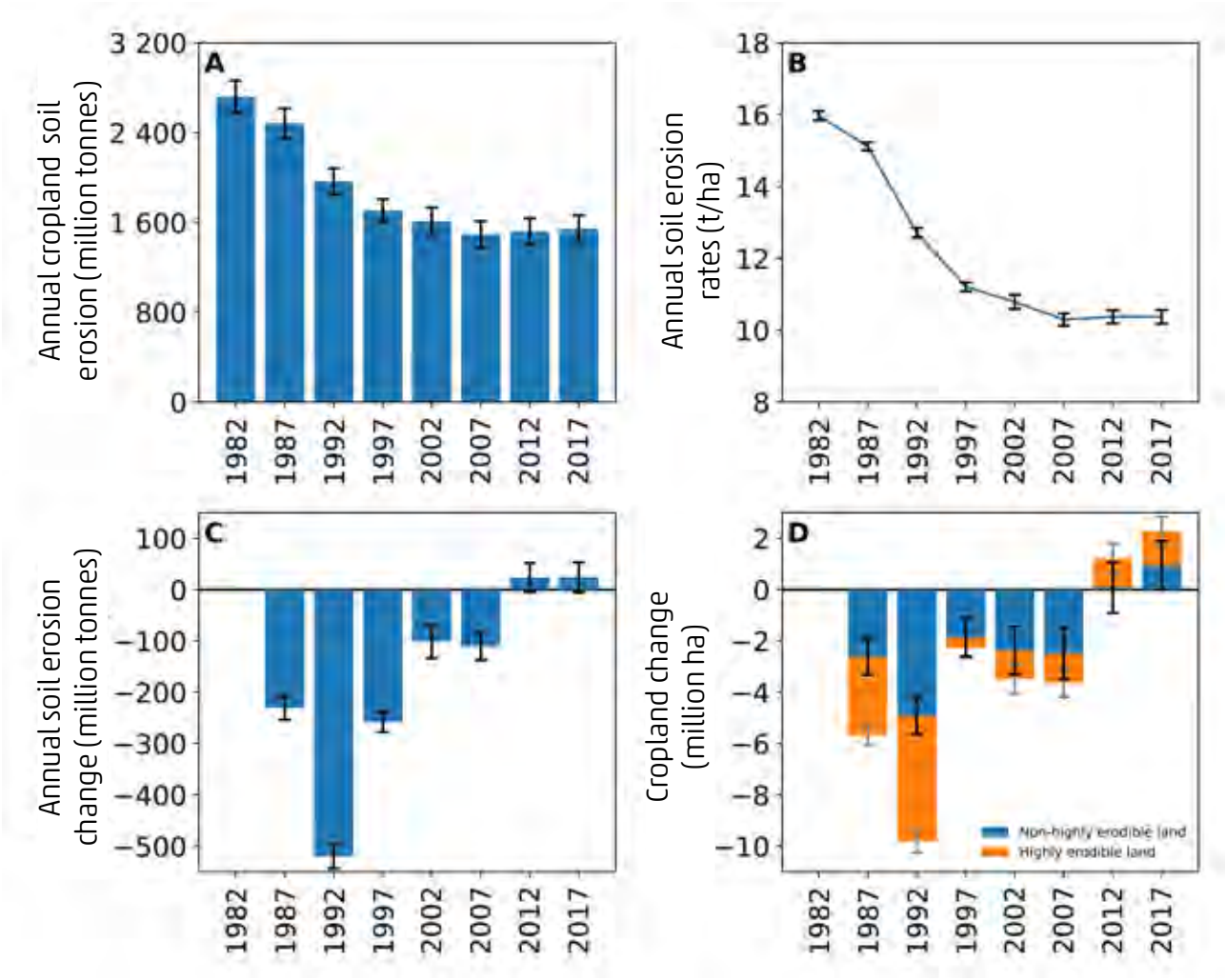
Refer to the disclaimer on page ii for the names and boundaries used in this map.

◊ *Source:* **Lobb, D.A., Li, S., McConkey, B.G. & Martin, T.C.** 2023. Soil Erosion Risk Indicator. In: *AAFC*. Ottawa, Agriculture and Agri food Canada (AAFC). [Cited 16 October 2023]. https://agriculture.canada.ca/en/environment/resource-management/indicators/soil-erosion-risk-indicator

In the United States, temporal trends in nationwide soil erosion on private crop- and pastureland are evaluated as part of the USDA National Resources Inventory (USDA, 2020), which is mandated by congressional legislation. The inventory is issued every five years, though it has produced annual assessments of soil erosion since 2000 (Schnepf and Flannagan, 2016). The most recent iterations of the inventory estimate water (sheet and rill) erosion using the Revised Universal Soil Loss Equation 2 (USDA, 2008) and estimate wind erosion using the Wind Erosion Equation (USDA, 2020).

Nationwide, erosion rates in the United States are estimated to have declined from 16.0 t/ha annually in 1982 to a low of 10.3 t/ha in 2007 (Figure 12.4). However, from 2007, estimated annual erosion rates increased slightly to 10.4 t/ha for both 2012 and 2017. As of 2017, water erosion is estimated to cause approximately 60 percent of the erosion, with the remaining 40 percent attributed to wind (USDA, 2023) (Figure 12.4). Although national average estimates of erosion rates have remained relatively constant over the last decade, the total magnitude of erosion has increased from 1.495 billion tonnes per year in 2007 to 1.543 billion tonnes per year in 2017 (USDA, 2023) (Figure 12.4).

Figure 12.4. Temporal trends in A) cropland soil erosion, B) soil erosion rates, C) soil erosion change, and D) cropland change for the United States

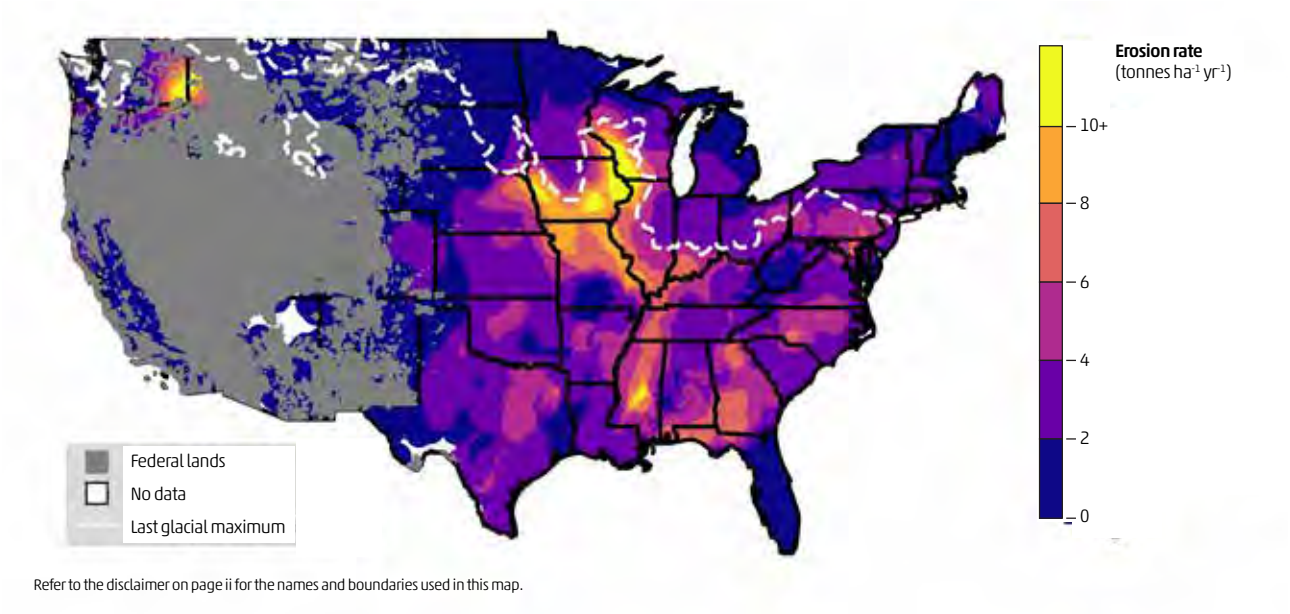


◊ *Source:* **USDA.** 2023. *Cropland Soil Erosion Trends.* NRCS National Resources Inventory, 2017. [Accessed on 16 October 2023]. https://publicdashboards.dl.usda.gov/t/FPAC_PUB/views/RCADVErosionbyStateNRI20171/ErosionTrends?%3Aembed=y&%3AisGuestRedirectFromVizportal=y. Licence: CC BY 4.0.

The increase in erosion magnitude is associated with a cropland area increase of 3.41 million ha, 72 percent of which are considered highly erodible land (USDA, 2023) (Figure 12.5). The areas with the highest estimated erosion rates in the United States are in the midwestern states where maize and soybean are the predominant crops and the Palouse region in the northwest where wheat is the predominant crop (Figure 12.5). Tillage erosion is not considered in the National Research Assessment, but regional-scale modelling for the Midwestern United States suggests present-day rates of tillage erosion are approximately 11 t/ha annually (Kwang *et al*., 2023), a value that is similar to the estimated nationwide erosion rate, due to wind and water (USDA, 2023). Hence the sum of erosion from all three sources (water, wind and tillage) is likely to be substantially higher than from wind and water alone.



Figure 12.5. Predicted erosion rates from water (sheet and rill erosion) for the United States in 2017

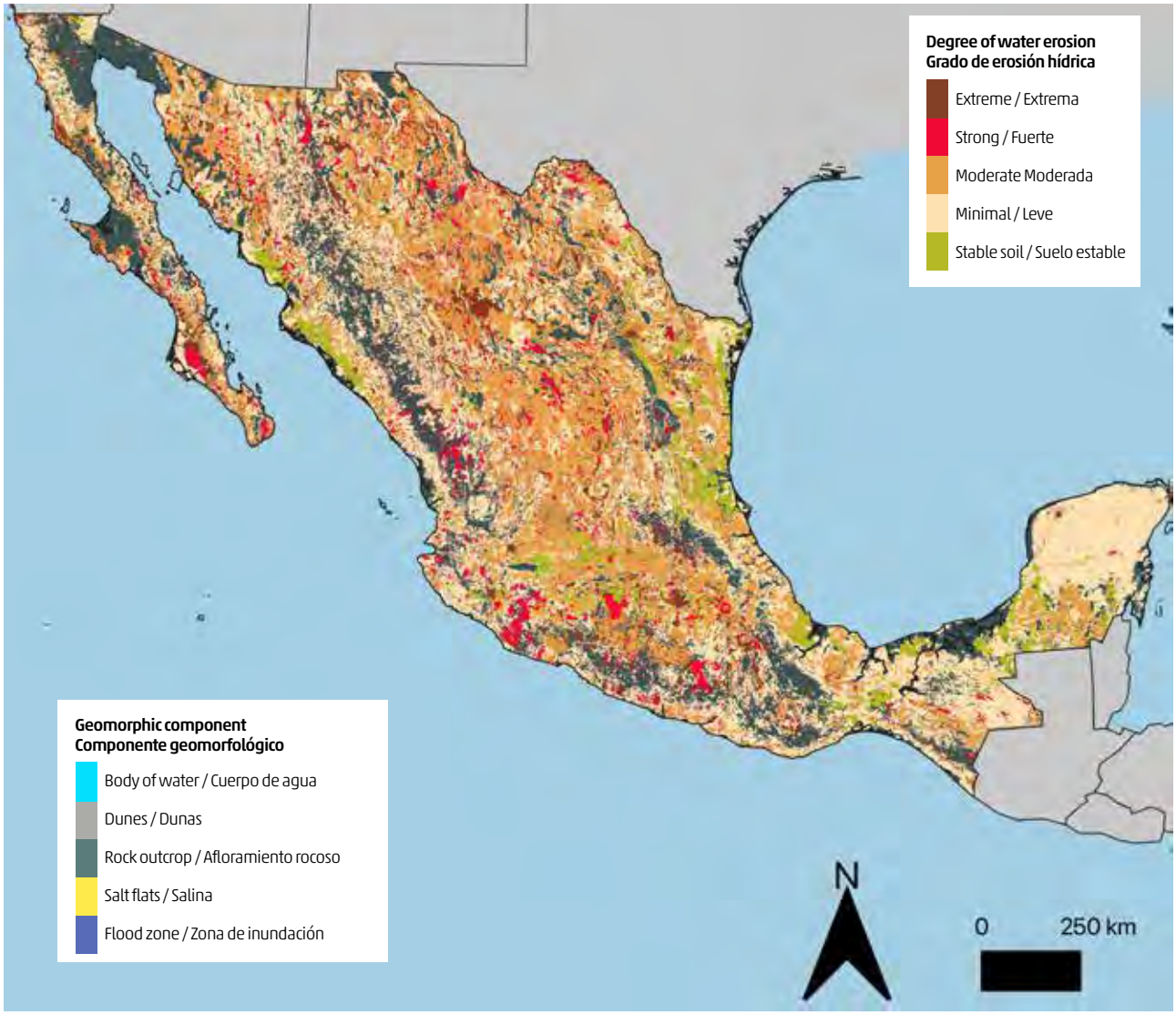


Source: **USDA.** 2020. *National Resources Inventory (NRI).* [Accessed on 16 October 2023]. https://www.nrcs.usda.gov/nri Licence: CC-BY-4.0.

In Mexico, soil erosion has been assessed via evaluation of high-resolution satellite imagery and field-based information to generate a national erosion map (Bolaños Gonzalez *et al.*, 2016). Mapped polygons classify erosion as extreme, strong, moderate, slight, or stable and primarily identify areas with evidence of water erosion. Nationwide, 76 percent of the land area has been eroded by water and approximately 13 percent of the area has evidence of extreme and strong erosion (Bolaños Gonzalez *et al.*, 2016). In the fourteen northern states of Mexico, between 8 and 18 percent of the land area has evidence of extreme and strong soil erosion, with the remainder of the land area exhibiting signs of moderate erosion (7 to 57 percent), slight erosion (21 to 49 percent), or excluded from the erosion assessment (3 to 42 percent) (Figure 12.6) (Bolaños Gonzalez *et al.*, 2016).



Figure 12.6. Soil erosion severity in Mexico based on satellite image analysis and field observations



Refer to the disclaimer on page ii for the names and boundaries used in this map.

◊ *Source:* **Bolaños González, M.A., Paz Pellat, F., Cruz Gaistardo, C.O., Argumedo Espinoza, J.A., Romero Benítez, V.M. & de la Cruz Cabrera, J.C.** 2016. Mapa de erosión de los suelos de México y posibles implicaciones en el almacenamiento de carbono orgánico del suelo [Soil erosion map of Mexico and possible implications for soil organic carbon storage.]. *Terra Latinoamericana,* 34(3): 271–288.

◊ **CGG-SAGARPA y COLPOS.** 2009. *Documento de Referencia para la Estimación de la Erosión Actual del Suelo en México [Reference Document for Estimating Current Soil Erosion in Mexico].* Mexico City, Ministry of Agriculture, Livestock, Rural Development, Fisheries, and Food (SADER).

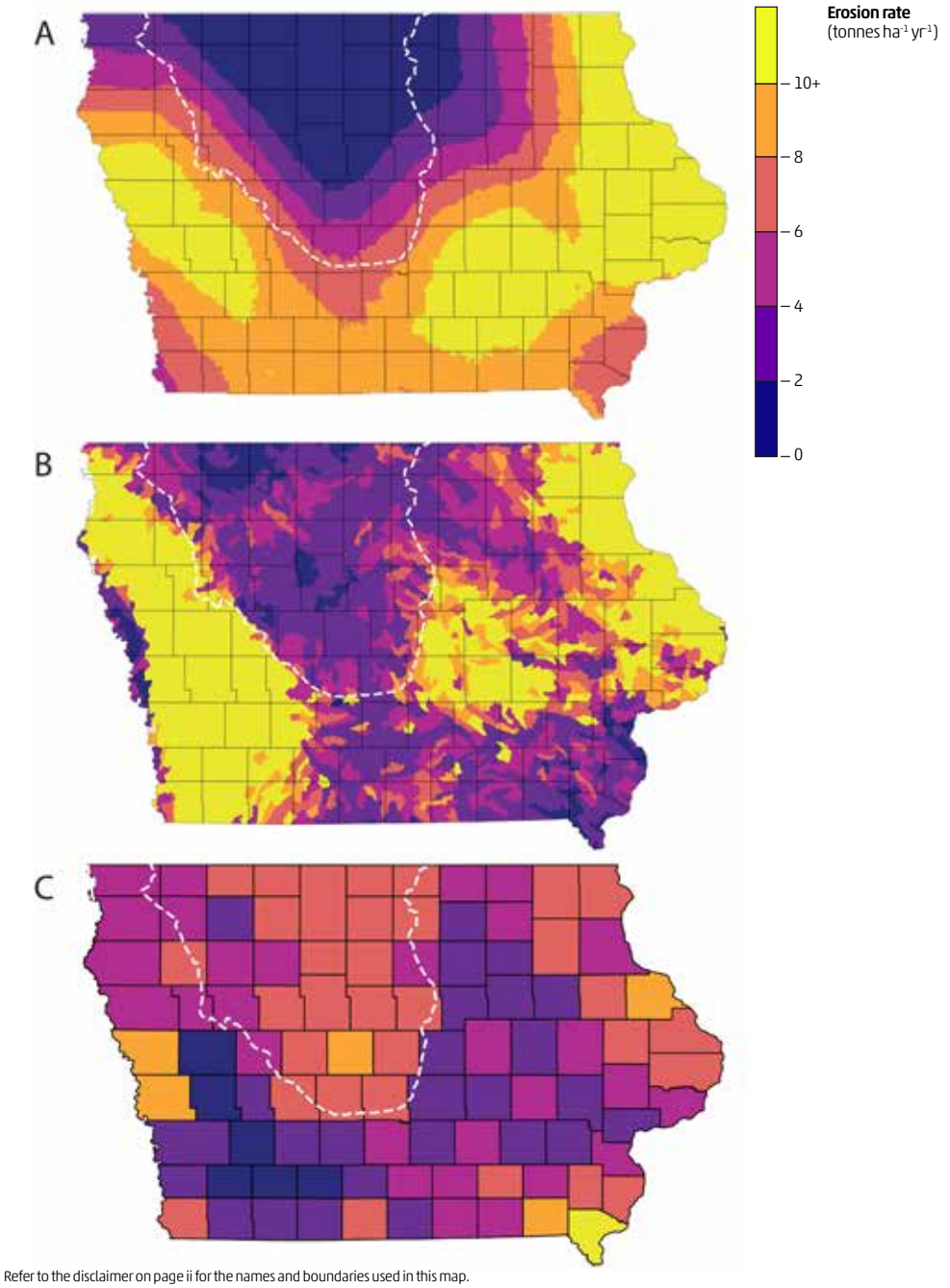
Since 2001, there has been an expansion of intensive, industrial agriculture, particularly in northern Mexico, where the conversion of forest, shrubland and grassland for agriculture has caused erosion rates to increase (Santini, Cuervo-Robayo and Adame, 2022). The states of Chihuahua, Coahuila, Durango and Sonora in northern Mexico have the largest documented areas of water erosion (overall, more than 10 million ha) (Bolaños Gonzalez *et al.*, 2016). The widespread erosion has reduced crop productivity, with approximately 28 percent of farmers reporting issues with soil fertility (INEGI, 2019; Ortiz-García *et al.*, 2022). Seventy percent of the agricultural land area is classified as family farming due to small farm size or subsistence production, with many of these farmers cultivating steep slopes, combined with issues of poverty, which prevent investment in soil conservation practices, which exacerbates erosion and loss of fertility (Ortiz-García *et al.*, 2022). Increased rates of soil erosion are particularly associated with slash and burn practices on traditional farms, followed by intensive tilling, which also leads to the loss of nutrients. Government initiatives such as “Mi parcela no se quema” (“My agricultural land does not burn”) have therefore been implemented to promote

alternative agricultural practices (Santini, Cuervo-Robayo and Adame, 2022). In recognition of the need to sustainably manage agricultural soils and reverse land degradation, Mexico has recently developed a National Soil Strategy for Sustainable Agriculture (Ortiz-García *et al*., 2023). Although tillage erosion is not recognized in Mexico's assessments, small, intensively tilled fields on steep land are no doubt subjected to extreme soil losses by tillage erosion.

The USLE and RUSLE2 models, used to predict erosion rates in Canada and the United States, are based on decades-old, primarily empirically based technology. Recent efforts to use alternative erosion models have focused primarily on the intensively farmed Midwestern region of the United States. The Daily Erosion Project (DEP) (Gelder *et al*., 2017; Iowa State University, 2024) is a regional soil erosion estimation effort focused on Iowa, Nebraska, and parts of six other midwestern states. The project uses remote sensing-based estimates of crop residue cover and tillage practices in combination with high-resolution topographic data and radar-based precipitation data to estimate daily values of runoff, soil detachment and hillslope soil loss using the Water Erosion Prediction Project (WEPP) model (Laflen, Lane and Foster, 1991). Comparing different approaches to estimate erosion rates for the State of Iowa, which has the highest proportion of cropland of the 50 states in the United States (74.7 percent, or 10.8 million ha) (Bigelow and Borchers, 2017), is useful because multiple models make estimates for the entire state. For the five-year period ending in 2017, DEP estimated an average state-wide annual erosion rate of 4.7 t/ha, whereas the National Resources Inventory estimated a mean annual water erosion rate of 12.6 t/ha. The approximately three-fold difference in erosion rates is an indicator of the level of uncertainty associated with water erosion estimates in the United States (Figure 12.7). Unlike Canada, nationwide erosion estimation for the United States does not consider erosion caused by tillage, nor is the inclusion of tillage erosion planned for the next generation of United States soil erosion estimate technology (Weltz *et al*., 2020). However, recent studies indicate tillage erosion is widespread and a major contributor to soil degradation in the Midwestern United States.

Tillage erosion and water erosion are focused on different parts of hillslopes, with tillage erosion occurring predominantly on convex hilltops and upper slopes and water erosion focused on lower backslopes (Li *et al*., 2007). Spatial variability in the timing of the last glacial advances in North America has generated landscapes with different hillslope topography, with hilly landscapes having more convex slopes in the areas that were most recently glaciated, and more fluvially dissected topography in areas with older glaciations (Thaler, Larsen and Yu, 2021). The differences in glacial history and topography cause spatial patterns of predicted water and tillage erosion for the Midwestern United States to differ considerably at the regional scale (Figure 12.7). The National Resources Inventory and DEP water erosion models estimate low rates of erosion in the area with glacial till-derived soils (ice-covered during the last glaciation) and high rates of erosion for loess-derived soils outside of the last glacial ice limit (white dashed line in Figure 12.7a and Figure 12.7b). However, the recently glaciated area has the lowest rate of adoption of low intensity and no-till farming practices in the region (Azzari *et al*., 2019), which in combination with the hilly topography, leads to high rates of predicted tillage erosion (Kwang, Thaler and Larsen, 2023) (Figure 12.7c). These results indicate that by not accounting for tillage erosion, the National Resources Inventory underestimates the total magnitude of soil erosion in the United States.

Figure 12.7. Soil erosion rates for the State of Iowa, United States, predicted by different models



Refer to the disclaimer on page ii for the names and boundaries used in this map.

Note: The white dashed line shows the extent of the most recent Pleistocene ice advance. The parent material for soils north of the ice limit is predominantly glacial till, whereas loess is the dominant parent material for the areas to the south of the most recent ice advance. The water erosion models (A and B) predict low rates of erosion for the hummocky glacial landscape north of the ice limit, whereas the tillage erosion model (C) predicts erosion rates are highest north of the ice limit.

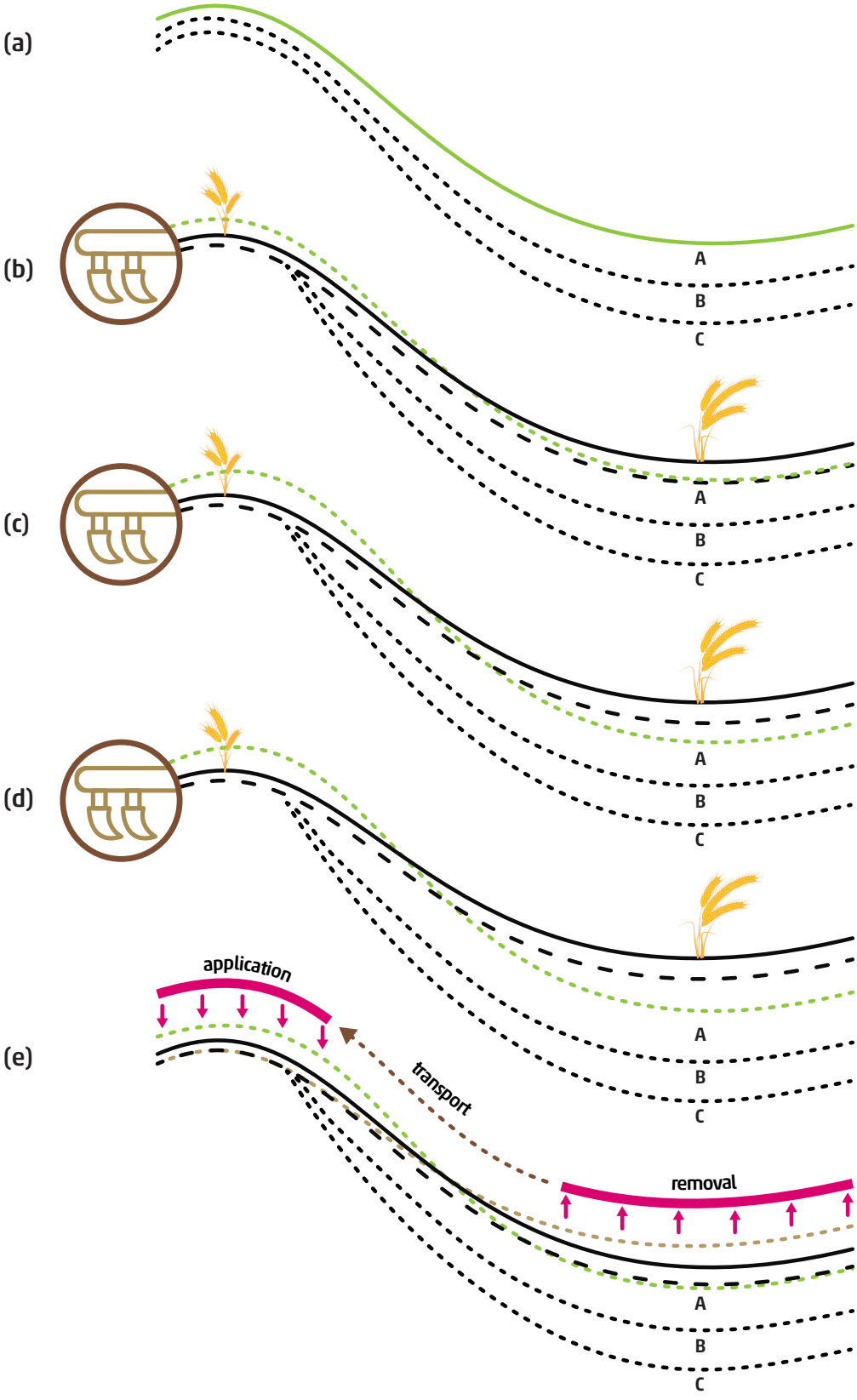
◊ *Sources:* A. **USDA.** 2020. *National Resources Inventory (NRI).* [Accessed on 16 October 2023]. https://www.nrcs.usda.gov/nri. Licence: CC-BY-4.0.

◊ B. **Iowa State University.** 2024. *Daily Erosion Project. [Cited 20 November 2024]. https://www.dailyerosion.org*

◊ C. **Kwang, J.S., Thaler, E.A. & Larsen, I.J.** 2023. The Future of Soils in the Midwestern United States. *Earth's Future*, 11(5): e2022EF003104. https://doi.org/10.1029/2022EF003104

The practice of soil landscape restoration is an old concept (Lowdermilk, 1935), that has recently gained attention in North America as a means of rapidly restoring productivity to eroded hilltop soils, as the soil transported to lower slopes by tillage erosion is still highly productive. Soil landscape restoration (Figure 12.8) is the process of excavating soil from depositional areas and transporting it to hilltops where it is spread over degraded soil (Lobb, 2011, 2015; Ernst, 2019). Plot-scale measurements made over six years at a site in west-central Minnesota in the United States demonstrated that soil landscape restoration effectively redistributed soil nutrients and increased crop yield relative to unrestored sites (Schneider *et al.*, 2023).

Figure 12.8. Soil erosion on a typical hillslope in the formerly glaciated North American mid-continent



Notes: A. Natural grassland prior to cultivation. B. After a few decades of cultivation. C. After several decades of degradation. D. An advanced state of degradation, like much of the region today. E. The concept of soil landscape restoration, where soil transported from upper to lower slopes by tillage erosion is excavated with earth moving equipment and spread over degraded soils on upper slopes.

◊ *Source:* Adapted from **Lobb, D.A.** 2011. Understanding and managing the causes of soil variability. *Journal of Soil and Water Conservation*, 66(6): 175A–179A. http://dx.doi.org/10.2489/jswc.66.6.175A

The models used to estimate erosion in Canada and the United States do not include gully erosion (which can cause deep erosion and transfer soil out of fields to waterbodies [Li *et al.*, 2016]), or interactions among erosion processes, such as the role of tillage in transporting soil to topographic locations susceptible to water erosion. The difficulties of including gully erosion or interactions between soil transport processes are rooted in the historical development of the USLE and its revised forms, as well as the WEPP model, which both estimate erosion at the scale of two-dimensional hillslopes. Simulating gully erosion or process interactions requires a three-dimensional representation of topography that evolves in time and space in response to soil detachment, transport and deposition. High resolution topographic data from airborne laser swath-mapping offers a potential solution. Such data can be used as initial conditions for erosion modelling, not just in individual hillslopes or fields, but for hundreds of thousands of hectares of cropland (e.g. Kwang *et al.*, 2023). However, to capitalize on the availability of high-resolution topographic and other landcover and weather datasets, soil erosion prediction will require new model frameworks (e.g. Weltz *et al.,* 2020).

Annual rates of erosion in North America do not reflect the degree that soil has been degraded by historical erosion, or the resulting magnitude of lost soil that continues to affect soil function. The cumulative loss of soil organic matter due to erosion occurring since the initiation of farming has caused crop yields to decline throughout Canada and the United States. In Canada, reductions in crop yield caused by soil degradation led to cumulative economic losses of CAD 33.5 billion between 1971 and 2015, with a maximum annual loss of CAD 1.9 billion in 2013 (Badreldin and Lobb, 2023). Although the percentage of cropland with the highest erosion rate classification (>33 t/ha annually) declined from 5.5 percent in 1971 to 1.5 percent in 2015, the cost of erosion in Canada increased over the same time period, rising from an annual average of USD 0.2 billion (1971 to 1988) to USD 1.6 billion (2008 to 2015) (Badreldin and Lobb, 2023).

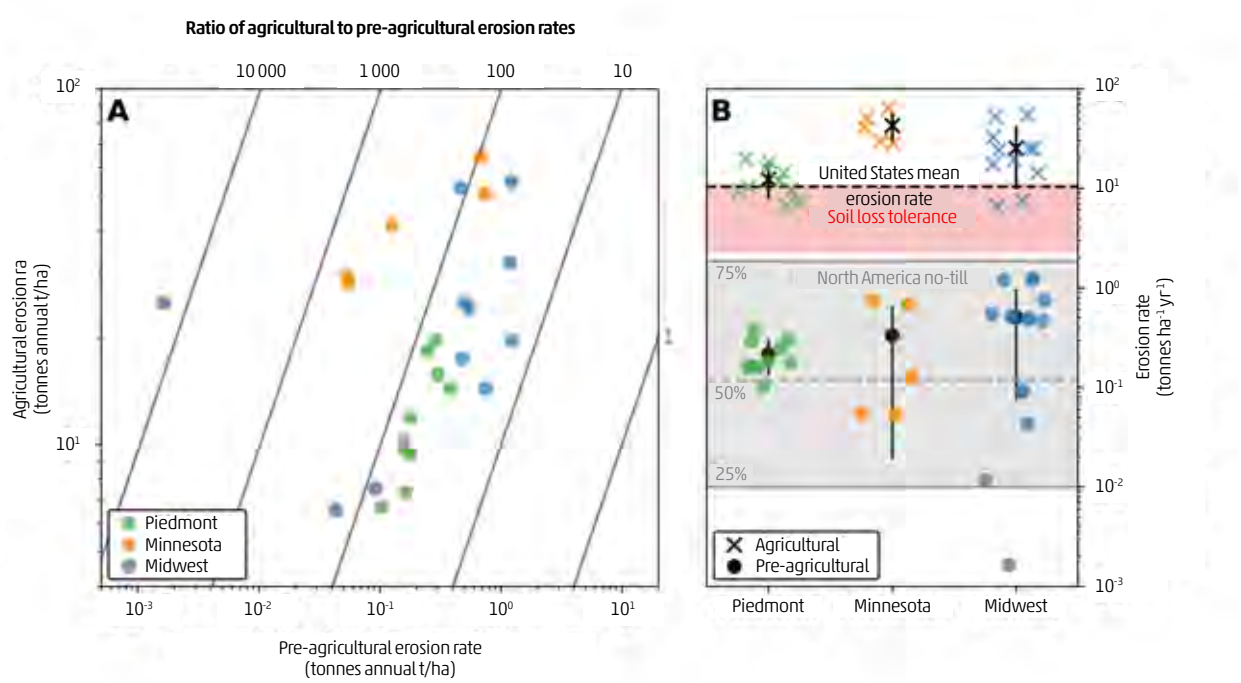
In the Midwestern United States, economic losses due to lost productivity are estimated to be of similar magnitude to those from Canada, where erosion has removed organic matter-rich soils from approximately one-third of the agricultural land area (Thaler, Larsen and Yu, 2021). The removal of 1.4 billion tonnes of organic C from hillslopes is estimated to have reduced maize yields by 6 percent, resulting in annual losses of USD 2.8 billion. Soil erosion has reduced the value of non-irrigated maize produced in Mexico by 4 percent to 7 percent, which equates to onsite costs of approximately USD 45 to 90 million (Cotler, López and Martínex-Trinidad, 2011).

The economic loss values for both Canada and the Midwestern United States are minimum estimates as they only account for areas where organic matter-rich soil horizons have been completely eroded, and not areas where thinning of the topsoil has occurred (Thaler, Larsen and Yu, 2021; Badreldin and Lobb, 2023). The economic losses also do not include the cost of increased fertilizer application. One-third of the N fertilizer used in United States maize production is used to offset the organic matter loss caused by soil erosion (Jang *et al.*, 2021). The cost of N fertilizer application to degraded soils in the Midwestern United States is on the order of USD 0.5 billion annually (Basso *et al.*, 2019; Jang *et al.*, 2021). In 1990, the sum of nationwide on- and offsite costs of soil erosion in the United States was estimated to be USD 44 billion annually (Pimentel *et al.*, 1995). There is a need for updated estimates of the nationwide cost of soil erosion in the United States and Mexico, as well as estimates of the offsite costs of soil erosion in Canada and Mexico.

Upper soil horizons that are rich in organic matter are critical for crop productivity. Such horizons can only persist if agricultural erosion rates are comparable to rates of preagricultural soil erosion and soil production, otherwise soil thickness declines over time (Montgomery, 2007; Wilkinson and McElroy, 2007). Several recent studies documenting the degree which agriculture has accelerated erosion, using cosmogenic nuclides to measure preagricultural soil erosion rates. Farming in the Appalachian Piedmont region of the eastern United States led to average annual erosion rates of 12.4 t/ha between the 1700s and the early twentieth century (Trimble, 1977), more than 50 times

greater than the average preagricultural erosion rate of 0.2 t/ha annually (Reusser, Bierman and Rood, 2015). In west-central Minnesota in the United States, agricultural annual erosion rates of 42.9 t/ha are approximately 130 times greater than the preagricultural erosion rate of 0.33 t/ha per year (Jelinski *et al.*, 2019). Soil erosion rates estimated by measuring 137Cs over fifty years were of a similar magnitude, suggesting there has been little reduction in erosion rates (Jelinski *et al.*, 2019). Surveys of the elevation difference between native prairies and adjacent agricultural fields at sites throughout the Midwestern United States indicate erosion rates to have been 25.5 t/ha annually (Thaler, Larsen and Yu, 2022), more than 40 times greater than preagricultural annual erosion rates of 0.59 t/ha measured for the native prairies (Quarrier *et al.*, 2023). Agricultural erosion rates in the United States are therefore a factor of 10 to >100 times greater than preagricultural soil erosion rates (Figure 12.9a). Across North America, measured soil erosion rates from conventional farming practices generally exceed soil loss tolerance values, whereas soil erosion rates measured for no-till farming practices are comparable to preagricultural erosion rates (Figure 12.9b), highlighting the potential for no-till practices to reduce erosion rates to sustainable levels.

Figure 12.9. Changes in soil erosion rates caused by agriculture and no-till farming



Note: A. Agricultural versus preagricultural, or long-term, "background" erosion rates for three sites in the United States showing that agriculture has increased erosion rates by a factor of 10 to >1 000. B. Agricultural erosion rates and preagricultural erosion rates relative to USDA soil loss tolerance values, and the mean predicted national erosion rates and the interquartile range of no-till erosion rates from the compilation of Montgomery (2007).

Measured soil erosion rates from conventional farming practices generally exceed soil loss tolerance values. Soil erosion rates measured for no-till farming practices are comparable to preagricultural erosion rates.

Source: See References.

Although the current mean annual erosion rate in the United States falls within the range of values considered tolerable by the USDA (Figure 12.9b), the average erosion rate and soil loss tolerance values are orders of magnitude higher than measured preagricultural or soil production rates, which is consistent with prior studies that have advocated that soil loss tolerance values are inadequate for sustaining soil (e.g. Montgomery, 2007; Nearing *et al.*, 2017). However, no-till farming reduces soil erosion rates to levels that are comparable to preagricultural soil erosion rates (Figure 12.9b) (Montgomery, 2007). Overall, soils in the United States are eroding at rates many times greater than rates of soil formation. The situation in Canada and northern Mexico is not substantially different from that in the United States, where current policies and practices lead to the continued demise

of soil resources by promoting agricultural productivity at the expense of soil health.

More widespread utilization of no-till farming could greatly reduce rates of soil erosion (Kwang, Thaler and Larsen, 2023). However, returning the productivity of eroded soils to predegradation levels will require policies that promote the adoption of agricultural practices that increase organic matter content in degraded soils, such as use of cover crops (Reilly *et al*., 2023), and crop rotations (Montgomery, 2017; Macray and Montgomery, 2023). It should also be emphasized that efforts to conserve and restore soil health and productivity must recognize the complexities of controlling three forms of erosion and their cumulative effects: for example, conservation tillage practices that protect soil from wind and water erosion may not protect soil from tillage erosion.

12.3.2 Nutrient mismanagement in the North America region

Nutrient mismanagement is a major issue for SSM. It includes the over- or underapplication of nutrients relative to plant demand and can impact soil erosion, C stocks and biodiversity. Much of North American arable lands are intensively managed, and the excessive use of nutrients in the region is of primary concern (Chai, Pannell and Pardey, 2023; Zuidema *et al*., 2023) although nutrient mining (the failure to replace nutrients or management resulting in negative nutrient balances) also occurs (Reid, Schneider and Joosse, 2019). The following analysis focuses on croplands which are sources of nutrient surpluses (primarily N and P). In 2022, cropland in Canada and the United States accounted for 32 percent of global maize production and 35 percent of global soybean production (FAO, 2022a). Over 80 percent of croplands in North America are in the United States, where maize production consumes >45 percent of annual synthetic fertilizer use and is often grown in rotation with soybean. Thus, the United States maize-based production systems present a useful case study for examining regional nutrient mismanagement.

Extensive water monitoring shows that N and P from arable lands continue to compromise water quality. In the most recent assessment, the United States Environmental Protection Agency (US EPA) found 46.6 percent of the United States river and stream miles were in poor condition with respect to total N, a value unchanged from previous assessments, while 41.8 percent were in poor condition from P loads, an improvement of almost 18 percent when compared to the same assessment five years earlier (US EPA, 2022). Subregions with intensive row crop and animal agriculture remain negatively impacted. Many concentrated animal feed operations (CAFOs) are co-located with maize-based crop production systems. Miralha, Sidique and Muenich (2022) identified 16 260 CAFOs in the Midwestern corn belt states. Poor ratings for river and stream miles in this subregion are approximately 70 percent for N and 43 to 60 percent for P.

Nitrogen fertilizer application to soils can lead to the microbial production of N_2O, a potent greenhouse gas with 273 times the global warming potential of CO_2. Concern and focus on agricultural N_2O emissions have increased. In North America there are 231 000 tonnes per year of direct N_2O emissions from synthetic fertilizers (81 percent from United States) and 35 000 tonnes per year from manure (89 percent from United States) applied to soil (FAO, 2022b). Direct N_2O emissions from United States soils account for 96 percent of total United States N_2O emissions and the majority of greenhouse gas emissions from crop production (Lu *et al*., 2022a).

Within North America, estimation of partial nutrient balance (e.g. fertilizer plus biological N fixation inputs minus harvest outputs) has proved useful in understanding N and P mismanagement and links between the magnitude of N and P surplus at the farmgate and environmental losses, trade-offs and ecosystem service degradation (Table 12.8). Frei *et al*. (2021) showed that 2 to 8 percent of the United States land area contributed 75 percent of nutrient flux to streams and lakes and recommended the prioritization of N and P reductions in catchments that contributed disproportionately high loads to surface waters. Kim *et al*. (2021) used meta-modelling to identify hotspots with high mitigation potential in the United States maize-producing states, and found

that implementing a 10 percent fertilizer reduction targeted at 19.9 percent of cropland could reduce nitrate leaching by 9.6 percent, but corn belt yields would decrease by only 0.6 percent. Maize production in the United States has been associated with 60 percent higher nitrate leaching losses for any given level of N balance when compared to studies from outside of North America (Tamagno *et al.*, 2022). This was attributed to higher levels of N inputs for a given N balance or the prevalence of subsurface artificial drains that dominate the region (Valayamkunnath *et al.*, 2020). Zhao, Nafziger and Pittelkow (2017) demonstrated that managing N to increase fertilizer recovery by 30 percent (such as reducing N application rates by 25 percent) could reduce N^2O emissions by 30 percent, but highlighted yield trade-offs and economic losses.

Table 12.8. Cropland nitrogen, phosphorus and potassium balances (kg/ha) for the United States, Canada and North America

			United States			Canada			North America		
	Pool[1]	Source[2]	P1	P2	Diff.[3]	P1	P2	Diff.	P1	P2	Diff.
N	Inputs	Synthetic fertilizer	57.3	58.7	1.4	49.4	62.4	13.0	55.8	59.4	3.7
		Manure	9.6	10.0	0.4	5.7	5.6	−0.1	8.8	9.1	0.3
		Atmospheric deposition	7.5	7.2	−0.3	4.4	4.4	0.0	6.9	6.6	0.3
		Biological N fixation	36.6	44.8	8.2	15.4	22.5	7.0	32.6	40.5	7.9
		Seed	1.2	1.4	0.2	1.6	1.9	0.3	1.3	1.5	0.2
		Total	112.1	122.1	9.9	76.5	96.7	20.2	105.3	117.1	11.8
	Removal	Crop	71.6	86.0	14.4	48.1	60.5	12.4	67.1	81.1	14.0
	Surplus		40.5	36.0	−4.5	29.4	36.2	7.8	38.2	36.1	2.1
	Loss	Leaching	7.6	7.9	0.3	6.2	7.4	1.2	7.4	7.8	0.4
		Volatilizing	20.0	20.7	0.7	16.8	20.4	3.6	19.4	20.6	1.2
P	Inputs	Synthetic fertilizer	10.5	10.7	0.3	7.5	11.0	3.5	9.9	10.8	0.9
		Manure	3.4	3.6	0.2	1.8	1.7	−0.02	3.1	3.2	0.1
		Seed	0.2	0.2	0.03	0.3	0.4	0.1	0.2	0.3	0.03
		Total	14.1	14.6	0.5	9.5	13.1	3.5	13.2	14.3	1.1
	Removal	Crop	12.5	14.7	2.2	9.2	11.5	2.3	11.9	14.1	2.2
	Surplus		1.6	−0.2	−1.8	0.3	1.6	1.3	1.3	0.2	−1.2
K	Inputs	Synthetic fertilizer	19.3	20.9	1.6	5.9	10.8	4.9	16.7	19.0	2.2
		Manure	11.7	12.1	0.4	6.4	6.1	−0.3	10.7	11.0	0.3
		Seed	Not available								
		Total	31.0	33.0	2.1	12.3	16.9	4.6	27.4	29.9	2.5
	Removal	Crop	24.9	29.6	4.7	13.8	17.2	3.4	22.8	27.2	4.4
	Surplus		6.1	3.5	−2.7	−1.5	−0.3	1.3	4.6	2.7	−1.9

Notes: Values shown are the annual average for 2005–2014 period (P1) and 2015–2021 period (P2) and the difference change between the two periods. Boldface differences are significant (P ≤0.05).
[1] Nutrient balance pools include all inputs, removal in the harvested portion of the crop and the surplus or difference between inputs and removal.
[2] Source pools for inputs are synthetic fertilizers, animal manures, and the nutrient content of planted seed (N and P only). For N, additional input sources include atmospheric deposition and biological N fixation (BNF) and losses via leaching and volatilization are estimated.
[3] Annual estimates from the two time periods were compared with a 2-sample t-test.

◊ *Source:* **FAO.** 2022. *FAOSTAT: Cropland Nutrient Balance.* [Accessed on 1 February 2024]. https://www.fao.org/faostat/en/#data/ESB. Licence: CC-BY-4.0.

In Canada, estimates of cumulative P balance have been used to predict risk posed to water by legacy P that is slowly desorbed from agricultural soils (Reid and Schneider, 2019). Recent studies of regional soil quality have explored the impact of N balance on SOC in maize-based systems with some studies showing no immediate change with fertilizer rate reduction (e.g. Bailey *et al.*, 2023) and others showing agronomic N rates were needed to maintain SOC (Poffenbarger *et al.*, 2017). Although most studies of nutrient balances do not directly measure both N_2O emission and NO_3^- leaching, some evidence suggests increasing SOC increases N_2O emissions while decreasing leaching (Eagle *et al.*, 2017).

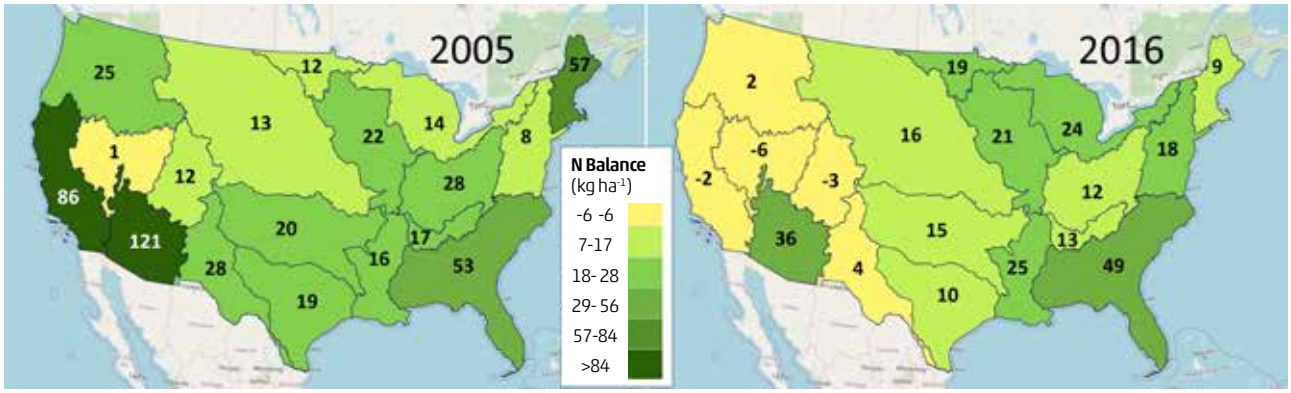
Nutrient imbalances at the field and farm scale remain a reflection of a complex array of socioeconomic factors governing real and perceived uncertainties associated with weather, input costs, expected yields, time lag to return on investment and trade-offs associated with dual prioritization of stewardship and profitability. The N and P surpluses in North America are considered to be limited by transport not source, with transport processes linked to soil properties and anthropogenic modifications, including artificial tile drainage (King *et al.*, 2015). Regional analyses can identify hotspots that drive the persistent surpluses and are critical to the effective use of nutrient balances in policy strategies to effect meaningful changes in land management (Lu and Tian, 2017).

Where cropland surplus nitrogen persists and known unknowns

For North America as a whole, the comparison between estimated annual N inputs to cropland since 2015 and those in the preceding decade (2005 to 2014) shows that the total annual N inputs increased by approximately 12 kg/ha (11 percent) but annual crop removals increased by 14 kg/ha, resulting in similar annual estimates of surplus across the two time periods (38.2 kg/ha for 2005–2014 and 36.1 kg/ha since 2015 (FAO, 2022c). Since 2015, a slight decrease in total N surplus in the United States has been counterbalanced by increases in the Canadian surplus. The FAOSTAT cropland nutrient balance (CNB) estimates for N are accompanied by estimates of losses to volatilization and leaching (FAO, 2022c). Estimates for N losses as NH_3, N_2O and NO_3^- are similar for Canada and the United States, and the combined North America annual losses since 2015 are 20.6 kg/ha (volatilized) and 7.8 kg/ha (leached) of cropland. Collectively, volatilization and leaching losses account for 80 percent of North American annual N surpluses, leaving 7.7 kg/ha without attribution. Tier 1 estimates of total (direct and indirect) emissions at the farmgate (FAO, 2022b) also suggest little change from the last assessment, a direct reflection of the stability in annual cropland N surpluses. Canadian emissions associated with synthetic fertilizer use appear to have increased but regional emissions are dominated by United States emissions averaging 246 kg/ha, a non-significant increase over the preceding decade.

The FAOSTAT CNB estimates have acknowledged limitations but are considered directionally correct (FAO, 2022d). For subregion analysis, the International Plant Nutrition Institute (IPNI) developed the Nutrient Use Geographic Information System (NuGIS) (TFI and PNC, n.d.). The NuGIS database aggregates annual (1982–2016) cropland nutrient balances, theoretically using the same data sources as FAOSTAT but retaining subregion watershed and geopolitical referencing. Analysis of large subregion watersheds indicate N balances have changed most in the west where cropland surpluses in the lower Colorado region have declined from 122 kg/ha in 2005 to 35.6 kg/ha in 2016, reflecting both increased crop removal and decreased synthetic fertilizer use (Figure 12.10).

Figure 12.10. Annual cropland nitrogen balances (kg/ha) for the United States in 2005 and 2016



Refer to the disclaimer on page ii for the names and boundaries used in this map.

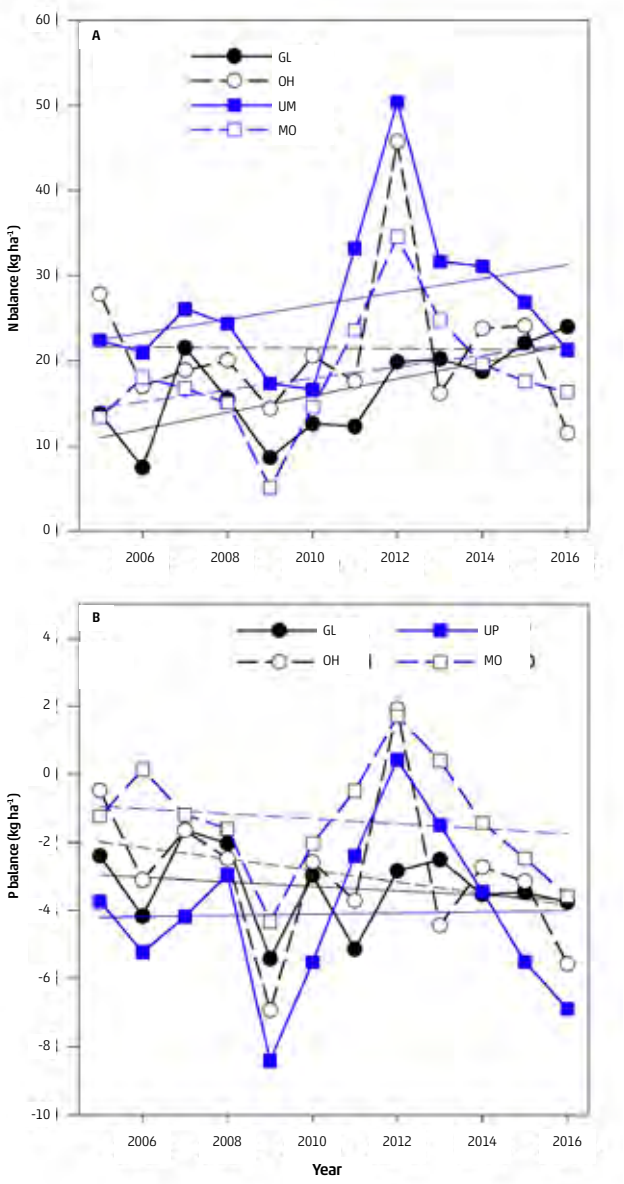
Note: Values shown are for the 18 HUC 2 watersheds in the conterminous United States.

Source: **TFI & PNC.** n.d. *NuGIS: Nutrient Use Geographic Information System. [Accessed on 3 January 2024]. https://nugis.tfi.org/. Licence: CC-BY-3.0*

In the California region of the United States, crop removal has been relatively constant since 2005, but annual N balances declined from 86 to −2 kg/ha, reflecting steady decreases in synthetic fertilizer use. For the major watersheds draining the Midwestern United States corn belt and other major grain producing states in the northern and central plains, year-to-year variation is pronounced but overall trends are flat or only weakly positive (Figure 12.11a and Figure 12.11b). Since 2005, annual surpluses have ranged from 16.4 kg/ha in the Great Lakes Region to 26.9 kg/ha in the Upper Mississippi Region of the United States.



Figure 12.11. Annual cropland nitrogen (A) and phosphorus (B) balances and trend lines for the HUC 2 watersheds from 2005 to 2016 in the United States

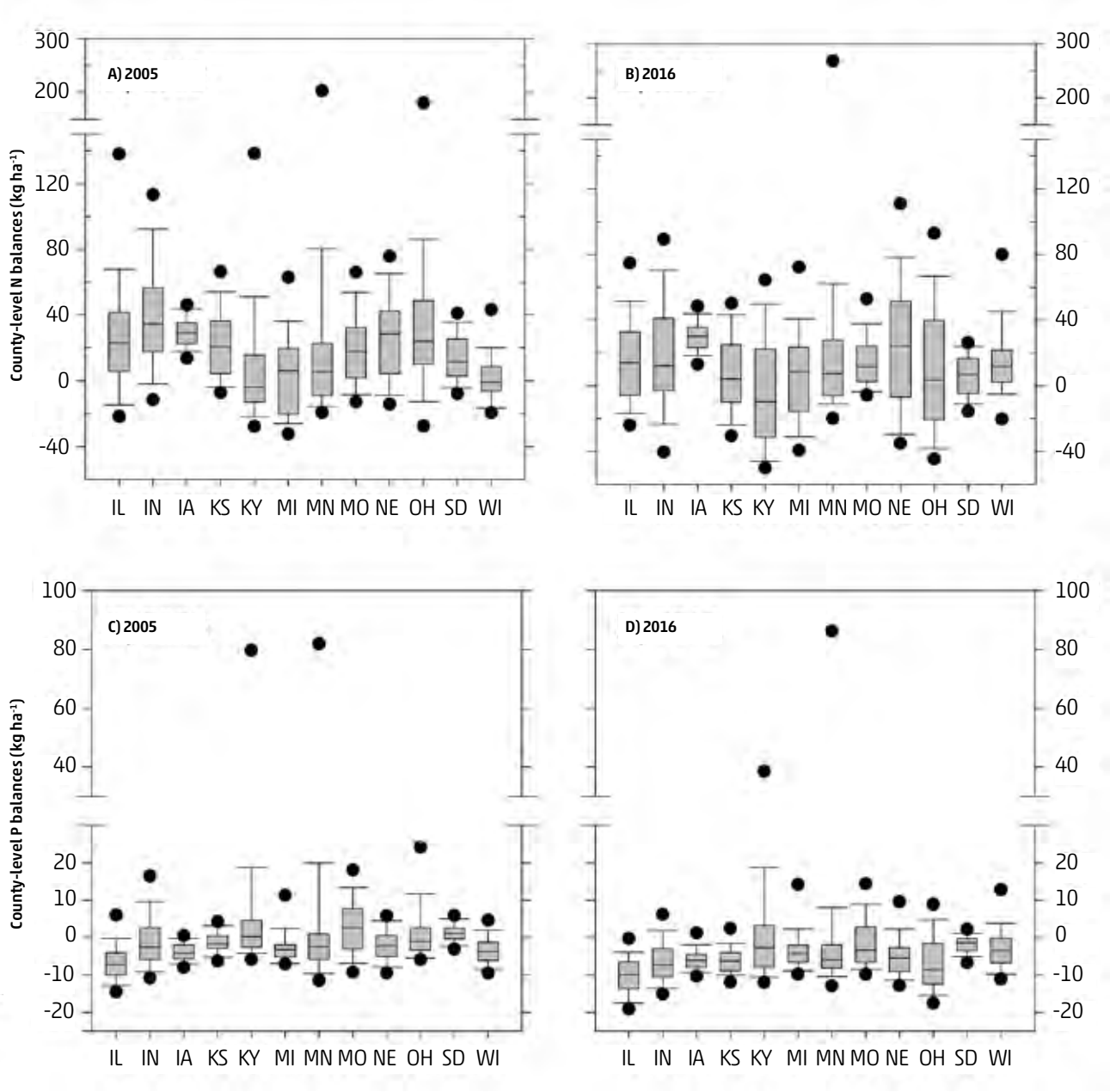


Note: GL = Great Lakes, OH = Ohio, UP = Upper Mississippi, and MO = Missouri (MO) HUC 2 watersheds from 2005 to 2016 (last year of available data).

Source: **TFI & PNC.** n.d. *NuGIS: Nutrient Use Geographic Information System.* [Accessed on 3 January 2024]. https://nugis.tfi.org/. Licence: CC-BY-3.0.

The NuGIS data can be resolved at the county level within states and watersheds, allowing state-level estimates and identification of notable hotspots. As with country-level estimates, the downscaled estimates are likely directionally correct with noted limitations. In 2005, for the United States, N surpluses were greatest in the states of Ohio and Minnesota, at 54 kg/ha (Figure 12.12). The greatest median N balances were in Iowa (30 kg/ha, both years). On average, balances declined by 7 kg/ha over the decade, and median values declined in all states except Wisconsin. A noted challenge in interpreting NuGIS data is that synthetic fertilizer values are for recorded sales and distribution cannot be assumed. Another key criticism of the NuGIS dataset has been that it is not crop-specific and cannot be used to show the impact of an intensively managed cropping system on balances in states that also have less N-intensive crops.

Figure 12.12. Distributions of county-level nitrogen (A and B) and phosphorus (C and D) balances (kg/ha) for states within the Great Lakes, Ohio, Upper Mississippi, and Missouri HUC 2 watersheds in United States



Note: United States states shown are as follows: IL = Illinois, IN = Indiana, KS = Kansas, KY = Kentucky, MI = Michigan, MN = Minnesota, MO = Missouri, NE = Nebraska, OH = Ohio, SD = South Dakota, and WI = Wisconsin. Boxplots show median, 25th and 75th percentiles (extent of grey box), 10th and 90th percentiles (whiskers) and 5th and 95th percentiles (black circles) for all counties within a state. The dotted red reference line indicates a zero balance.

◊ *Source:* **TFI & PNC.** n.d. *NuGIS: Nutrient Use Geographic Information System.* [Accessed on 3 January 2024]. https://nugis.tfi.org/. Licence: CC-BY-3.0.

Without crop specificity and a better characterization of farmer practices, the magnitude of management opportunity for N balance modification is unknown and an impediment to designing effective policies. At present, approximately 60 percent of the United States cropland area used for maize uses a return on fertilizer investment or Maximum Return To Nitrogen (MRTN) approach (Sawyer *et al.*, 2006), that recommends rates based on the price ratio of fertilizer to grain. As the fertilizer to grain price ratio decreases across a realistic range of ratios from 0.2 to 0.05, the N fertilizer at the MRTN can change by as much as 100 kg/ha, with those fertilizer inputs achieving 96 to 99 percent of the maximum, N-unlimited yield.

Analysis of on-farm trials in Illinois by Zhao, Nafziger and Pittelkow (2017) found the MRTN rates for maize corresponded to a fertilizer surplus of 67 kg/ha for N. It was speculated that actual farmer-applied N rates are likely to be 30 kg/ha higher, in keeping with the predictions of farmer practice discussed previously (Table 12.9). The Corn Nitrogen Rate Calculator (CNRC) tool indicated that MRTN rates in Illinois and Indiana range from 109 to 251 kg/ha of N depending on price ratio, preceding crop, and subregion within the state (CNRC, n.d.; Sawyer *et al.*, 2006). Further, the calculator estimated the MRTN as a range of N rates, generating a profit within USD 2.47 per ha of the maximum from applying too little N or too much N. Applying too little N (low MRTN), sacrifices USD 2.47 per ha of profit as more N would lead to a profitable increase in grain yield. Applying too much N (High MRTN), would sacrifice USD 2.47 per ha of profit as the extra N would not make an economical increase in grain yield. This range from low to high MRTN is typically 30 kg/ha of N. Recent surveys found the majority of farmers (including a disproportionately high number managing large areas) either applied the high MRTN or an additional 6 to >11 kg/ha of N beyond the high MRTN (Houser, 2022). Simply reducing rates from high to low MRTN has been suggested to reduce N surpluses to ≤70 kg/ha, and losses to leaching by 20 kg/ha (Tamagno *et al.*, 2021). However, this reduction also increases risk of yield loss.

Table 12.9. Example nitrogen balance estimates from the major maize-producing states of Indiana and Illinois in 2012 in the United States

	2012 state estimates		Example budgets for farmer practice in Indiana (IN) and Illinois (IL)			
Pool	IN	IL	Putnam County (IN)	Whitley County (IN)	Champaign County (IL)	Perry County (IL)
N rate (kg/ha)[1]	226	185	241.9	264.3	218.4	241.9
N in P fertilizer (kg/ha)[2]	n/a	n/a	39	38.4	45.4	32.8
Atmospheric deposition (kg/ha)[3]	n/a	n/a	7.2	7.2	7.2	7.2
Net biological N fixation (kg/ha)[4]	31	28	16.4	17.2	20.7	14.6
Total N inputs (kg/ha)[5]	257	213	304.5	327.1	291.7	296.5
N needs/removal (kg/ha)[6]	134	135	139.3	128.0	151.9	111.9
N fertilizer surplus (kg/ha)[7]	92	50	102.6	136.3	66.5	130
Total surplus (kg/ha)[8]	122	78	165.2	199.1	139.8	184.6
Fertilizer efficiency (%)[9]	59.3	73.0	57.6	48.5	69.6	46.3
Total efficiency (%)[10]	52.1	63.4	45.7	39.1	47.9	37.7

Notes: State estimates are derived from Xia, Kwon and Wander (2021). The example farmer practice budgets are for an intensively managed maize–soybean rotation in counties that were identified as having N surpluses of 50 to 100 kg N/ha (without manure or N credits.)

[1] Data fusion estimates of total application from Xia, Kwon and Wander (2021) for 2012 and a farmer practice of applying at least the high MRTN rate (Sawyer *et al.*, 2006) are based on farmer surveys (Houser, 2022).

[2] Based on the reported rate for maize in the states in 2018 (USDA ERS, 2019). It assumes 47 percent P2O5 and 14 percent N, reflecting the prevalence of monoammonium phosphate (MAP) and diammonium phosphate (DAP) use.

[3] United States values from Table 12.8

[4] Empirical estimates of N credits are for 2012 data (Xia, Kwon and Wander, 2021). For counties, the yield-scaled estimate of biological nitrogen fixation (BNF) in excess of N that is removed in soybean grain in 2021 (USDA & NASS, 2024). Estimated as 7.35 percent of grain N after Herridge *et al.* (2022).

[5] Sum of rows A–D.

[6] State estimates of crop need from Xia, Kwon and Wander (2021) which is not directly equivalent to FAOSTAT crop removal. County estimates of N removed in grain is after Ludemann *et al.* (2022) and reported maize yields in 2022 2021 (USDA & NASS, 2024).

[7] Difference between rows A and F.

[8] Difference between E and F.

[9] Fraction in F of A.

[10] Fraction in F of E.

◊ *Sources:* See References.

Effective use of the N balance approach to improve regional nutrient stewardship also likely requires a more nuanced understanding of inputs from the biological fixation of atmospheric N^2 (BNF) and legume N returned to soil in crop residues. Peoples *et al*. (2021) suggested that due to FAOSTAT adopting an updated, yield-scaled approach to estimating BNF in major grain legumes, the estimated contributions for several crops including soybean had been increased by 13 to 15 percent on a land area basis. However, Herridge *et al.* (2022) found only a 3 percent increase in soybean BNF but noted the dominant role soybean now plays in global BNF estimates, a reflection of both yield and land area increases. Given the growing importance of soybean in North America production systems as well as current sustainability narratives that encourage legume inclusion in cereal systems (Barrios *et al.*, 2020, Liu *et al.*, 2023), legume influences on system N balances require clarification.

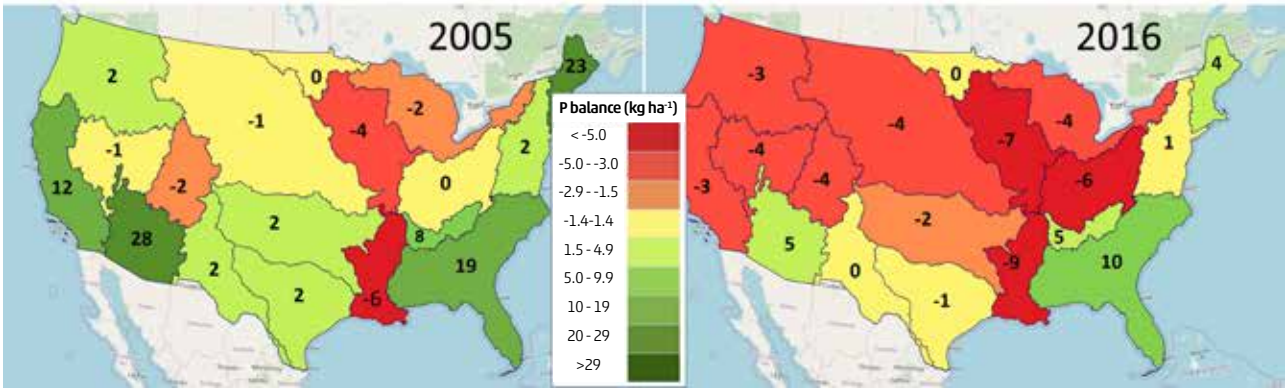
Other ambiguities and knowledge gaps in down-scaling include N fertilizer applied to legumes and P management with ammoniated P fertilizers. Recent United States records indicate that an additional 30 kg/ha of synthetic N fertilizer is applied annually to soybean (USDA ERS, 2019). In the United States, N fertilizer impacts on soybean yields are minimal and not profitable (Mourtzinis *et al*., 2018). Approximately 90 percent of the country's phosphate fertilizer consumption is ammoniated (11 to 20 percent N by mass [USDA ERS, 2019]). In many regional production systems, soil P fertility is managed with post-harvest, autumn fertilizer applications and this N (potentially an additional 40 kg/ha to maize), apparently remains unrecognized or generally overlooked in regional N balance estimates and farmer recommendations alike (Decock, 2014; Margenot and Lee, 2023). In summary, the general adherence to current recommendations could result in total N inputs to North American farm fields of 300 kg/ha, with surpluses of 200 kg/ha, as illustrated for maize with corresponding low fertilizer (45 to 60 percent) and total use (35 to 50 percent) efficiencies (Table 12.9) that are well below desired targets.

Legacy phosphorus surpluses continue to drive environmental degradation

Balances for cropland P in the North America region indicate that surpluses are minimal or even negative (TFI and PNC, n.d.). In the United States, the average annual P crop removal since 2015 (14.6 kg/ha) now exceeds total inputs from synthetic fertilizer and manure (14.3 kg/ha), and cropland P balances are slightly negative (Table 12.8). In contrast, Canadian cropland surpluses remain positive, increasing in most recent years when compared to the preceding decade, which reflects a 46.6 percent increase in synthetic fertilizer use. Across North America, cropland P surpluses are only 0.2 kg/ha, representing a decline of 1.2 kg/ha from the preceding decade.

For subregions within countries, NuGIS balances suggest some major crop producing regions now have notable negative P balances. The major watersheds draining the Midwestern United States corn belt had negative annual P balances from −0.5 kg/ha in Ohio to −3.7 kg/ha in the Upper Mississippi in 2005: declining further to P balances from −3.6 kg/ha in Missouri to −6.9 kg/ha in the Upper Mississippi in 2016 (Figure 12.13). However, year-to-year variability is pronounced with no clear trend in the last decade (Figure 12.12b). Further, examination of county level data for states within subregions reveals the same highly skewed distributions in balances observed for N (Figure 12.12). Median P balances were all negative in 2016 and lower in every corn belt state when compared to 2005. In contrast, spatially explicit analyses of grain producing areas within the Canadian provinces in 2018 identified localized annual surpluses of 5 kg/ha in southern Alberta, 9 kg/ha in Ontario and 16 kg/ha in Manitoba (Wang, Qi and Bennett, 2022). Although still positive, balances in Ontario were much lower than the peak surpluses observed in 1978.

Figure 12.13. Annual cropland P balances (kg/ha) for the United States in 2005 and 2016



Refer to the disclaimer on page ii for the names and boundaries used in this map.

Note: Values shown are for the 18 HUC 2 watersheds in the conterminous United States.

Source: **TFI & PNC.** n.d. *NuGIS: Nutrient Use Geographic Information System.* [Accessed on 3 January 2024]. https://nugis.tfi.org/. Licence: CC-BY-3.0.



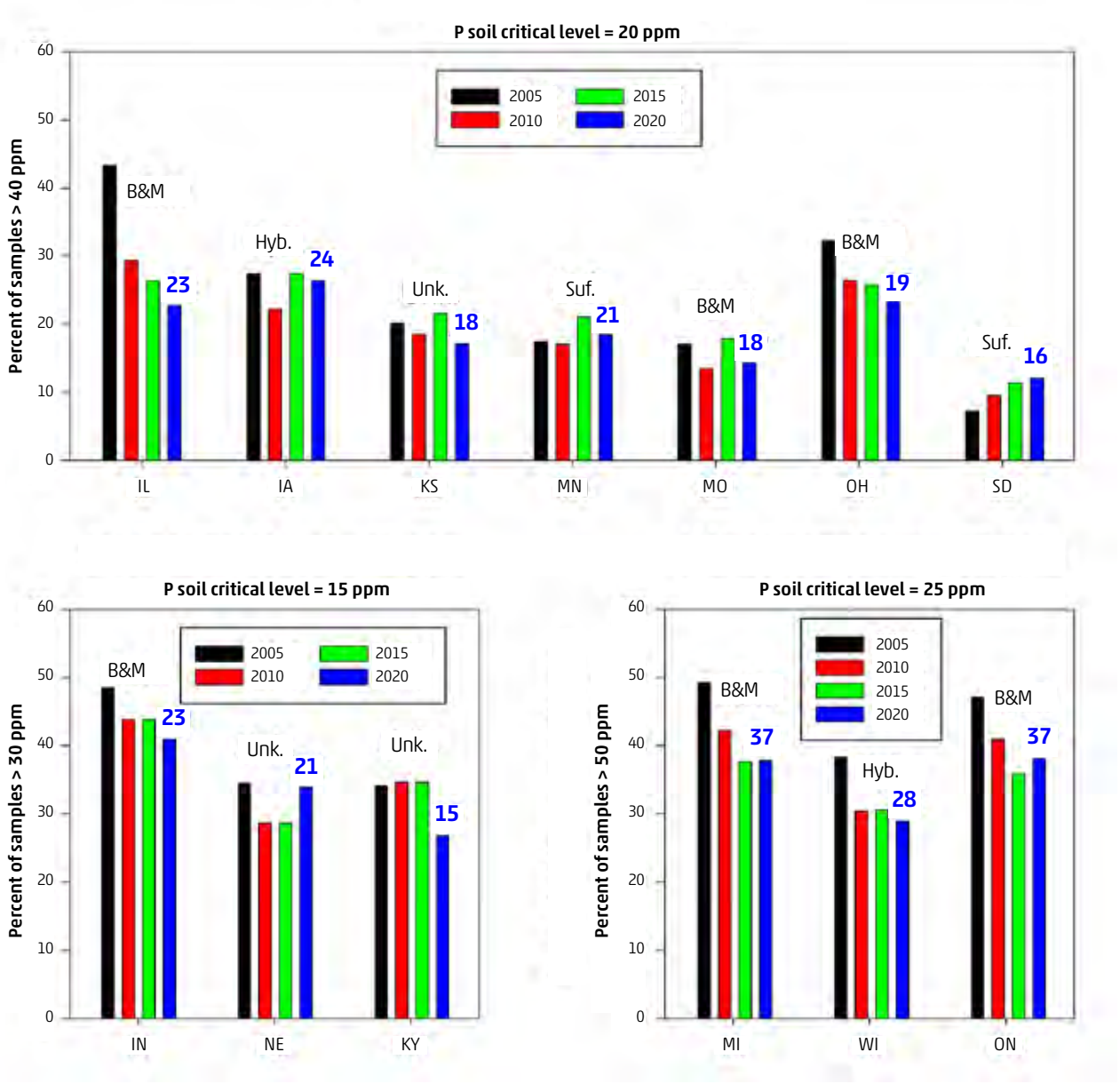
A high-yielding maize–soybean rotation will remove 3.4 kg of P per tonne of maize and 8.1 kg of P per tonne of soybean grain (FAO, 2022a), and, at the farm gate, periodic large surpluses are to be expected and are not an indicator of poor stewardship on soils without long histories of year-over-year surpluses. However, many cropland soils in North America have long histories of surpluses. Because North American farmers routinely test soils as part of fertilizer programmes, and as a requirement to receive benefits from conservation programmes, soil test results can be useful in assessing the extent of persistence of legacy P (Reid, Schneider and Joosse, 2019). Since 2001, The Fertilizer Institute (TFI) has compiled a soil test summary (STS) of voluntarily contributed results from regional public and private labs in five-year increments (TFI, 2020). For maize and soybean in North America, the soil test P (STP) level required for agronomic sufficiency, or critical level (CL), varies by state or province but the 2020 survey shows median STP values are at or above the CL (Figure 12.14). This indicates that the majority of farms tested are unlikely to observe a yield response to fertilizer in the near term. More importantly, however, the proportion of samples testing at levels two-fold or more than the CL remains substantial, although with some incremental decreases since 2005. Recent research demonstrates that the agronomic CL can correspond to thresholds at which desorption of P from soils becomes an environmental concern (Brouder and Volenec, 2023; Welikhe *et al.*, 2020).

Legacy phosphorus surpluses continue to drive environmental degradation

Balances for cropland P in the North America region indicate that surpluses are minimal or even negative (TFI and PNC, n.d.). In the United States, the average annual P crop removal since 2015 (14.6 kg/ha) now exceeds total inputs from synthetic fertilizer and manure (14.3 kg/ha), and cropland P balances are slightly negative (Table 12.8). In contrast, Canadian cropland surpluses remain positive, increasing in most recent years when compared to the preceding decade, which reflects a 46.6 percent increase in synthetic fertilizer use. Across North America, cropland P surpluses are only 0.2 kg/ha, representing a decline of 1.2 kg/ha from the preceding decade.

For subregions within countries, NuGIS balances suggest some major crop producing regions now have notable negative P balances. The major watersheds draining the Midwestern United States corn belt had negative annual P balances from −0.5 kg/ha in Ohio to −3.7 kg/ha in the Upper Mississippi in 2005: declining further to P balances from −3.6 kg/ha in Missouri to −6.9 kg/ha in the Upper Mississippi in 2016 (Figure 12.13). However, year-to-year variability is pronounced with no clear trend in the last decade (Figure 12.12b). Further, examination of county level data for states within subregions reveals the same highly skewed distributions in balances observed for N (Figure 12.12). Median P balances were all negative in 2016 and lower in every corn belt state when compared to 2005. In contrast, spatially explicit analyses of grain producing areas within the Canadian provinces in 2018 identified localized annual surpluses of 5 kg/ha in southern Alberta, 9 kg/ha in Ontario and 16 kg/ha in Manitoba (Wang, Qi and Bennett, 2022). Although still positive, balances in Ontario were much lower than the peak surpluses observed in 1978.

Figure 12.14. The percent of soil samples in the 2005, 2010, 2015 and 2020 soil test samples that tested at levels twice the critical level for phosphorus in soils



Notes: United States states shown are as follows: Il = Illinois, IN = Indiana, KS = Kansas, KY = Kentucky, MI = Michigan, MN = Minnesota, MO = Missouri, NE = Nebraska, OH = Ohio, SD = South Dakota, WI = Wisconsin and the Canadian province ON = Ontario.

All the states and province are aggregated by their respective critical levels (15, 20 and 25 ppm) for a maize–soybean rotation. State/province fertilizer philosophies of build and maintain (B&M), sufficiency (Suf.) and hybrid (Hyb. [a combination of B&M and Suf.]) are shown. States without a clearly identifiable philosophy for recommendations as determined by Lyons *et al.* (2023) are identified as unknown (Unk.). The number shown in blue is the median soil test P value observed in each state in 2020.

◊ *Sources:* **TFI.** 2020. *Soil Test Levels in North America.* [Accessed on 3 January 2024]. https://soiltest.tfi.org/. Licence: CC-BY-3.0.

◊ **Lyons, S. E., Clark, J. D., Osmond, D. L., Parvej, M. R., Pearce, A. W., Slaton, N. A. & Spargo, J.T.** 2023. Current status of US soil test phosphorus and potassium recommendations and analytical methods. *Soil Science Society of America Journal*, 87(4): 985–998. http://dx.doi.org/10.1002/saj2.20536

In North America, high levels of STP often reflect cropland soils proximate to CAFOs where manure disposal requirements have led to historic applications far in excess of plant demand (Miralha, Sidique and Muenich, 2022; Reid, Schneider and Joosse, 2019). In the United States, state regulations for thresholds at which manure applications must cease are often far in excess of 100 parts per million STP and many times above the agronomic critical level (Qin and Shober, 2018). Elsewhere, historic and even existing recommendation frameworks that are yield-protective and attempt to build and maintain soils at non-limiting P levels have encouraged excess P application as a risk minimization strategy. In the United States, this build-and-maintain philosophy of soil management has been pursued in Illinois, Indiana, Michigan, Missouri and Ohio, and whereas Minnesota and South Dakota pursue a more conservative sufficiency approach of matching application rates with crop removal (Lyons *et al.*, 2023; Culman *et al.*, 2020). The build-and-maintain approach is apparent in the comparatively higher percentages of samples testing greater than two times above the CL (Figure 12.14). In Ontario, Canada, the high percentages of samples testing twice above the CL reflects the legacy of build-and-maintain applications, even though present-day rates are now lower (Reid, Schneider and Joosse, 2019). Creating both spatially specific and crop-specific maps of STP levels that can be reconciled with P balances at the farm gate could facilitate better targeting of mitigation programmes.

Other challenges

Meta-analysis has emerged as an important tool for synthesizing studies to quantify management efficacy at scale (e.g. Tamagno *et al.*, 2019; Eagle *et al.*, 2017). Such syntheses are foundational to effectively and efficiently target policies and conservation practice implementation to systems and regions where nutrient mismanagement is driving the most environmental damage (Chai, Panenell and Pardey, 2023). Although now mandated by many funding agencies, sharing of experimental data is not yet common practice in agriculture and accessibility of research data remains a barrier to high quality meta-analyses. A much larger challenge, however, remains the access to the farm management and assessment data that characterizes farmer practices (Brouder *et al.*, 2019). The NuGIS database is no longer being updated and its curation is uncertain. The challenges associated with fusing data range from limited or no data available on as-applied farmer rates and missing state-level data, to discrepancies among theoretically redundant data sources and a host of uncertainties associated with estimating crop N sourced from manure (Xia, Kwon and Wander, 2021).

Tools for visualizing data are rapidly emerging, but their impactful use requires easy access to high quality data on nutrient balances. Given the current investments in programmes assessing C in soils and agricultural greenhouse gas fluxes, the immediate goals should be to improve the quality and the ease of use of knowledge bases tracking cropland nutrient balances, along with that of general soil fertility.



12.3.3 Ecoregional assessment of soil carbon change in the North America region

North America includes diverse ecoregions, from deserts in the south to tundra in the north, from temperate forests in the east to mountain and coastal forests in the west, with grassy plains between (US EPA, 2024) (Figure 12.15). Soil C stocks vary widely, from <10 to >100 kg/m^2 (Sothe *et al.*, 2022), reflecting differing intensities of soil-forming factors (Jenny, 1941) (Table 12.10). The total soil C reserves in Canada have been estimated to be 572 billion tonnes of C to a 2 m depth (Sothe *et al.*, 2022), of which substantial portions are in the northern boreal and permafrost regions (Turestsky *et al.*, 2020). In all regions, soil C reserves have been influenced by human activity, often with accelerating intensity, though the extent, direction and mode of such influence vary. For example, while soil C in croplands, grazing lands and some forestlands are subject to direct human influence, soil C in northern peatlands and boreal forests is influenced mostly indirectly, through anthropogenic climate change (Antala *et al.*, 2022).

Figure 12.15. Ecological regions of the North America region



Refer to the disclaimer on page ii for the names and boundaries used in this map.

◊ *Source:* **US EPA.** 2024. *Ecoregions of North America.* [Cited 16 March 2025]. Washington, DC. https://www.epa.gov/eco-research/ecoregions-north-america

Table 12.10. Summary table for percentage of soil organic carbon stocks in different ecoregions and summary of trajectory, dominant threats and corrective measures

Ecoregion or land use	North America soil C stock[1] (%)	Current trajectory (overall)	Dominant threats	Availability of direct corrective measures[2]	Mitigation practices
Permafrost	30	↘	Climate change	0	Minimizing climate change
Boreal forest and peatland	25	↘	Climate change and fire	1	Minimizing climate change, avoided cultivation and sustainable harvesting
Temperate forest	15	→	Unsustainable management and cultivation	1	Sustainable harvesting, avoided cultivation and fire management
Cropland	10	→	Erosion, disturbance, bare fallowing and inadequate residue return	3	Reduced disturbance, perennialization, intensification of biomass return, regrassing, reforestation, agroforestry, and biochar
Grassland	15	↘	Cultivation and erosion	2	Sustainable grazing, avoided cultivation and nutrient management
Wetlands	3	→	Drainage	2	Restorative practices
Other	2	Variable			Various

[1] Authors' consensus estimate (approximate).

[2] Numerical scale from 0 (no direct measures available) to 3 (widespread, demonstrably effective measures available).

In this summary, the dominant threats, current trajectories, opportunities for enhancement and uncertainties for each broad ecoregion are presented, focusing largely on managed lands, where restorative practices can be directly implemented. Preserving soil C in other ecoregions is at least as important – indeed, soil C stocks in North America are dominated by those in unmanaged lands – but averting further losses is often a global mandate (notably mitigating climate change) and is considered elsewhere. Although inorganic C in arid soil may also be artificially modified (Sharififar *et al.*, 2023), The focus in this case is on SOC. For clarity, SOC is first considered in six prominent groupings and then summarized with crosscutting issues relevant to all areas.

Arctic permafrost

Soils of the Arctic circumpolar permafrost region (including also some boreal forests) contain about 1 460 to 1 600 billion tonnes C to a depth of 3 m (Schuur *et al.*, 2022), of which a major share occurs in North America. This vast pool of soil C, representing a substantial portion of North American soil C stocks, has accumulated over many centuries due to cool temperatures and waterlogging that have inhibited litter C decay. Climate change is the major threat to C stocks in this region.

Boreal forests

Boreal forests occupy a broad swath in upper latitudes across North America, from Alaska to Newfoundland and Labrador, covering more than 6 million km^2 (Natural Resources Canada, 2021;

Wells *et al.*, 2020) (Figure 12.14). Owing to their expanse and remoteness, only a portion of boreal forests are subjected to direct human management, such as harvesting and fire suppression. Estimates of soil C stocks, including surface organic layers (litter, fermented and humic [LFH], or O soil horizons), are prone to large uncertainties because of low density and variable depth of sampling. Sothe *et al.*, (2020) estimated that peatlands account for about a third of Canada's total soil C stocks to a 2 m depth.

Changes in soil C stocks reflect differences between C inputs (primarily plant litter) and losses (primarily from respiration, fire, and dissolved organic C [DOC] fluxes to rivers), are all influenced, either directly or indirectly, by human activities. For arctic permafrost, the prominent factor is ongoing climate change, which affects soil C through multiple pathways. First, climate warming often enhances net primary productivity, leading to an increased litter C return to soil, although any gains may be offset by accelerated respiration (Castañeda-Gómez *et al.*, 2023) so that warming may accelerate C cycling without changing soil C stocks (Paré *et al.,* 2022; Ziegler *et al.*, 2017). Second, climate change can increase the frequency and intensity of wildfires (Wang *et al.*, 2020), leading to C losses but conceivably, also to the enhanced stability of the remaining SOC (Pelligrini *et al.*, 2022). Third, climate change can influence soil C through migration of insect pests such as the mountain pine beetle and spruce budworm, affecting C cycling (Pureswaran *et al.*, 2015; Woodman *et al.,* 2021).

Current trajectories of soil C in boreal forests are highly variable and uncertain. Some modelling analyses predict that boreal forests will undergo a shift from C sink to source (Miquelajauregui, Cumming and Gauthier, 2019). An assessment of major Arctic rivers found no increases in DOC fluxes over recent decades (Tank *et al.,* 2023).

Boreal biomes also include expanses of peatlands, both forested and unforested (Webster *et al.,* 2018). These organic soils hold substantial reserves of soil C, with an estimated 190 billion tonnes C to a 2 m depth in Canada, according to Sothe *et al.* (2022). The boreal peatlands and associated forests of northern Canada are among Earth's "irrecoverable C reserves" (Noon *et al.,* 2022).

Because of their expanse and vulnerable SOC reserves, boreal forests play crucial roles in maintaining the global C cycle. Concerted research is needed to understand C fluxes, quantify net C exchanges, address emerging issues, such as invasive earthworms (Lejoly, Quideau and Laganière, 2021) and develop appropriate management strategies.

Temperate forests

Temperate forests stretch from the southern boundary of the boreal forest zone across Canada, through the United States and into the montane forests of northern and central Mexico. They once covered most of the eastern half of the continental United States (the Rocky Mountain, Sierra Nevada and Cascade ranges in the west, and the Pacific coast region from central California to southeast Alaska). Coniferous species are dominant in the western montane forests, on the northern edge of the eastern temperate forests and in intensely managed plantations in the southeast United States, while deciduous species predominate in eastern forests. Soil C stocks (0–30 cm) are highly variable, ranging from <2 kg/m^2 in warm, dry and rocky soils in parts of montane forests in the southwestern United States and northern Mexico, to >14 kg/m^2 in cold, moist environments of the Pacific northwest (Thompson *et al.,* 2020).

Many temperate forests in southeast Canada and the eastern United States were converted to annual cropland and grassland following the European settlement, leading to substantial losses in topsoil C (Houghton and Nassikas, 2017). However, since the mid-nineteenth century, cropland abandonment has led to forest regrowth, which, along with reduced harvest of primary forests, generated a substantial net C sink, predominantly in wood biomass but also in SOC stocks. The current C sink on forested land in the continental United States is estimated at ~200 million tonnes per year (US EPA, 2024).

Opportunities for management-induced SOC gains in forests are probably limited, except for those in early phases of regrowth (Johnson and Curtis, 2001), as most forest C enhancement projects focus on increasing biomass C stocks (Kaarakka *et al.*, 2021). The amount, form and distribution of soil C pools are vulnerable to altered disturbance regimes linked to climate change, such as increased fire frequency and severity (Adkins *et al.*, 2019), invasive species such as earthworms (Frelich *et al.,* 2006; Jennings and Watmough, 2016), and changes in species composition (Morin *et al.*, 2018).

Grasslands and grazing lands

Grasslands co-evolved with large herbivores (Milchunas and Lauenroth, 1993) and are used by humans in the region primarily for the grazing of domestic livestock. A large share of grassland in North America has also been cultivated for annual crops due to the favourable climate and fertile soil, leading to substantial losses of SOC stocks (Paustian *et al.*, 1997; Sanderman, Hengl and Fiske, 2017).

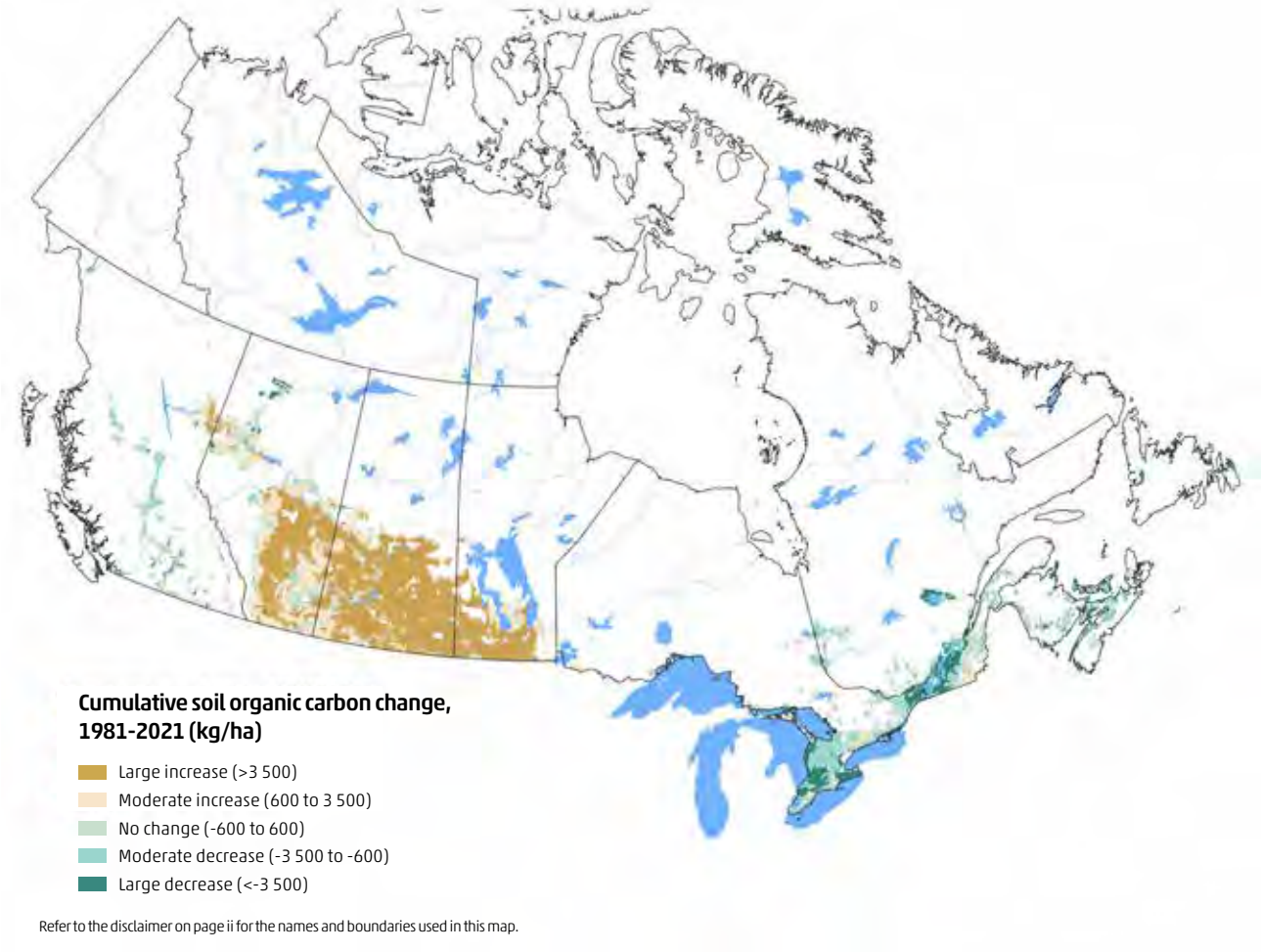
Well-managed grasslands maintain high soil C stocks, but many have been overgrazed, which, together with poor nutrient and vegetation management (Bardgett *et al.*, 2021), has resulted in losses of soil C (Eze, Palmer and Chapman, 2018). Although severely degraded grasslands are not easily recovered, SOC in moderately degraded grassland can often be restored (Conant *et al.*, 2017; Bai and Cotrufo, 2022). Adding fertilizer, enhancing vegetative biodiversity and sowing legumes, can increase productivity and therefore increase the C inputs to soils, yielding SOC gains of around 0.5 t/ha per year (Conant *et al.*, 2017). Rotational grazing (such as adaptive multipaddock [AMP] grazing) may increase SOC stocks even on non-degraded pastures of mesic grasslands (Conant, Six and Paustian, 2003; Mosier *et al.*, 2021; Teague *et al.*, 2011), but may not improve SOC stocks in more widespread semi-arid and arid systems (Briske *et al.*, 2011).

Bai and Cotrufo (2022) estimated an annual "achievable" sink capacity of 3.4 billion tonnes CO^{2eq}, with the widespread adoption of favourable management globally, but this sink capacity would attenuate after two to three decades.

Croplands

Croplands are ecosystems devoted to arable farming for food and other commodities and occupy about 6 percent of Canada's terrestrial area and 29 percent of the United States area (World Bank 2021; Winters-Michaud *et al.*, 2024). Historically, croplands have lost appreciable soil C (Lal, 2018; Paustian *et al.*, 2019a), but losses have now typically waned, and, in some regions, soil C is gradually increasing. In Canada, for example, soil C is typically increasing slowly in the prairie region (~80 percent of the national cropland area) (Cerkowniak *et al.*, 2016) but is declining in eastern Canada, where perennial crops are giving way to annual crops (Samson, Angers and Poirier, 2023; AAFC, 2022) (Figure 12.16). Gains of soil C in the prairie region may be attributable partly to increasing crop yields that enhance C inputs to soil (Fan *et al.*, 2017). A modelling study by Ogle *et al.* (2023a) estimated that SOC stocks in United States croplands increased annually from 1996 to 2015, with highest rates in 1996 (16.6 million tonnes annually) and lowest in 2015 (10.6 million tonnes annually) (Figure 12.17).

Figure 12.16. Soil organic carbon change in Canada between 1981 and 2021



Refer to the disclaimer on page ii for the names and boundaries used in this map.

◊ Source: **Cerkowniak, D., McConkey, B.G., Smith, W.N. and Bentham, M.J.** (2026). *Soil Organic Matter Indicator: Agri-Environmental Indicators Report - The Environmental Sustainability of Canadian Agriculture, Census Year 2021*. Agriculture and Agri-Food Canada, Figure 3, p. 9. Available at: https://agriculture.canada.ca/atlas/data_donnees/agriEnvironmentalIndicators/supportdocument_documentdesupport/en/Soil%20Organic%20Matter%20Technical%20Chapter%202021.pdf

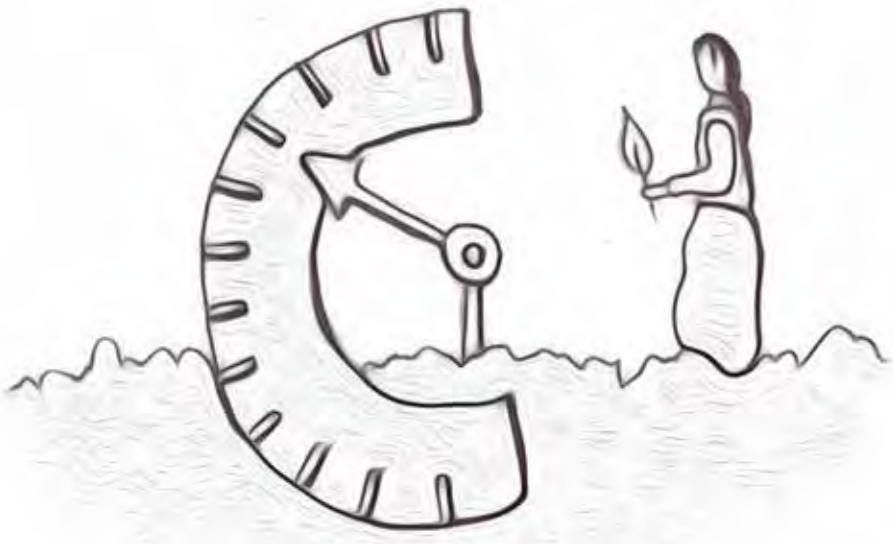
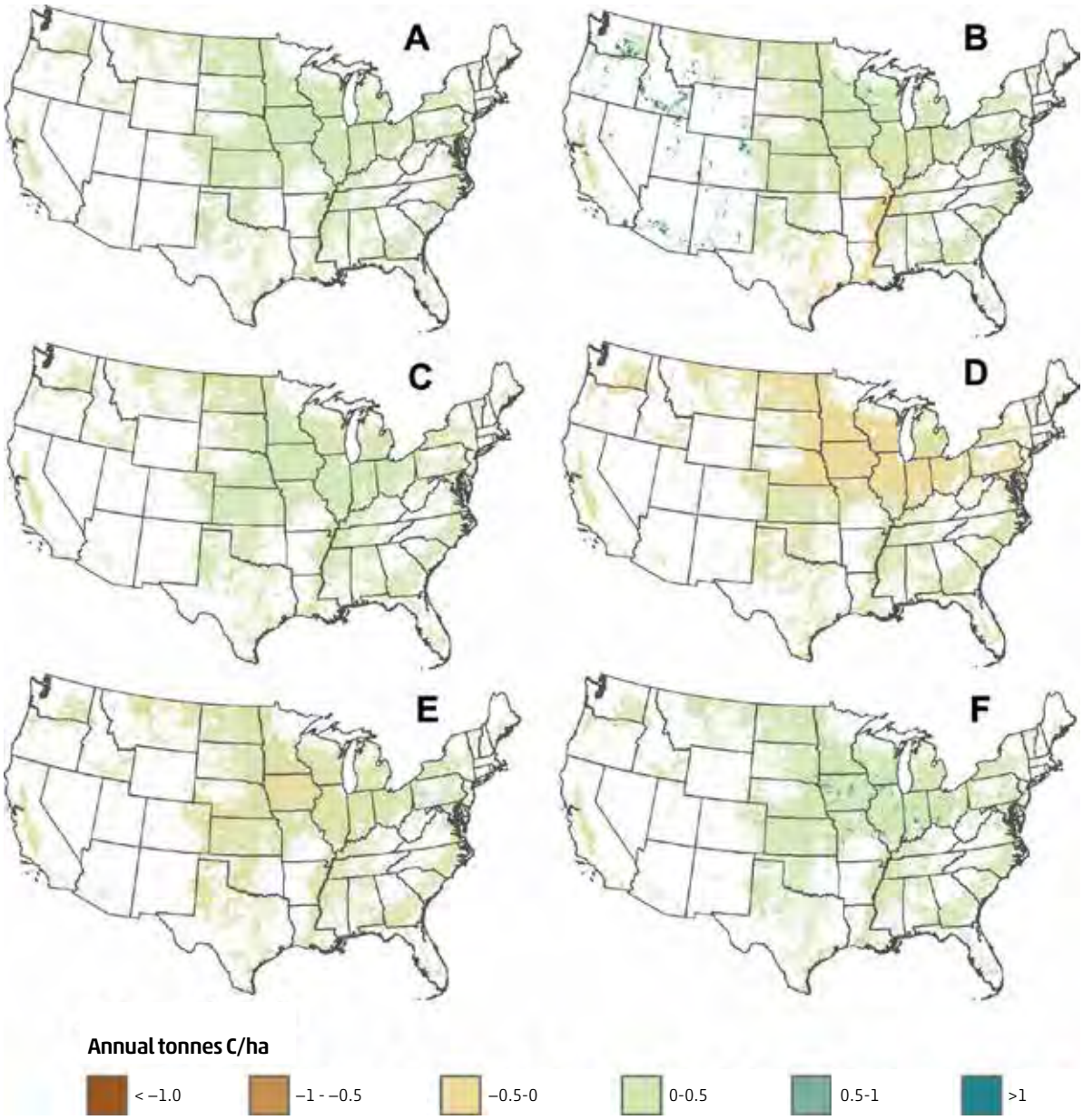


Figure 12.17. Average annual soil organic carbon stock changes (tonnes of carbon per hectare) from 1995 to 2015 in the United States an interpolated 5 km grid for different management practices



Refer to the disclaimer on page ii for the names and boundaries used in this map.

Notes: A: Conservation tillage. B. Manure amendments. C. Winter cover crops with herbicide termination. D. Winter cover crops with tillage termination. E. Hay and pasture in rotation with annual crops. F. Set-aside cropland from production. United States values from Table 12.8.

◊ *Sources:* **ESRI.** *n.d. ArcGIS release 10.7.1.* Redlands, USA. https://www.esri.com/en-us/home

◊ **Ogle, S.M., Breidt, F.J., Del Grosso, S., Gurung, R., Marx, E., Spencer, S., Williams, S. & Manning, D.** 2023a. Counterfactual scenarios reveal historical impact of cropland management on soil organic carbon stocks in the United States. Scientific Reports, 13(1): 14564. https://doi.org/10.1038/s41598-023-41307-x

Croplands, due to past losses and intensity of management, are seen as potential opportunities for enhancing soil C reserves (Lal, 2020). Broadly speaking, soil C in a given place increases when photosynthetically derived C inputs exceed losses via respiration, DOC flow and erosion (Paustian *et al.*, 2019a). There are many studies measuring potential gains in soil C from "improved" agricultural practices and several broad strategies are presented here, with selected examples.

One approach is to reduce the intensity of disturbance, notably by adopting no-till systems. Such systems can often increase soil C (Sperow, 2020; Blanco-Canqui *et al.*, 2021; Prairie, King and Cotrufo, 2023), especially in semi-arid conditions (Liang *et al.*, 2020), but benefits are variable when the full soil profile is considered (Franzluebbers and Steiner, 2016; Franzluebbers *et al.*, 2021d; Wuest *et al.*, 2023).

A second strategy is to use perennial species, which have longer photosynthesizing seasons and allocate more C below ground (Ledo *et al.*, 2020; Singer, Franzluebbers and Karlen, 2009; Crews, Carton and Olsson, 2018; Crews and Rumsey, 2017). The most productive perennials are often used as forage for livestock, so potential soil C gains (or losses) are related to livestock density (Samson, Angers and Poirier, 2023). In some regions, agroforestry and planting of hedgerows help re-perennializing croplands and augment soil C (Stevenson *et al.*, 2024; Chiartas *et al.*, 2022). Perennial food grain crops offer promising prospects (Kim *et al.*, 2022), but economic and agronomic feasibility remain years away.

Third, intensifying cropping systems can often increase SOC by mimicking perennials (King and Blesh, 2018). Double cropping and the use of cover crops, for example, can increase C inputs to soil (Seifert and Lobell, 2015; Farmaha, Sekaran and Franzluebbers, 2022; Zhang *et al.*, 2020), but competition for water sometimes limits its adoption (Ghimire *et al.,* 2018). Soil C can also be enhanced by using improved crop rotations, plant varieties, plant nutrition, irrigation, drainage design, and other innovations (Paustian *et al.*, 2019a; Fan *et al.*, 2019; Castellano *et al.*, 2019; Ashton *et al.*, 2023). This approach could be abetted by developing genotypes designed to allocate more C below ground (Rizi and Mohammadi, 2023; Poffenbarger *et al.*, 2023). Often the most effective way of restoring soil C in croplands is by permanently re-establishing native vegetation, such as grasses (Bremer *et al.*, 2022; Núñez and Schipanski, 2023).

In all cases, approaches for conserving soil C need to be carefully tuned to local conditions and ecological histories. For example, the benefits of no-till farming for soil C depends on soil and climatic conditions (Liang *et al.*, 2020). Soil C responses to improved practices are also usually temporary, typically waning after one or several decades, as soils approach a new quasi-equilibrium (Minasny *et al.*, 2017; Thapa *et al.*, 2023).

Modelling studies project that soil C can often be maintained or increased in croplands, despite climate change and other threats, although rates and direction of change can vary across ecosystems (FAO, 2022e). For example, estimates of potential increases in soil C with improved management include: 45 to 98 million tonnes C per year for United States croplands (a total of 157 million ha) (Chambers *et al.*, 2016); 140.1 million tonnes of CO_2 per year over twenty years for the conterminous United States (as shown by Sperow [2020], using 2019 IPCC factors); ~14 billion tonnes CO_{2e} by the year 2100 for United States croplands (as estimated by Robertson *et al.* [2022], partly by SOC accrual); and 19.4 million tonnes CO_{2e} annually for United States croplands (Aragon *et al.*, 2024).

Despite widespread opportunities for enhancing soil C in North American croplands, uncertainties and obstacles remain, including socioeconomic barriers (Amundson and Biardeau, 2018), and potential trade-offs between soil C gain and other GHG emissions (Schlesinger, 2022). Future gains may also be suppressed by growing demands for alternate uses of plant biomass, reducing its return to soil (Franzluebbers, 2015a).

Other land uses

Other ecoregions and land-use categories, while occupying smaller areas, also hold significant soil C reserves subject to human influence and management. These include:

- Urban and peri-urban lands: With rapid urbanization, increasing areas of land now fall under urban use, resulting in potential loss of soil C but also creating opportunities for regenerating soil C in green spaces (Lorenz and Lal, 2015) (see Section 12.3.5).

- Wetlands: Restoring wetlands, previously drained for cultivation, can recover lost C, though the net benefit for reducing radiative forcing may be diminished by enhanced emission of CH_4 (Hemes *et al.*, 2015).
- Deserts and other arid lands: Deserts generally have low soil C stocks per unit area but still hold significant stocks collectively because of their expanse (Lal, 2019b). Restoring lost soil is constrained by low NPP.

Crosscutting and overarching themes

Impacts of climate change on soil carbon

Climate change and rising CO_2 concentrations may elicit both negative and positive feedback on soil C stocks (Amundson *et al.*, 2015). For example, elevated temperatures and CO_2 may promote net primary productivity, thereby enhancing SOC. Conversely, SOC may decline under a changing climate due to moisture constraints on NPP, migration of pests and increasing fire intensity, or, more directly, because of accelerated decay of existing C reserves (Hursh *et al.*, 2017; Haaf, Six and Doetterl, 2021; Crowther *et al.*, 2016; Hicks Pries *et al.*, 2017; Johnson and Henderson, 2018).

Persistence of stored carbon in managed soils

A critical assumption underlying the mitigative benefits of soil C sequestration is that newly stored C will be withheld from the atmosphere for long time periods and therefore the "sequestered" C will exhibit "permanence". However, recently augmented SOC is not permanently "locked away" but depends on continual replenishment (Kleber *et al.,* 2021) and the continuation of practices that led to SOC buildup.

Land-use change

Land-use change is a major driver of C stock change, globally (Sleeter *et al.*, 2018; Zhang *et al.*, 2021). The coopting of forests, grasslands and wetlands for food production almost invariably elicits substantive soil C losses (Sanderman, Hengl and Fiske, 2017; Amundson, 2022; Liang *et al.*, 2023). Reversing the process by revegetating cultivated lands to their precultivation history through reforestation, regrassing or reflooding, is often the most effective way of restoring previously lost SOC, but carries a risk of "leakage", whereby increasing SOC in one place may cause degradative land-use change elsewhere to replace lost productivity (Reay, 2020; Ogle *et al.*, 2023b; Don *et al.*, 2024).

In some regions, ongoing climate change may induce the gradual creep of land uses. For example, as northern regions warm, a previously inhospitable climate may become amenable to cropping. Or, conversely, lands once favourable for farming may be abandoned because of heat or drought stresses. Such gradual transitions, while difficult to foresee, may merit further scrutiny and study. The area of United States cropland is now apparently increasing again, largely at the expense of grasslands and wetlands, with concomitant loss of soil C (Spawn, Lark and Gibbs, 2019; Johnston, 2013). Avoiding C-depleting land-use change may be at least as important as augmenting stocks in managed lands (Fuhrman *et al.*, 2023).

Systems-based analysis of soil carbon changes

Many of the anthropogenic effects on soil C, whether leading to gains or losses, exert affiliated responses in emissions of other GHGs, notably CH_4 and N_2O (Ogle *et al.*, 2023b). For example, growing perennial forages to build SOC may elicit higher enteric CH_4 emissions (Beauchemin *et al.*, 2022), or adding N to promote C gains may prompt higher N_2O emissions (Guenet *et al.*, 2022). The net influence of any regime on climate must consider soil C changes in concert with emissions of CH_4, N_2O and energy-derived CO_2 (You *et al.*, 2024), demanding a systems perspective. Several

farm- and landscape-level decision support systems have been developed for this purpose (Kröbel *et al.*, 2021; Paustian *et al.*, 2020b), but their reliability under diverse landscapes over the long term and under future climatic conditions is still uncertain.

Monitoring the soil carbon status on working and unmanaged lands

The critical roles that SOC plays in governing soil productivity and mitigating climate change necessitate reliable, systematic and long-term monitoring of SOC (Oldfield *et al.*, 2022). Soil C stocks exhibit high spatial variability, and annual changes are usually small relative to background amounts, so that changes can be definitively quantified only after five to ten years (Ellert, Janzen and Entz, 2002).

In the United States, around 3 600 forest soil monitoring sites have been part of the Forest Inventory and Analysis (FIA) system since 2001 (Domke *et al.*, 2017). In 2024, the USDA announced plans for a full-scale national system on agricultural soils. Mexico has a national monitoring network (INFyS) for forest biomass and soils using around 26 000 points, with some soil measurements in agricultural lands, though forests are the primary focus (CONAFOR, 2012). An urgent mandate will be to expand and solidify the nascent networks, especially in Canada and the United States (Lawrence *et al.*, 2023) and to extend coverage to unrepresented land covers, particularly the C-rich systems that are vulnerable to C losses, such as wetlands, peatland and tundra.

Integrated systems, using dynamic models calibrated with ongoing measurements and supported by ancillary measurements have been proposed for quantifying regional soil C stock changes (Paustian *et al.*, 2019, Smith *et al.*, 2020; Lawrence *et al.*, 2023). Recent implementations of such iterative approaches show promise for the robust and cost-effective measurement of soil C change at scales from fields to regions (e.g. Guan *et al.*, 2023; Ogle *et al.*, 2023a).

Developing and widely implementing robust, long-term measuring strategies remain one of the most urgent prerequisites for minimizing threats to soil C in North America. Without such systems in place, it will be not possible to know how much soil C is being lost (and where), and the purported benefits of adopting C-conserving practices on managed lands will not be verifiable.

Barriers to adoption of soil carbon-conserving practices

Despite widespread recognition of the merits of preserving and augmenting soil C, the adoption of beneficial practices is often constrained by economic and other social barriers (Amundson and Biardeau 2020; Sykes *et al.*, 2020). For example, managers may be hesitant to change practices because of initial investment costs, higher economic risks or the inertia of familiarity. Consequently, maximizing benefits of soil C conservation and sequestration is more than a technical, scientific undertaking, as it also demands the concerted involvement of communicators, educators and policymakers.

The vast soil C reserves of North America occupy a wide range of ecosystems, each with their own stresses and evolving changes. Among the largest and most vulnerable pools of SOC are the northern peatland and permafrost reserves. Preserving these C troves, and avoiding potentially calamitous feedback on radiative forcing, relies mostly on reducing GHG emissions elsewhere. In other ecosystems, such as managed croplands, grasslands and forests, adopting prudent practices can often not only prevent further losses, but also help recover past losses, thereby withdrawing atmospheric CO^2 and enhancing ecosystem functions. Thus, the current narrative of SOC in North America offers both threats of growing dangers and also pathways for restoring biospheric resilience. In all cases, researchers, land managers and policymakers have an acute and urgent need for clearer measurements of the soil C trajectory in all of our ecosystems.

12.3.4 Soil biodiversity loss in the North America region

Soil biodiversity is the life that inhabits and interfaces with soil in a significant way. This includes countless billions of microbes, invertebrates, plants, and vertebrates that interface with each other and utilize, transform, and create soil in the process. However, a common definition of soil biodiversity that is useful to the policy community is still needed (Orgiazzi, 2022). Most soil life resides in the top metre of the soil surface and is responsible for many of the ecosystem services that humans, plants and the rest of life on Earth rely on for survival. These include, but are not limited to, nutrient cycling, filtration of rainwater, pest control in livestock and crop agriculture, erosion mitigation, and mitigation of a changing climate.

One of the challenges of discussing soil biodiversity change in North America is the large knowledge gap regarding the status of soil biodiversity and its associated functions, making it difficult to state the magnitude and types of change in soil biodiversity with certainty. Currently, soil biodiversity is not monitored using standardized indicators and protocols, nor are any resultant data harmonized and stored in freely available repositories for public access. Aside from being a large gap in North American environmental policy (especially for the United States and Mexico), industrial and agricultural sectors remain unprepared for current and emerging threats to the life in soil and the ecosystem services that the economies and healthcare systems of North America rely on.

Threats to soil biodiversity and resultant effects

Land-use change and agricultural intensification

Recent advances in soil ecology have uncovered the primary role of soil multidiversity (such as diversity across different trophic groups and body sizes) in driving multiple functions and services in agricultural ecosystems, relating to crop yield, nutrient provisioning, element cycling and pathogen control (Jiao *et al.*, 2022). Agricultural intensification leads to rapid losses of predatory soil species at the top of the soil food chain (Hu *et al.*, 2023; Puissant *et al.*, 2021). These effects are pervasive and detected in key soil community attributes such as the prey-to-predator ratio, which is consistently higher in cultivated versus non-cultivated land across North America biomes (such as temperate and montane grasslands, conifer and broadleaf temperate forests, boreal forests, and tundra) (Figure 12.18). The resulting functional homogenization of soil communities further impacts the physical and chemical conditions of the soil and the environment, including its capacity to resist and recover from changes in climate (Eisenhauer, Bonn and Guerra, 2019; Martínez-García *et al.*, 2017). For instance, biodiversity manipulation studies indicate that loss of soil functional diversity caused by agricultural intensification reduces litter decomposition and soil respiration rates, two key processes that drive C sequestration and nutrient availability to plants (De Graaff *et al.*, 2019).

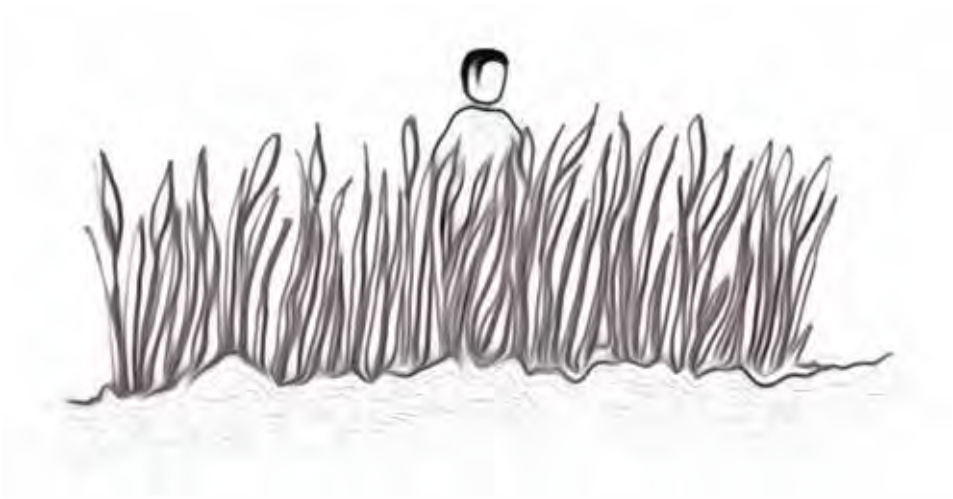
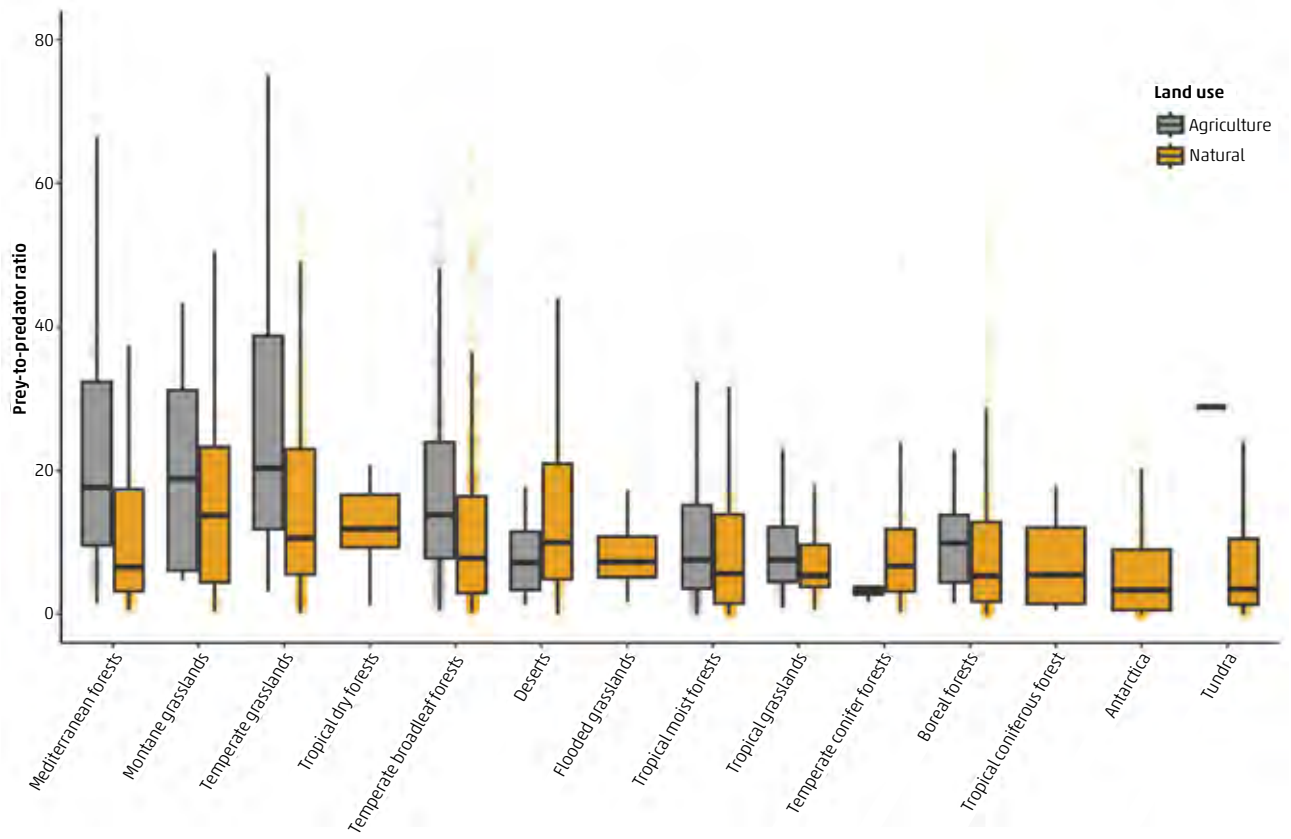


Figure 12.18. Land-use associations with nematode prey-to-predator ratio



Note: Graph shows median and interquartile range of nematode prey-to-predator ratios (n = 6 825) in agriculture and natural ecosystems of biomes.

◊ *Source:* **Van den Hoogen, J., Geisen, S., Wall, D.H. Wardle, D.A., Traunspurger, W., De Goede, R.G.M., Adams, B.J.** *et al.* 2020. A global database of soil nematode abundance and functional group composition. Scientific Data, 7(1): 103. https://doi.org/10.1038/s41597-020-0437-3

Agricultural intensification strategies based on crop diversification and organic amendments have been shown to simultaneously promote soil biodiversity and soil health in European crop systems (Walder *et al.*, 2023). A similar pattern for cropland and rangeland systems in North America under intensification using crop diversification, reduced tillage and rotational grazing may exist, but it remains untested how these so-called regenerative approaches impact soil biodiversity and the functions and services they deliver across large scales in the region. Evidence from studies assessing dead organic matter as indicators of soil biological function point to promising outcomes in systems incorporating perennial polycultures into crop rotations (Sprunger, Martin and Mann, 2020). However, it must be noted that new management strategies must be adjusted based on the particular ecosystem and changing climatic conditions.

Climate change-induced drought and floods

Droughts and deluges alter the functional composition of soil communities, but the direction of the effects varies by trophic level and local environment. For example, a rainfall manipulation study across grassland types in the Great Plains of North America (Franco *et al.*, 2019), showed that groups of soil nematodes in higher (predators) and lower (root and microbial feeders) trophic levels of the food chain inversely responded to year-to-year changes in rainfall. Additionally, the magnitude of the responses was greater in moist compared to arid grassland ecosystems. In the moist grassland where responses were stronger, drought induced greater changes in nematode abundances compared to the effects of deluges, leading to increased abundances of root- and microbial-feeding

nematodes and decreased abundances of predatory nematodes (Franco *et al.*, 2019). Increases in the abundance of root-feeding nematodes in soils can aggravate drought effects on plant growth and above and below ground plant biomass allocation (Franco *et al.*, 2020).

During drought conditions, the increased activity of soil engineers, such as earthworms, ants and termites which manipulate the soil matrix, can relieve the impacts of water stress on plants by enhancing soil water retention and diversity in soil nutrient distribution (Ashton *et al.*, 2019; Johnson *et al.*, 2016). The large-scale implications of this process for buffering ecosystems against desertification have been demonstrated through mathematical modelling by Bonachela *et al.*, (2015), which showed the strong connection between above- and below-ground ecosystem processes.

Increased distribution of invasive species

Non-native, introduced species, both invasive and naturalized (introduced species that integrate into a new environment without causing harm to the ecosystem) come in myriad forms, and originate from all branches of the tree of life. Introduced organisms may negatively impact the biodiversity of soils. Included among invasive species are organisms ranging from plants to microbes (pathogenic bacteria and fungi), mesofauna (nematodes and microarthropods such as Collembola and mites), to macrofauna (such as earthworms, insects and planarians), and vertebrates (rodents, other fossorial mammals and reptiles). Importantly, non-native organisms may have indirect effects on soil biodiversity by modifying habitat or vegetation characteristics that render soils inhospitable to the native biota, or direct effects through pathogenicity, competition or predation interactions.

Plant invasions almost always impact the diversity or community structure of the underlying soil biota. This occurs through various means, including allelopathy, decreasing soil heterogeneity, and through pathways such as litter inputs and rhizosphere inputs. The effects of plant invasions on species richness are variable but are almost always associated with changes in community composition and functional attributes and could be made worse by climate warming. For example, Anthony *et al.* (2020) showed in an experiment in Harvard Forest in the United States, that an invasion of garlic mustard (Allaria petiolate) changed fungal communities by significantly reducing ectomycorrhizal fungi and increasing saprophytes and plant pathogens in soils that were also subject to enhanced warming regimes.

The rate of introduction of non-native invasive organisms is predicted to increase due to increases in international travel and trade (Seebens *et al.*, 2021) and has been noted for various groups of soil organisms, including ants (Bertelsmeier *et al.*, 2018), termites (Blumenfeld *et al.*, 2021) and earthworms (Chang *et al.*, 2021). These three groups are important when they invade because they are classified as ecosystem engineers, with the capacity to change habitats physically and chemically from the native state, often with consequences for the native biota (Ferlian *et al.*, 2020). The effects of invasive ants and earthworms on native soil invertebrate diversity are relatively well known (Coyle *et al.*, 2017). For example, Achury, Holway and Suarez (2021) showed that with increasing spread of the invasive ant Linepithema humile over a period of 21 years, there was a corresponding decrease in the total species richness (−71 percent) of native ants at their sites in southern California, in the United States. Likewise, when non-native earthworms invade, there can be negative impacts on native soil fauna. This has been documented particularly in North and South America (Jochum *et al.*, 2021; Demetrio *et al.*, 2023) and is a recognized problem globally.

There are large economic costs associated with invasive species worldwide, totalling hundreds of billions of United States dollars in losses and control measures (Cuthbert *et al.*, 2022). The most cost-effective method for minimizing costs remains the prevention of introductions, as invaders are often difficult to manage once they become established. Failing prevention, the next most cost-effective strategies for invasive species control are early detection and eradication (Reaser *et al.*, 2020). Prevention is particularly relevant for invasive organisms that live in soil, as their low visual detectability makes them more likely to escape notice until well established.

Ways to protect and remediate soil biodiversity in North America

Threats to soil biodiversity in North America are not considered high priorities in federal, regional or local agricultural policy contexts, but this needs to change. The path forward lies in a cross-national reassessment of priorities in research, environmental and economic decision-making. Vital to this reassessment are the following key initiatives (FAO, 2020):

- Mainstream soil biodiversity into strategies to achieve sustainable development.
- Establish soil information and monitoring systems that include soil biodiversity as an indicator of soil health.
- Improve knowledge of the soil microbiome and strengthen knowledge of the roles of soil invertebrates in provisioning ecosystem services.
- Develop a framework for information sharing so land managers can connect projects to each other, to funding sources, and open data-sharing mechanisms and create a network of knowledge.
- Develop ambitious, objective-driven partnerships between research and industry to support the advancement of national environmental policies in soil biodiversity protection.

Key steps to achieving soil biodiversity mainstreaming in North America include ensuring that soils and soil biodiversity are explicitly incorporated into national and regional policies and strategies related to economic development, agriculture, infrastructure, energy, and other sectors. Conducting economic valuations of ecosystem services provided by soil biodiversity may help policymakers understand the benefits derived from nature and integrate them into cost–benefit analyses of development projects, including sustainable land-use planning (Pascual *et al.*, 2015; Bartkowski *et al.*, 2020). The mainstreaming of soil biodiversity across various sectors will require recognizing the interconnectedness of soil biodiversity with social, economic, and environmental aspects of the UN's SDGs in general and SDG 15 in particular. This includes regular monitoring and reporting on the status of soil biodiversity to tracking the impacts of development projects and effectiveness of policies to protect soil biodiversity. Initiatives are already underway to establish standardized data collection protocols and develop effective indicators (i.e. FAO's Global Soil Biodiversity Observatory [GLOSOB] and GEOBON's Soil Biodiversity Observation Network [Soil BON]), but increased capacity for data management and stakeholder involvement alongside training to ensure accurate and reliable data are needed.

Regular monitoring of soil health can guide farmers, land managers and policymakers to inform the maintenance and enhancement of soil productivity and ecological balance. At the same time, increased knowledge about the soil microbiome and the functional contributions of soil invertebrates will help link soil biodiversity indicators directly to provisioning ecosystem services, such as food production (Thiele-Bruhn *et al.*, 2012). It is known that soil biodiversity plays a crucial role in provisioning ecosystem services, yet the complex interactions of soil biodiversity among microorganisms, plants, animals and abiotic factors make elucidating this difficult.

To understand how soil invertebrates improve soil health, a combination of approaches to ecological research are needed. Engagement that involves local peoples, Indigenous Peoples and stakeholders in research and monitoring can provide valuable insights into traditional knowledge, in particular Indigenous Peoples' food and knowledge systems, and how these contribute to sustainable land management. Specific areas of research include studying the interactions between soil invertebrates and soil microorganisms and holistic, *in situ* studies of the soil food web, which represents the interactions between all soil life.

The loss of and lack of conservation of soil biodiversity are issues in North America as well worldwide, but some multinational unions, such as the European Union, now recognize the importance of soil

biodiversity in soil health and SSM by including it in massive policy and funding initiatives (EC, 2021). Threats exist across national and state boundaries, and, therefore, must be tackled with political coordination across borders (Van der Putten *et al.*, 2023). Policy and land conservation initiatives need to purposefully address the outcomes of threats to soil biodiversity and the cascading and interacting consequences of these threats, such as those posed to SOC storage and nutrient cycling.

Conservation initiatives for wildlife do not extend to soil biodiversity in the region (Zeiss *et al.*, 2022) and mechanisms need to be in place to protect these critical organisms and their functions. Mechanisms of protection, conservation and restoration should revolve around the discovery of the functions of novel soil biodiversity in key ecosystem processes that preserve soil carbon and regulate nutrient cycling. Allocating adequate financial resources for soil biodiversity conservation is key to achieving these goals, while recognizing that across the gradient of North America (arctic, boreal, temperate, subtropical and tropical regions), the magnitude and relative strength of different threats vary. For instance, arctic and boreal regions are anticipated to experience greater impacts and changes from climate warming, while growing demands for land use that coincide with soil erosion, soil pollution and soil disturbance are greater in southern latitudes. Arctic and boreal regions of North America are undersurveyed, with less known about soil biodiversity functions than in other regions (Guerra *et al.*, 2020). Decreasing C reserves in the arctic and boreal regions coincide with substantial recent land cover change (CEC, 2024). Understanding the role soil biodiversity plays in C sequestration is imperative to mitigating climate change. At the same time, soil biodiversity is richest in moist temperate and tropical systems that face significant threats from human activities such as agriculture conversion and climate change (Guerra *et al.*, 2022).

Addressing gaps and actionable items

While the protection and conservation of soil biodiversity is an issue around the world, North America is one of the last continents to fully recognize the role of soil biodiversity in soil health and the necessity of a sustainable management plan (Van der Putten *et al.*, 2023). Threats exist across national and state boundaries, and therefore cannot be easily addressed, given political heterogeneity. Policy and land conservation initiatives need to purposefully address the outcomes of threats to soil biodiversity and the cascading and interacting consequences of these (e.g. soil C storage and nutrient cycling). Conservation initiatives for wildlife do not so far extend to soil biodiversity (Zeiss *et al.*, 2022) and mechanisms need to be in place to protect these critical organisms and their functions. Mechanisms of protection, conservation and restoration should revolve around the discovery of the functions of novel soil biodiversity in key ecosystem processes that preserve soil C and regulate nutrient cycling. Allocating adequate financial resources for soil biodiversity conservation is key to achieving these goals, while recognizing that across the gradient of North American ecological zones (arctic, boreal, temperate, subtropical and tropical), the magnitude and relative strength of different threats vary. For instance, arctic and boreal regions are anticipated to experience greater impacts and changes from climate warming, while growing demands for land use that coincide with soil erosion, soil pollution, and soil disturbance are greater in southern latitudes. Arctic and boreal ecoregions of North America are undersurveyed, with less known about soil biodiversity function than other regions (Cameron *et al.*, 2021; Guerra *et al.*, 2020). Understanding the role soil biodiversity plays in C sequestration is imperative to mitigate the effects of climate change. At the same time, soil biodiversity is richest in moist temperate and tropical systems that face significant threats from human activities such as agriculture conversion and climate change (Guerra *et al.*, 2022).

Overcoming challenges associated with the creation of funding schemes for strategic long-term research projects on soil monitoring and research will facilitate the monitoring, data-sharing and knowledge-building of soil biodiversity and links with ecosystem services and functions. Soil biodiversity and related ecosystem functions have been politically neglected for many years, and

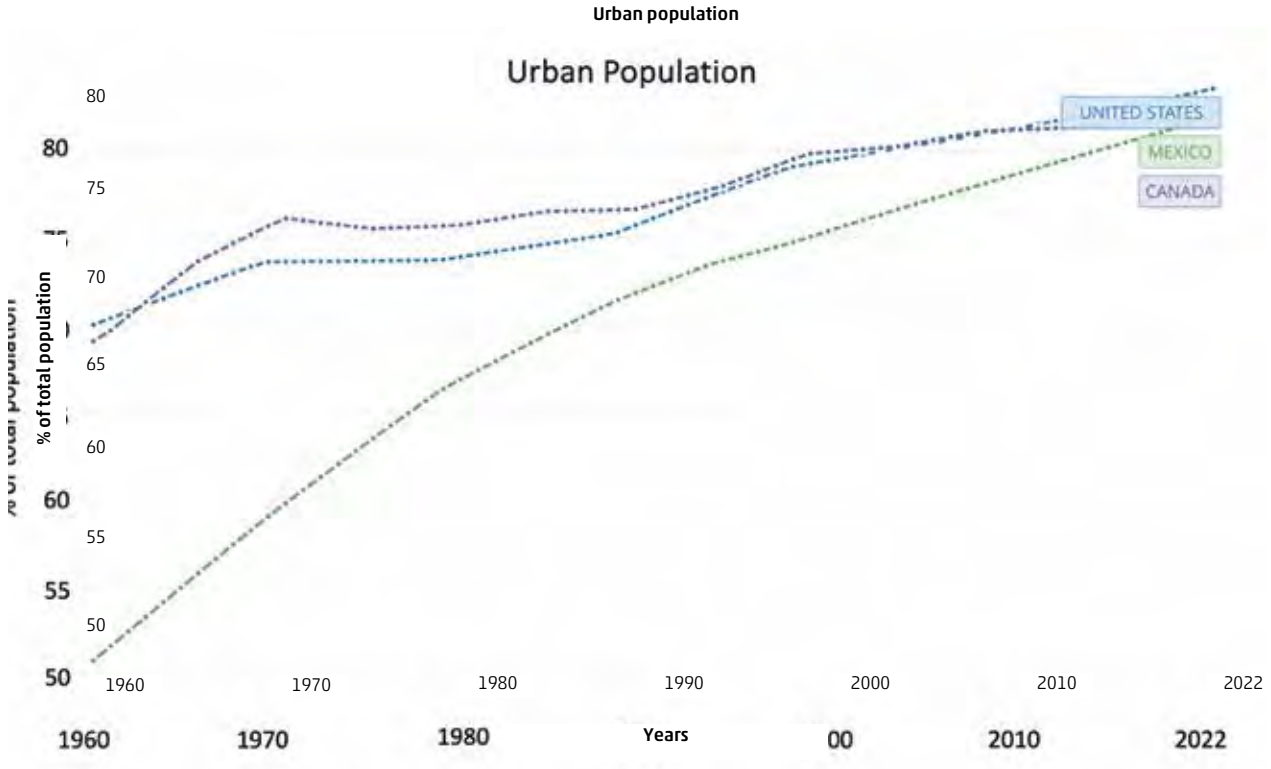
while this is now changing (Zeiss *et al.*, 2022), knowledge of soil biodiversity distribution still lags markedly behind above ground knowledge. To establish that soil health underpins soil biodiversity, research and policy priorities must be realigned to envision agricultural areas as a part of the ongoing ecological conservation of soil organisms and their ecosystem functions. This will establish a strong social and business case for supporting soil biodiversity functioning that will influence mainstreaming conservation efforts among multiple stakeholders.

12.3.5 Urbanization and soil sealing

Urban expansion and Its impact on land use and forest conversion

North American urbanization has transformed land use, encroaching upon forests and diminishing SOC. Concomitant effects of urbanization including soil sealing and contamination (e.g. PFAS) pose further sustainability challenges. According to the World Bank (2024), from 1960 to 2023, urban populations in Canada, Mexico and the United States had increased by 82 percent, 82 percent and 83 percent respectively. In 2015, urban land use was 13 983 km² in Canada, 28 588 km² in Mexico and 186 573 km² in the United States (World Bank, 2015). Population densities in 2021 stood at 4 people per km² in Canada, 65 people per km² in Mexico and 36 people per km² in the United States (Figure 12.19).

Figure 12.19. Trends in urban population (1960–2022) in North America



◊ *Source:* **World Bank.** 2022. *Urban population (% of total population) - Mexico, United States, Canada, North America.* [Accessed on 15 February 2024]. https://data.worldbank.org/indicator/SP.URB.TOTL.IN.ZS?locations=MX-US-CA-XU. Licence: CC BY 4.0.

A major trend in North America urbanization is expansion into forested and agricultural lands. Over half of the urban development has occurred in forested areas, marking a transformation of unmanaged habitats into urban settings. There is a broader pattern of land-use change, with forests increasingly being converted for urban development. The implications of urban expansion are multifaceted, affecting landscapes and ecosystem services including C sequestration and biodiversity (Güneralp *et al.*, 2020).

At the same time, there has been a decrease in urban population density, especially in smaller to medium-sized cities in North America. Density has dropped from about 3 500 people per km² to just 500 people per km². The decline in urban population density, coupled with urban expansion, suggests a sprawling development pattern, with cities extending their boundaries, often at the cost of natural resources and ecosystems (Güneralp *et al.*, 2020). Chicago, in the United States, exemplifies North American urbanization trends. Between 1975 and 1990, the city's urban land expanded by 50 percent, driven by suburban expansion and a resurgence in central urban areas, reshaping land use and land rents. There was considerable growth in the outer areas of the city with a more compact pattern of development from 1975 to 2015. However, from 1990 to 2000, the city experienced a period of no growth, and afterwards, urban density began to decrease due to migration of the inner-city population to the outer rings (Gerten, Fina and Rusche, 2019).

The shift to urban sprawl reduces C sequestration by replacing forests and farmlands with impervious surfaces, limits water-holding capacity due to increased runoff and reduced groundwater recharge, and worsens thermal impacts by expanding the urban heat island effect, as more surfaces absorb and radiate heat compared to denser, more compact areas (Zhou *et al.*, 2022; Chen *et al.*, 2022; Steenberg *et al.*, 2023; Feng, Zhang and Bourke, 2021).

Urbanization in North America, particularly in the northeast and Midwestern United States, has led to significant environmental changes due to soil sealing. Sealing involves covering the soil with impermeable materials such as concrete and asphalt, a widespread method in urban development for building construction and road paving. In Canadian cities, asphalt and concrete constitute as much as 40 percent of urban surface area, with Toronto's asphalt averaging 16.2 percent and concrete at 13.7 percent, not including rooftops, with these averages being much higher in city centres (Wang, Berardi and Akbari, 2016). The sealing materials have a low albedo, typically around 0.1 for asphalt and between 0.3 to 0.4 for concrete, contributing to urban heat island (UHI) effects (Wang, Berardi and Akbari, 2016). The creation of impervious surfaces disrupts the natural water cycle, hindering groundwater recharge and altering surface water flow as evidenced by a study by Hodgkins *et al.* (2019) which showed that increased impervious surfaces diminished the land's water absorption capacity, impacting surface temperatures and soil moisture levels.

Increased impervious surfaces also reduce the natural filtration and buffer capacity of soils, leading to greater groundwater contamination from urban pollutants like heavy metals, chemicals and runoff that bypasses natural soil filtration processes (US EPA, 2022).

Hodgkins *et al.* (2019) also reported that urbanized basins in the northeast and Midwestern United States experienced marked increases in peak-flow magnitudes that were directly linked to the level of urbanization. The redirection of water as surface runoff, a consequence of its inability to infiltrate impervious surfaces, leads to heightened erosion and sediment transport. The shift in hydrology not only endangers the replenishment of local groundwater sources but also presents significant challenges to stormwater management systems. Such changes increase the risk of flooding during periods of high precipitation, underscoring the need for careful urban planning and sustainable development practices.

In urban settings, soil is frequently moved, graded, compacted or contaminated, impeding the infiltration of surface moisture into deeper layers. The reduction in soil moisture leads to a higher conversion of incoming radiation into sensible heat, thereby exacerbating the UHI effect (Ramamurthy and Bou-Zeid, 2017). Urban areas can be 3 to 4 °C warmer than surrounding rural areas due to soil sealing. Studies such as Ramamurthy and Bou-Zeid (2017) have shown that vegetated areas exhibit a surface resistance 70 percent higher than that of sealed surfaces, further emphasizing the impact of soil conditions on local temperatures. Surface resistance, which is the ability of a surface to resist heat transfer, is much higher in vegetated areas compared to sealed surfaces, significantly influencing local temperature dynamics. Such studies highlight the profound

influence of soil moisture deficit on UHIs at a local scale, particularly in larger metropolitan areas where the soil moisture deficit can extend over several kilometres, potentially impacting regional climate patterns. While Ramamurthy and Bou-Zeid (2017) underscored the need for more accurate representation of urban soils in simulations, it is important to note that impervious surfaces like asphalt and concrete, which prevent water infiltration and increase runoff, further intensify the UHI effect by reducing soil moisture and promoting heat retention.

Urbanization also alters the composition of soils in urban areas. The natural mineral and organic components, derived from rocks, plants and animals, are augmented by manufactured artifacts like glass, brick, concrete and ash. These size-variable artifacts introduce additional elements such as trace metals and PAHs, deviating from the composition of natural soils (Gonzales *et al.*, 2021; Howard *et al.*, 2019; Marquez-Bravo *et al.*, 2016; Paltseva *et al.*, 2022). A study analysing artifacts, street dust and fly ash-impacted in Detroit, Michigan, in the United States, found that while these materials contained elevated concentrations of toxic trace elements, they did not significantly increase the toxicity of urban soil and street dust, due to low bioavailability (Howard *et al.*, 2019).

The presence of materials like concrete can alter the pH of the soil and contribute to the C content. The weathering of introduced materials can lead to the formation of amorphous clays and introduce non-native minerals, fundamentally altering soil structure and compromising its natural functions. The uniqueness of urban soils is further accentuated by the introduction of these anthropogenic materials, which influence soil properties such as pH levels, water retention and C dynamics, underscoring the need for specialized management and classification approaches (Huot *et al.*, 2017).

Soil classification in urban areas is complex due to the volume of artifacts present, often exceeding 10 percent by volume, which necessitates a different approach than that used for non-urban soils (IUSS Working Group WRB, 2015; Soil Survey Staff, 2022). In light of these complexities, Artesol has been proposed as a new soil order within the USDA soil taxonomy system, aiming to classify significantly and artificially altered soils typically found in urban or heavily modified environments, though it is not yet officially recognized (Riddle *et al.*, 2022).

The sealing process, along with the introduction of artificial materials, has led to changes in soil properties, water cycle dynamics, and local climate conditions, highlighting the need for sustainable urban planning and environmental management. In response to these challenges, several strategies for managing stormwater in urbanized areas have been developed (Table 12.11).



Table 12.11. Strategies for managing stormwater in urbanized areas

Strategy	Approach	Description
Green infrastructure implementation	Permeable pavements	They allow water to percolate, reducing surface runoff and aid in groundwater recharge.
	Green roofs	They absorb rainwater, provide insulation and manage water runoff.
	Rain gardens	They absorb rainwater runoff from roofs, pavements and streets, reducing the amount of runoff entering storm drains.
Preservation of natural landscapes	Wetlands and forests	Integrating and preserving wetlands within urban areas can act as natural buffers to flooding and water pollution. Wetlands filter pollutants and manage stormwater, thereby reducing the impacts of impervious surfaces on peak flows and water contamination.
	Grasslands and shrublands	Grasslands are essential for biodiversity, C sequestration and water cycle regulation, particularly in areas with intermediate rainfall. Shrublands, consisting of low-lying woody plants and shrubs, thrive in various climates and are crucial for habitat provision and soil erosion prevention.
Sustainable urban planning	Stormwater retention basins	The basins are designed to collect and hold rainwater, helping to control flooding and manage stormwater.
	Urban design integration	Incorporating stormwater management practices into the overall urban design can significantly mitigate the impacts of impervious surfaces.
Policy and regulation	Development policies	These are enforcing regulations that mandate the inclusion of stormwater management.
	Incentives for retrofitting	Offering incentives for existing developments to incorporate stormwater management infrastructure can enhance urban stormwater management.

◊ *Source:* **Levin, M.J. & Paltseva, A.** 2023. Management of park areas, sport fields, and school yards, including golf courses and public right of ways. In: M.J. Goss & M.A. Oliver, eds. *Encyclopedia of Soils in the Environment*. 2nd Edition, pp. 488–504. Amsterdam, Elsevier. https://doi.org/10.1016/B978-0-12-822974-3.00274-3

Dynamics and management of urban soil carbon

Impacts of urbanization on soil health and ecosystem services are evident. In the Midwestern United States, urbanization has resulted in a 40 percent reduction in SOM and a 30 percent decrease in water filtration. Moreover, C storage in urban soils is reduced by up to 60 percent compared to rural soils, indicating a substantial environmental trade-off (Hodgkins *et al.*, 2019).

A study examining two watersheds in Florida, the United States from 2003 to 2060 predicted that urbanization could reduce SOC storage by 16 to 26 percent in the urbanized 2060 scenario, primarily due to land-use change. The reduction in SOC was coupled with an assumption that C pools in 2060 will be 5 percent less than in 2003 (Delphin *et al.*, 2016). Additionally, the conversion of natural land to urban land in the Midwestern United States increases land surface temperature, alters vegetation cover and affects surface moisture, all of which are vital for maintaining soil health and quality (Jiang, Fu and Weng, 2015).

The urban landscape of New York in the United States exemplifies the profound impact of urbanization on soil characteristics, where 62.7 percent of urban soils are sealed, significantly altering their composition and ecological function. Similarly, in Vancouver, Canada, a low-density neighbourhood demonstrates the C storage potential of vegetated areas, with soils under lawns and trees storing around 96.5 t/ha, although this average drops to 34.5 t/ha across the neighbourhood due to impervious surfaces which cover 64.3 percent of the land (Lorenz and Lal, 2015). In Baltimore, Maryland, the United States, natural soils cover 38.2 percent of the land area and contain an average SOC of 10.4 kg/m^2 in the top metre of soil. In contrast, sealed soils, which cover 42.6 percent of the land area, contain less SOC at 3.3 kg/m^3. This comparison highlights the substantial difference in SOC storage capacity between natural and sealed urban soils (Pouyat *et al.*, 2020).

In urban settings like New York, the United States, different soil types play varying roles in C sequestration. Tidal marsh and woodland soils are notable for being substantial C reservoirs. Interestingly, even soils under turf areas have been found to store as much C as those in woodlands, highlighting the diverse potential of urban soils in C storage (Cambou *et al.,* 2018). The presence of black carbon, a byproduct of fossil fuel combustion, is common in urban environments, which is more resistant to microbial degradation than standard plant-derived organic matter. Black carbon tends to bind more readily with contaminants, making it a factor in urban soil composition (Cambou *et al.*, 2018).

Surface sealing changes the soil environment, particularly by reducing microbial activity. As a consequence, the C present prior to sealing becomes more resistant to decomposition. This resistance can potentially increase the soil's capacity to preserve C over long periods. However, once the soil is sealed, its interaction with the atmosphere and its ongoing ability to accumulate C are substantially reduced (Cambou *et al.*, 2018).

Understanding the effects of soil sealing is vital for urban planning and environmental sustainability. Incorporating the use of permeable and porous pavements has shown potential benefits for the urban C cycle, as they can positively influence soil moisture and gas exchange, which are critical to maintaining balanced C dynamics in urban soils (Fini *et al.*, 2017). To enhance urban soil C stocks, it is essential to focus on both increasing the inputs of organic and inorganic C and prolonging their retention in the soil. Effective strategies include the addition of organic matter, the creation of biochar from compost and the strategic burial of C materials, particularly before sealing areas with pavements and buildings (Lorenz and Lal, 2015). Burial of C materials may contribute to C sequestration and also enhance the overall health and functionality of urban soils. By acknowledging and addressing the impact of soil sealing, urban planners and environmental managers can make more informed decisions that balance development needs with environmental conservation.

Per- and polyfluoroalkyl substance contamination

Urbanization has been linked to the increased PFAS contamination in the environment. Per- and polyfluoroalkyl substances are artificial chemicals that have been used since the 1950s in various industries and consumer products, including firefighting foams (AFFF), non-stick cookware (such as Teflon™), waterproof clothing, food packaging (such as fast food wrappers), and cosmetics (such as mascara and foundation). Per- and polyfluoroalkyl substances are persistent, meaning they do not break down easily and can accumulate over time in the environment and human body. The expansion of urban areas heightens the risk of PFAS entering the environment through industrial discharges, use of products containing PFAS, and leaching from landfills into soils and waterways. The presence of PFAS in urban settings raises significant concerns for water quality and public health, as they are associated with serious health issues including cancers, hormone disruption and immune system effects (Brusseau, Anderson and Guo, 2020; Rankin *et al.*, 2016; Wang, Munir and Huang, 2023).

Research indicates that air and landfill leachates are the primary sources of PFAS in soil (Wang, Munir and Huang, 2023). Soil solution pH and cation levels affect PFAS sorption and desorption at the water–soil interface, though no single factor is decisive. Per- and polyfluoroalkyl substances migration from soil to plants, including wheat and maize, varies by PFAS type and plant species. Soil PFAS levels correlate with land use and are highest at primary exposure sites (industrial areas and fire-fighting sites) followed by secondary sites (landfill) and lowest at background sites (residential land). Legacy PFAS-perfluorooctanoic acid (PFCA) and perfluorooctane sulphonic acid (PFOS) are most prevalent (Wang, Munir and Huang, 2023).

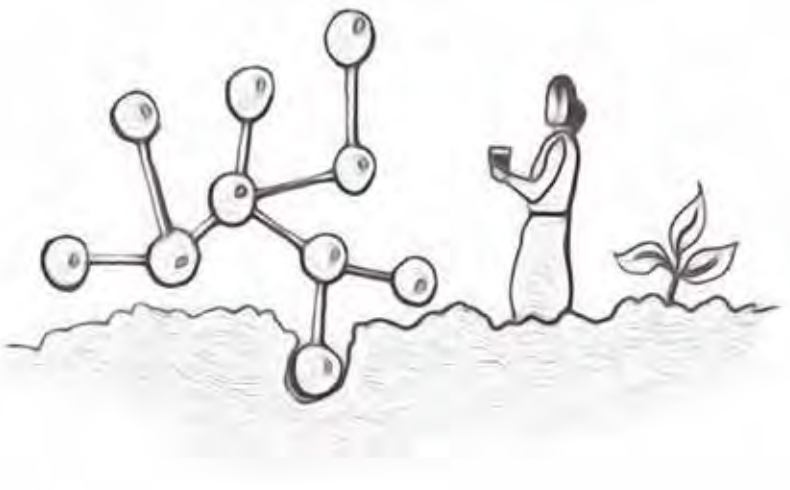
In North America, PFAS contamination in soils is a growing problem, particularly in urban and industrial regions. Studies have consistently found PFAS in surface soils across the continent, with detectable concentrations in all samples tested. In 33 soil samples from the region, Rankin *et al.* (2016) reported concentrations of perfluoroalkyl carboxylic acids (PFCAs) ranging from 145 to 6 080 picograms (pg)/g of dry soil (with a geometric mean of 1 820 pg/g of dry weight), and (perfluoroalkane sulphonic acids (PFSAs) ranged from 35 to 1 990 pg/g of dry weight (geometric mean: 410 pg/g of dry weight). When comparing these North American concentrations to concentrations representative of global soils, a distinct pattern emerges. The geometric mean concentrations are highest in Asia and North America, followed by Europe and Australia, and are lowest in Africa, South America and Antarctica. This trend mirrors the hemispheric distribution, with the higher concentrations in the northern hemisphere (Asia and North America) and the lower concentrations in the southern hemisphere (South America, Antarctica and parts of Africa) (Rankin *et al.*, 2016).

Tackling PFAS contamination requires a multifaceted approach involving urban planning, industrial regulation, and pollution control. Some cost-effective technologies that can be used to remediate PFAS-contaminated soil include the following (Wang, Munir and Huang, 2023):

- Immobilization: This technique involves adding sorbents such as activated C, biochar and minerals to the soil to stabilize PFASs and reduce their mobility and bioavailability.
- Separation: PFAS can be removed from the soil and transferred to a separate liquid phase through soil washing or liquefractionation. Soil washing involves the use of water or organic solvents to detach PFAS from the soil, while soil liquefractionation combines foam fractionation and soil liquefaction to separate PFAS.
- Destruction: Thermal treatment and chemical oxidation are options for destructing or transforming PFAS in soil. Thermal treatment involves heating the soil to vaporize and desorb PFAS, followed by thermal destruction at high temperatures to degrade them. Chemical oxidation relies on the use of chemical oxidants to degrade PFAS in the soil.
- Restriction and substitution: A critical long-term measure involves identifying and employing alternatives to PFAS, or enacting comprehensive bans on their use, to prevent further soil contamination. This preventive action serves as a cornerstone for safeguarding soil integrity and public health against persistent chemical pollutants.

It should be noted that the field applications of these technologies are still limited, and more research is needed to optimize their effectiveness and cost-efficiency for PFAS remediation in soil (Wang, Munir and Huang, 2023).

To manage the impact of urbanization on soils, cities should adopt permeable pavements and green infrastructure to reduce soil sealing and improve water infiltration. Governments need to regulate urban expansion to protect agricultural lands and forests while promoting sustainable land-use strategies. Stronger regulations on PFASs, alongside incentives for safer alternatives, will reduce soil and water contamination risks. Investments in soil remediation technologies, such as biochar application and soil washing, can help restore contaminated soils. Finally, collaborative monitoring and data-sharing initiatives will support better urban soil management and inform policy decisions for healthier, more sustainable cities.



12.4 State and trends in sustainable soil management

North American soils have demonstrated several notable improvements in state and trend (Figure 12.20). However, major concerns persist about the current and future capacity of soils to deliver critical ecosystem services and contribute to the UN's SDGs.

The information presented in this chapter was used to assess the state and trend of threats to soil functions through a management-based approach (Figure 12.20 and Table 12.12). Instead of directly measuring or modelling the physical severity of each soil threat, as was done in the 2015 SWSR report (FAO and ITPS, 2015), the current report assesses the level of adoption of SSM practices known to mitigate each soil threat.

The state reflects the effectiveness of management practices in place as of 2015 in preventing or reducing a specific soil threat. At the regional or subregional scale, changes in management occur at decadal or longer time scales and hence the state as of 2015 is relevant for 2015 to 2024 assessment period.

The trend indicates the direction of change in the adoption of SSM practices between 2015 and the finalization of this report (late 2024). In short, trend reflects the how SSM adoption has progressed over time.

Figure 12.20. Assessment of state and trend of threats to soil functions in North America through a management-based approach



Note: For more details, refer to Table 12.12.

Table 12.12 Summary of threats to soil functions in North America and the confidence in the assessment of their state and trend

Threat to soil function	Summary/comments	Confidence	
		Condition	Trend
Erosion	Soil erosion remains a significant threat. Tillage is increasing after decades of decline, which is of significant concern. Growth in herbicide resistance and crop residue production may explain the increase in tillage. Human-induced wind erosion is high in the Sonoran Desert of Northern Mexico and the high plains of the United States.	Medium	Medium
Nutrient mismanagement	Advances in crop genetics and management have continued to lower nutrient surpluses, although major hotspots of nutrient loss remain. Manure-based nutrient management is a particular challenge.	Medium	Medium
SOC change	Overall, SOC is increasing due to growth in crop residue production, agricultural land abandonment and reforestation. Climate change is a major threat, as a substantial proportion of SOC is found in permafrost and peatlands.	Medium	Low
Soil biodiversity change	There is insufficient knowledge about changes in soil biodiversity due to a lack of monitoring. In managed lands, trends are variable and associated with the implementation of conservation agriculture practices.	Low	Low
Urbanization and soil sealing	Urbanization continues to increase at the expense of prime farmland. Urban sprawl continues to encroach upon unmanaged lands with little or no mitigation efforts.	Medium	High

Note: The threats are presented in the order of most severe threat (erosion) to least severe (urbanization and soil sealing).

Although these threats do not operate independently and are not the only concerns about soil management, they represent broad processes that are critical to the future of soil security. Regionally, this assessment includes Canada and the United States as well as northern Mexico's rangeland ecosystems that are ecologically similar to those in the southwestern United States.

Erosion, nutrient mismanagement, C loss, biodiversity loss and urbanization threaten food production and environmental quality, and their sources are many. Some are a direct result of unsustainable approaches to soil management that can be improved at local or regional scales. However, others are a result of climate change and can only be solved through collective global action.

Erosion continues to be the primary threat to SSM. At present, the average rates of soil erosion are orders of magnitude greater than preagricultural rates of soil erosion or rates of soil production.

Elimination of tillage can reduce erosion rates to levels that are comparable to preagricultural levels, and in the long-term it can improve soil structure and reduce GHG emissions (Six *et al.*, 2004). However, recent data indicate that tillage is increasing, due to increased herbicide resistance and crop residue production (e.g. Lu *et al.*, 2022b). This disturbing trend represents a reversal of significant progress made over previous decades in the adoption of no-till. This trend is occurring in both Canada and the United States: in the United States, it is coincident with an increase in cropland area that is disproportionately classified as "highly erodible land".

The ability to manage these troubling trends is limited by the availability of measurement and modelling tools that can accurately estimate rates of soil loss. Current levels of uncertainty in erosion estimates are highlighted by three fold differences in the outcomes of various approaches to measuring and modelling erosion rates. One reason for the differences is that most tools do not capture all the processes that contribute to soil loss such as wind, water, and tillage erosion.

Tillage is a major factor driving erosion in North America not only because it is a major driver of erosion, but also because Canada and the United States produce roughly one-third of global maize and soybean crops. In addition to the increased threat of erosion, the production of high-yielding annual crops makes the region a large user of nutrients. Although nutrient mismanagement includes over- and underapplication of nutrients, in this region, overapplication of nutrients is of primary concern.

Excessive application of N and P leads to significant nutrient losses that degrade the environment. Nitrogen losses harm aquatic ecosystems, pollute drinking waters, degrade air quality, and contribute to GHG emissions. Phosphorus losses similarly harm aquatic ecosystems.

In the United States, river and stream quality assessments by US EPA (2022) indicate that nearly half of river and stream miles are in poor condition with respect to total N and this value has remained relatively static. However, the same assessments indicate major progress towards reducing P losses, with roughly 40 percent of river and stream miles in 2020 being in poor condition with respect to P, which represents an improvement of almost 18 percent when compared to the same assessment five years earlier. Nevertheless, losses of N and P to aquatic ecosystems are highly localized. In the United States corn belt, 70 percent of river and stream miles are in poor condition for N, yet less than 10 percent of United States land contribute to 75 percent of nutrient losses.

Partial nutrient balances (fertilizers inputs plus BNF minus harvest outputs) are a useful tool to understand N and P management and correlate with environmental outcomes. In North America, total N inputs have increased along with harvest outputs, resulting in relatively little change in annual surpluses of approximately 37 kg/ha of N. As is typical with all soil threats, there are notable differences within North America. In California, N balances are now negative (−2 kg/ha of N). Despite neutral P balances for North America as a whole, P balances are neutral or negative in the United States (consistent with the improvements in water quality) but remain positive in Canada despite recent improvements.

There are significant uncertainties in these nutrient balances owing to a lack of information about fertilizer inputs and BNF, as data sharing in agriculture is not yet common practice. The largest concern is access to farm management data.

Agricultural land-use affects nutrient management and soil C stocks. Recent gains in soil C may be partly attributable to increasing crop yields that enhance C inputs to the soil. As described previously, increasing crop yields (such as greater nutrient harvest) requires greater nutrient inputs and is coincident with increasing nutrient inputs. These increases in C could complement additional gains in cropland soil C from a variety of practices that are viewed as a significant opportunity to mitigate climate change, although croplands occupy only 6 percent of Canada and 44 percent of the terrestrial area of the United States.

Climate change is a major threat to North American soil C stocks on unmanaged lands. A substantial fraction of soil C in this region is found in permafrost and peatlands. In Canada, peatlands alone account for approximately one-third of total soil C. These C stocks have accumulated over millennia owing to cool temperatures and waterlogging: conditions that are under threat from climate change.

At present, the temperate forests of Canada and the United States represent a substantial C sink. Cropland abandonment and reforestation contribute to C accumulation both in trees and soils. Although opportunities to manage forest sinks are limited, an increase in fire frequency is a potential threat.

Grasslands offer more opportunity for soil C management. These lands co-evolved with large herbivores and can be used for grazing of domestic livestock while maintaining or improving soil C stocks. Improved practices on overgrazed grasslands can increase SOC stocks, such as fertilizer addition and rotational grazing, including vegetation and livestock management. Globally, there may be an achievable sink of 3.4 billion tonnes CO_{2eq} per year in the near term.

Without soil C monitoring, it is not possible to know the amount and location of C gains or losses. Like erosion and nutrient management, sufficient monitoring networks for soil C stocks are lacking. Integrated systems of measurements and modelling offer a robust, cost-effective solution.

Like soil C, land use is a major threat to soil biodiversity. Losses of soil biodiversity can negatively impact agriculture in many ways. Cultivation often reduces predator-to-prey ratios and rates of crop residue decomposition. Changes in precipitation have increased root-feeding nematodes and decreased predatory nematodes. Decreases in residue decomposition are particularly concerning, as crop residue production is increasing with yield, and crop residue decomposition is critical to nutrient availability and management. There is also a critical need to better understand the role soil biodiversity plays in soil C storage. Multidiversity (diversity across different trophic groups and sizes of soil organisms) is emerging as a critical factor driving soil function in agricultural systems.

Despite the important function of soil biodiversity, a common definition of biodiversity for policymaking is lacking. Monitoring systems and information sharing that link soil biodiversity to soil health would improve understanding and management of soil biodiversity. These features are critical as growth in international travel and trade are predicted to increase the introduction of invasive organisms. Worldwide, costs associated with invasive organisms total hundreds of billions of United States dollars. Early detection is often the best way to limit costs.

Urbanization, soil sealing and soil contamination integrates many soil threats. In this region, urban areas are expanding into forest and agricultural lands. At the same time, urban population density is decreasing. Together, these factors increase the cost of urbanization on natural resources. In addition to the change in land use, urbanization is often accompanied by soil sealing. For example, nearly 30 percent of the soil surface in Toronto, Canada is sealed by asphalt and concrete, not including rooftops. This sealing limits our ability to manage soil C and biodiversity, alters hydrology, and contributes to the UHI effect. Greater rates and volume of surface runoff can increase erosion of the soil that is not sealed.

Urban soils are also elevated in PFAS contaminants. These “forever” chemicals do not break down and accumulate over time in soils and human body. They can be transferred from soils to plants including wheat and maize. The contamination of urban soils by PFAS is a growing problem, and cost-effective strategies are required to remedy this issue. Nevertheless, new opportunities to address soil sealing and contamination have emerged. Biochar applications and soil washing can remove PFAS contaminants, turf can be managed to sequester large amounts of C (rivalling forests) and well-planned urban environments can make large reductions in sealed area compared to historical levels.

Common among the five threats to soil function addressed here is the lack of sufficient, systematic monitoring networks. Threats to soil function generate significant economic costs that affect human health and the environment. A lack of networks to monitor soil systems at landscape or watershed scales limits information for decision-making and our ability to anticipate and manage threats to soil function. Many of these threats interact and feedback with one another, so integrated monitoring networks will both reduce costs and increase benefits.

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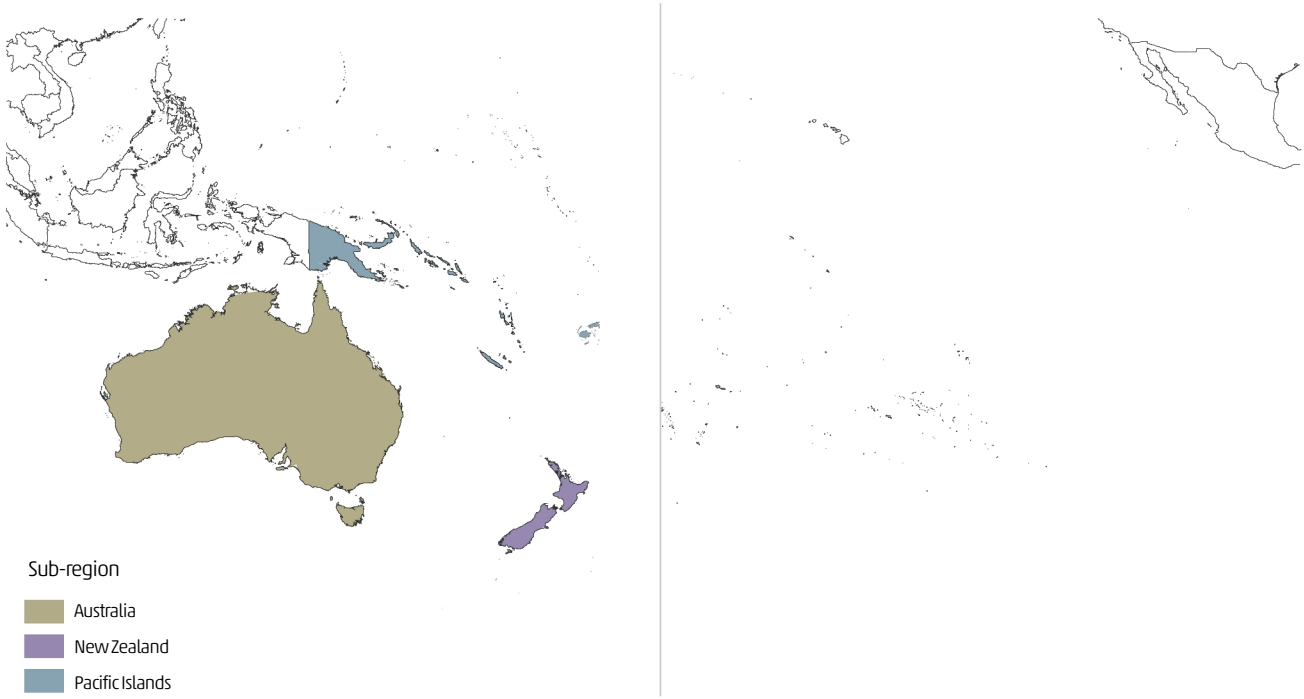
Chapter 13 | Regional assessment for the Southwest Pacific region

13.1 Regional overview

13.1.1 Characteristics of the region and subregions

The Southwest Pacific region is diverse and includes the three subregions of Australia, New Zealand and 22 Pacific Island countries (the latter comprising of over 1 700 small islands dispersed throughout the expansive reaches of the southwest Pacific Ocean) (Figure 13.1).

Figure 13.1. Map of the Southwest Pacific region



Refer to the disclaimer on page ii for the names and boundaries used in this map.

Source: Adapted from **UN Geospatial.** 2020. Map Geodata [Shapefile]. New York, USA, UN.

Australia

Australia is the sixth largest country in the world, with a total area of 7.7 million km^2 comprising of the Australian continent and thousands of small fringing islands (Geoscience Australia, 2014a). The Australian continent includes some of the oldest rocks in the world, with some over 300 million years old (Geoscience Australia, 2014b). The landforms of Australia are the result of tectonic movement and erosion by wind and water over many millions of years. About half of Australia's rivers flow inland, often ending in ephemeral salt lakes. Large tracts of Australia are flat, but the landscape also

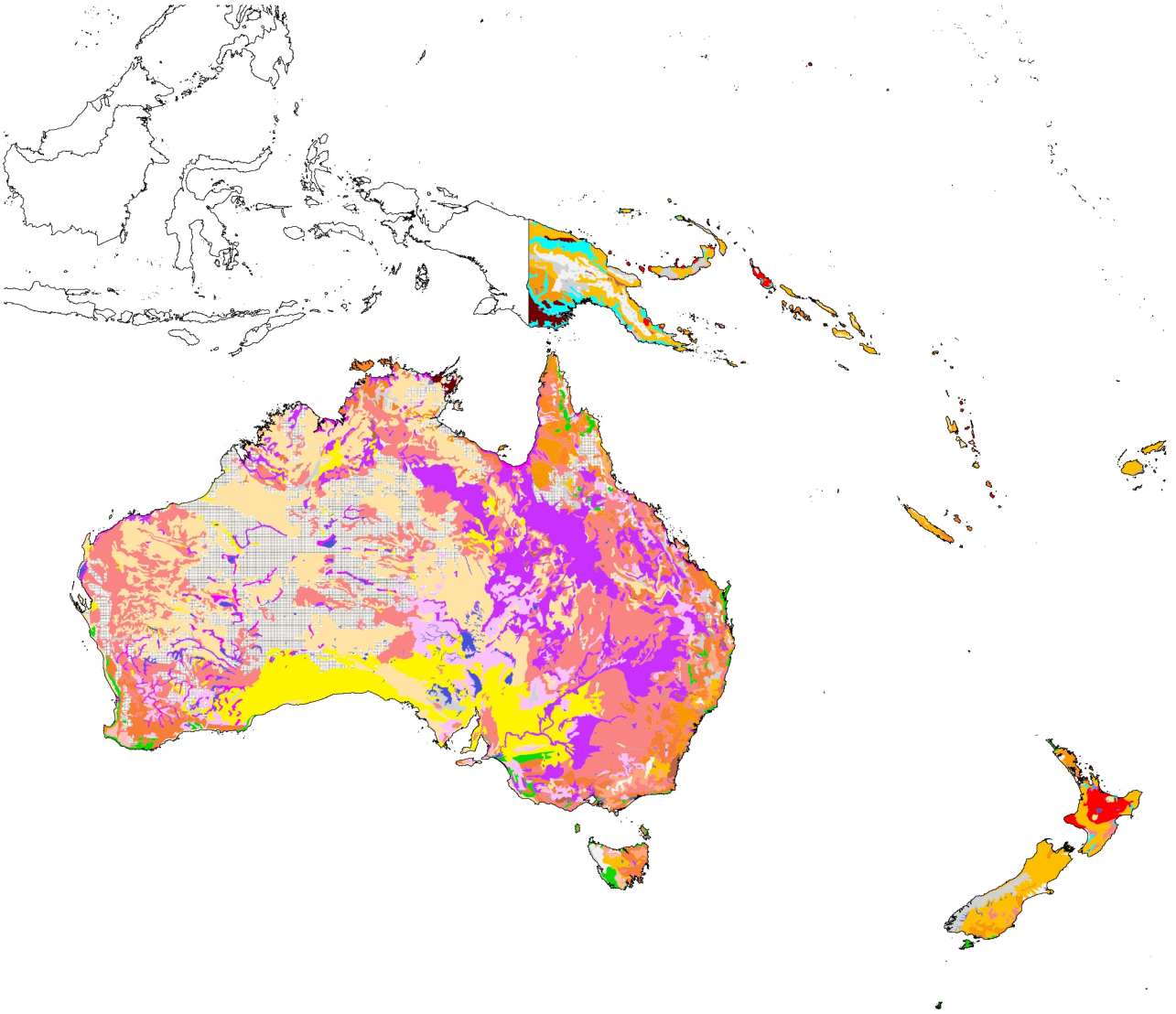
includes numerous ranges. The Great Dividing Range runs parallel to the east coast for 4 000 km, from northern Queensland to Victoria, separating rivers that flow inland towards the north and west from those that flow out to the Pacific Ocean. The mean annual rainfall in Australia ranges from less than 200 mm in the arid central region of Australia, north of Adelaide, to over 2 000 mm on the tropical coastal margins of northern Australia and in the west of Tasmania (Australian Bureau of Meteorology, 2008). Mean annual temperatures vary with latitude from about 11 °C in Hobart, Tasmania, to about 27 °C in Katherine, Northern Territory. Australia is prone to prolonged severe droughts as well as intermittent extreme high intensity rainfall events.

The wetter, more productive regions on the margins of Australia are dominated by Luvisols and Planosols, according to the WRB Soil Classification (IUSS Working Group WRB, 2022) (Figure 13.2). Arenosols, Regosols and Solonetz dominate the drier inland regions. There are also extensive areas of Vertisols in inland Queensland and the Northern Territory (FAO, 1992).

The cropland areas of Australia are found in the southwest corner of western Australia and along the eastern and southeastern coastal areas (Figure 13.3). Grazing is common throughout the vast interior area of grassland and sparse vegetation.



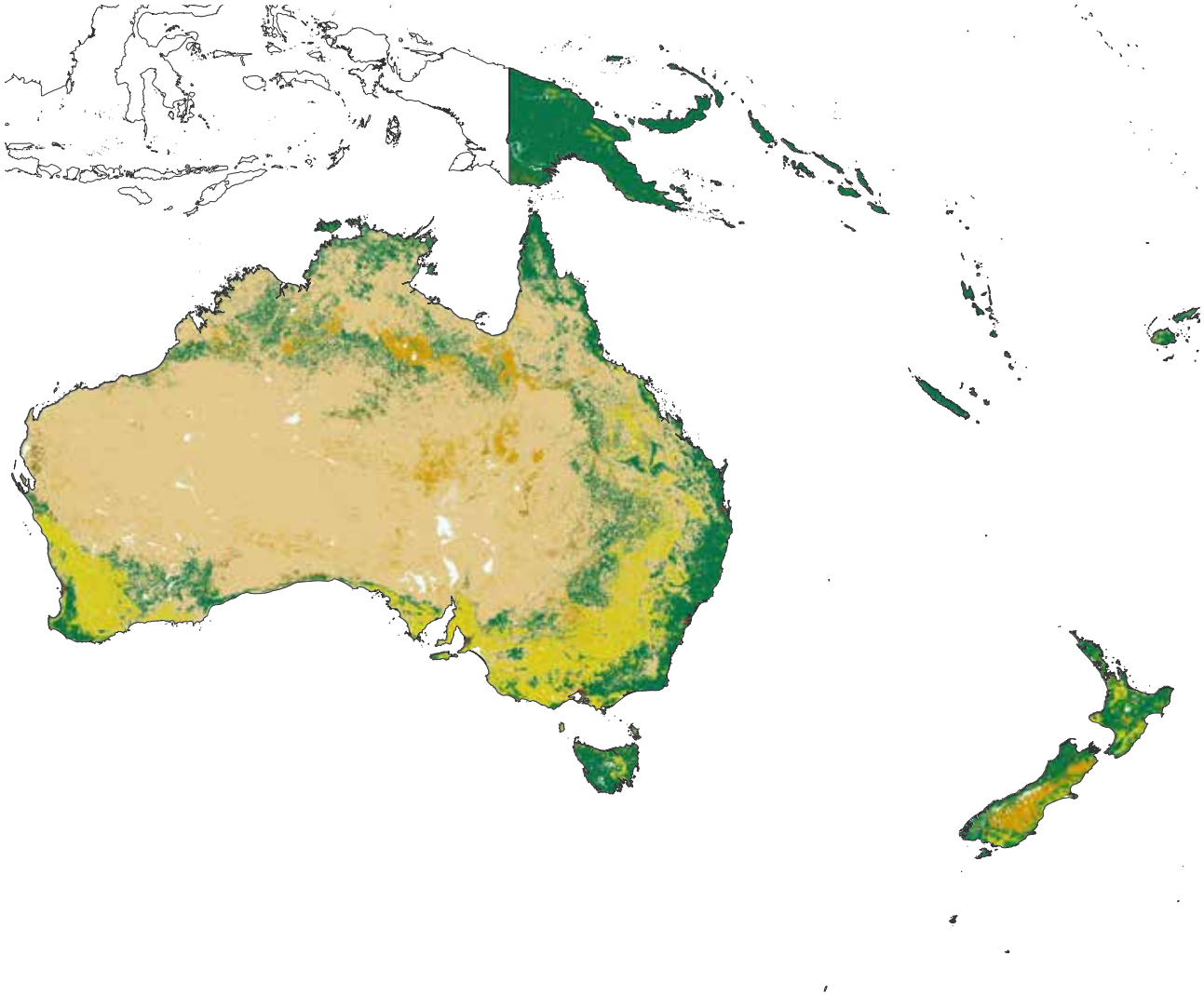
Figure 13.2. Land cover map of the Southwest Pacific



Refer to the disclaimer on page ii for the names and boundaries used in this map.

Source: **FAO.** 2024. *GAEZ: Land and Water Resources.* [Accessed 24 October 2024] https://gaez-services.fao.org/apps/theme-1/. Licence: CC BY 4.0.

Figure 13.3. Land cover map of the Southwest Pacific region



Refer to the disclaimer on page ii for the names and boundaries used in this map.

◊ *Source:* **FAO**. 2024. *GAEZ: Land and Water Resources.* [Accessed 24 October 2024] https://gaez-services.fao.org/apps/theme-1/. Licence: CC BY 4.0.

New Zealand

New Zealand has an area of 268 000 km^2 comprising of two main islands (North Island and South Island) and numerous smaller islands, located approximately 1 500 km east of Australia. New Zealand is situated on the highly active convergent boundary of the Pacific and Australian tectonic plates, with resulting rapid (up to 10 mm annually) tectonic uplift and active volcanism (Hewitt, Balks and Lowe, 2021). Because of this, New Zealand has a large amount of hilly and steep land with a wide variety of parent materials, some of which are particularly vulnerable to landslide erosion. New Zealand extends across latitudes from 34° to 47° south with a mean annual temperature near sea level ranging from 9 to 15 °C and cooler temperatures at higher altitudes. Mean annual rainfalls vary from about 300 mm in the inland basin of Central Otago, to over 6 000 mm at Milford Sound on the west coast of the South Island. Most of New Zealand has generous rainfall and mild temperatures

due to the moderating effects of the surrounding ocean (Hewitt, Balks and Lowe, 2021).

New Zealand has a diverse array of soils with all 12 soil orders represented (according to USDA soil taxonomy) (Hewitt, Balks and Lowe, 2021). In terms of the WRB soil classification, New Zealand is dominated by Cambisols with Andosols dominant in the volcanic regions of central North Island (FAO, 1992) (Figure 13.2). The forested areas occur in the North Island and along the west coast of the South Island with large areas of grazing land in both islands (Figure 13.3).

The Pacific Island countries

The Pacific Island countries can be characterized into two main groups: those predominantly formed on volcanic materials, which extend more than 30 m above sea level (high islands), and low-lying coral reef islands (low islands) (Table 13.1). Some islands are composite, with both volcanic materials and a coral fringe. A few, such as New Caledonia, are comprised of continental crust. The islands of the Southwest Pacific generally have tropical climates with a humid, wetter, cyclone-prone period from November to March and a drier winter period. Mean monthly temperatures are generally 25 °C all year round. The range of parent materials, climate, topography and surface age leads to a wide variety of soils within the Pacific Island countries. For example, Leslie (2010) identified Cambisols, Andosols, Phaeozems, Fluvisols, Regosols, Histosols, Gleysols, and Ferralsols in Samoa. The largest of the island countries (Figure 13.3) are dominantly tree-covered.

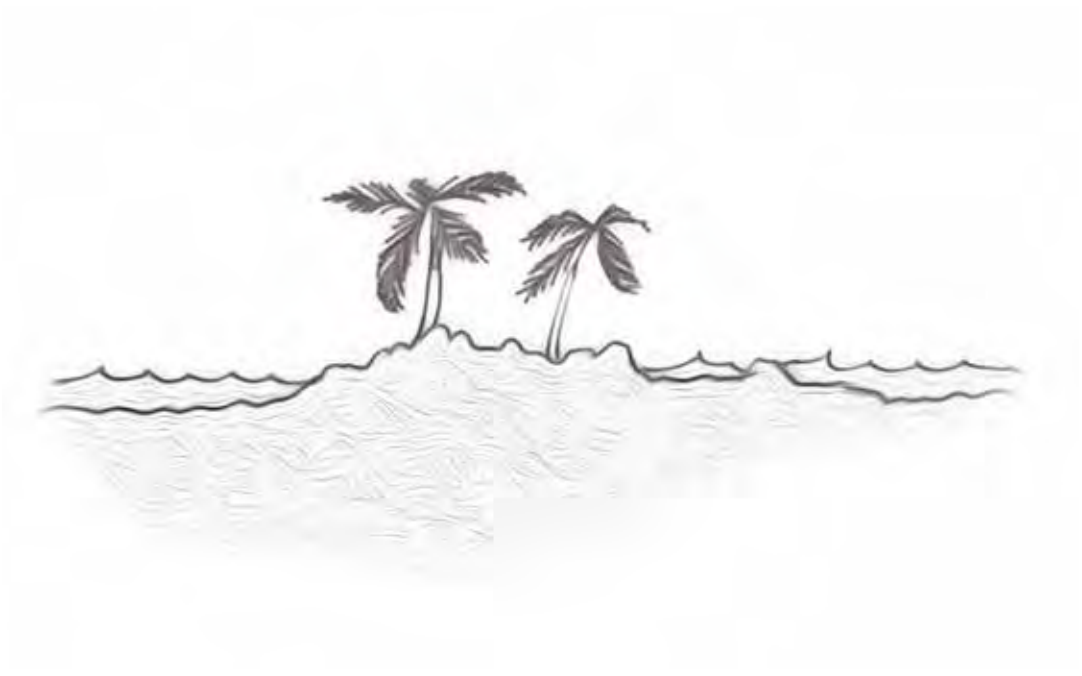


Table 13.1. Classification of Pacific Island countries based on maximum elevation above sea level and dominant and subdominant rock types

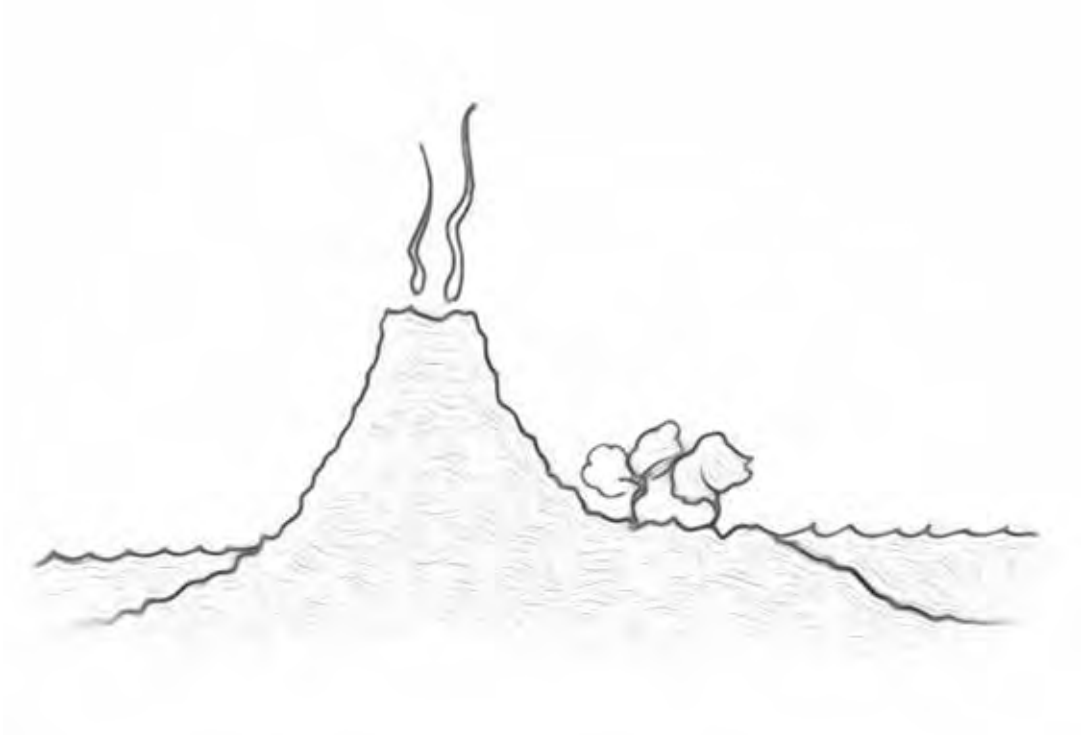
Country	Number of islands	Total area (km²)	Average island maximum elevation (m)	Dominant island type by area[1, 2,3]	Subdominant island type by area
American Samoa	7	222	396	Volcanic high island	Reef islands
Cook Islands	15	297	73	Composite high island	Reef islands
Fiji	211	20 857	134	Volcanic high island	Limestone high island
French Polynesia	125	3 939	154	Volcanic high island	Reef island
Guam	1	588	400	Composite high island	
Kiribati	33	995	6	Reef islands	Limestone low island
Marshall Islands	34	286	3	Reef islands	
Micronesia (Federated States of)	127	799	45	Volcanic high island	Reef island
Republic of Naoero	1	23	71	Limestone high island	
New Caledonia	29	21 613	121	Continental high island	Limestone high island
Niue	1	298	60	Limestone high island	
Northern Mariana Islands	16	537	444	Volcanic high island	Composite high island
Palau	33	495	58	Composite high island	Limestone high island
Papua New Guinea	437	67 754	135	Composite high island	Volcanic high island
Samoa	7	3 046	504	Volcanic high island	
Solomon Islands	415	29 675	88	Composite high island	Volcanic high island
Tokelau	3	16	5	Reef islands	
Tonga	124	847	56	Limestone high island	Volcanic high island
Tuvalu	10	44	4	Reef islands	
Vanuatu	81	13 526	330	Composite high island	Volcanic high island
Wallis and Futuna Islands	14	190	94	Volcanic high island	Reef island

[1] High islands rise to a maximum elevation of at least 30 m above mean sea level.

[2] Composite islands are composed of less than 80 percent volcanic and less than 80 percent calcareous(limestone) rock types.

[3] Continental islands are composed of at least 80 percent continental (not of oceanic origin) rocks.

◊ *Source:* **Nunn, P.D., Kumar, L., Eliot, I. & McLean, R.F.** 2016. Classifying Pacific islands. *Geoscience Letters, 3(1): 7. https://doi.org/10.1186/s40562-016-0041-8*



13.1.2 Summary of state from 2015 State of the World's Soil Resources report

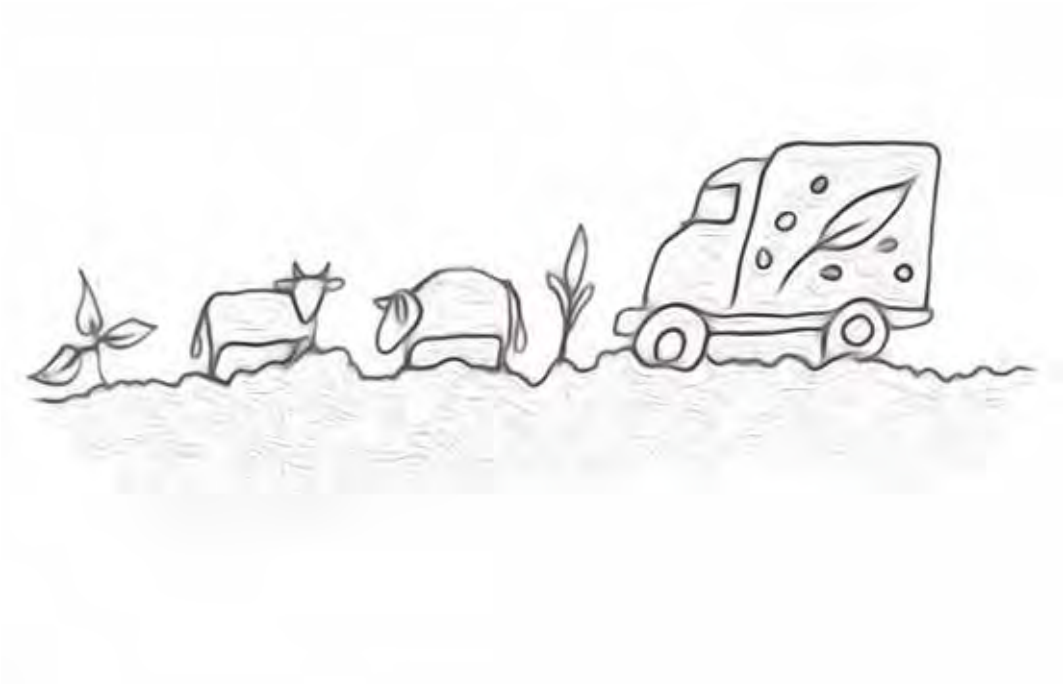
The assessment of state and trend for the Southwest Pacific region in the 2015 SWSR report showed marked differences among the three subregions (Australia, New Zealand and the Pacific Island countries) (FAO and ITPS, 2015).

Soil acidification was judged to be the most serious threat due to its very severe effect in southern Australia, with a strong potential to irreversibly damage soils. The greatest threats in areas with tropical soils were leaching and the removal of soil chemical elements in harvested products.

Soil erosion was the second most severe threat and was widespread throughout the Southwest Pacific region. In Australia and New Zealand, improved land management had reduced rates of erosion in some areas, but unsustainable rates of soil loss were associated with logging and land clearing in several Pacific Island countries.

Land-use conversion from pastures or forests to cropland was the major driver of SOC loss throughout the region. The intensification of agriculture in New Zealand and Australia was also leading to significant environmental harm due to the increased use of synthetic fertilizers and an increase in the number of ruminant animals. In some of the Pacific Island countries, there was evidence of nutrient mining due to insufficient nutrient inputs.

Contamination was generally judged to be in good condition in the region, although legacy contamination issues were being managed in Australia and New Zealand. In some Pacific Island countries there were still some issues to address.



13.2 Changes in the drivers of threats to soil functions

The indirect drivers of threats to SSM in the southwest Pacific region discussed here include population growth, land-use change and land tenure issues in the Pacific Island countries. Direct drivers include nutrient mismanagement, with nutrient mining of concern in some Pacific Island countries, along with some land management practices that lead to soil erosion.

13.2.1 Population growth

Population growth is uneven across the Pacific Island countries (UN DESA, 2023). Most Pacific Island countries are projected to have growth rates below the global averages for 2021 to 2025 and 2025 to 2035, with absolute population decreases occurring in American Samoa, Palau, and the Wallis and Fortuna Islands. Population increases above the global averages are projected for Kiribati, Papua New Guinea, Samoa, Solomon Islands and Vanuatu.

In 2015, the Federated States of Micronesia, Samoa, Solomon Islands, Tonga and Vanuatu had relatively low urban populations (<30 percent) (UN DESA, 2018). The percentage growth rate of urban population is projected to be high (>15 percent) between 2015 and 2035 in Fiji, Kiribati, Niue, Solomon Islands, Tuvalu and Vanuatu. Australia and New Zealand already have highly urbanized societies (>85 percent urban) and are projected to have low increases (<5 percent) in urban populations through to 2035.

Land degradation in the Pacific Island countries is caused by increasing populations, the pressure of food production and the increasing economic drive for commercial production for export and import substitution (Wairiu, 2016). In most high Pacific islands (i.e. those with maximum elevations at least 30 m above sea level), extended cropping periods on the same piece of land with reduced fallow periods have resulted in the reduction of soil fertility, with a consequent reduction in crop yields (Bourke *et al.* 2005; Government of Samoa, 2007; Government of Vanuatu, 2007a). In Fiji, competition and pressure for land through population growth, the expansion of cash cropping and grazing on the flatter lands have forced subsistence gardens onto steeper slopes. Some gardens have experienced soil loss, especially when traditional mulching was not being practiced, and fallow periods were too short. Soil loss measurements clearly demonstrate that the agricultural productive base in many sugar cane-growing areas and with ginger on slopes, is declining at a rate that is well above what would be regarded as economically sustainable (Government of Fiji, 2007).

Population growth in Australia and New Zealand was higher than global averages in the 2015–2021 period but is projected to be close to global averages through to 2035.

As demonstrated by UN DESA (2018), the countries undergoing significant population increases over the next decade are also projected to have increases in the percentage of the population in urban areas. The growth in urban population puts further stress on agricultural and natural lands surrounding the urban areas. In New Zealand and the Pacific Island countries, population growth and associated urban expansion continues to cause the loss of the limited areas of highly productive soils.

13.2.2 Land-use changes

The land-use changes and nutrient use data for the Southwest Pacific are presented using both maps and tables. The maps display data for Australia, New Zealand and the larger of the Pacific Island countries (Fiji, New Caledonia, Papua New Guinea, Solomon Islands, and Vanuatu). Data for the remaining Pacific Island countries are presented in tabular format.

Australia experienced a substantial increase in cropland area (a gain of 253 00 ha) in the 2015–2022 period (FAO, 2024a) (Figure 13.4). New Zealand had a higher percentage increase in cropland (11 percent gain) but a lower areal gain (60 000 ha). Papua New Guinea also had a substantial increase a gain of 83 000 ha for a 7.3 percent increase.

Figure 13.4. Country map of change in cropland area between 2015 and 2022



Change to cropland from 2015 to 2022

100 000 ha gain to 1 million ha gain | 100 000 ha loss to 100 000 ha gain

Refer to the disclaimer on page ii for the names and boundaries used in this map.

◊ *Source:* **FAO** 2024. *FAOSTAT: Land Use.* [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

The majority of Pacific Island countries reported no change in cropland area between 2015 and 2022 (FAO, 2024a) (Table 13.2). The greatest percentage changes were in French Polynesia (a gain of 11.8 percent), with substantial losses in Guam (a loss of 11.1 percent), Northern Mariana Islands (a loss of 66 percent) and Samoa (a loss of 32.3 percent).

Table 13.2. Land area, cropland land area and change in cropland between 2015 and 2022 for Pacific Island countries

Country	Land area (2022) (ha)	Cropland area (2022) (ha)	Change in cropland area 2015–2022 (%)
American Samoa	20 000	3 000	6.6
Cook Islands	24 000	2 000	0.0
French Polynesia	347 000	29 000	11.8
Guam	54 000	8 000	–11.1
Kiribati	81 000	34 000	0.0
Marshall Islands	18 000	7 000	0.0
Micronesia (Federated States of)	70 000	2 000	0.0
Republic of Naoero	2 000	> 1000	0.0
Niue	26 000	4 000	0.0
Norfolk Island	4 000	> 1000	0.0
Northern Mariana Islands	46 000	> 1000	–66.0
Palau	46 000	2 000	0.0
Samoa	278 000	43 000	–32.3
Tokelau	1 000	1 000	0.0
Tonga	72 000	31 000	0.0
Tuvalu	3 000	2 000	0.0
Wallis and Futuna Islands	14 000	6 000	0.0

◊ *Source:* **FAO** 2024. *FAOSTAT: Land Use.* [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

Only four countries in the Southwest Pacific region report changes to the area of permanent meadows and pastures in the 2015–2022 period (FAO, 2024a). Australia reported a substantial gain of 15 147 000 ha, representing a gain of 4.8 percent. New Zealand reported a substantial decrease in the area of permanent meadows and pastures of 1 025 000 ha (representing a 10.1 percent decrease). The Northern Mariana Islands reported a decrease of 1 000 ha (representing a 69 percent decrease) and Samoa had a decrease of 6 000 ha (representing a 47 percent decrease).

Australia, and to a lesser extent, Fiji and New Zealand, reported very large areas of reforestation in the 2015–2025 period (Figure 13.5). Globally, Australia had the third-highest area of reforestation (a gain of 910 000 ha in this time period) after China and India (FAO, 2024a). Substantial deforestation was reported in Papua New Guinea , with smaller areas in the Northern Mariana Islands, New Caledonia, Samoa and Solomon Islands (Table 13.3).



Figure 13.5. Country map of change in forestland area between 2015 and 2022

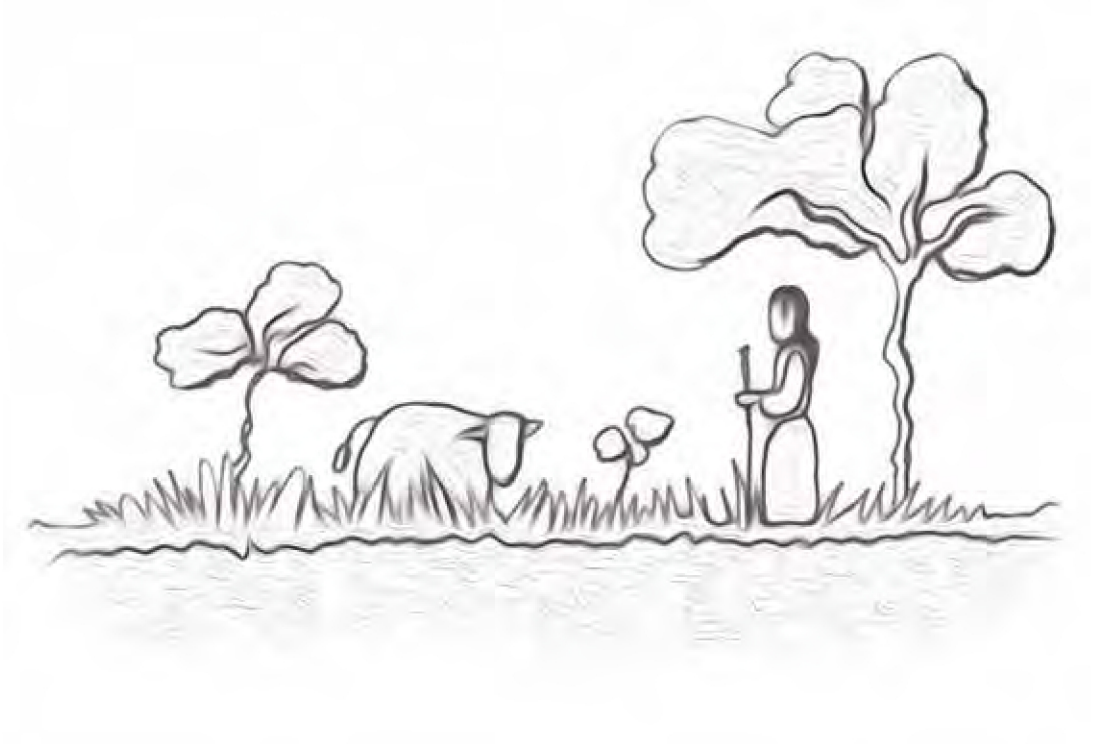


Change in forestland from 2015 to 2022

100 000 ha gain to 1 million ha gain | 100 000 ha loss to 100 000 ha gain | 100 000 ha loss to 1 million ha loss

Refer to the disclaimer on page ii for the names and boundaries used in this map.

Source: **FAO** 2024. *FAOSTAT: Land Use.* [Accessed 5 November 2024] https://www.fao.org/faostat/en/#data/RL. Licence: CC BY 4.0.

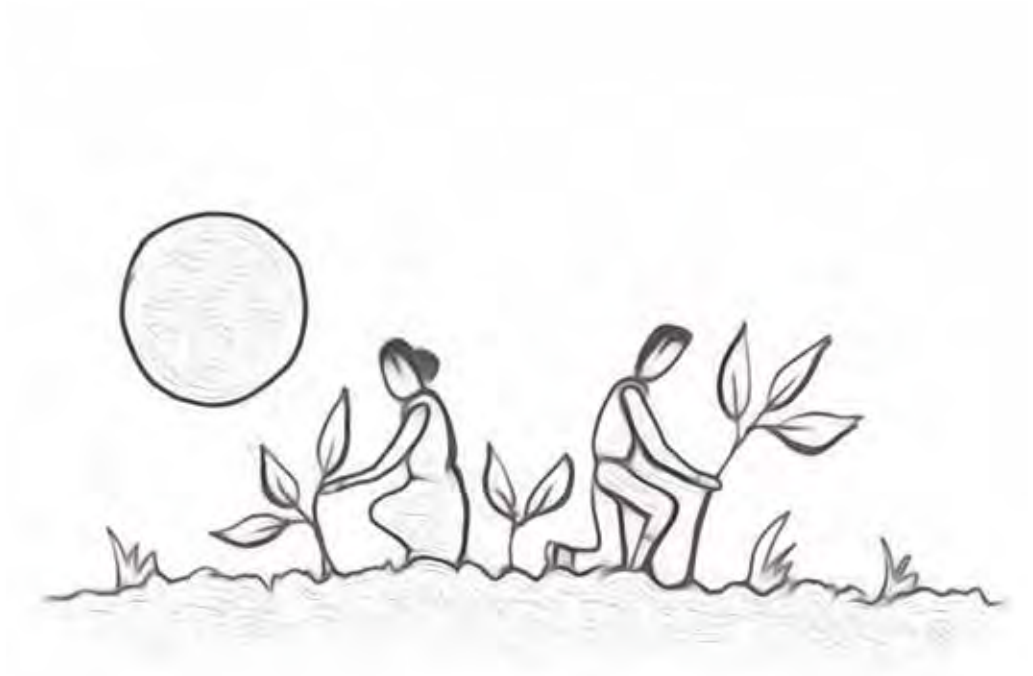


In the Pacific Island countries (Table 13.3), a substantial loss of forestland occurred in the Northern Mariana Islands (a percentage decrease of 17 percent) and a substantial gain in Guam (a gain of 12 percent) between 2015 and 2022. Eleven countries reported no changes in the area of forestland.

Table 13.3. Land area, forestland area and change in forestland between 2015 and 2022

Country	Land area (ha)	Forestland area (%)	Change in forestland area 2015–2022 (%)	Areal change in forestland (2015–2022) (ha)
American Samoa	20 000	85.4	−1.2	−210
Cook Islands	24 000	65.0	0.0	0.0
French Polynesia	347 000	43.1	0.0	0.0
Guam	54 000	51.9	12.0	3 000
Kiribati	81 000	1.5	0.0	0.0
Marshall Islands	18 000	52.2	0.0	0.0
Micronesia (Federated States of)	70 000	92.1	0.3	210
Republic of Naoero	2 000	0.0	0.0	0.0
Niue	26 000	72.7	0.4	70
Norfolk Island	4 000	12.3	0.0	0.0
Northern Mariana Islands	46 000	53.0	−17.4	−5 130
Palau	46 000	90.4	1.4	580
Samoa	278 000	57.8	−2.0	−3 360
Tokelau	1 000	0.0	0.0	0.0
Tonga	72 000	12.4	0.0	0.0
Tuvalu	3 000	33.3	0.0	0.0
Vanuatu	1 219 000	36.3	0.0	0.0
Wallis and Futuna Islands	14 000	41.6	0.0	0.0

◊ *Source:* **FAO** 2024. *FAOSTAT: Land Use.* [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.



13.2.3 Nutrient management

Most of the Pacific Island countries reported average nutrient balances between 2017 and 2021 at levels indicative of yield gaps or nutrient mining. Ten countries reported negative N balances, ten countries had negative P balances and nine countries reported negative K balances (FAO, 2024b) (Table 13.4). Negative balances of all three nutrients occurred in Kiribati, the Federated States of Micronesia, Papua New Guinea, Solomon Islands, and Vanuatu.

High surpluses of P were reported in the Cook Islands, Fiji, New Caledonia, and Tuvalu, while N and K levels were within or close to acceptable ranges for these countries. According to FAO (2024b) (Table 13.4), both Australia and New Zealand had levels of soil N and P within acceptable levels. Australia showed a slight K deficit in the 2017 to 2021 period.

Table 13.4. Average values (2017 to 2021) for nitrogen, phosphorus and potassium nutrient use in the Southwest Pacific and accumulated phosphorus residual from 1961 to 2019

Country	N outputs[1] (kg/ha)	N nutrient balance[2] (kg/ha)	P nutrient balance[3] (kg/ha)	Accumulated P residual[4] (kg/ha)	K nutrient budget[5] (kg/ha)
Pacific Island countries					
Cook Islands	45.8	62.7	36.8	1071.0	34.9
Fiji	55.6	10.4	7.0	808.0	−2.2
French Polynesia	50.0	−23.0	−1.0	107	−5.5
Kiribati	90.1	−78.6	−12.6	−77.0	−31.9
Micronesia (Federated States of)	585.8	−388.1	−25.2	43.0	−55.9
Republic of Naoero	95.9	−41.5	6.0	295.0	6.4
New Caledonia	50.1	82.2	32.9	538.0	95.3
Niue	14.4	−2.5	0.4	−1.0	−1.0
Papua New Guinea	46.8	−10.5	−2.4	−56.0	−25.6
Samoa	30.2	−7.1	3.1	158.0	−4.9
Solomon Islands	37.6	−21.6	−4.5	−178.0	−23.3
Tonga	39.4	−7.6	3.2	190.0	1.5
Tuvalu	20.6	46.7	22.7	495.0	42.1
Vanuatu	41.6	−31.4	−3.9	−69.0	−8.5
Australia	38.7	17.4	3.6	490.0	−1.6
New Zealand	45.5	28.1	3.8	3378.0	26.3

Notes: Yellow highlighted cells indicate nutrient mining and brown highlighted cells indicate nutrient excess.

[1] A very low annual nitrogen surplus (<20 kg/ha) may be associated with nutrient mining (EU Nitrogen Expert Panel, 2015)

[2] Negative values for N nutrient balance indicate mining of N from soil organic matter. Nitrogen surpluses greater than 80 indicate a high risk of N pollution.

[3] Negative values for P nutrient balance indicate mining of P from soil. Annual values for P nutrient balance greater than 6.9 kg P/ha may be unsustainably high (Zou, Zhang and Davidson, 2022).

[4] Negative values for accumulated P residual indicate a depletion of soil P reservoir since 1961. Positive values indicate a surplus of P additions (Zou, Zhang and Davidson, 2022).

[5] Negative values for K indicate mining of K from soil.

◊ *Sources:* **FAO.** 2024. *FAOSTAT: Fertilizers by Nutrient.* [Accessed on 9 January 2024]. https://www.fao.org/faostat/en/#data/RFN. Licence: CC-BY-4.0.

◊ **EU Nitrogen Expert Panel.** 2015. *Nitrogen Use Efficiency (NUE) - an indicator for the utilization of nitrogen in agriculture and food systems.* Wageningen, Kingdom of the Netherlands, Wageningen University. https://research.wur.nl/en/publications/nitrogen-use-efficiency-nue-an-indicator-for-the-utilisation-of-n

◊ **Zou, T., Zhang, X. & Davidson, E.A.** 2022. Global trends of cropland phosphorus use and sustainability challenges. *Nature,* 611(7934): 81–87. https://doi.org/10.1038/s41586-022-05220-z

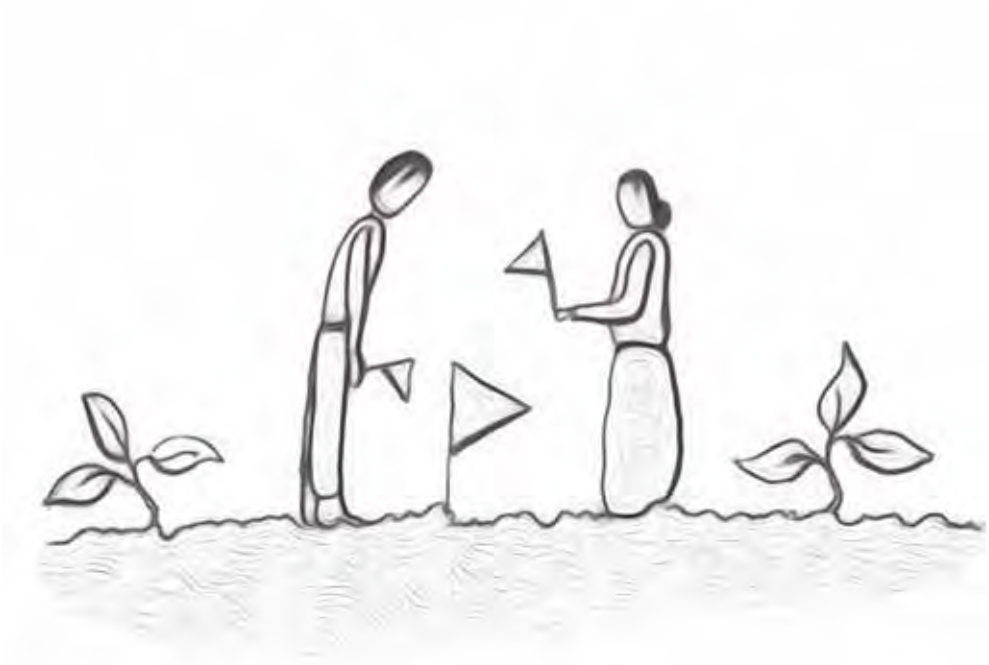
13.2.4 Other drivers in the Pacific Island countries

Inappropriate agricultural practices

Inappropriate agricultural practices continue to be a significant driver of land degradation in Pacific Island Countries (PICs). Expansion of cultivation onto steep slopes, shortening of traditional fallow periods, removal of protective vegetation cover, and increasing agricultural intensification have accelerated soil erosion, nutrient depletion, and sediment transport to freshwater and coastal ecosystems (IPBES, 2018; Thaman, 2014). These pressures are compounded by population growth, increasing food demand, and the transition from subsistence farming to market-oriented production systems. Agricultural intensification associated with crops such as coffee in Papua New Guinea, taro in Fiji, kava in Tonga and Vanuatu, and other export commodities has increased pressure on fragile landscapes, particularly where soil conservation practices are inadequately implemented (Wairiu, 2017; IPBES, 2018; UNEP, 2019; Kiup *et al.*, 2025). The resulting loss of soil fertility and ecosystem services threatens both agricultural productivity and the resilience of Pacific food systems (FAO, 2020).

Land tenure

Land tenure arrangements play a critical role in determining land management outcomes across PICs. Much of the region operates under customary land ownership systems, which provide important social and cultural benefits but can also create challenges for long-term agricultural investment and sustainable land management where tenure security is uncertain (Crocombe, 1995; Fingleton, 2008). In some cases, short-term lease arrangements may discourage investments in soil conservation and rehabilitation, contributing to land degradation. Fragmentation of landholdings, increasing population pressure, and competing demands for agricultural, residential, and commercial development have further intensified pressures on coastal zones, wetlands, and watershed areas. These changes have altered hydrological processes, increased erosion and sedimentation, and reduced the capacity of ecosystems to provide critical environmental services (World Bank Group, 2024; ACIAR, 2023; McGree *et al.*, 2022).



13.3 Regional status and trend of threats to soil functions for Australia

13.3.1 Indigenous Peoples' viewpoint

In Australia's State of the Environment 2021 report, it was stated that:

> *Indigenous knowledge and connections to Country are vital for sustainability and healing Australia. Indigenous people have cared for Country across generations for tens of thousands of years. With decreasing health of Country, Indigenous people continue to seek a larger role in managing its recovery back to health. Indigenous people seek greater participation in Australia's environmental management system. Respectful use of Indigenous knowledge, recognition of Indigenous knowledge rights, and Indigenous and non-Indigenous knowledge systems working together will lead to positive change. A renewed emphasis on engagement across all sectors of society is required to reverse environmental decline and to achieve ecologically sustainable development that underpins future prosperity and the wellbeing of future generations. Renewed focus on restoration of the landscape, and greater recognition and empowerment of Indigenous land management practices, where possible, across large parts of Australia can help us to heal Country and find new ways to gain a broad range of benefits* (Cresswell, Janke, and Johnston, 2021, p. 12).

13.3.2 Soil erosion in Australia

Soil erosion is a widespread threat to environmental values (Lal, 2001; McCloskey *et al.,* 2021). Soil erosion reduces land productivity by removing soil and with it soil C, N and P. In New South Wales in Australia, both water and wind erosion have contributed to the nutrient transport rate with 2.4 percent of the surface soil (0–5 cm) removed from the total stock (Yang *et al.,* 2023). Soil water-holding capacity and nutrient status are important for the productivity of crops and pastures. Soil erosion also causes offsite degradation by exporting sediment and nutrients, and contaminants bonded to the sediments, which can degrade aquatic and marine environments (Bui, Hancock and Wilkinson, 2011; McTainsh and Strong, 2007). Gully erosion has been identified as the largest source of fine silt and clay sediments in catchments draining to the Great Barrier Reef lagoon (McCloskey *et al.*, 2021; Wilkinson *et al.,* 2023). Australia has also been identified as the third-largest global dust source, producing 20 percent of the global dust emission with an estimated annual emission of 5.9 +/– 3.0 million tonnes (Chappell *et al.*, 2023).

There are minimal data on actual extent of soil erosion across Australia (e.g. Freebairn and Wockner, 1986; O'Loingsigh *et al.*, 2014), due to the expense of acquisition and the expansive area needing to be covered. Modelling can address these issues and incorporate important spatial and temporal risk factors to provide nationally consistent, validated estimates of hillslope soil erosion (Yang *et al.*, 2023). Wind erosion estimates have explored several avenues (Cattle, 2016; De Deckker, 2020; Marx *et al.*, 2018; O'Loingsigh *et al.*, 2014; Metcalfe and Bui, 2017). Current estimates utilize combinations of aerosol monitoring network (Leys *et al.*, 2008; Leys *et al.*, 2018), satellite data (Chappell *et al.*, 2023; O'Loingsigh *et al.*, 2017) and modelling (Nguyen *et. al.*, 2019; Chappell *et al.*, 2023).

Teng *et al.* (2016) used RULSE (Renard *et al.*, 1995) to estimate hillslope water erosion for Australia. The greatest zones of water erosion risk were along the coastal regions of Queensland, New South Wales, Victoria, Tasmania and in the north of Western Australia. These areas have high rainfall

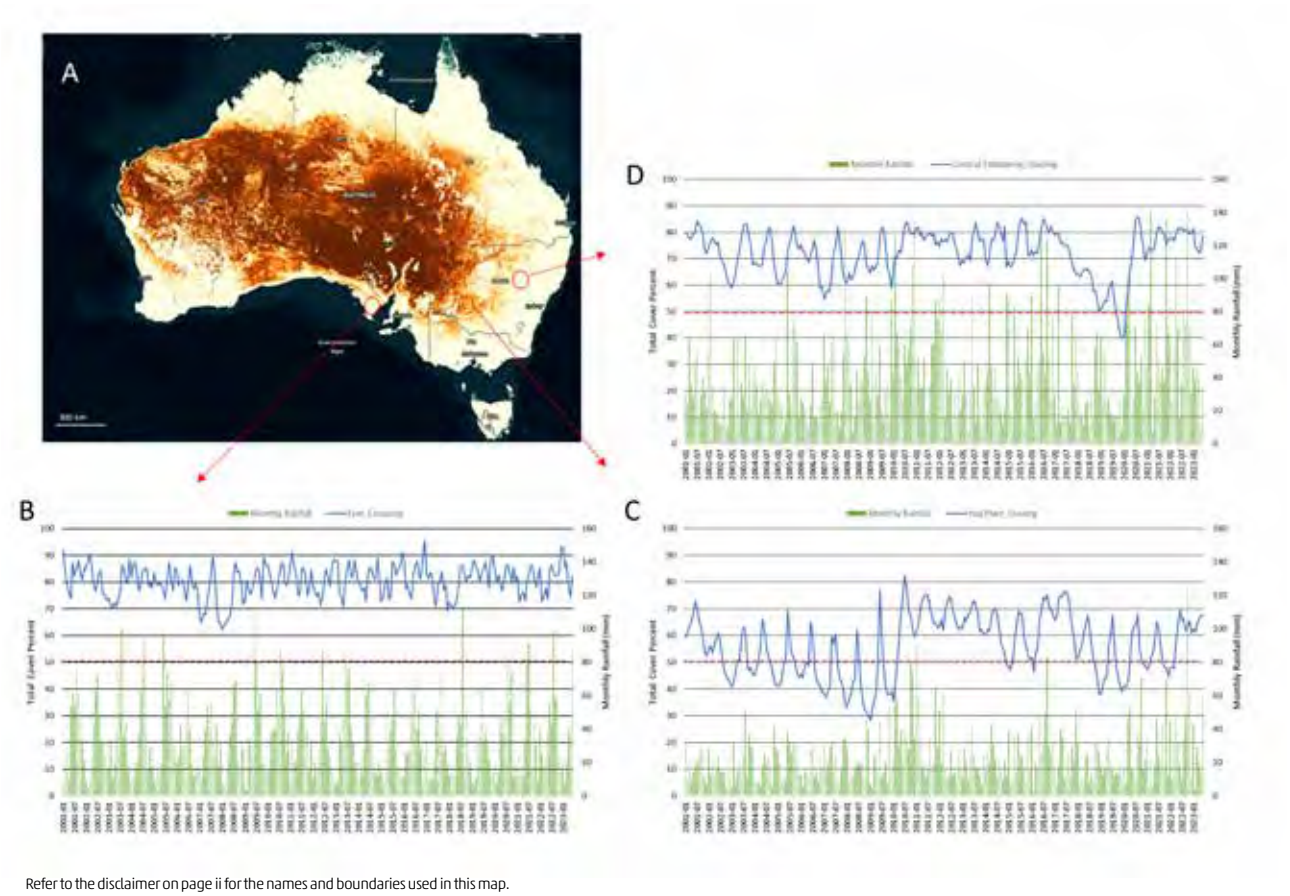
erosivity, steep slopes and inherently vulnerable soil types. Gully erosion is also widespread across Australia (Hughes and Prosser 2012). Solonetz (sodic soils) (IUSS Working Group WRB, 2022) that cover 28 percent of Australia are more vulnerable to gully erosion, and this soil type is often used for agriculture (for example, 73 percent of Victoria's agricultural land) (Naidu, Sumner and Rengasamy, 1993).

Wind erosion activity in Australia, like water erosion, is highly episodic. It is influenced strongly by climate drivers that impact vegetation growth. Hence, vulnerable areas for wind erosion occur throughout the arid and semi-arid rangelands of inland Australia, or where annual rainfall is consistently below 600 mm and high wind speeds (>65 km/hr) are prevalent.

Managing wind and water erosion risk requires maintaining more than 50 percent ground cover (Freebairn and Wockner, 1986; Silburn *et al*., 2011) (Figure 13.6). By using ground cover as a proxy measure of erosion risk, the effects of changing land management practices on ground cover can also be assessed. The recent modelled estimates of water erosion are generally smaller than previous continental estimates (Lu *et al*., 2003; Teng *et al*., 2016). The wind and water erosion potential in Australia are shown in Figure 13.6 and Figure 13.7 for land uses and regions over the 2000–2023 period. There are positive signs that ground cover has been maintained at above the required minimum 50 percent cover. Central to improvement has been the increased awareness by farmers of the importance of retaining ground cover (Leys *et al*., 2023a, 2023b). The erodible cropping regions of southern Australia are more protected from wind erosion due to the widespread adoption of conservation farming strategies, resulting in persistent ground cover throughout much of the year (Figure 13.6A and Figure 13.7B). In contrast, grazing lands in Hay Plain Rangelands of New South Wales (Figure 13.3D) or the Darling Downs cropping lands of Queensland (Figure 13.7D) demonstrate frequent low cover, highlighting the need for improved management to maintain ground cover in a drying environment, or in areas more prone to high intensity rain events and flooding (Figure 13.6B and Figure 13.7D). Climate-induced shifts in drought extent and intensity have increased the vulnerability of management systems to erosion risk, as shown during the 2019 drought in Australia (Figure 13.6C).



Figure 13.6. Wind erosion potential between 2001 and 2023 in South Australia and New South Wales in Australia



Refer to the disclaimer on page ii for the names and boundaries used in this map.

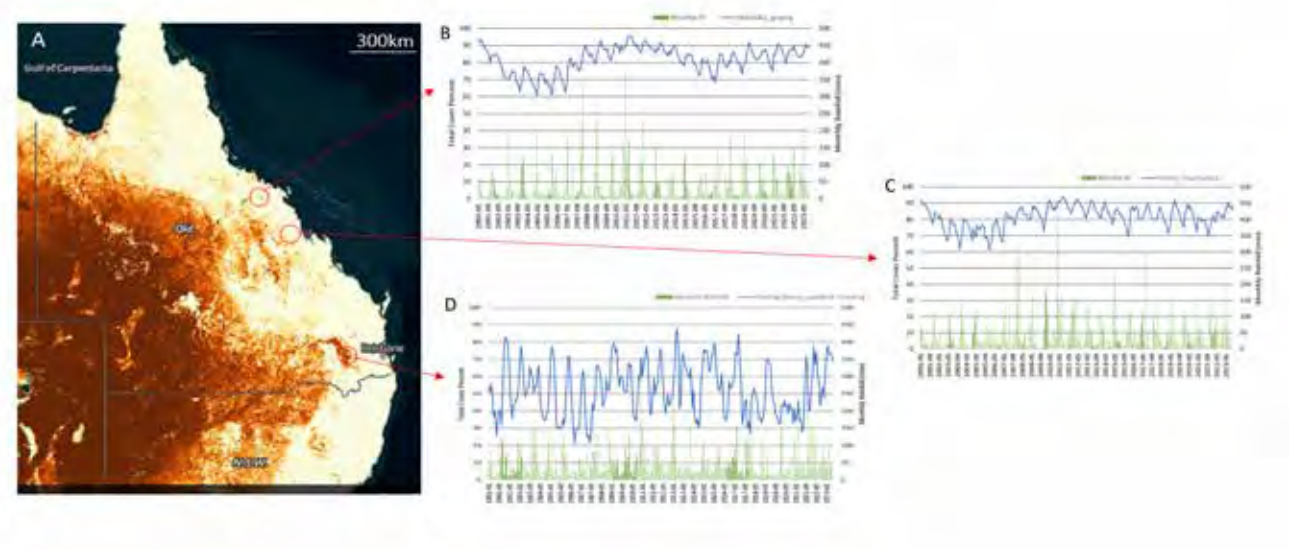
Notes: A) Long term summer average total cover below 50 percent (January 2000 to January 2023), B) Eyre Peninsula monthly total cover for cereal cropping land-use, C) Central Tablelands monthly total cover for grazing native pastures land-use; D) Hay Plain rangelands monthly total cover for grazing native pastures land-use.

The green line is local monthly rainfall, the blue line is monthly total cover (MODIS), and the red dotted line highlights the 50 percent total cover target for wind erosion.

Source: **CSIRO.** 2023. GEOGLAM RAPP Rangeland and Pasture Productivity. In: *GEORAPP.* Canberra. [Cited 20 September 2023]. https://www.geo-rapp.org/



Figure 13.7. Water erosion potential between 2001 and 2023 in Queensland, Australia



Refer to the disclaimer on page ii for the names and boundaries used in this map.

Notes: A = Long term summer average total cover below 70 percent (January 2000 to January 2023), B = Burdekin Natural Resource Management (NRM) region monthly total cover for grazing land-use, C = Fitzroy NRM region monthly total cover for grazing native pastures land-use, and D = Darling Downs monthly total cover for cropping.

The green line is local monthly rainfall. the blue line is monthly total cover (MODIS), and the red dotted line highlights the 50 percent total cover target for water erosion.

Source: **CSIRO.** 2023. GEOGLAM RAPP Rangeland and Pasture Productivity. In: *GEORAPP*. Canberra. [Cited 20 September 2023]. https://www.geo-rapp.org/

Government investment in producing open source, satellite-derived estimates of ground cover has been crucial in reducing erosion risk, as land managers are better informed on the effects of grazing management on feed availability and maintaining at least 50 percent ground cover (Guerschman *et al.*, 2016; Leys *et al.*, 2020, 2023a, 2023b).

13.3.3 Soil carbon loss in Australia

Since 1788, an estimated 91 million ha of woodland and forest, mallee, heath and grassland has been cleared in Australia, primarily for agricultural and settlement purposes (NLWRA, 2001). After native vegetation has been cleared for agriculture there is in a resulting rapid loss of SOC stocks (for example, land-use change for cropping in Australia has released an estimated 51 percent of the SOC stored in the top 10 cm of soil) (Luo, Wang and Sun, 2010). The clearing from the 1960s onwards of 58 percent of the Brigalow Belt bioregion for agriculture – occupying nearly 37 million ha in Queensland and New South Wales – has resulted in a 46 percent loss of SOC (Thornton and Shrestha, 2021). The loss in SOC is primarily from the biological and abiotic oxidation of C through soil disturbance, including the loss in soil structure and the increased erosion of topsoil, and the decline of C inputs when woodland is converted to cropping. Across Australia, SOC losses are estimated at 1.8 million tonnes for 0–30 cm depth, 5.4 million tonnes for 0–100 cm depth and 10 million tonnes for 0–200 cm depth since the advent of land conversion to agriculture, particularly with increases in cropping since the 1960s (Sanderman, Hengl and Fiske, 2017).

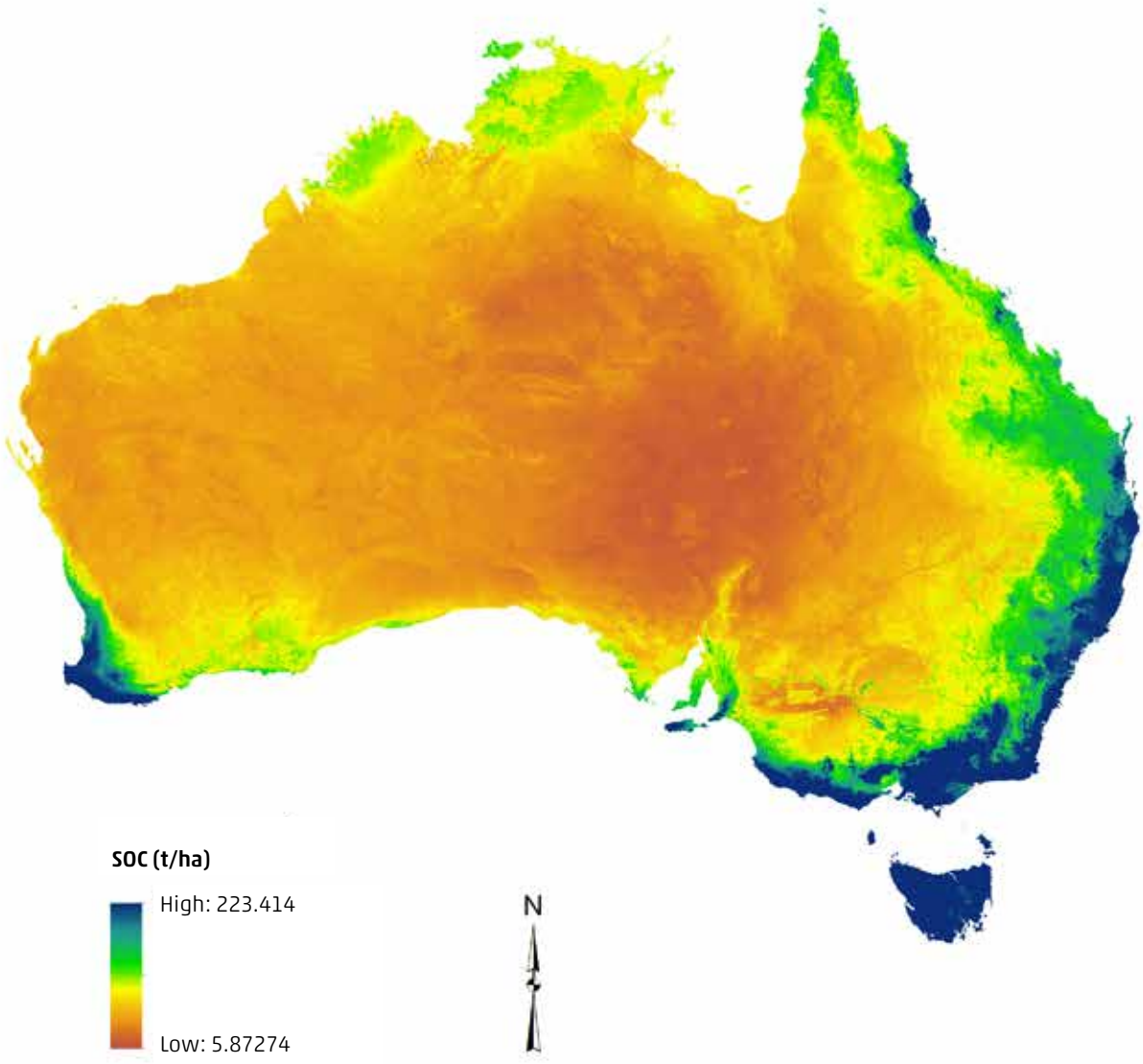
It has recently been estimated that the topsoil layer (0–30 cm) in Australia contains around 28 million tonnes of C, comprising of an average of 27.6 million tonnes of C in terrestrial ecosystems and an average of 0.35 million tonnes of C in wetland (blue C) ecosystems (Viscarra Rossel *et al.*, 2014; Walden *et al.*, 2023). This amount of C makes up approximately 3.5 percent of the total global stock (793 million tonnes at 0–30 cm depth), with Australia accounting for 5.2 percent of the global land area (Padarian *et al.*, 2022). Viscarra Rossel *et al.* (2019) interrogated the different C fractions

across Australia's soils to determine the potential vulnerability of stocks to change. It was estimated that in the 0–30 cm layer, the mineral-associated C fraction stored a total of 20.8 million tonnes of C, the resistant organic carbon (ROC) (comprising of aromatic structures such as charcoal from fires) stored 9.1 million tonnes of C and the particulate organic carbon (POC) fraction stored 5.4 million tonnes of C.

Soils of southern Australia are considered to be the most vulnerable to soil C loss (Viscarra Rossel *et al*., 2019) due to their higher total C stocks and relatively large POC fractions, which are expected to be more readily mineralized compared to mineral-associated C or ROC. Soils with greater mineral-associated C and ROC commonly occur in central and northwest Australia and the semi-arid climates of northern Australia, and were shown to be potentially less vulnerable to C loss, but also typically contain lower C stocks (Figure 13.8). The key environmental, landscape and edaphic factors that influenced the vulnerability of the POC fraction were solar radiation and the mean annual temperature, elevation, soil pH, ground cover, and total N. Total N was also an important factor in the mineral-associated C pool due to the tight C:N ratio that stabilised organic matter typically exhibits, alongside climate, clay type, P availability and ground cover.



Figure 13.8. Distribution of soil organic carbon stocks from 0–30 cm



◊ *Source:* Adapted from **Viscarra Rossel, R.A., Webster, R, Bui, E.N. & Baldock, J.A.** 2014. Baseline map of organic carbon in Australian soil. *Global Change Biology,* 20(9): 2953–2970. http://dx.doi.org/10.4225/08/5397E11CC3383

Emerging soil management techniques such as the application of biochar contribute directly to the ROC pool and could result in an increase of the SOC storage ceiling over longer time frames by stabilizing C inputs (Weng *et al.*, 2022), but uncertainty remains on the level of change. There is also an increasing interest in the role of Fe minerals in stabilizing and protecting C in soil (Song *et al.*, 2022), which occurs through processes such as adsorption and co-precipitation of rhizodeposits which protect them from microbial mineralization, or through physical protection via aggregation. Soil conservation practices that may contribute to increased SOC include maintaining ground cover and ensuring continual C inputs into soil, minimizing soil disturbance, increasing plant species diversity and integrating livestock with crop production (e.g. introducing a pasture phase in continuous cropping systems) (Khangura *et al.*, 2023). However, most of these practices are well-accepted and currently practiced across Australia's dryland cropping regions, thus making further improvements in SOC unlikely.

Recognizing the potential benefits for climate change mitigation and soil health from enhancing SOC stocks, soil management has been included in the Australian emissions trading scheme. Landholders can earn Australian Carbon Credit Units (ACCUs), based on documented increases in SOC, when they implement eligible activities over 25 years (Australian Government, 2021).

13.3.4 Soil biodiversity loss in Australia

Soil biology has been and remains an active field of research in Australia (CSIRO, 1983; Pankhurst Doube and Gupta, 1994; Abbott and Murphy, 2007; Phillips *et al.*, 2021; DAWE, 2021). Importantly, the general public, land managers and policymakers have taken a greater interest in the importance of soil biology since 2015. This heightened awareness has also seen concerted efforts by researchers, farmers and industry to either improve or regenerate soil biology, often in combination with further attempts to capitalize on the essential ecosystem functions that soil biology delivers.

Much of the previous and current research on soil biodiversity focuses on agricultural and forestry production systems, primarily as a result of undergoing regular human interventions for sustainable production and expected economic benefits. However, the importance of remnant and pristine landscapes providing evidence of soil biology change has resulted in their inclusion and focus in some major investigations, such as the Biome of Australia Soil Environments (BASE) (Bissett *et al.*, 2016), part of the Australian Microbiome Initiative (AMI). The AMI is a publicly-accessible database of microbial diversity across a geographically expansive and diverse range of Australian terrestrial ecosystems. As of 2024, it has provided insight into the climatic and anthropogenic drivers of soil biology changes within landscapes at local, regional and continental scales (Araujo *et al.*, 2016; Lem *et al.*, 2022; Peddle *et al.*, 2022; Viscarra Rossell *et al.*, 2022).

There has also been a concerted effort to utilize DNA sequencing tools, so that the evolution, distribution and status of soil biodiversity can be understood across Australia (Bissett *et al.*, 2016; Araujo *et al.*, 2016; Wakelin *et al.,* 2021). Woodman *et al.* (2008) provided some preliminary assessments for Australia and outlined strategies for developing a longer-term system for assessing soil biodiversity.

Soil pH, organic C, clay and iron oxide mineralogy, and regional climate are major drivers of variations in the composition and diversity of soil microbiomes. Native vegetation soils are a recognized reservoir of these organisms for the adjoining agricultural soils. Environmental preferences of dominant fungal taxa indicate three dominant ecological clusters associated with mesic, forest and dryland biomes worldwide, including Australia.

Since the beginning of the twenty-first century, research has examined soil biodiversity responses to agriculture practices and compared the soil DNA with that of archived and fresh soils from across Australia (DAWE, 2021). Such research hopes to identify fundamental changes in soil biological communities and determine the influence of farming systems and other environmental factors. In the past the focus was solely on the loss of the soil biodiversity in the region, but the level of detail and abundance of data available provides insights into population shifts, associated functional changes and the levels of resistance and resilience in the soil ecosystem.

A comprehensive assessment of the status of soil biodiversity in the Australian region is still lacking, but progress is occurring, and some general inferences can be drawn.

Characterization of soil biodiversity remains inferior to that of above-ground biodiversity, with focus largely on the microbiome and less emphasis on fauna studies, with the exception of some DNA approaches (Delgado-Baquerizo *et al.,* 2017). The faunal component of the soil food web and overall soil biological functions are still underrepresented, especially for meso- and macrofauna.

A range of pressures and stressors such as loss of SOC, acidification, physical disruption through cultivation or other means, land-use change (such as pasture–crop to intensive grain cropping) and the multifarious impacts of climate change (such as fire, flooding and sea level rise) directly affect soil biodiversity and functions. While the adoption of conservation agricultural practices has resulted in significant changes in microbial and faunal communities and functions, large areas in the region have and will continue to increasingly experience these pressures and stressors. These effects may be greater in lower SOC soils in both rainfed cropping regions and intensive agricultural systems.

In a global study that included Australia, Guerra *et al.* (2022), found a mismatch between above ground biodiversity and soil biodiversity, with tropical soil ecosystems having a low alpha diversity (i.e. mean species diversity in a site at a local scale measured by species richness or evenness), and temperate ecosystems showing the greatest species richness.

13.3.5 Nutrient mismanagement in Australia

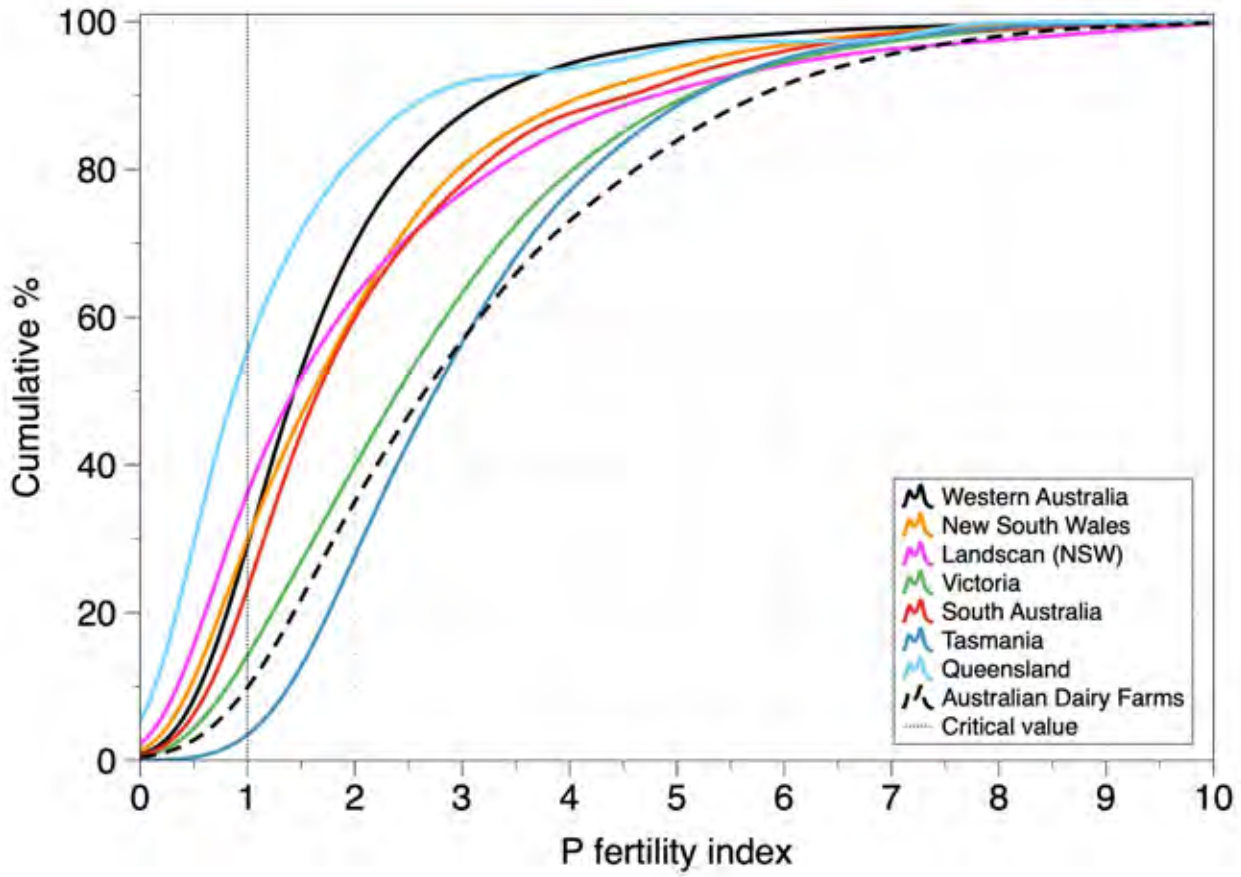
The Australian Bureau of Agricultural and Resource Economics and Science (ABARES) report assessed nutrient and soil reaction changes in three major land uses over the last decade: cropping, grazing (both extensive and intensive) and sugarcane production, showing that agricultural practices have altered soil nutrients and acidity across 62 million ha of Australia (ABARES, 2023). The longstanding goal is to reduce offsite nutrient impacts by minimizing surplus nutrients beyond plant needs.

In southern Australia, grain production is reliant on rainfed production (275–700 mm annually). Since the middle 1990s, the proportion of agricultural land used for growing grain has increased by 20 percent to a total of 60 percent, with a corresponding decline in land to sown pasture, especially legume-based pastures (Scanlan *et al.,* 2023). This change in agricultural management has disrupted the supply of N to crops from mineralization of organic matter and increased nutrient removal from soils. Consequently, the application of N fertilizer for crop production has increased (Angus *et al.*, 2019). Soil testing data from farms also suggests that P balance in grain cropping is substantially positive or in excess. Positive P balance is the result of overfertilizing, where P inputs in fertilizer exceed P outputs in products. By contrast, negative K balance is quite common, leading to K rundown, particularly in subsoils (Ma *et al.*, 2022). Compared to other regions, micronutrient deficiencies are not common in Australia, and where needed, most growers top up levels of micronutrients (Zn, Cu and Mn) in grain cropping.

Soil acidification is a widespread threat to crops due to product removal and nitrate leaching. However, in Western Australia, after a three decade-long extension programme, topsoil acidity is being ameliorated by lime application, but subsoil acidity continues to be present, as broadcast application of lime in no-till farming will only alleviate topsoil acidity. The area affected by subsoil acidity may be greater than recorded, as conventional soil testing usually only samples the first 20 cm of soil (Condon, Burns and Li, 2020). To address subsoil acidity, increasing numbers of growers are undertaking strategic deep tillage to incorporate lime to depth (Davies *et al.*, 2019).

Phosphorus balance for most grazing regions in Australia remains positive. For systems with positive P balance, soil P stores and plant-available soil P continues to increase beyond agronomic targets, especially in Tasmania and Victoria (Figure 13.9). The two areas that exhibited higher percentages (40 to 60 percent) of P deficiency were the Brigalow Belt region in Queensland (Peck *et al.*, 2015) and parts of New South Wales (Burns *et al.*, 2014) (Figure 13.9). Long-term soil testing trends in the Peel-Harvey catchment in Western Australia (Weaver and Summers, 2021) show increased soil P fertility beyond agronomic requirements (Figure 13.10). Elevated soil P increases the risk of P loss and eutrophication of the Peel-Harvey estuary, an internationally listed RAMSAR wetland. Around 70 percent of soil tests show high soil P status in the southwest of Western Australia, K and S deficiency are common, and 85 percent of paddocks are affected by soil acidity. The stratification of soil P on grazing land can increase the risk of dissolved P loss via surface runoff, due to surface application of fertilizer, and manure and pasture returns to the soil surface. Landholder compliance with the results of soil testing can contribute to significant reductions in offsite P losses by modifying their fertilizer applications (Weaver, Summers and Neuhaus, 2023).

Figure 13.9. Cumulative frequency distributions of soil phosphorus fertility index from soil test records of grazing land within different states, and for specific projects



Note: The agronomic critical value is defined at a P fertility index of 1, where below 1 is deficient and above 1 is in excess.

◊ *Source:* Adapted from **Burns, H., Weaver, D.M., O'Keefe, P., Howard, L. & Phillips, N**. 2014. *An assessment and benchmarking of phosphorus nutrient use efficiency and industry management practice in southern Australia.* Sydney, Australia, Meat & Livestock Australia Limited. https://www.mla.com.au/contentassets/625b2148ac7841afac4be7300a45f363/b.pue.0105_final_report.pdf

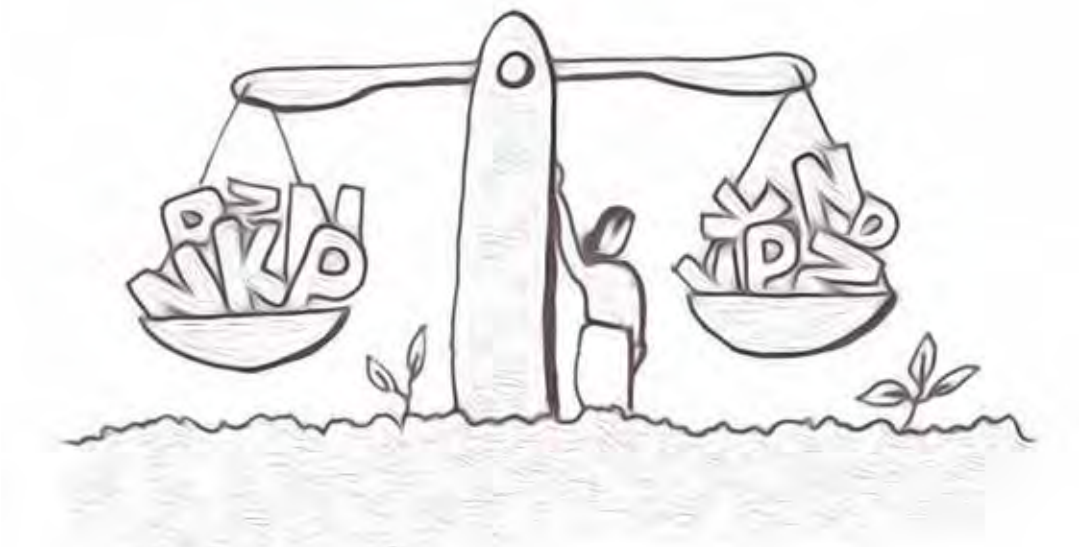
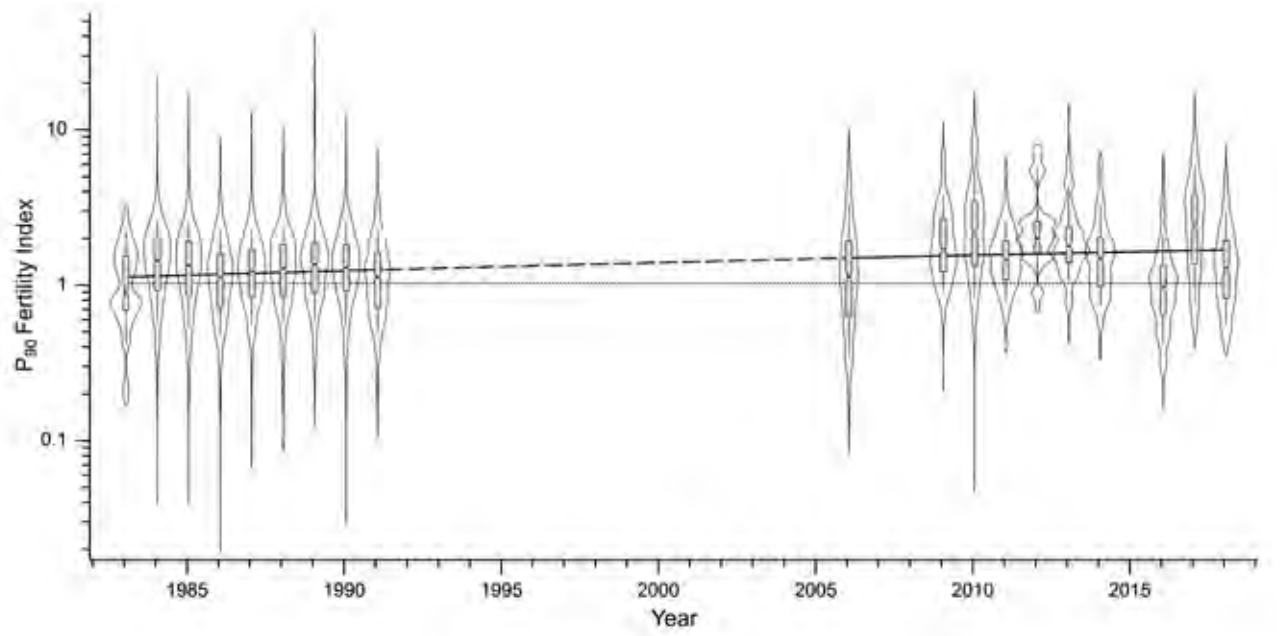


Figure 13.10. Combined violin and notched boxplots showing distribution and temporal variation of the ninetieth percentile fertility index over the period of sample collection

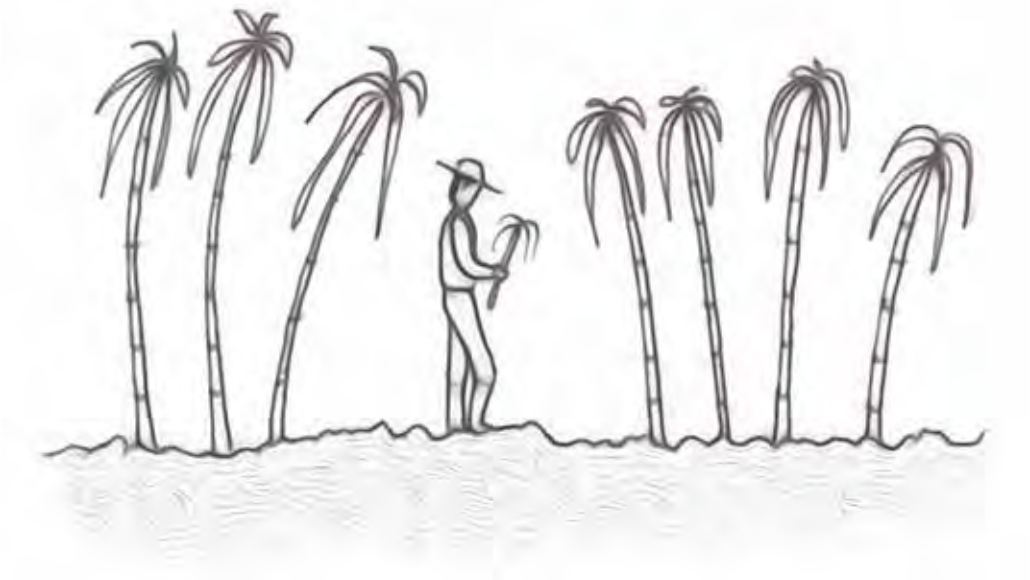


Note: The solid and long dashed line shows the linear fit with 95 percent confidence interval (grey shading) to median P90 in each year. The dotted line shows the optimal P90 = 1 for reference.

◊ *Source:* **Weaver, D. & Summers, R.** 2021. Phosphorus status and saturation in soils that drain into the Peel Inlet and Harvey Estuary of Western Australia. *Soil Research*, 59(7): 699–714. https://doi.org/10.1071/SR20259

Runoff from agricultural land has negatively affected the World Heritage-listed Great Barrier Reef of Australia, with local, national and international consequences. The Great Barrier Reef Protection Amendment Bill 2009 (Queensland Government, 2009) requires sugarcane growers in Great Barrier Reef catchments to use a prescribed soil test methodology before each crop cycle to determine appropriate site-specific N and P application rates. This legally limits fertilizer application rates to prevent nutrient surplus while still optimizing sugarcane yield. The 2025 water quality target is for a 60 percent reduction in anthropogenic end-of-catchment dissolved inorganic nitrogen (DIN) loads, calculated from the 2013 baseline (Queensland Government, 2018). The latest report card of April 2022 showed a cumulative reduction (28 percent) of catchment DIN loads to June 2020 based on modelling.

Other jurisdictions currently support a voluntary approach to nutrient management. For example, Fertilizer Australia's role is to engage with government and other stakeholders to develop and implement policies which support the sustainable growth of Australian agriculture. Their Fertcare® programme is intended to promote the skills and knowledge of fertilizer and soil ameliorant product suppliers and ensure that high quality advice is delivered to farmers, so they can optimise productivity, while at the same time minimizing environmental and food safety risks.



13.3.6 Salt accumulation in Australia

Australian soils are inherently saline, sodic, or both. Cumulatively, 357 million ha of mainly inland Australian soils are estimated to be naturally sodic and saline (primary salinity) (Northcote and Skene, 1972). Australia's soils, especially those in the southwest, are inherently saline, as they lie in the path of a constant flow of westerly winds, laden with oceanic aerosols that have been depositing salts for millennia (Hingston and Gailitis, 1976). Added to this, aeolian dust storms have spread saline clay deposits (parna) across many Australian landscapes, following arid phases and megadroughts.

A further 2 million ha of saline soils (dryland salinity) occur in the hinterland between the arid and coastal regions and are caused by the historical clearing of native vegetation for agriculture (Table 13.5). Most of this land has become salt-affected as a result of changes to the water balance of rainfed farmland. Following extensive clearing and above-average rainfall starting in the 1950s, groundwater levels rose, and evaporation and salt accumulation started in previously arable areas. By the 1980s, dryland salinity had come to prominence in most southern Australian states.

The first Australia-wide estimate of dryland salinity was conducted by the Prime Minister's Science, Engineering and Innovation Council (PMSEIC) (PMSEIC, 1999) (Table 13.7), using a variety of data sources and methods. The most recent assessment was Western Australia's Land Monitor project using satellite (25 m pixel) and other digital data to map paddock-scaled salinity (Caccetta *et al.*, 2022), while Victoria and South Australia used local air photo mapping. Previously, states had relied on landholder estimates captured in Australian Bureau of Statistics (ABS) surveys (Table 13.7). There have been no coordinated estimates since 2008 for the whole country (Table 13.7).

Table 13.5. Estimates of previously productive land (ha) now affected by salinity in the states and territories of Australia between 1999 and 2022

State	1999[1]	2002[2] (ha) (% of farms)	2008[3]	2022
NSW/ACT	120 000	124 000 (7.4%)	61 993[4]	not reassessed
Victoria	120 000	139 000 (13.4%)	149 912[5]	not reassessed
Queensland	10 000	107 000 (3.4%)	40 000	not reassessed
South Australia	402 000	350 000 (21.6%)	350 000	not reassessed
Western Australia	1 802 000	1 241 000 (51%)	1 077 000[6]	1 750 000[7]
Tasmania	20 000	6 000 (9%)	73 900	not reassessed
Northern Territory	n/a	2 000 (2%)	n/a	n/a
Total	2 476 000	1 969 000	1 753 805	

Note: NSW/ACT = New South Wales/Australian Capital Territory.

Source: [1] **PMSEIC (Prime Minister's Science, Engineering and Innovation Council).** 1999. Dryland Salinity and its Impacts on Rural Industries and the Landscape. Canberra. https://kiriganaicom.files.wordpress.com/2016/09/pmseic_dryland-salinity-and-its-impacts-on-rural-industries-and-the-landscape.pdf

[2] **ABS (Australian Bureau of Statistics).** 2002. 4615.0 - Salinity on Australian Farms, 2022. In: *ABS*. Canberra. [Cited 2023]. https://www.abs.gov.au/ausstats/abs@.nsf/Latestproducts/4615.0Main%20Features12002?opendocument&tabname=Summary&prodno=4615.0&issue=2002&num=&view=

[3] Unlinked figures collated by author based on state presentations at the second International Salinity Forum in 2008 in Adelaide, South Australia

[4] **DECC NSW (Department of Environment and Climate Change New South Wales).** 2009. *Salinity Audit: Upland catchments of the New South Wales Murray–Darling Basin.* Sydney, Australia. https://www.environment.nsw.gov.au/research-and-publications/publications-search/salinity-audit-upland-catchments-of-the-new-south-wales-murray-darling-basin

[5] **Allan, M.J.** 1994. *An assessment of secondary dryland salinity in Victoria.* Technical Report No. 14. Melbourne, Australia, Department of Conservation & Natural Resources. https://www.ccmaknowledgebase.vic.gov.au/kb_resource_details.php?resource_id=2135 (Victoria total: 239 912 ha excludes areas of primary salinity).

[6] **George, R., Kingwell, R., Hill-Tonkin, J. & Nulsen, B.** 2005. *Salinity Investment Framework: Agricultural land and infrastructure.* Report 270. Perth, Australia, Department of Primary Industries and Regional Development. https://library.dpird.wa.gov.au/cgi/viewcontent.cgi?article=1252&context=rmtr

[7] **Caccetta, P.A., Simons, J., Furby, S., Wright, N. & George, R.** 2022. *Mapping salt-affected land in the South-West of Western Australia using satellite remote sensing.* Series Number EP2022-0724. Melbourne, Australia, Commonwealth Scientific and Industrial Research Organisation (CSIRO). https://library.dpird.wa.gov.au/lr_publishedrpts/3/

Recent hydrologic studies in the Murray–Darling Basin have shown that following a wetter phase in the mid-twentieth century (when water tables rose steadily and salinity expanded), the hydrologic system has been replaced by mostly stable or falling water tables (DECC NSW, 2009; Fu, Rojas and Gonzales, 2022). This was attributed climatically to the recent long phase of El Nino, following the so called "millennium drought".

In Western Australia, the driving force remains the legacy of clearing 18 million ha of vegetation for agriculture and the resultant orders-of-magnitude impact on the water balance (McFarlane *et al.*, 2016). More recently, some areas have equilibrated, although this is variable depending on the hydrological zone and nature of the landscape (Raper *et al.*, 2014). In more recently cleared sandy catchments, rising water tables and salinity continue to affect arable land. In addition, land use has changed (Raper *et al.*, 2014). A doubling of the cropped area since the 1990s, the adoption of soil moisture retention strategies and the use of chemical weed control as a water management tool, appears to have increased the recharge rates into the regolith, especially in wetter years.

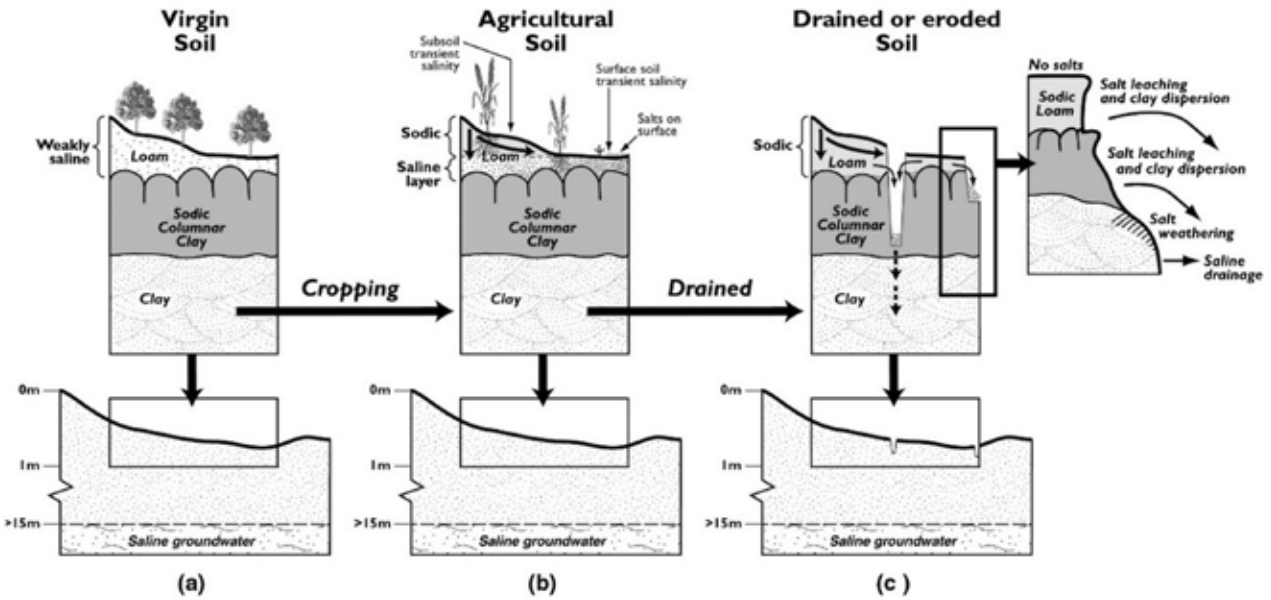
Sodic (dispersive) soil is often structurally unstable under wet conditions, reducing pore space and limiting water and root passage. Hence waterlogging is common but, when dry, soils become too dense to allow easy water infiltration or root accessibility (So and Aylmore, 1973; McKenzie and McBratney, 2001) (Figure 13.11). Sodium accumulation in soils is common and non-transient in Australia's highly weathered landscape, which has perhaps the largest proportion of any continent constrained by dispersive soils.

Historical estimates suggest that as much as 250 million ha of Australia and 60 percent of Australia's cropping soils are sodic (Rengasamy, 2002). Recent estimates now suggest that 68 percent of Australia's wheat-cropping area is constrained by sodicity, with lost productivity costing AUD 1.3 million annually (Orton *et al.*, 2018). Under current climate conditions, losses associated with sodicity greatly exceed those related to acidity and salinity. Expected changes in climate, with higher intensity rainfall and longer periods between rain events (Dey *et al.*, 2019), sodic constraints to water infiltration and storage in soils will increase productive losses in Australian farming systems.

An estimated 16 million ha of acid sulphate soils in Australia is encountered in inland environments (Fitzpatrick, Powell and Marvanek, 2011). With climate change, the effects of sodicity and sulphurization will potentially become more pronounced (Figure 13.11 and Figure 13.12). Both natural processes (such as the millennium drought) (Figure 13.12b) and anthropogenic processes caused by humans (such as the excavation of wetlands), and cattle (such as pugging [cattle treading]) (Figure 13.12c) cause the subaqueous inland acid sulphate soils with hypersulphidic material to dry. Drying and exposure result in the decline of the water table and exposure of the hypersulphidic material to air or oxygen, which induces the oxidation of iron sulphides (FeS_2) and the formation of sulphuric acid or sulphuric material, as shown in Figure 13.8c. Exposure allows microbial mediated oxidation of reduced inorganic sulphur (RIS) in hypersulphidic material and the subsequent release of acidity (H^+), which transforms hypersulphidic material (pH >4) to sulphuric material (pH <4). The initial form of acidity occurs as soluble or readily exchangeable acidity (i.e. sulphuric acid) in the fluid-filled soil pores. Secondary minerals, such as jarosite, sideronatrite and schwertmannite may also form within soil profiles, which act as "stored" acidity (they are sparingly soluble and may produce acidity upon dissolution during rewetting) (Figure 13.12d).

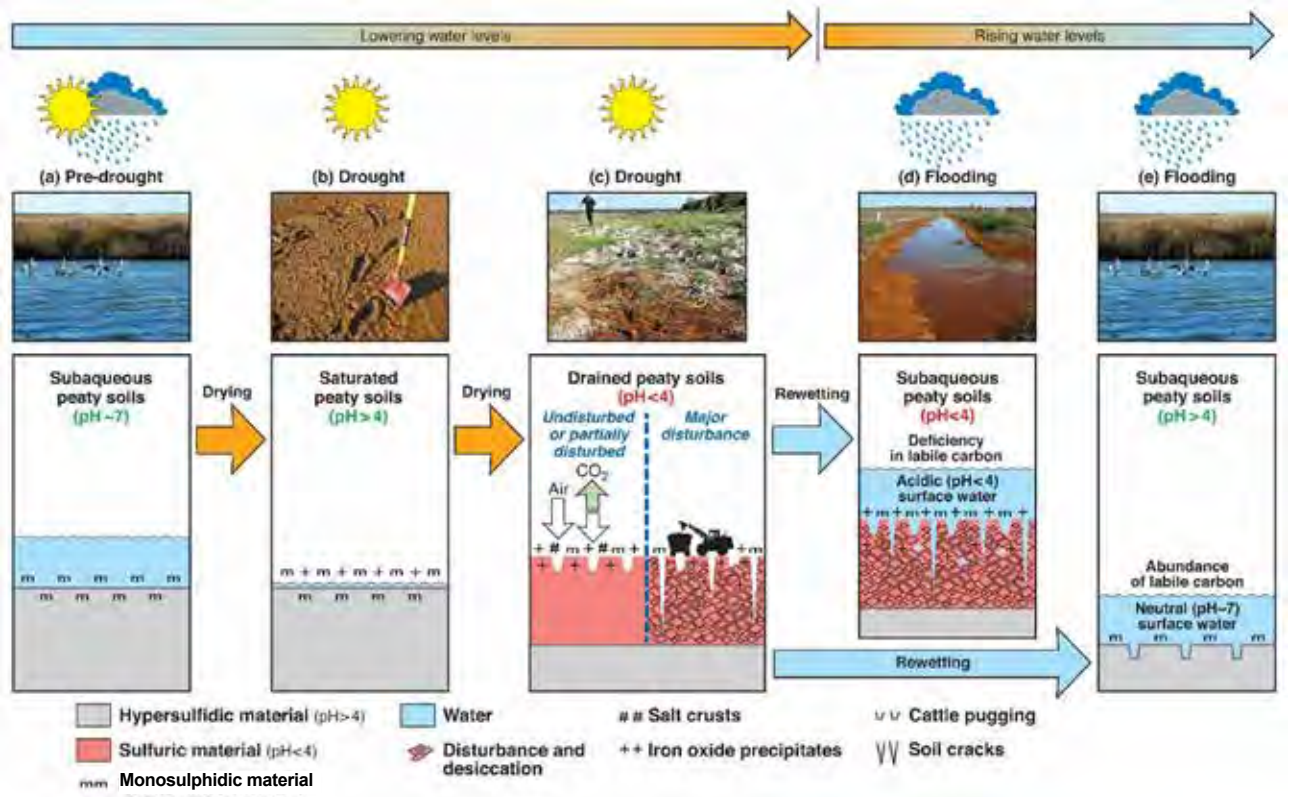


Figure 13.11. Soil regolith model showing salt transport and erosion processes leading to the formation of subsoil and surface soil transient salinity (not associated with the saline groundwater tables) in a sodic duplex soil (Sodosol)



◊ *Source:* Adapted from **Fitzpatrick, R.W.** 2008. Soils and Natural Resource Management. In: K.M. Scott & C.F. Pain, eds. *Regolith Science*, pp. 307–339. Dordrecht, Kingdom of the Netherlands, Springer.

Figure 13.12. Conceptual model illustrating transformation processes of acid sulphate soil materials caused by drying (droughts) and rewetting (flooding) conditions



◊ *Source:* Adapted from **Fitzpatrick, R.W., Shand, P. & Mosley, L.M.** 2017. Acid sulfate soil evolution models and pedogenic pathways during drought and reflooding cycles in irrigated areas and adjacent natural wetlands. *Geoderma*, 308: 270–290. https://doi.org/10.1016/j.geoderma.2017.08.016

The expected changes in climate have resulted in increasing government and industry investment into amelioration strategies to address and improve sodic soils. However, since constraints often occur in the subsoil, potent, scalable and cost-effective solutions remain a challenge. Among the reclamation measures applied for acid sulphate soils have been a reduction in the intensity and depth of drainage ditches, and laser-levelling of the land to make the drainage measures more efficient while contributing to lessened release of acidity to waters leaving the farm.

13.3.7 Soil pollution in Australia

Australian soils are inherently saline, sodic, or both. Cumulatively, 357 million ha of mainly inland Australian soils are estimated to be naturally sodic and saline (primary salinity) (Northcote and Skene, 1972). Australia's soils, especially those in the southwest, are inherently saline, as they lie in the path of a constant flow of westerly winds, laden with oceanic aerosols that have been depositing salts for millennia (Hingston and Gailitis, 1976). Added to this, aeolian dust storms have spread saline clay deposits (parna) across many Australian landscapes, following arid phases and megadroughts.

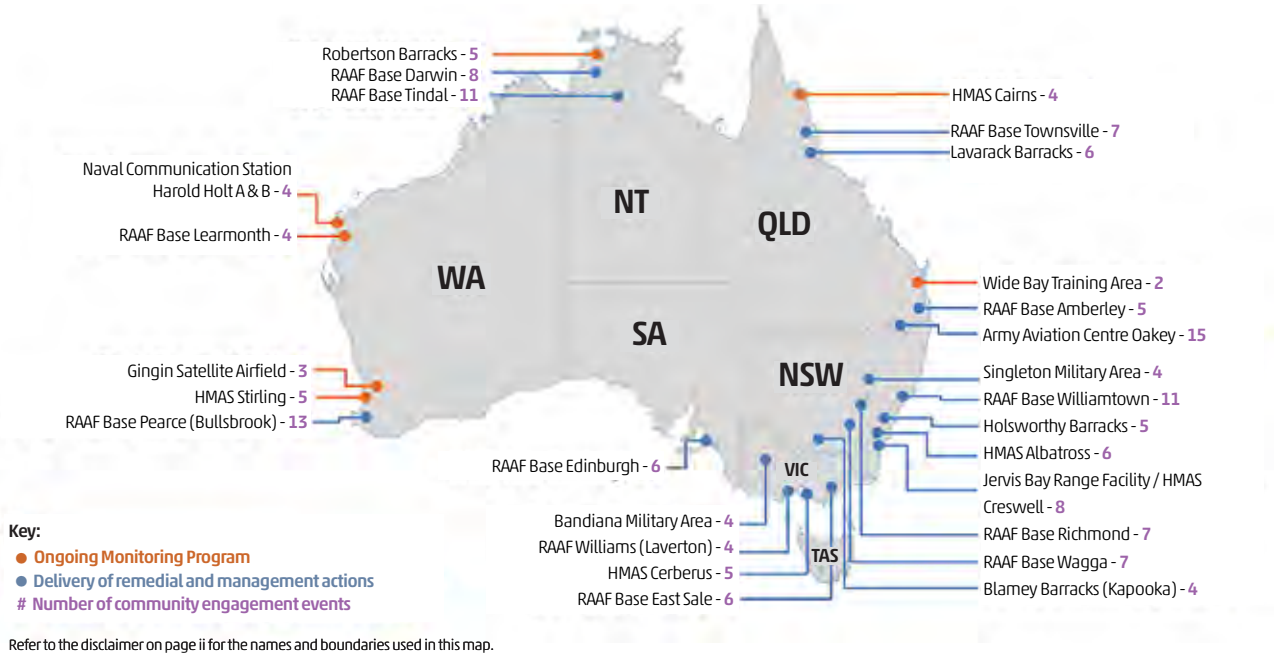
Over 50 percent of Australia's land is used for agriculture (Cresswell, Janke and Johnston, 2021) and millions of tonnes of pesticides, fertilizers and soil amendments are applied (e.g. Ludemann *et al.*, 2022). Many soils still suffer residue contamination from now banned legacy pesticides, such as horticultural soils contaminated with As, Cu and Pb (Merry, Tiller and Alston, 1983; Wightwick *et al.*, 2013), and up to 10 000 former cattle and sheep dip sites are impacted by organochlorine pesticide and residues (Sarakar *et al.*, 2007). Contemporary pesticides such as neonicotinoids present concerns with off-target impacts and accumulation in certain soils (Goulson, 2018). Controls exist to reduce the impact of trace impurities (such as Zn, Cd and Ni) in fertilizers (De Vries and McLaughlin, 2013) and certain chemical contaminants in organic waste-derived soil amendments, including biosolids and composts. However, there are emerging contaminant concerns for soil amended with recycled waste products, include brominated flame retardants (BFRs) (McGrath *et al.*, 2020), PFAS and certain pharmaceuticals and personal care products (Kumar *et al.*, 2022). Risks include accumulation, food chain contamination, endocrine disruption and antibiotic resistance (Wu *et al.*, 2023). Agricultural soils are also increasingly impacted by micro- and nanoplastic pollution from sources such as mulching films, plastic netting and composts. Recent work estimates that 3 700 million tonnes of plastics are released to Australia's agricultural land in biosolids and 140 million tonnes to landscape topsoil each year (Okoffo *et al.*, 2020). How these microplastics and associated chemicals are transformed and transported in the soil, and the impacts on soil health and food safety remains little understood (Nelis *et al.*, 2023).

While land classified as "mining and waste" makes up less than 1 percent of Australia's land surface (Cresswell, Janke and Johnston, 2021), past and present mining activity has resulted in a substantial yet undefined contamination footprint (Werner *et al.*, 2020). Tens of thousands of orphaned or abandoned mine sites are contaminated from tailings, mine wastes and ore-processing activities (Salmi, Bekele and Schmid, 2022), and pose ongoing risks to the environment and humans, although some sites are now under active management or rehabilitated (Campbell *et al.*, 2017).

Despite Australia's low intensity industrialization, many industrial sites are contaminated, some with widespread offsite pollutant movement into soils and waterways. A recent review of human health risks associated with contaminated land in Victoria between 2006 and 2018 identified that the greatest risks were presented by service stations, fuel depots, dry cleaners, gasworks, mechanical parts manufacturers, imported landfill and unknown offsite sources generating regional groundwater pollution (Kabay, Papaleo and Reichman, 2023). Per- and polyfluoroalkyl substances contamination of soil and water has emerged as a significant threat across multiple locations where PFAS were used, including airports, defence bases, firefighting training sites, mining, landfills, and construction sites (Cresswell, Janke and Johnston, 2021; Chamberlain *et al.,* 2022) (Figure 13.13).

Hundreds of locations are potentially impacted across Australia, and a National PFAS Investigation Management Program has been established for a coordinated approach (HEPA, 2020).

Figure 13.13. Australian Government Defence's National PFAS Investigation and Management Program



◊ *Source:* **HEPA.** 2023. PFAS INVESTIGATION AND MANAGEMENT PROGRAM SNAPSHOT - JUNE 2023 [poster]. Canberra. https://www.defence.gov.au/sites/default/files/2023-06/PFAS%20website%20snapshot.pdf

Landfill sites are important potential sources of mixed soil contamination, including PFAS (Gallen *et al.*, 2016). While approximately 600 landfill sites are registered, up to 2 000 facilities are unregulated (Cresswell, Janke and Johnston, 2021) (Figure 13.13). The National Pollutant Inventory listed 580 facilities (including landfills) as legally emitting 46 chemicals into the ground in Australia during the 2021–2022 period (DCCEEW, 2023), with litter and illegal waste dumping also being a concern.

Between 10 and 35 percent of urban garden soil is contaminated with metals and metalloids, particularly Pb, from vehicle emissions, building materials (including paint) and industry (Kandic *et al.*, 2019; Laidlaw *et al.*, 2018; Taylor *et al.*, 2021; Kristensen, Taylor and Flegal, 2017). Asbestos contamination is also a concern (Khatib *et al.*, 2023; Gray, Carey and Reid, 2016), and many roadside soils are impacted by contamination from vehicle emissions (De Silva *et al.*, 2016; De Silva *et al.*, 2021; Kristensen *et al.*, 2017; Rouillion, Gore and Taylor, 2013).

13.3.8 Urbanization and soil sealing in Australia

Soil sealing is defined as the loss of soil resources due to covering by an impermeable material (Evangelista *et al.,* 2023). Other impacts identified as a result of soil sealing include changes to agriculture, local perishable food availability, flood mitigation, habitat and biodiversity, catchment health and water yield and quality, and links to cultural place (Lawton and Morrison, 2022; Buxton and Butt, 2020).

Permanent soil loss to sealing and capping is a result of land-use change, and urban expansion on the borders of major cities. Land conversion for housing in Australia is the main driver of soil capping and sealing processes, while a parallel threat is through transport and energy infrastructure, as well as mining developments. Another emerging threat is the expansion of solar energy and transmission line development, which often occurs in regional and rural areas of Australia, far from major urban centres.

The quantification of soil sealing and capping as a soil threat has received limited attention, although recognized in nationwide State of Environment reporting (Williams *et al.*, 2021). Using ABS data, Lawton and Morrison (2022) showed there had been a loss of 66 percent of lands for primary production in Western Sydney in Australia between 2011 and 2016 and this conversion has continued. There are also concerns over the ongoing loss of land for fresh food production (ISF, 2023). The primacy of housing provision continues to be the driver of urban sprawl, with all capital and regional cities in Australia showing increasing growth. Greenfield development is considered the main strategy to accommodate urban growth, rather than high-density housing in existing urban areas (Williams *et al.*, 2021). In Victoria, the peri-urban area contributes up to 41 percent of Melbourne's food needs through vegetable and poultry production, yet at the current rate of land conversion, this could drop to 18 percent by 2050 (Carey *et al.*, 2016).

Whilst urban agriculture is compensating in a small way to the loss of agricultural land by utilizing suitable areas for food production, there is limited opportunity in many new urban release areas due to block and verge size, with the house and hard surfaces covering most available land area (Hume, Summers and Cavagnaro, 2021). Land-use histories of urban environments can also have implications for soil health, such as heavy metal contamination through fruit and vegetable production and consumption (Salomon *et al.*, 2020). Urban agriculture sites also have significant nutrient imbalances, due to excessive inputs of organic amendments (Salomon *et al.,* 2020). Little is known about the impact of soil sealing from urbanization on soil biodiversity and the implications for soil health and ecosystem services (Salomon and Cavagnaro, 2022).

The expansion of mining, renewable energy and infrastructure developments in Australia have an impact on soil resources and lands, through disturbance, capping and sealing. Whilst the Government of New South Wales has claimed that only 1 percent of rural land is estimated to be impacted by renewable energy developments to 2051 (55 000 ha), there is no acknowledgement of the potential impacts on soil quality and land value (NSW Government, 2023).

Whilst most solar energy developments allow for limited grazing use, the design and operation of the solar developments affect the land and soils through changes in development, use and management. The land and soil impact of decommissioning will further disturb soil, and rehabilitation may be required to restore soil values.

There is limited attention paid to the impact of soil sealing and capping as an issue in Australia. The National Soil Strategy (DAWE, 2021) in Australia seeks to highlight soil as a valued resource that needs to be conserved and managed to deal with food security and climate change adaptation. The need for soil and land values to be considered in strategic planning processes is essential.

13.3.9 Soil governance in Australia

Government policies and processes need to guide and facilitate the actions of individuals in managing land as a public good resource that also provides private benefits. Most of the land in Australia continues to be vulnerable to land degradation due to agricultural activity, with more than 54 percent in grazing lands and a much smaller proportion (4 percent) under cropping (Cresswell, Janke and Johnston, 2021). The ability to promote sustainable land management is hampered by social networks being less active in rural areas, and the growing gap between intention and action. The reduced presence of state and federal government agencies at a local scale has led to the loss of locally-relevant knowledge and increased expectations that markets and private enterprise will fill the knowledge gap, but there is no evidence that this has occurred.

The National Soil Strategy is Australia's first national policy on soil and has set a 20-year plan in motion to value, manage and improve its soil (DAWE, 2021). There are three overarching goals:

- to prioritize soil health;
- to empower soil innovation and stewards; and
- to strengthen soil knowledge and capability.

The National Soil Action Plan (DAFF, 2022) sets out the foundational goals to guide the focus of partner activities and investments in soil for the next five years. The plan has four priorities, including the identification and development of a soil workforce and the capabilities needed to meet current and future challenges for Australia and the region. This priority includes increasing the number of trained soil scientists that can collect accurate soil information, as well as interpret the significance of the soil results for the land manager. As of 2024, this is only utilized by a quarter of farming businesses in Australia, and only in low densities and over small areas of land (Lobry de Bruyn and Andrews, 2016).

A survey of Australian soil science research, development and education staff re-enforced the limited investment in soil knowledge exchange, with the most staff in research (39 percent) and postgraduate training (26 percent), with 12.5 percent working in extension and 6 percent in teaching (DAFF, 2011). The successful training of the next generation of soil scientists requires mentoring by the soil scientists remaining in the workforce, and this opportunity is quickly disappearing (Lobry de Bruyn *et al.*, 2022). The loss of experience and expertise in soil science with an aging workforce needs to be addressed before the opportunity is no longer viable. Alternatively, the use of existing or developing new knowledge networks or online information held in nationwide databases may address these concerns (Lobry de Bruyn, Jenkins and Samson-Liebig, 2017). However, the static nature of existing data sets and the limited investment in new soil information suggests that relevant and credible soil information on the impact of best management practices on soil at the local scale, has a lower likelihood of turning information into actionable knowledge without considerable co-innovation taking place (Lobry de Bruyn and Prager, 2024).



13.4 Regional status and trend of threats to soil functions for New Zealand

13.4.1 Indigenous Peoples' viewpoint

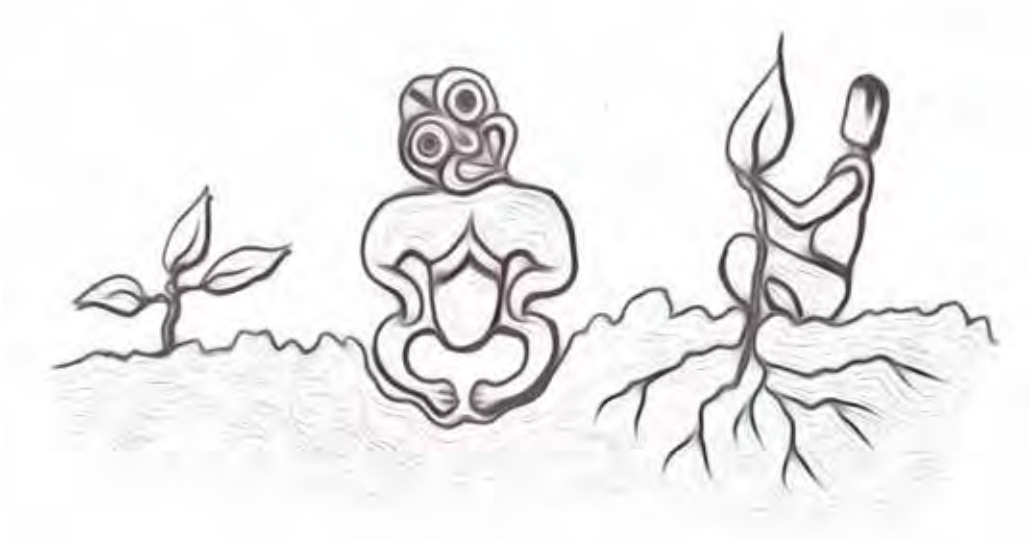
> *"Te toto o te tangata he kai, te oranga o te tangata, he whenua, he oneone."*
> *"While food provides the blood in our veins, our health is drawn from the land and soils."*
> (Indigenous Māori proverb.)

In New Zealand, the Māori Indigenous Peoples' view of the world places Māori at the heart of natural ecosystems and processes as part of a *whakapapa* (ancestral lineage) that links humans to all other organisms (down to microbes), through an intricate, hierarchical web. This gives humans a lower rather than dominant status in the natural and built world. This is often termed an Indigenous Peoples' view (Ataria *et a*l., 2023).

From a biocentric perspective, Indigenous Peoples place their cosmogony and belief systems, the environment, and biodiversity at the center of their restoration practices, in both natural and semi-natural ecosystems (FAO, 2024c).

The Māori Indigenous Peoples' perspectives of soil (*oneone*) and soil health incorporate the indivisible connection between the wider environment (*taiao*), ecosystems, and human health, meaning the state of the soil within the environment affects human well-being (Harmsworth and Awatere, 2013; Stronge *et al.*, 2020; Ataria *et al.*, 2023). Thus, all people are related via *Papa-tū-ā-nuku* (earth) and *Rangi-nui* (sky), who connect us, our world and the universe. Māori worldviews emerge from this connection, and Māori are often referred to as *tangata whenua* (people of the land) (Hutchings, Smith and Harmsworth, 2018). Māori gardeners work with the environment and use gardening practices to improve the *mauri*, or life force, of soil (Hutchings, Smith and Harmsworth 2018; Hutchings, 2020).

The wider society of New Zealand is becoming more aware of core Māori values and principles, and increasingly, the relational and intrinsic values of soil are recognized in addition to economic values (MfE, 2023a; Stronge *et al.*, 2023). Soil health issues require society to negotiate unavoidable trade-offs between different soil uses (Bünemann *et al.*, 2018). Therefore, a more holistic approach to address the sustainability of our soil resources is needed in the country that is consistent with the worldview of the Māori Indigenous Peoples. Weaving together western-centric perspectives and Māori knowledge – incorporating a range of diverse people, sectors, their values and spirituality, and the appreciation of well-being that people obtain from soil – will likely result in improved soil management, with wider benefits of environmental and human health, and sustainability.



13.4.2 Soil erosion in New Zealand

Erosion rates are naturally high in parts of New Zealand due to erodible geology, high rates of tectonic uplift and high rainfall. Human-accelerated soil erosion remains a problem in New Zealand. Soil erosion modelling estimate that around 182 million tonnes of soil are lost yearly into waterways (Stats NZ, 2024). Forty-four percent of eroded soil in waterways has been estimated to come from pastures (exotic grasslands) (MfE and Stats NZ, 2018). Hillslope mass movement (landslides, earthflow and gully erosion) is expected to increase in soft rock hill country by the middle of the twenty-first century, primarily through more extreme and frequent storms, leading to regional increases in coastal sediment loads ranging from 1 to 233 percent (Neverman *et al.*, 2023). Cyclone Gabrielle in February 2023 caused an estimated 300 million tonnes of landslides, with an economic cost of approximately NZD 1.5 billion (conservatively estimated at NZD 5 per tonne of eroded soil) (McMillan *et al.*, 2023).

Current policies on freshwater management address sediment loads but may not be adequate with continued climate change impacts (Neverman *et al.*, 2023). Following Cyclone Gabrielle, amendments to the national environmental standards for plantation forestry have been underway. These standards include managing erosion risks from forestry operations (Panel for the Ministerial Inquiry into Land Use, 2023). In addition, the Hill Country Erosion Programme is a partnership between the Ministry for Primary Industries (MPI), councils and landowners, providing funding and technical capacity to deliver erosion control work (MPI, n.d.).

13.4.3 Soil carbon loss in New Zealand

The mineral soils of New Zealand have naturally high SOC content, with mean SOC stocks in topsoil (0–30 cm) estimated as 98.7 tonnes/ha (Minasny *et al.*, 2017). Agricultural land is predominantly grazed grassland, contributing to high SOC content (Schipper *et al.*, 2017). As most mineral soils in New Zealand are generally considered to be at or close to C saturation levels, the major challenge is preventing SOC losses (Whitehead *et al.*, 2018).

Land-use change between 1990 and 2016, where the change from grasslands to forest exceeded the change from forests to grasslands, resulted in SOC stocks decreasing by an average of 3.3 tonnes/ha (range 0.78–7.5 tonnes/ha) nationally for the areas where land-use change occurred (Whitehead, McNeill and Mudge, 2021). This amounted to a total decrease in SOC stocks in 0–30 cm depth of 8.4 million tonnes (Whitehead, McNeill and Mudge, 2021). A new programme, the National Soil Carbon Benchmarking and Monitoring System for agricultural land, monitors changes in SOC stocks to 60 cm. Initial benchmark results showed that the highest SOC stocks were in high-producing grasslands, while the lowest were in cropland and horticulture (Mudge *et al.*, 2022).

Mounting evidence suggests that some land management practices, such as intensive grazing, irrigation and forage cropping have led to a loss of SOC stocks (Mudge *et al.*, 2021; Schipper *et al.*, 2017; Wall *et al.*, 2020; Wall *et al.*, 2023). Mass movement erosion has also been shown to affect SOC stocks in New Zealand. Seventy years following mass movement events, SOC stocks at eroded sites were still 20 to 40 percent below those measured at uneroded sites (Basher *et al.*, 2018).

Organic soils in New Zealand cover about 259 000 ha (1 percent of the total land area), but have the highest SOC density (Hewitt, Balks and Lowe, 2021; Ausseil *et al.*, 2015). They are generally associated with wetlands and over the past 200 years have been extensively drained and converted to grassland (168 000 ha) and cropland (10 000 ha) (MfE, 2023b). Decreased water table depth is the primary driver of C loss from organic soils (Campbell *et al.,* 2021). Although organic soils drained for agriculture cover <1 percent of land area, they account for around 8 percent of net agricultural emissions (Pronger *et al.,* 2022). The average SOC stocks to 30 cm were estimated at 102 tonnes/ha for organic wetlands and 121 tonnes/ha for mineral wetlands. The SOC stocks at full peat depth (a national average of 3.9 m) were estimated at 1 348 tonnes/ha (Ausseil *et al.,* 2015).

13.4.4 Nutrient mismanagement in New Zealand

Plant-available P remains relatively high in monitored soils, especially under dairy farming and cropping, with an increasing trend detected for Olsen P in cropping and grass-fed meat farming sites between 1996 and 2018 (MfE and Stats NZ, 2021; Stevenson and McNeill, 2020). Sixty-three percent of soil samples collected between 2001 and 2015 were enriched beyond agronomic targets (McDowell *et al.*, 2020).

Total N in monitored soils is generally within the target ranges. Despite best management practices (Monaghan *et al.,* 2021) and national policy direction (MfE, 2023c), including a cap on the use of synthetic N fertilizer (MfE, 2023d), many New Zealand rivers are still showing excess N and P concentrations, indicating a risk of environmental harm (Stats NZ, 2022a, 2022b). Continuing agricultural intensification, along with lag and legacy effects from historic high fertilizer additions to soil since the beginning of the twenty-first century are likely to continue affecting water quality into the near future (Larned *et al.,* 2018; MfE and Stats NZ, 2021; Monaghan *et al.*, 2021).

To address the increasing demand for more accurate soil pH data, a digital soil pH model was developed for New Zealand (Roudier *et al.*, 2020), and a 0–7.5 cm product was generated based on this model. Soil quality monitoring shows a few sites with acidification (Stats NZ, 2021). Nevertheless, soil acidification and associated Al toxicity are critical issues for pasture productivity in high and hill country areas of New Zealand (Hendrie *et al.*, 2021; Whitley, Moir and Almond, 2019). In productive lowland pasture, acidification is overcome by liming. However, recent studies have shown that liming of high and hill country pastures is unlikely to increase plant P availability from mineralization and thus is unlikely to improve soil fertility (Hendrie *et al.,* 2021).

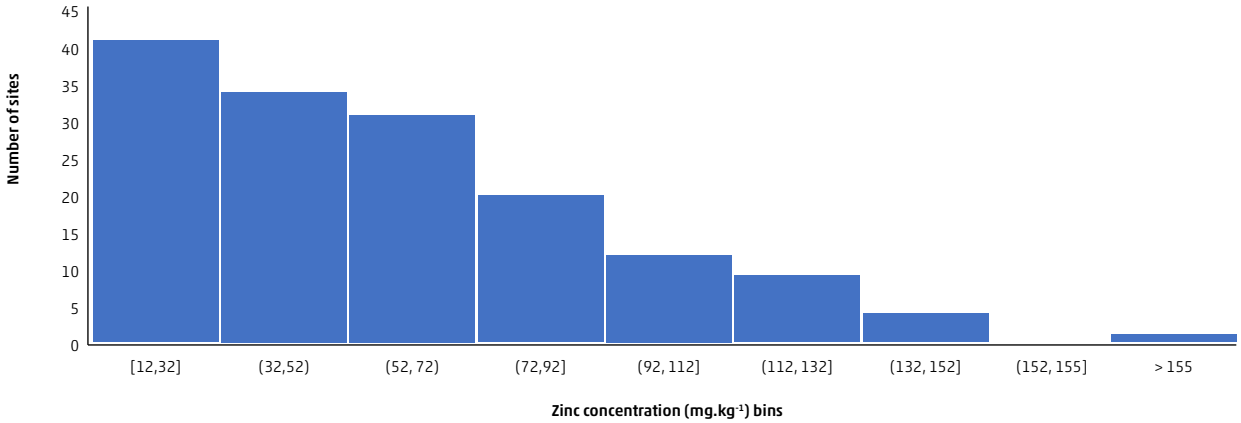
Soil acidity may also be impacted by acid sulphate soils. Acid sulphate soils were identified in the north of North Island during the 1970s (Dent, 1980, 1986; Metson *et al.,* 1977). They are commonly formed during historic sea level rise and inundation of land close to coastal areas where sulphates in the seawater mix with land sediments containing iron oxides and organic matter. They are typically buried >1 m below the current soil surface and remain inert when left undisturbed in a waterlogged state. However, when soils are drained or excavated, or the water table drops significantly during extended dry periods, sulphides can be oxidized, resulting in sulphuric acid being generated upon rewetting. Wider investigations relating to infrastructure and risk mapping projects in the Northland, Auckland and Waikato regions on North Island have resulted in several recent but unpublished reports (Trewin, 2018; Easton *et al.,* 2018; Farrey, 2015). Although infrastructure damage and environmental impacts from acid sulphate soils have been identified, the risk posed by these soils in New Zealand is still uncertain due to limited research.

Some regions of New Zealand have been monitoring a range of micronutrients for up to twenty years, with national reporting of some micronutrients being published in 2021 (MfE and Stats NZ, 2021; Stevenson and McNeill, 2020). Boron (B) is highly mobile in soil and easily washed out in drainage, which can lead to B deficiency (Skinner, Han and Singh, 2003). Volcanic ash soils have very low natural Co levels, historically resulting in the wasting disease “bush sickness”, which caused animals to starve, despite adequate food (Grange and Taylor, 1932). Copper deficiency may be an issue for farm animals on unfertilised soils due to the naturally high organic C content in New Zealand soils that binds available Cu. Although Cu toxicity in animals has been reported if Mo intake is too low, Cu toxicity has also occurred where dairy cows have been dosed with Cu to avoid deficiency but then fed palm kernel as a supplementary feed product. Palm kernel, which can have high concentrations of Cu, combined with the direct Cu dose, can induce Cu toxicity (Dias, 2010). Some acidic sands and fibrous peat soils may be deficient in Mo, and, when combined with increasing Cu, there is some risk of Mo-Cu imbalances affecting primary production (Gowariker *et al.*, 2009). Zinc deficiency may be an issue for gleyed or sandy soils due to their naturally low Zn contents. However, there are multiple sources of Zn contamination, raising levels of Zn in soils (Alloway, 2008).

Monitoring in the Waikato region from 2001 to 2020 (Taylor, 2025) shows levels rarely exceed the

guideline concentrations for strong acid-extractable B (>10 mg/kg), Co (volcanic soils) (>4 mg/kg) or Mo (sandy or gleyed soils) (>1 mg/kg). In contrast, a small proportion of monitored sites in pasture or horticulture land uses exceeded the soil guideline value of 130 mg/kg for protecting ecological receptors for Zn (Figure 13.14).

Figure 13.14. Distribution of zinc at 153 soil quality monitoring sites in the Waikato region



Note: Values in parentheses are the lower and upper Zn concentration for each bar.

Source: Adapted from **Taylor, M.D.** 2025. *Trends in selected trace element monitoring data in the Waikato region 2001-2020.* Technical Report TR25-1. Hamilton, New Zealand, Waikato Regional Council.13.4.6 Soil pollution

13.4.5 Soil pollution

Point source discharges of pollutants to soil are strongly regulated and managed in New Zealand, but diffuse source discharges are less tightly controlled. Concerns about the lack of tracking and monitoring of over 30 000 chemicals currently approved for use and how they impact the environment have been expressed. Routine testing is still limited to about 200 chemicals (PCE, 2022). New Zealand uses very high rates of pesticide per hectare (averaging 8.8 kg/ha in 2020) (FAO, 2023). However, the assessment of these pesticides on human and environmental health is complex, due to data gaps (PCE, 2022). Monitoring of some trace elements has been included in the National Environmental Monitoring Standard for soil quality (NEMS, 2022).

The accumulation of Cd from impurities in P fertilizer is managed by the Cadmium Management Strategy (CWG) (see CWG, 2019). The strategy was recently updated to ensure that "cadmium in rural production poses minimal risks to health, trade, land-use flexibility and the environment over the next 100 years" (Rys, 2020, p.1). However, in rural areas, the accumulation of other contaminants (Zn, Cu, Ch and As), such as from livestock treatments, agricultural pesticides, and fence post treatments (copper chrome arsenate [CCA] treatment continues without specific management). Ecological soil guideline values for protecting ecosystem receptors have been developed, and their implementation into policy has been explored for priority contaminants (Cavanagh and Munir, 2019; Cavanagh and Harmsworth, 2023).

In urban soils, the accumulation of Pb (such as from fuel and paint), Zn (such as from galvanised metal and tyre abrasion), As (from CCA-treated timber) and Cu (from CCA-treated timber, brake linings and metal) are recognized. The duration of urbanization influences soil contamination, with housing age and quality driving Pb pollution in New Zealand (Martin *et al.*, 2023). Other known sources of soil contamination in urban areas are historic gasworks sites (PAHs, phenol, cresol and cyanide), petroleum and drycleaning solvent leaks and spills, and the application of pesticides (MfE and Stats NZ, 2018). Citizen science is identifying community soil values and mapping Pb and other metal concentrations in home gardens through initiatives such as Soilsafe Aotearoa (Sharp *et al.*, 2024).

Many modern-day chemicals are released into the environment, potentially causing adverse environmental or human health effects (PCE, 2022), including chemicals commonly found in household and personal care products, pharmaceuticals and plastics. A national strategy to manage emerging organic contaminants, the Aotearoa-NZ Strategy for Emerging Contaminants, has been developed to safeguard people and ensure that food export products continue to meet trade requirements (Emerging Contaminants Advisory Panel, 2018). However, there is a lack of generally accepted threshold values for toxicity assessment and guidance for priority setting.

In the Indigenous Peoples' ecocentric and biocentric view, humans are part of the interconnected web of animate and inanimate (e.g. chemical waste) phenomena, where everything is linked and reliant on interconnections and interdependencies, including soil.

This means that if one part of the environment suffers or deteriorates, the whole system is affected, which has widespread ramifications for ecosystem health, plant health, animal health, and human health, amongst others. The Māori world view always seeks to restore balance or equilibrium in an environmental system, including different levels of energy or vitality often referred to as restoring the mauri (life essence). The relationships and responsibilities span temporal and spatial scales. Indigenous knowledge is fundamental for transformative, equitable and sustainable approaches to chemicals and waste management, pollution prevention, and ecological restoration (Ataria *et al.,* 2023).

From a biocentric perspective, as highlighted by the FAO Indigenous Peoples Unit, Indigenous Peoples place their cosmogony and belief systems, the environment, and biodiversity at the centre of their restoration practices, in both natural and semi-natural ecosystems (FAO, 2024c).

13.4.6 Urbanization and soil sealing

The Māori Indigenous Peoples were the first in New Zealand to utilize some of the best food growing land (Singleton, 1988; Harmsworth, 2020). By the mid-nineteenth century, the settler government enforced the individualization of land titles, leading to land ownership fragmentation, in contradiction with the collective and customary rights of Indigenous Peoples over their lands, territories and resources (UN General Assembly, 2007). Large areas of Māori land were confiscated, with a particular focus on prime agricultural land (Kingi, 2008). Fifteen percent of land in New Zealand was classified as highly productive within land-use capability (LUC) class 1, LUC class 2 and LUC class 3. Land-use capability classes have increasing physical limitations as they move up the scale towards LUC class 8 (Lynn *et al.*, 2009)). Land in LUC class 1, LUC class 2 and LUC class 3 is particularly good for food production because it has the best soil for multiple rural land uses, is flat or gently sloping, and has an optimal growing climate. Protecting this land is important as it contributes to food security, while losing this land and soil to development is irreversible. Despite its importance, there is increasing competition for this land (Mackay *et al.*, 2011).

Population growth and the demand for more housing have put pressure on the availability of highly productive land, given the lower costs of developing flat land (Greenhalgh *et al.*, 2017). Land fragmentation has been apparent since at least the 1950s (Hunt, 1959), resulting in less land for agricultural or horticultural production (MfE and Stats NZ, 2021). Conversely, consumers demand local produce and the economic drivers of primary export industries incentivize land protection for food security.

The fragmentation of highly productive land and subsequent loss to commercial food production is escalating. For example, in the Auckland region, the number of land parcels of <8 ha with a dwelling increased by 74 percent on LUC class 1, 32 percent on LUC class 2 and 44 percent on LUC class 3 between 2002 and 2019 (Curran-Cournane *et al.*, 2021). Urbanization is expected to continue due to increasing pressures on highly productive land for housing and commercial development (MfE, 2022). To address urban development pressures, the National Policy Statement for Highly

Productive Land was gazetted in September 2022 with the primary objective of protecting the most productive land for primary production for intergenerational use and opportunities (MfE, 2022).

13.4.7 Other topics: Soil monitoring to assess sustainability of land uses in New Zealand

Monitoring of soils in New Zealand to assess the sustainability of current land-use activities and the effectiveness of policies, is required under legislation (the Environmental Reporting Act [New Zealand, 2015] and the Resource Management Act [New Zealand, 1991]). Monitoring of soil quality started in the 1990s, following national guidelines (Hill *et al.*, 2003; Hill and Sparling, 2009), before being upgraded by the National Environmental Monitoring Standards (NEMS) (NEMS, 2022). Sites are revisited every five years and trends are reported at both regional (Taylor, 2021) and national levels (MfE and Stats NZ, 2021). Target ranges are used to establish whether soil quality indicators are within the optimum range for productivity and environmental outcomes (Sparling, Lilburne and Vojvodic-Vukivic, 2008). These target ranges are currently being reviewed to better inform the balance between production and environmental outcomes (Stevenson, Drewry and Cavanagh, 2022).

Between 2014 and 2018, 80 percent of the monitored sites failed to meet the targets for at least one of the seven soil quality measures (pH, total C, total N, anaerobic mineralizable N, Olsen P, bulk density, and macroporosity) (MfE and Stats NZ, 2021)). Nearly half of all sites and 65 percent of dairy sites showed signs of compaction. While there were no improvements from 1995 to 2018 for meat or dairy farming, there were also no further declines in compaction observed (Stevenson and McNeill, 2020). Half of dairy, cropping and perennial horticultural sites had nutrient levels greater than agronomic needs. Soil C was below the target range at 26 percent of cropping sites (MfE and Stats NZ, 2021).

Although government and industry guidelines exist to promote SSM, no legislation or policy in New Zealand currently addresses soil health directly. However, freshwater management policy now has some guidance on practices that negatively affect soil compaction, such as damage to soil structure in wet conditions through pugging (MfE, 2023e).

13.4.8 Other comments

> *"Whatungarongaro te tangata, toitū te whenua."*
> *"People come and go, but the land endures."* (A Māori Indigenous Peoples' proverb).

Soils in New Zealand are experiencing an array of complex and interconnected challenges that may become exacerbated by storm events associated with climate change. Some impacts (such as severe erosion) are irreversible, and it is important to take an intergenerational perspective when considering, for example, the increasing vulnerability of the food system that soil supports. Taking an intergenerational and holistic perspective when considering soil is inherent and core to Māori values and principles (Hutchings, Smith and Harmsworth 2018; Hutchings, 2020).

13.5 Regional status and trend of threats to soil functions for the Pacific Island countries subregion

13.5.1 Indigenous Peoples' viewpoint

Indigenous Peoples constitute the majority of the population in most Pacific Island nations, maintaining profound generational connections to their lands and marine environments. These Indigenous Peoples are defined by distinct customary laws, traditional belief systems and specialized ecological knowledge systems that underpin their socioeconomic structures and resource management (UNPFII, 2015.)

Many Pacific Island countries are too small to feature on global maps, being characterized by their inhabitants as *vanua* or *fonua* (small landmasses) in a huge ocean (*moana*). Traditionally, land and the environment have connectedness to humans and culture, as demonstrated in the constitution of Tonga's motto: "*Ko e ʻOtua mo Tonga ko hoku tofiʻa*" ("God and Tonga are my inheritance").

An unpublished survey was conducted as part of the unpublished soil strategy for Tonga (S. Halavatau, personal communication, 24 February 2025) on the relational values of soils, humans and society confirmed the connectedness of soil to community by hereditary, its use, and livelihoods and culture. Some statements from the community are given as follows:

> *I am the eldest and I'm responsible to look after and preserve that land. Close to my heart. It sustains life and all humankind's development in terms of food supply. I have deep cultural connections to the land and soil. For me soil represents heritage and identity. Humans support and provide the necessary resources for soil to preserve its structure, physical characteristics, chemical composition, and other qualities that contribute to soil health. In turn, soil supports healthy plant life, clean water, and a beautiful human environment.*

In an article, Siwatibau (1984) stated that:

> *Pacific Societies identified closely with earth, with the physical manifestation of the earth-whether rocks, mountains or animals. In such a society the environment was not something separate from oneself–but an integral part of one's self, providing the physical manifestation of the vital link between the living and the dead* (Siwatibau, 2024, p.365).



13.5.2 Characteristics of the Pacific Island countries subregion

The islands of the Southwest Pacific region are highly diverse in their age, lithologies, soils and land management histories (Table 13.1) (Neall and Trewick, 2008; Nunn *et al.*, 2016). Diversity exists between the different nations but also within a given island chain. The variety of landforms results from the combination of volcanism from tectonic activity at the boundary of tectonic plates, subsequent reef island (atoll) development ringing the volcanic (higher) islands and, in some cases, the uplift of emerged reefs to form limestone islands (Nunn *et al.*, 2016) (Table 13.1).

Soil systems in the Pacific Island countries are changing at an unprecedented rate due to a combination of natural and anthropogenic factors. Wairiu (2016), SPC (2011), Halavatau (1998), and others have reported on changes, which now compromise the capacity of many soils to provide crucial ecosystem services, including production of food, fibre and wood, climate regulation, clean water and habitat for a wide diversity of plants and soil organisms. Tipping the balance in favour of natural resource management requires not only an increased capacity to meet recurrent expenditure, but also a greater awareness of the social costs of environmental degradation and pollution (Hunt, 1996).

13.5.3 Soil erosion in the Pacific Island countries subregion

Wind erosion is not an issue in the Pacific Island countries, but the geophysical nature of the islands renders some of the higher islands vulnerable to erosion due to rainwater (Morgan *et al.*, 2019). The mountainous islands (such as Guadalcanal Island in Solomon Islands, and the larger islands of Fiji and Vanuatu) were formed by volcanoes and have steep land areas which are susceptible to water erosion. The atoll islands (such as Kiribati, the Marshall Islands and Tuvalu) were formed by corals growing on extinct seamounts or volcanoes that have eroded or subsided and extend up to only a few metres above sea level. The low-lying islands are therefore particularly vulnerable to sea level rise and coastal erosion. Thus, erosion is taking place on two fronts: soil erosion due to rainfall events, especially on sloping and steep lands, and coastal erosion which may be exacerbated by sea level rise.

Overgrazing and overstocking of livestock have resulted in bare patches and has led to soil erosion, especially on steep marginal areas. Overgrazing has become evident in Fiji, Samoa, Solomon Islands and Vanuatu. In Fiji, goat-grazing areas are overstocked and show bare eroded patches due to the farmer's need to recoup expenses quickly and an ignorance of controlled grazing techniques (Government of Fiji, 2006). Overgrazing was also observed amongst smallholder livestock farmers in Samoa (Government of Samoa, 2006), and Solomon Islands (Government of Solomon Islands, 2006). In Vanuatu, deterioration in land quality was exacerbated by the commercialization of livestock farming, with unfenced paddocks, no protection of riparian zones and overstocking (Government of Vanuatu, 2007b).

Mining causes land degradation through extensive land clearing, rendering land prone to erosion. In Solomon Islands, Gold Ridge Mining cleared a large tract of natural forest for an open-pit mine and associated earth work, resulting in excessive runoff and sedimentation into local river systems (Government of Solomon Islands, 2006). Effects of mining are more pronounced in Papua New Guinea where more than a dozen open-pit mines are in operation (Government of Papua New Guinea, 2006). In Fiji, an environmental impact survey of the Mount Kasi open-pit gold mine indicated significant impacts for the surrounding environment, including the Yanawai River and coral reef ecosystems. Long-term environmental consequences are evident at the mine at Savusavu, near the Yanawai River on the southern coast of Vanua Levu, which ceased operation in 1998, especially where mine tailings were deposited (Government of Fiji, 2006). In Vanuatu, the expansion of mining operations for aggregate for construction is a major and growing issue surrounding the urban

areas of Port Vila and Luganville (Government of Vanuatu, 2007b). The areas cleared for mining also always remain exposed, resulting in runoff and degraded soils.

13.5.4 Soil carbon loss in the Pacific Island countries subregion

The SOC status of Pacific Island countries' soils is influenced by land-use practices which contribute to SOC reduction, such as the mechanization of ploughing that can cause oxidation of organic C, the use of plant residues which are often burnt, and agricultural intensification resulting in shortened fallow periods and loss of biodiversity. Overstocking of animals can also result in removal of vegetation cover, increasing soil erosion and the reduction in SOC.

In the higher islands with mostly volcanic soils (Table 13.1), the effects of cropping on soil biological quality are generally similar, in that cropping leads to a decline in topsoil organic matter and microbial biomass. Morrison and Gawander (2016) observed that topsoil changes could be related to changes in organic matter and associated ion exchange properties. The major changes occurred in the first few years after clearing, but some changes continued for many years. Subsoil samples (below 30 cm) showed an increase in organic C after cultivation, probably due to soil mixing during land preparation, organic matter movement during cropping and the decay of crop roots. Some studies also found a change in the C source over time from natural vegetation to cultivated plant species, with C_4 C fixation pathways (Manu *et al.*, 2014) and that water-stable aggregates declined with the increasing intensity of cultivation. Overall, careful management of topsoils is essential for maintaining soil fertility and hence crop productivity on these highly weathered soils.

The most vulnerable Pacific Islands are the atolls with inherently low soil fertility and productivity that rely heavily on SOC to improve soil nutrient supply and water-holding capacity (ACIAR, 2017).

13.5.5 Soil biodiversity loss in the Pacific Island countries subregion

The driving forces behind changes in soil biodiversity include land-use change, overuse and exploitation at the ecosystem level, as well as change in the natural environmental conditions caused by burning or overcultivation. Soil biodiversity usually declines in connection to other soil degradation processes, such as the loss of SOM, the application of pesticides and chemical fertilizers, soil contamination, soil compaction, and insufficient crop residues (Tibbett, Fraser and Duddigan, 2015). Although knowledge on current trends in soil biodiversity is limited, it is commonly agreed that soil biodiversity is reduced by agriculture intensification which has often been accompanied by monocropping, mechanization and the increased use of chemicals (Tibbett, Fraser and Duddigan, 2020).

13.5.6 Nutrient mismanagement in the Pacific Island countries subregion

Nutrient balance is defined as the difference between the nutrient inputs to the farming system (mineral and organic fertilizers) and nutrient outputs, such as the uptake and removal of nutrients in crops and losses to leaching. As discussed in Section 13.2.3, most Pacific Island countries have nutrient deficits, as nutrient removal exceeds nutrients added though fertilizers and soil amendments such as compost.

The downward spiral of soil fertility in the Pacific Island countries is largely caused by nutrient decline as documented in semi-intensive farming systems such as for sugar cane in Fiji where substantial topsoil fertility decline over 30 years was caused by inadequate soil conservation and poor nutrient management practices (Morrison *et al.*, 2016). They concluded that soil and farm nutrient budgets need developing to assist farmers with management decisions. Nutrient decline is not confined to sugarcane production but occurs more broadly across farming systems in the Pacific Island countries, especially where intensification and crop export from the field are occurring. Strongly weathered soils from Fiji are of low fertility, being low in phosphate and other nutrients (Curtin, Naidu and Syers, 1991; Naidu, 1985).

In Pacific Island countries, traditional cropping systems are based on a long bush-fallow system to maintain soil fertility. In the case of atolls, this involves recycling of large amounts of organic material in pits or heaps. However, smallholders who have intensified crop production to supply growing urban and export markets have typically failed to replenish soil nutrients and organic matter adequately, leading to falling yields and problems with the soil-borne diseases that characterize declining soil health (ACIAR, 2014). The commercial production of taro in Fiji and Samoa, ginger and kava in Fiji, and squash in Tonga has resulted in increased soil erosion, reduced yields and decreased household food security (SPC, 2014). In Papua New Guinea, the total value created by agriculture makes up 36 percent of the gross domestic product (GDP), with exports in 2011 equivalent to USD 1.28 billion (nearly six times that of Fiji). Since the late nineties, Vanuatu's agricultural products have made up around 80 percent of total export earnings; by far the highest of any Pacific Island country (Taylor *et al.* 2016).

Acidification is caused by rainfall and leaching, acidic parent materials, organic matter decay, the harvest of high yielding crops, and the nitrification of ammonium. The extent of acidification is poorly documented in the soils of the Pacific Island countries.

13.5.7 Salt accumulation in the Pacific Island countries

The salinization of soil in the Pacific Island countries, especially on low-lying atolls through seawater intrusion and sea sprays, is a serious threat to food security. Flash flooding, seawater inundation and tsunamis can cause waterlogging and increase salinity and sodicity. The 2022 volcanic eruptions and associated tsunami in Tonga caused salinization and sodification through the inundation associated with the tsunami (EBSCO, 2025). Such problems are likely to be exacerbated by sea level rise and increased storminess associated with climate change.

13.5.8 Soil pollution in the Pacific Island countries

There are two types of soil contamination or pollution: "diffuse" and "point source".

Diffuse source pollution (from activities such as fertilizer and pesticide applications) is the main issue affecting the soils of Pacific Island countries . However, the magnitude of soil diffuse contamination is unknown due to the complexity of the diffuse sources and lack of data on contaminant input. There is data on triple superphosphate application but there is no information on Cd loading in farmed soils. Preliminary investigation by R. Naidu (personal communication, 15 October 2024) shows low but appreciable levels of 0.01 M $Ca(NO^3)^2$ extractable Cd in sugarcane farm soil in Fiji.

There is data on herbicide applications for some countries, but no data available on pesticide concentrations in soils (R. Naidu, personal communication, 15 October 2024). There is a knowledge gap on the magnitude of soil diffuse pollution in the region. Some countries have data on heavy metals, however there are few data on pesticide residues and even less on emerging contaminants, despite being recognized as important issues.

Reports on point source pollution in the region are also limited (R. Naidu, personal communication, 15 October 2024). There are reports on the impacts of heavy metal contamination from Au and Cu mining activities on the surrounding areas, such as reports on impacts of Ok Tedi mine in Papua New Guinea. Other potential point source pollution sites include waste dumps, chemical stores, septic tank discharges and old industrial sites.

Large farms that raise livestock, including cows, pigs and chickens, are also sources of point source pollution, through feeds, chemicals, hormones, and untreated or mismanaged animal wastes (R. Naidu, personal communication, 15 October 2024). There is also a danger to local populations on some Pacific Island countries from the abandoned military bases and weapons buried after the First and Second World War.

With the increase in urban populations and expansion onto productive land, inadequate strategies are in place for the management of household wastes, including poorly designed landfills, which can lead to significant degradation of land and water bodies, especially where landfills are on coasts (R. Naidu, personal communication, 15 October 2024). The use of livestock manure as an alternate source of nutrients continues to adversely impact soil health and quality across the whole region.

Unregulated discharges from point sources can result in water pollution and unsafe drinking water and can restrict activities like fishing and swimming (R. Naidu, personal communication, 15 October 2024). Some chemicals discharged by point sources are harmless, but others are toxic to people and wildlife. Whether a discharged chemical is harmful to the aquatic environment depends on several factors, including the type of chemical, its concentration, the timing of its release, weather conditions, and the organisms living in the area.

13.5.9 Urbanization and soil sealing in the Pacific Island countries subregion

Encroachment of villages and development infrastructure have increased soil sealing in the region in recent years. Most villages in the Pacific Island countries were established during the nineteenth century and were nearly all built on, or adjacent to, land highly suited to horticulture and cropping. Over time, the villages developed into towns and cities, resulting in the sealing of good agricultural lands. For economic reasons, it is unlikely that these good quality soils will ever be allowed to regain their biological function. Loss of the most productive soils to urban development is occurring throughout the region and is particularly significant in fast-developing islands like Fiji, and some smaller Pacific islands. The problem will intensify in coming decades with increasing populations, and increased road, urban and other infrastructure development, yet currently, the rate of sealing is not being monitored.

Urban development is increasing, and most arable lands in most Pacific Island countries are now being developed for different uses, such as housing, garment industries, and tourism. For example, about 500 ha of first-class sugarcane land in the corridor between Nadi Town and Nadi Airport in Fiji was taken for non-agricultural purposes (Government of Fiji, 2007), and the best agricultural land at Lunga on the Guadalcanal plains in Solomon Islands was converted to residential and commercial plots (Government of Solomon Islands, 2006). In most Pacific Island countries, soil sealing as a result of urban and infrastructure expansion is becoming an important issue (Government of the Cook Islands, 2004). In the Cook Islands, tourism infrastructure development has concentrated on the foreshore, with 64 percent of the foreshore's issues related to the construction of tourist facilities and the protection of properties (Government of the Cook Islands, 2006). Three issues associated with tourism development that are contributing greatly to growing land degradation problems are building too close to the sea, the complete clearance of natural vegetation (and subsequent replacement with ornamental species not suited to the environment), and the increased need for the disposal of liquid waste like domestic wastewater and sewage (Government of the Cook Islands, 2006).

13.5.10 Climate change impacts in the Pacific Island countries subregion

Climate change can negatively impact land, leading to land degradation. This includes increasing temperature within the soil profile, changing rainfall patterns, sea level rise resulting in coastal erosion, inundation and salt intrusion, and the increasing intensity of extreme events such as cyclones and droughts. In Kiribati, most pandanus (*Pandanus tectorius*) trees are being lost through excessive coastal erosion, saltwater inundation and sea level rise. Breadfruit (*Artocarpus altilis*) and banana crops also suffer from drought stress resulting in lower yields (Government of Kiribati, 2007). In Tuvalu, groundwater salinization – associated with sea level rise and the overpumping of aquifers – is destroying the traditionally important swamp taro pit gardens (Webb, 2007). The World Bank (Bettencourt *et al.*, 2006) reported that between 1950 and 2004, about 207 extreme events were recorded in the Pacific region, with the cost of climate-related disasters on agricultural lands and crops estimated to range from USD 13.8 million to USD 14.2 million. In Fiji, the 1997 and 1998 drought events resulted in 50 percent loss in sugarcane production, with total losses in the industry of around USD 50 million, while other agriculture damage and losses, including livestock death and damage to fertile land, totalled around USD 7 million (McKenzie, Kaloumaira and Chand, 2005). The frequency and intensity of El Niño and La Niña events and the accompanying extremes such as high intensity rainfall, floods, tropical cyclones and droughts are posing threats to land resources. In Tuvalu, extreme weather has highlighted the vulnerability of swamp taro to dry conditions and salinity problems have been exacerbated, illustrating the effect of multiple climate-related stresses (Webb, 2007). The combination of the high temperatures experienced during the drought, and the increasing salinity of the pulaka pits, adversely affected the growth of the swamp taro (Government of Tuvalu, 2006).

13.5.11 Examples of innovative sustainable soil management practices in the Pacific Island countries subregion

Improved fallow systems

In Pacific Island countries, an increasing population pressure has resulted in reducing the traditional "bush fallow" phase (i.e. where cultivated land reverts to early stage of natural vegetation succession) from 15 years to an average of about 3 years and in many places it has been reduced to almost continuous cultivation. This has resulted in the depletion of SOC and soil nutrients, declining soil water stable aggregates and water-holding capacity, and the increased incidence of pests and diseases (ACIAR, 2015, 2017). The volcanic ash soils of the higher islands are strongly P-fixing and most are deficient in P for crop production (Halavatau, 1998).

One solution is using a leguminous cover crop. The velvet bean (*Mucuna pruriens*) is a tropical leguminous cover crop recently introduced into the Pacific. It nodulates well and grows vigorously without inoculation or fertilizer application on mineral-depleted farmlands. Carbon buildup in soils derived from Mucuna pruriens has been studied (Halavatau and Asher, 2003; Lal, 2013; Anand, 2016), with results showing that productivity can be improved while also improving SOC content and the resilience of local production systems.

Use of targeted compost on Pacific atolls

Soils of atoll island countries like Kiribati, the Marshall Islands and Tuvalu are inherently very poor in available macronutrients like N, P and K, and available micronutrients like Cu, Fe and Mn, impacting the countries' ability to produce crops. This was finally addressed through the development of composts that specifically targeted the low levels of macro- and micronutrients (ACIAR, 2014).

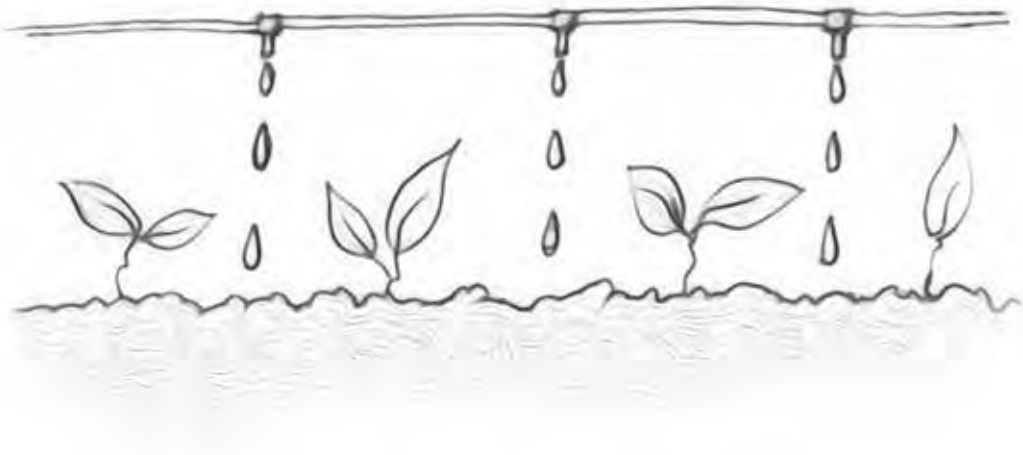
Improving nutrient management

Nutrient use efficiency in Pacific Island countries is still far below requirements. The CSIRO Pacific Soils Project looked at nutrient balance sheets for crops in the islands (CSIRO, n.d.). The project has resulted in countries adopting fertilizer recommendations based on soil testing, and has supplied countries with soil testing kits to aid the improved use of fertilizers.

Good water management is vital for good nutrient management

Most agricultural production in the Pacific region is rainfed, with very little irrigation (Burke, 1997). However, when growing commercial crops like watermelon and tomato that require irrigation, farmers irrigate manually without knowing whether they have given enough, not enough, or too much water. There is little knowledge of the link between watering and nutrient availability, and water management and fertilizer (nutrient) management are rarely considered in an integrated way.

The Australian Centre for International Agricultural Research (ACIAR) project, Improving soil health, agricultural productivity, and food security on atolls (ACIAR, 2022) analysed soils and compost ingredients, then developed targeted compost and conducted compost rates trials for the different atoll islands involved. Understanding that soil water is a constraint for production, the project looked at water management, taking into consideration the geomorphology of the islands, to decide where wicking-based systems or bucket drip irrigation should be employed.



13.6. Progress in and impediments to the adoption of sustainable soil management

The content in Section 13.6 has been developed based on the material presented in the previous sections. The criteria used in the evaluative statements is presented in Chapter 7 of this report.

13.6.1 Summary of state of and trend in sustainable soil management

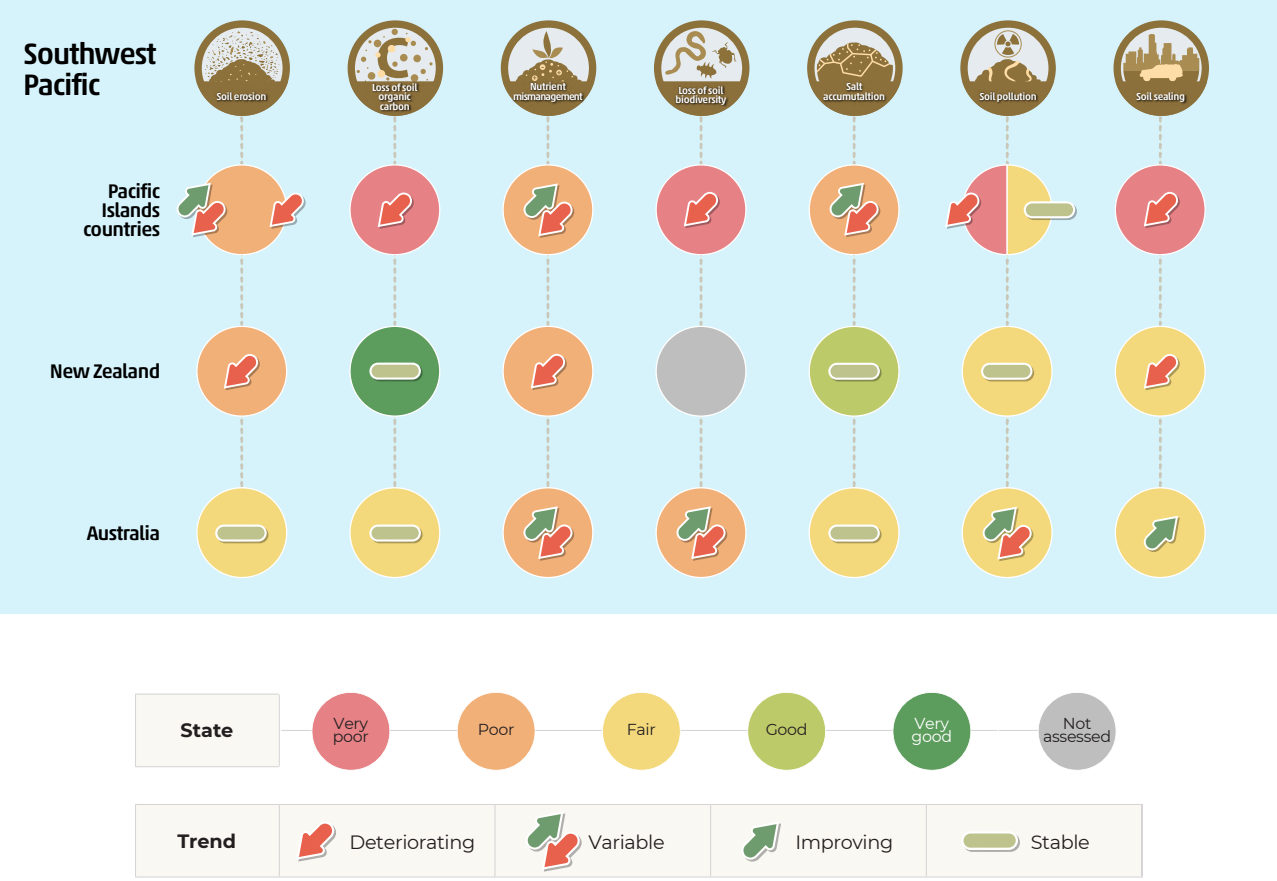
The state reflects the effectiveness of management practices in place as of 2015 in preventing or reducing a specific soil threat. At the regional or subregional scale, changes in management occur at decadal or longer time scales and hence the state as of 2015 is relevant for 2015 to 2024 assessment period.

The trend indicates the direction of change in the adoption of SSM practices between 2015 and the finalization of this report (late 2024). In short, trend reflects the how SSM adoption has progressed over time.

The state and trend in SSM in the Southwest Pacific region are mixed (Figure 13.15 and Table 13.6): while some countries have shown improvement in some respects, challenges persist for others. For example, while Australia and New Zealand have improved soil erosion monitoring and management, some problems remain in a few Pacific Island countries. Similarly, soil erosion linked to logging remains an issue in some Pacific Island countries, while reforestation has significantly increased in Australia, New Zealand and Fiji.



Figure 13.15. Assessment of state and trend of threats to soil functions in the Southwest Pacific region through a management-based approach



Note: For more details, refer to Table 13.6.



Table 13.6. Summary of threats to soil functions in the Southwest Pacific region and the confidence in the assessment of their state and trend

Threat	Subregion and description	Rank[1]	Confidence in	
			State	Trend
Soil erosion	Pacific Island countries: Mountainous volcanic islands (e.g. Fiji, Guadalcanal and Vanuatu) have steep lands and high rates of water erosion and a deteriorating trend often associated with overgrazing. Atoll islands and coastal areas on volcanic islands are threatened by water erosion and coastal erosion due to sea level rise.	2	*Atolls* Medium *High islands* High	*Atolls* Medium *High islands* High
	New Zealand: Severe erosion is experienced in areas with highly erodible geology and increasing frequency of extreme storm events.	1	High	High
	Australia: Over 80% of cropping lands are cultivated using minimum or no-till while retaining crop residue. Satellite data over 20 years confirm high levels of ground cover for previously eroded regions. There is high erosivity in coastal regions of eastern Australia.	1	High	High
Soil carbon loss	Pacific Island countries: There is high SOC loss due to inappropriate management practices (e.g. burning fallows, mechanization, and agriculture intensifications resulting in shortened fallow).	3	High	High
	New Zealand: Many managed soils in New Zealand have high SOC, particularly pastoral (grassland) soils. Soil organic carbon losses occur under intensive vegetable growing.	3	Medium	Medium
	Australia: Many managed soils in Australia have low SOC, particularly cropping and rangeland soils in drier regions. Maintenance of ground cover of over 50% is essential to reduce SOC loss by erosion.	2	Medium	Low
Soil biodiversity change	Pacific Island countries: Although knowledge on current trends in soil biodiversity is very limited in the Islands, it is commonly agreed that it is reduced by agriculture intensification (e.g. monocropping, mechanisation and increased use of chemicals).	9	High	High
	New Zealand: Very limited data prevent an assessment.	N/A	Low	Low
	Australia: There are limited data but a high percentage of minimum and no-till cropland should lead to improved biodiversity.	3	Low	Low
Nutrient mismanagement	Pacific Island countries: There is significant nutrient decline due to intensive agriculture, reduced fallow systems and SOM decline.	4	Low	Medium
	New Zealand: Intensive pastoral and vegetable growing land have high N and P inputs and losses to the environment.	2	High	High
	Australia: There are widespread issues due to soil acidification by fertilizers, especially for vulnerable soil types with low buffering capacity. Low levels of liming and poor nutrient balances intensify the issue.	5	High	High
Salt accumulation	Pacific Island countries: There is soil salinization, especially on low-lying atolls due to sea water intrusion, sea spray and inundation by tsunamis, and is a serious threat to food security.	8	Low	Low
	New Zealand: Not a substantial issue in the country.	5	Medium	Medium
	Australia: Historically, this has been a problem in the southern agricultural zone, especially in the southwest of Western Australia, having a high effect on productive lower landscape positions.	4	High	High

Threat	Subregion and description	Rank[1]	Confidence in	
Soil pollution	Pacific Island countries: There is a lack of data on pollution in mountainous islands. Diffuse source pollution is believed to be the main issue but there is significant point contamination from mining sites, waste dumps, chemical stores, septic tank discharges, and old industrial sites. Atolls are stable and in a fair state due to legislative controls.	7 7	High	High
	New Zealand: Legacy areas, especially in industrial and urban centres, exist and mapping of these areas in progress. Many known contaminated areas have been remediated. Information is needed on emerging pollutants.	6	Medium	Low
	Australia: Legacy sites have been identified and many known contaminated areas have been remediated. There are oncerns about emerging contaminants (e.g. PFAS and microplastics)	6	Medium	Low
Urbanization and soil sealing	Pacific Island countries: Historical settlement patterns and rapid growth of settlements have led to a high loss of cropland and horticulture.	6	High	High
	New Zealand: Urbanization and soil sealing are continuing to affect highly productive agricultural land.	7	High	High
	Australia: Urbanization and soil sealing are continuing to encroach on agricultural land. The effects of expansion of renewable energy (e.g. large-scale solar farms) is largely unknown.	7	High	High
Soil acidification	Pacific Island countries: There is high fertilizer use in some islands (e.g. Fiji) on acidic soils.	1	Medium	Medium
	Australia: There are widespread issues due to soil acidification by fertilizers, especially for vulnerable soil types with low buffering capacity. Low levels of liming and poor nutrient balances intensify the issue.	5	High	High
Soil compaction	Pacific Island countries: Compaction is associated with machine traffic and mechanization.	5	Low	Low
Soil governance	Australia: There is a lack of rural social networks, a gap between intention and action and a reduced presence of government leading to reduced information flow. There have been positive developments for medium action on the National Soil Strategy	8	High	High

Note: The assessment is based on the evidence provided in this chapter.

[1] This ranking indicates the relative severity of threats within each subregion, with 1 denoting the most severe threat and 9 the least severe.

Other issues remain serious across the subregions, such as the impact of climate change, nutrient management (with overuse in many parts of Australia and New Zealand and soil nutrient harvesting in many Pacific Island countries) and urbanization impacting soil resources. However, in many cases, it has been difficult to assess and quantify the management responses to the threats and trends due to a lack of monitoring and reporting.

There have been some significant improvements in the mapping and monitoring of soil quality, and policy interventions (such as nutrient management in North Queensland in Australia) and management interventions (such as monitoring the use of nutrients and grazing pressure) have improved soil health outcomes in many regions. Further improvements in interventions (such as better nutrient management, climate change mitigation and adaptation, and increased ground cover) will improve soil health and contribute to many aspects of national economic and environmental security. The concept of soil health and monitoring has been integrated in New Zealand policy and management for a significant period. However, key threats to soil health remain, including soil erosion (although driven by geology and rainfall), acidification, pollution, inappropriate land management, the loss of organic C and urbanization.

There have been significant improvements in monitoring and managing soil health in New Zealand

in recent years, and policy and management interventions have improved some aspects of soil health including the reduction of soil erosion. However, key challenges remain, exacerbated by climate change and extreme weather events, and require further policy, social and management interventions to improve soil health for sustainability of ecosystem functions that soil provides.

Since the European Union's Development of Sustainable Agriculture in the Pacific (DSAP) project (Hunter, 2007) (which promoted the use of cover crops, compost and bucket irrigation), there have been positive moves in the Pacific Island countries subregion towards managing soils sustainably. With support from ACIAR, the Pacific Island countries are developing appropriate technologies to manage their soils. Countries like Tonga are now developing a national soil strategy and soil management technologies have been developed in the islands which are currently making a difference in the region.

It was proposed and endorsed at the Pacific Week of Agriculture 2023 that soil security (and climate change) be elevated to the same level as food, water and energy securities (SPC, 2023). This has initiated countries in the region to develop national soil strategies, with Tonga completing its national soil strategy, and other countries such as Fiji, Kiribati, Naoero, Samoa, Tuvalu, and Vanuatu to follow.

There have been improvements in governance and policy related to SSM. For example, the National Soil Strategy of Australia is a national policy which includes a 20-year plan to value, manage and improve soil (DAWE, 2021). Soil monitoring in New Zealand is required under legislation to assess the sustainability of current land-use activities and the effectiveness of policies (the Resource Management Act [New Zealand, 1991] and the Environmental Reporting Act [New Zealand, 2015]). However, better coordination between national policies within the region is needed which may help develop region-specific policy intervention.

State of the environment monitoring and reporting in both Australia and New Zealand includes factors related to soil, and so a database providing some medium-term insights is starting to emerge.

For example, monitoring and reducing soil erosion has been successfully implemented in both Australia and New Zealand, utilizing tools such as ground cover, no-till cropping, and better management of grazing pressures, with reforestation also increasing. New information on the status of soil biodiversity is increasing both in Australia and New Zealand but is limited in many other Pacific countries mainly due to a lack of infrastructure and expertise.

Some progress on the impacts of urbanization and intensification of agriculture have emerged but their full impacts on soil health, biology and functions remain unquantified and potentially remain a major problem for all countries in the regions. Similarly, data on the impacts of climate change and extreme weather on SOC are improving but their effects on soil biodiversity and ecosystem functions remain limited. There is limited if any information available on how climate change, loss of biodiversity, management practices, and pollution interact to impact soil and its functions.

13.6.2 Recommendations for future priorities for action

Learning from successful outcomes such as with tackling soil erosion and reforestation in Australia and New Zealand, and applying this to other parts of the regions can improve outcomes for sustainable soil resource management. However, this will require the explicit consideration of the local environment, socioeconomic conditions and cultural aspects, along with the availability of resources, infrastructure and expertise. Country- or location-specific improved nutrient management can benefit many aspects of soil, including biodiversity, overall health and the functions it provides. Developing social interventions to reduce nutrient overuse in Australia and New Zealand and to increase nutrient use is in many Pacific Island countries, where nutrients are limited, can improve food security and environmental sustainability through improving soil health.

Increasing soil C is important for many reasons, including climate change mitigation and improving soil health. The promotion of agronomic practices that increase soil C (such as the use of biochar, cover crops and perennial crops) will be important, but clear guidelines for implementation need to be provided.

Soil biology, which is a critical determinant of soil functions and sustainability, has historically received limited attention but has seen some significant progress in recent years, particularly in Australia and New Zealand. This can be used as baseline data on which a regular monitoring programme can be instituted. However, many Pacific Island countries have limited information on soil biodiversity (as well as pollution and overall soil health monitoring), and there is an urgent requirement for the development of infrastructure and expertise for biodiversity and soil health monitoring. Regional coordination and resources should be explored to develop these facilities wherever required.

There is progress being made in soil resource governance across the region. Expediting the development of national strategies, along with coordination between different countries, will enhance cooperation to achieve the shared goals of SSM under a rapidly changing climate and intensifying extreme weather events. This cooperation should explicitly address the inequity in infrastructure and expertise, building and retaining capacities in traditional and emerging tools (such as genomics, satellites and artificial intelligence), and the creation of a regional knowledge hub for better monitoring, managing and restoring soils across the region.



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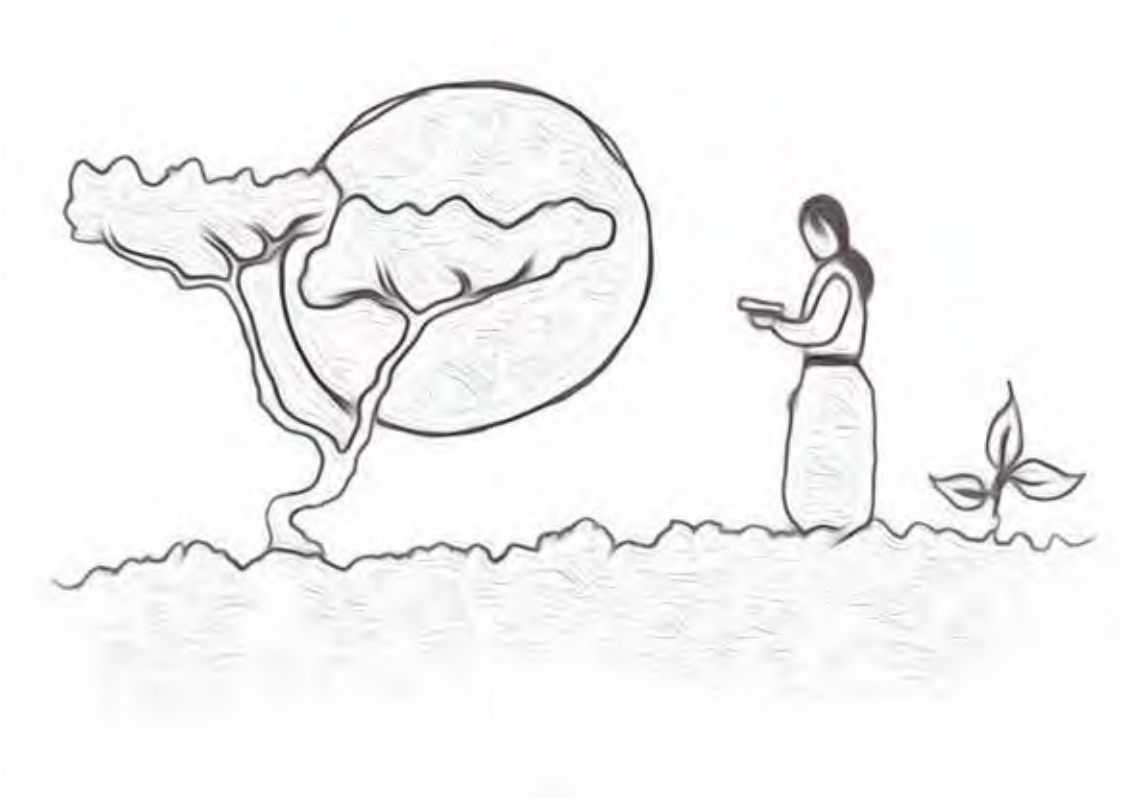
Chapter 14 | Regional assessment for the sub-Saharan Africa region

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14.1 Regional overview

14.1.1 Characteristics of the region and subregions

Introduction

For the purposes of this report, sub-Saharan Africa is divided into five subregions: equatorial forest, deciduous woodland, savannahh and highlands, desert and semi-desert, and islands of Africa (Figure 14.1). The names for the first four are from the simplified map of biomes (Miller *et al.,* 2016). The boundaries for the subregions correspond generally to the Global Agro-Ecological Zoning (GAEZ) 4.0 mapping project (FAO, 2022a) for sub-Saharan Africa (Figure 14.1). The equatorial forest Agro-Ecological Zoning (AEZ) corresponds to the lowland humid tropics AEZ and the highland humid tropics AEZ. It also includes an area of dominantly hydromorphic (wetland) soils of the Cuvette Centrale in the central Congo Basin. The deciduous woodland area generally corresponds to the lowland subhumid tropics AEZ. The savannahh and highlands subregion corresponds to the lowland semi-arid tropics and warm semi-arid sub-tropics AEZs and the semi-arid, subhumid and humid highland tropics AEZs. The desert and semi-desert subregion corresponds to the desert/arid climates AEZ.

Also, across sub-Saharan Africa, the traditional lands and territories of many Indigenous Peoples' groups - encompassing dry deciduous woodlands, Sahelian savannahs and hyper-arid zones - are managed under customary land tenure systems. These practices, maintained by hunter gatherers and nomadic pastoralists, play a documented role in stabilizing fragile topsoil and protecting localized resource dynamics within the continent's major agricultural frameworks (FAO and IWGIA, 2022).

In the following summaries for each subregion, the information on soils (Figure 14.2) and land cover (Figure 14.3) is drawn from the *Soil Atlas of Africa* (Jones *et al.,* 2013), information on farming systems comes from Dixon *et al.* (2001) and OECD and FAO (2022) and on the aridity index from Hill *et al.* (2018) (Figure 14.4).

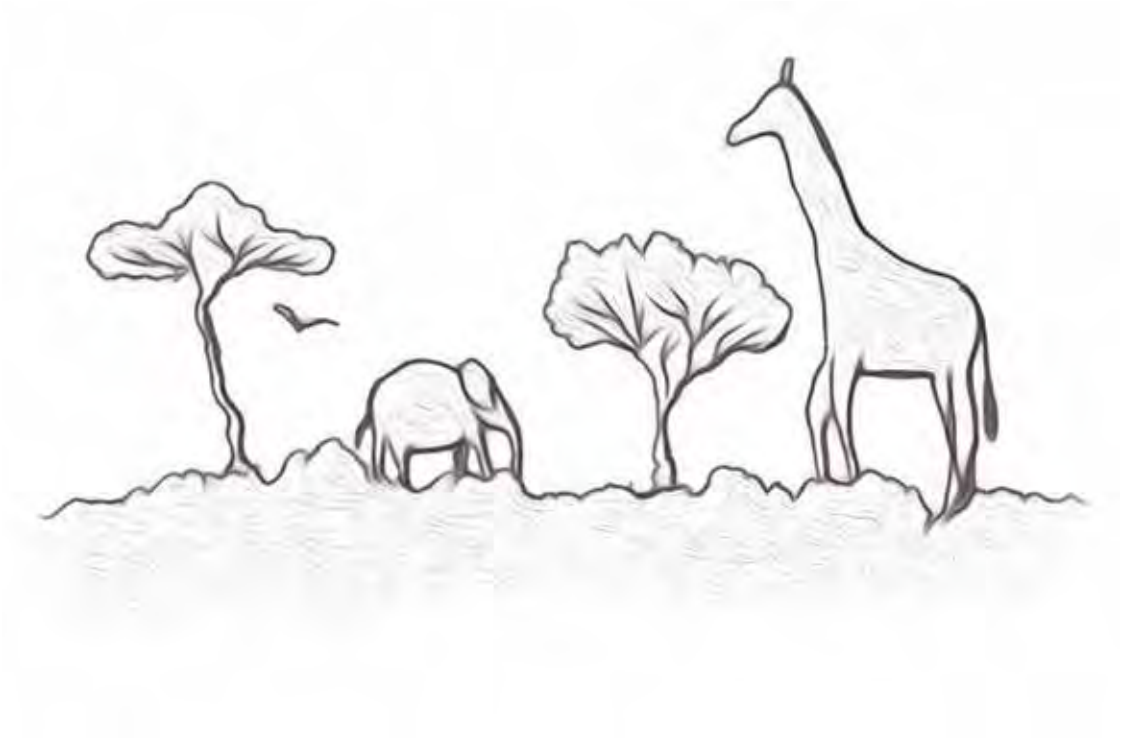
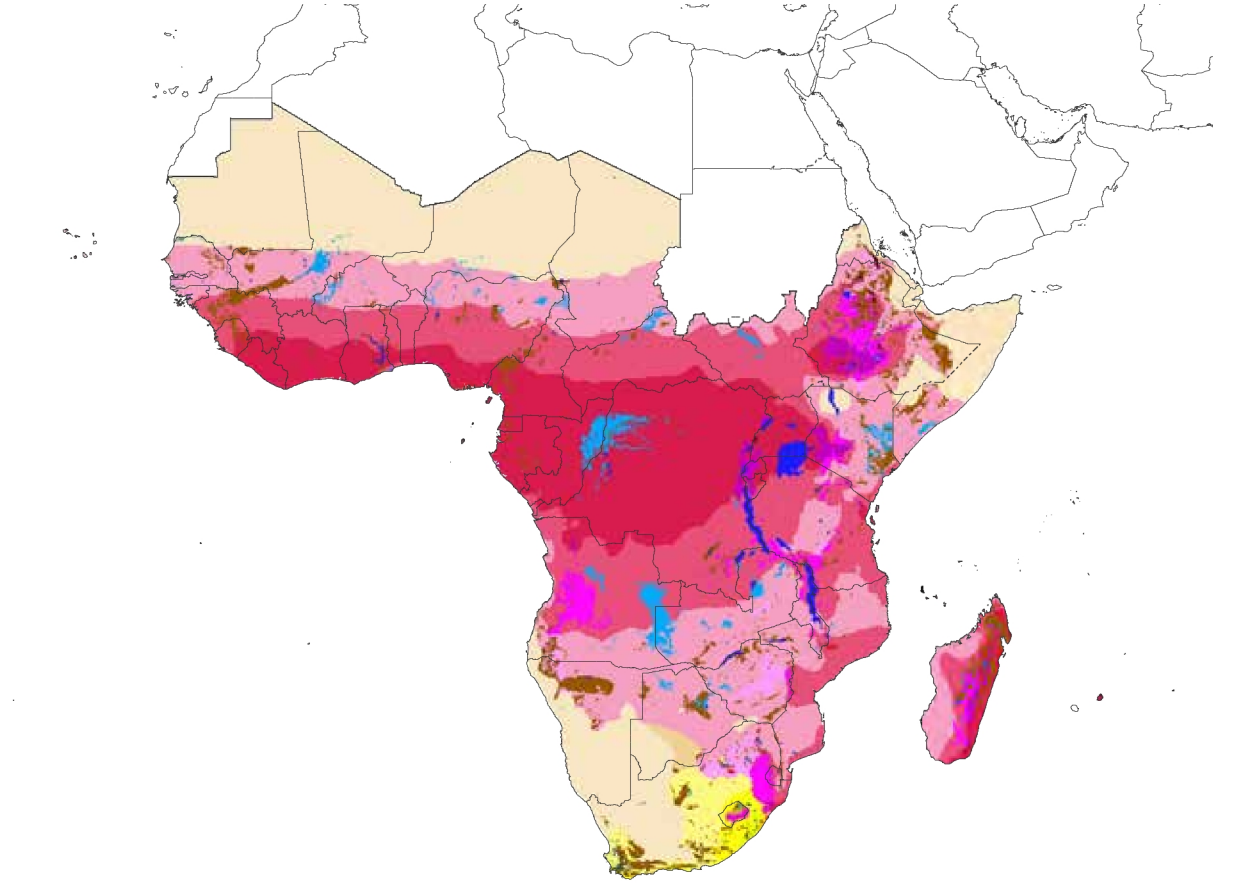


Figure 14.1. Map of Global Agro-Ecological Zoning (GAEZ 4.0) in sub-Saharan Africa



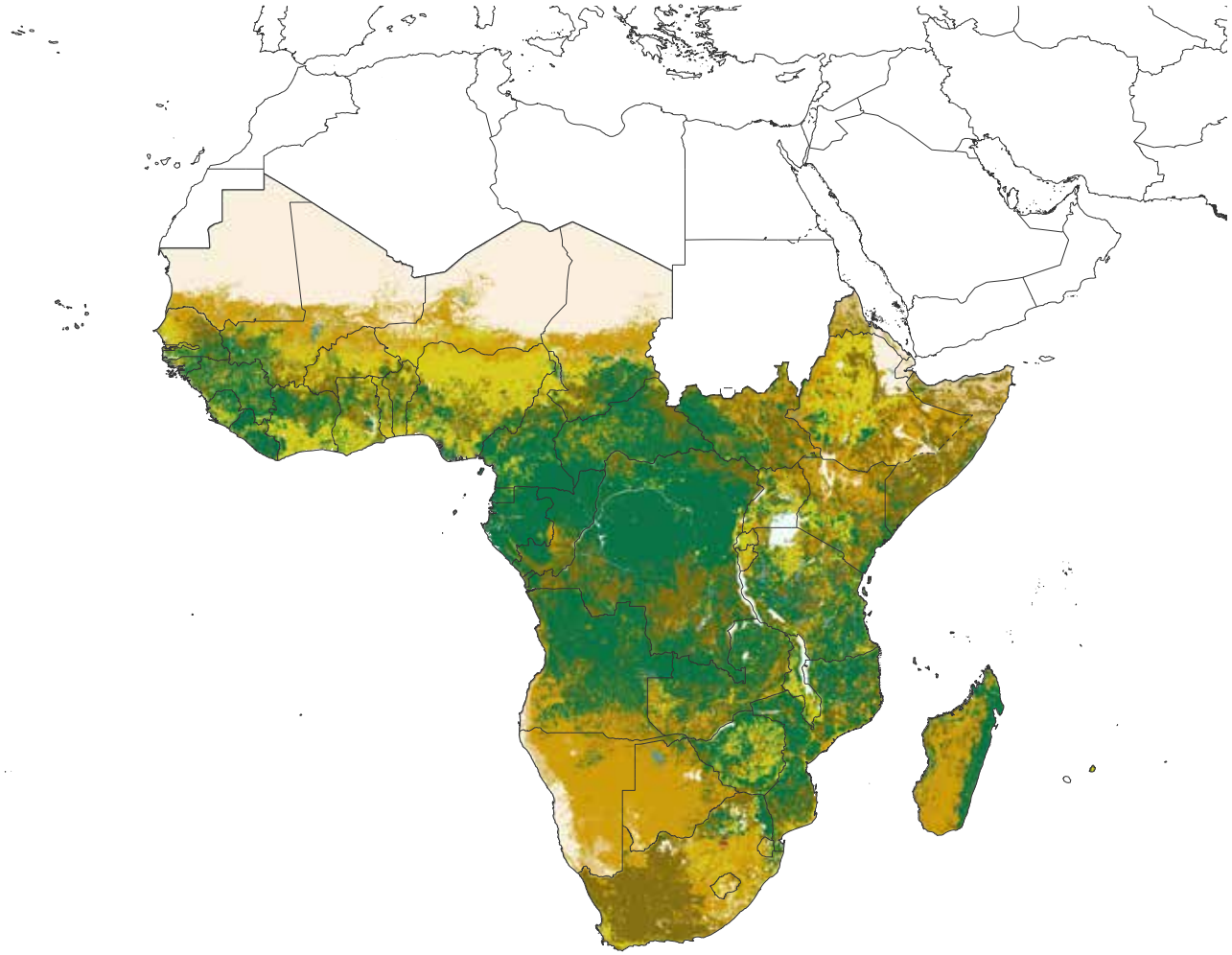
Agro-Ecological Zoning

- Tropics lowland (semi-arid)
- Tropics lowland (subhumid)
- Tropics lowland (humid)
- Tropics highland (semi-arid)
- Tropics highland (sub-humid)
- Tropics highland (humid)
- Subtropics warm (semi-arid)
- Subtropics warm (sub-humid)
- Subtropics warm (humid)
- Subtropics moderately cool (semi-arid)
- Subtropics moderately cool (sub-humid)
- Subtropics moderately cool (humid)
- Subtropics cool (semi-arid)
- Subtropics cool (sub-humid)
- Cold no permafrost (moist)
- Cold no permafrost (wet)
- Dominantly very steep terrain
- Land with severe soil/terrain limitations
- Land with ample irrigated soils
- Dominantly hydromorphic soils
- Desert/arid climate
- Dominantly built-up land
- Dominantly water

Refer to the disclaimer on page ii for the names and boundaries used in this map. Final boundary between the Republic of Sudan and the Republic of South Sudan has not yet been determined. Final status of the Abyei area is not yet determined.

◊ *Source:* **FAO**. 2024. *GAEZ: Land and Water Resources*. [Accessed 24 October 2024] https://gaez-services.fao.org/apps/theme-1/. Licence: CC-BY-4.0.

Figure 14.2. Land-cover map of Africa



Refer to the disclaimer on page ii for the names and boundaries used in this map. Final boundary between the Republic of Sudan and the Republic of South Sudan has not yet been determined. Final status of the Abyei area is not yet determined.

Source: **FAO**. 2024. *GAEZ: Land and Water Resources.* [Accessed 24 October 2024] https://gaez-services.fao.org/apps/theme-1/. Licence: CC-BY-4.0.

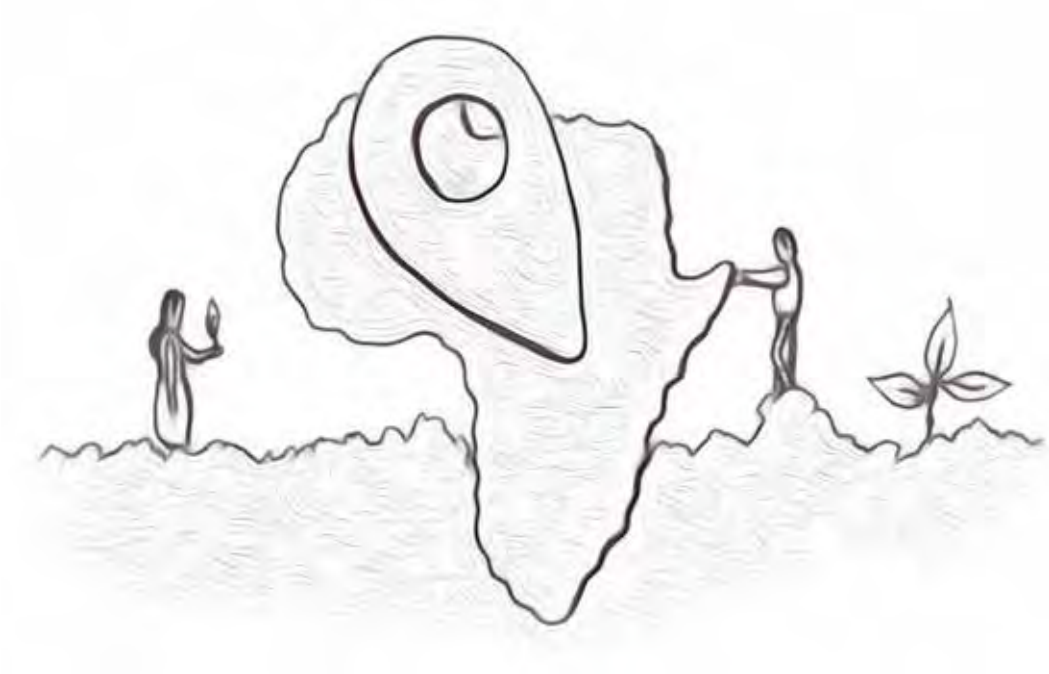
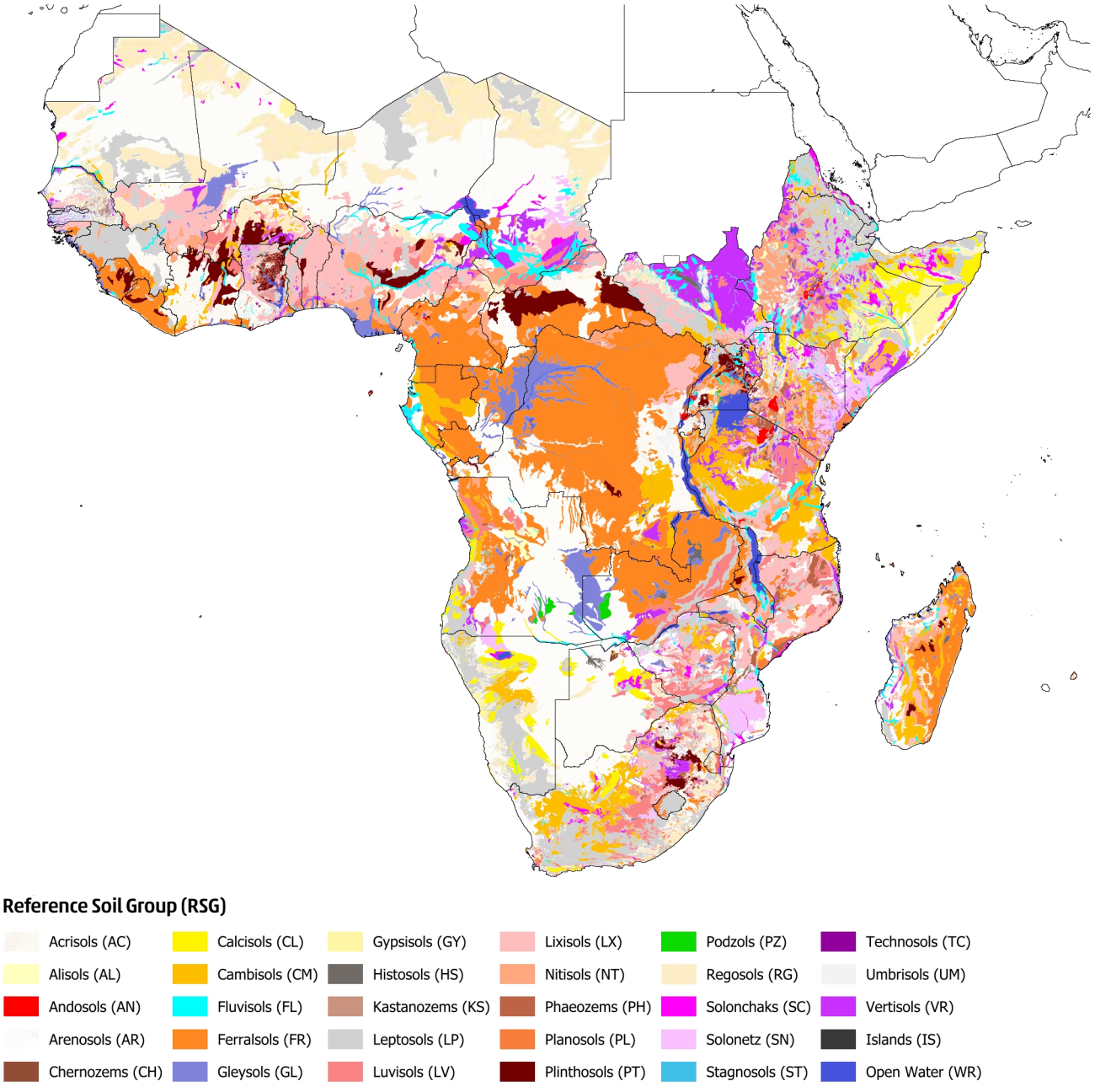


Figure 14.3. Soil map of Africa



Source: **FAO**. 2024. *GAEZ: Harmonized World Soils Database version 2.0*. [Accessed 24 October 2024] https://gaez.fao.org/pages/hwsd. Licence: CC-BY-4.0.

The equatorial forest subregion

The equatorial forest subregion occupies most of Central Africa and extends along the coastal area of the Gulf of Guinea in West Africa (Figure 14.1). The land cover in Central Africa is dominantly closed evergreen forest with a mosaic of forest and croplands on the northern and southern fringe of the subregions and throughout West Africa (Figure 14.2). The whole region is mapped as humid on the Global Aridity Index (Zomer, Xu and Trabucco, 2022). The soils of the equatorial forest subregion are dominated by Ferralsols in Central Africa and in the westernmost coastal areas of West Africa (Figure 14.3). Coastal areas of nations bordering the Gulf of Guinea have a complex mixture of Acrisols, Alisols and Lixisols (Figure 14.3). A large area of wetland soils (Gleysols) is associated with the vast peatlands of the Congo Basin (the "Cuvette Centrale") (Crezee *et al.*, 2022).

All non-wetland soils have agricultural production constraints (Jones *et al.,* 2013). The central African part of the subregion has a forest-based farming system with a limited area of cultivation (Dixon *et al*., 2001; OECD and FAO, 2022a) and the farmers principally practice shifting cultivation. The northern and southern fringes and coastal areas of West Africa have a tree crop farming system, producing industrial tree crops (cocoa, coffee, palm oil and rubber), with field crops such as yams and maize interplanted between the tree crops (Dixon *et al*., 2001).

The deciduous woodland subregion

The deciduous woodland subregion primarily occurs in two bands located north and south of the equatorial forest subregion (Figure 14.1). The land cover is principally open deciduous woodland mixed with cropland (Figure 14.2). The dominant farming system in the northern zone is a root crop system, changing to a cereal–root crop mixed system in its northern fringe (Dixon *et al*., 2001). Central and Eastern Africa changes from a root crop system adjacent to the equatorial forest subregion to a maize mixed land system in plateau and highland areas at higher elevations in Eastern Africa (Dixon *et al*., 2001). The most important root crops are yam and cassava, with legumes also being commonly grown. The maize mixed system includes maize, tobacco, cotton and livestock raising.

Annual precipitation in the deciduous woodland subregion is generally 1 000 to 1 500 mm and the subregion is generally classified as dry sub-humid in the Global Aridity Index (Zomer, Xu and Trabucco, 2022).

The soils of the northern band of the deciduous woodland subregion are a complex mixture of Plinthosols, Acrisols and Lixisols, reflecting the complex underlying rock types in the area. The eastern and central parts of the southern band have Ferrasols and an area of sandy Arenosols (Figure 14.3). In the eastern portion, the Rift Valley has steep slopes down to Lake Malawi and Lake Tanganyika with thin Leptosols in places and Luvisols and Lixisols on more stable surfaces. The eastern part has large areas of Lixisols, Acrisols and weakly developed Cambisols. Additionally, the small area of the subregion in Mozambique has rich alluvial soils associated with the Zambezi Delta and floodplain (Jones *et al*., 2013).

The savannah and highlands subregion

The savannah and highlands region is a mosaic of shrublands and grasslands, between the deciduous woodland subregion and the desert and semi-desert subregion (Figure 14.1). In this report, the highlands areas of sub-Saharan Africa are included with the savannah. The highlands covered in the report are the Ethiopian highlands, the highlands surrounding the East Africa Rift Valley and the highlands formed by the Drakensberg and Maloti mountain ranges in Southern Africa (Jones *et al.,* 2013).

The savannah subregion contains three main farming systems (Dixon *et al*., 2001; OECD and FAO, 2022a). The northern band is dominated by a cereal–root crop mixed system shifting to an agropastoral millet/sorghum system along the drier northern boundary. The cereal–root crop mixed system is also found in areas of Central and Southern Africa. A wide range of crops are grown, and intercropping is common. In an agropastoral millet/sorghum system, livestock and crops are of equal importance. The savannah subregion in Southern Africa includes parts of the maize mixed system and the large commercial and smallholder farming system in South Africa and southern Namibia. Both the large commercial and smallholder farms are principally mixed cereal livestock enterprises.

Throughout the savannah, precipitation is limited, ranging from 250 to 750 mm per year (Jones *et al*., 2013). The aridity index ranges from semi-arid to arid at the boundary with the desert and semi-desert subregion. The soils of the savannah reflect the drier conditions, with Arenosols and Regosols

dominating the northern band except for the large area of Vertisols in South Sudan (Figure 14.3). Arenosols dominate the southwestern area of the savannah, with a complex mixture of Luvisols, Cambisols and Leptosols in the southeastern and south central areas. The soils of the high plateau of South Africa include large areas of fertile Luvisols.

The highlands areas have two dominant farming systems (Dixon *et al.*, 2001; OECD and FAO, 2022a). The northern area of the Ethiopian highlands and the southern highlands in Lesotho and South Africa have a highland temperate mixed system with a wide variety of cereals, root crops, legumes and livestock. The southern area of the Ethiopian highlands and the highlands of Central Africa (found in Burundi, Rwanda and Uganda) are dominated by the highland perennial system, with perennial crops such as coffee, banana and plantain as well as cassava and cereals and a large livestock component. The highland perennial farming system has the highest population density of any of the farming systems.

The desert and semi-desert subregion

Sub-Saharan Africa has a large extent of desert and semi-desert lands including the Kalahari and Namid deserts in the south, the desert in the Somali Peninsula in the east and the continent-spanning desert and semi-desert at the northern boundary of the region (Jones *et al.*, 2013). The semi-arid and southern part of the hyper-arid class are referred to as the Sahel (Jones *et al.*, 2013). The area north of the Sahel is classed as hyper-arid (Hill *et al.*, 2018). The southern semi-desert areas are dominantly semi-arid, with arid land to the west and a small zone of hyper-arid land along the coast (Hill *et al.*, 2018). The soils of both the northern and southern zones are dominated by sandy Arenosols, shallow and rocky Leptosols and weakly-developed Cambisols (Figure 14.3). The soils in the Somali peninsula mainly have soils developed in a variety of marine sediments and are dominated by Gypsisols, Calcisols, and shallow Leptosols.

The major farming systems of the semi-arid zone are agropastoral millet and sorghum in the marginally wetter areas and pastoral in the drier parts (Dixon *et al.*, 2001; OECD and FAO, 2022a). The driest parts of the subregion have very limited pastoral activity with some scattered irrigation projects.

The islands of Africa subregion

A separate subregion was set up for the islands of Africa as they are often overlooked in regional summaries. The islands include Cabo Verde, Comoros, Madagascar, Mauritius, Mayotte, São Tomé and Príncipe and Seychelles.

Madagascar has a range of biomes almost as great as the whole continent. The three highland massifs that extend along the long axis of the island create a rain shadow in the western and southern part of the island with dry sub-humid and arid areas in the southwest corner (Hill *et al.*, 2018). Annual precipitation on the eastern coast is >3 500 mm, the southwest of the island is >1 000 mm and the central-northern part of Madagascar varies between 2 000 and 2 500 mm (Jones *et al.*, 2013).

The soils of the eastern and central, wetter areas are dominated by Ferrosols, with some Cambisols on younger land surfaces (Figure 14.3). A series of escarpments on the west coast give rise to a complex pattern of Vertisols, and Nitosols. Sandy Arensols, Regosols and Leptosols dominate in the drier southwest corner of the island. The dominant farming system in Madagascar is the rice–tree–crop system, with banana and coffee cultivation complimented with rice, maize, cassava and legumes, while cattle numbers are low (Dixon *et al.*, 2001).

The soils of Comoros, Mauritius, Seychelles and Réunion reflect the volcanic origin of the islands and the tropical and sub-tropical maritime climates (Jones *et al.*, 2013). In Comoros, much of the population works in agriculture, with banana and cassava being the major crops. The input levels of

fertilizer from all sources and pesticides are low and mechanization is limited (FAO, 2023b). Mauritius is dominated by sugarcane cultivation (98 percent of all cultivation and covering 55 percent of the area of the island). Other crops include fruit, tea and vegetables (Sumner *et al.*, 2021). Sugar cane is also the dominant crop in Réunion and is grown in coastal areas (Billen *et al.*, 2024). Areas of grassland and forage support livestock production in the western and south-central highland areas of the island.

The Atlantic islands of Cabo Verde have a wide variety of land conditions across the islands. Maize, beans and citrus fruits are important rainfed crops and sugar cane is a dominant irrigated crop (Monteiro *et al.*, 2020). The agriculture sector in São Tomé and Príncipe is small, and cocoa, coffee and palm oil production are the most economically-important agricultural activities (IFAD, 2024).

14.1.2 Summary of state from 2015 Status of the world's soil resources report

The 2015 SWSR report (FAO and ITPS, 2015) provided an assessment of sub-Saharan Africa. Erosion was judged to be the most serious threat to sustainable soil management (SSM), affecting 77 percent of the area. Water erosion was the dominant form of degradation and occupied nearly 46 percent of the sub-Saharan Africa area.

Soil organic matter loss was identified as the second major soil threat. Following land conversion from forest or woodland to agriculture, soils frequently continue to lose carbon (C) over time. The annual burning of grazing pasture during the dry season in most subhumid and semi-arid locations depletes SOM (which is emitted as CO_2) and was considered to be a major source of SOM loss.

Soil nutrient depletion was recognized as the third major soil threat in sub-Saharan Africa. Soils throughout much of sub-Saharan Africa were heavily weathered and thus naturally poor in soil nutrients, depleting by drainage during the raining season. Natural replenishment of nutrients during fallow periods was insufficient to maintain soil productivity over the long term due to increasing constraints on land and shorter fallow periods. Soil degradation was found to be responsible for the insufficient nutrient replacement in agricultural systems on land with low to moderate potential.

Soil biodiversity loss was ranked as the fourth most serious soil threat in sub-Saharan Africa in the 2015 SWSR report. Biodiversity loss occurs in several ways, with the most important factors affecting soil biodiversity including habitat fragmentation, resource availability (the amount and quality of nutrients and energy sources), temporal heterogeneity (e.g. seasonal effects), spatial heterogeneity (spatial differences in the soil), climate variability and interactions within the biotic community.

Soil contamination and pollution were highlighted as another threat after soil biodiversity loss. In most sub-Saharan Africa countries, pesticides and synthetic fertilizers had a harmful impact on the ecosystem. However, soil pollution resulting from pesticide usage in sub-Saharan Africa was found to be less problematic compared to other parts of the world, due to the generally low levels of application. As the use of fertilizers, pesticides and herbicides increased to raise productivity, measures would be required to ensure that damaging effects on soils and the wider environment were avoided.

The 2015 SWSR report also mentioned that extremely acidic soils in the region were found only in a small area around the Niger Delta and sporadically along the coastal plains of West Africa. The remaining acid soils occupied around 15 percent of the continent and were mainly found in the humid parts of semi-arid zones and in subhumid regions. Many Acrisols and some Lixisols had acid surface and subsurface horizons which, when combined with water stress conditions, made such soils extremely difficult to manage under low input conditions. In West Africa, the annual addition of Saharan dust by the Harmattan winds raised the pH of the surface horizons. The problem was

therefore less acute, although subsoil acidity remained. Another region of acidic soils lay south of the Tropic of Capricorn and included parts of South Africa where acidity was one of the main factors limiting production.

Salinization and sodification were widespread soil threats in arid and semi-arid areas due to rainfall being insufficient to drain salts and excess sodium ions (Na^+) from the rhizosphere. Sub-Saharan Africa had around 32 375 000 ha of salt-affected soils.

Waterlogging, compaction, crusting and soil surface sealing because of urban expansion were also recognized as threats to SSM in sub-Saharan Africa.

The 2015 SWSR report highlighted soil erosion, C loss and soil nutrient depletion as the most critical soil problems in sub-Saharan Africa. These threats were found to be interconnected and were well developed in the report. However, the report failed to break down the threats to soil in each stratified zone, thereby reducing the effectiveness of the control and actions to combat the soil threats. The current report will therefore consider subregions in order to give a more nuanced view of the status of the threats to soils in sub-Saharan Africa.



14.2 Changes in the drivers of threats to soil functions

14.2.1 Population growth

The most important indirect driver of threats to soil functions is population growth. The African continent has been at the epicentre of rapid population growth, recording the largest relative increase globally. Population growth in sub-Saharan Africa is projected to be the highest globally between 2022 and 2050 (UN DESA, 2022). With a population of 283 million in 1960, the continent has experienced a significant surge, reaching over 1.2 billion in 2024: an unprecedented five-fold increase. Experts predict that Africa's population is set to rise by an additional 950 million individuals, reaching 2.1 billion by 2050.

Eight countries are driving over half of the projected global population surge between 2022 and 2050, with five being in Africa. These are the Democratic Republic of the Congo, Egypt, Ethiopia, Nigeria, and the United Republic of Tanzania, and are predicted to have a population growth rate of between 2 and 3 percent per year over the 2022–2050 period (UN DESA, 2023). In 1990, Nigeria was the world's tenth most populous nation, with a population of 94 million, and is predicted to be the fourth most populous with a population of 375 million by 2050. All of the countries of sub-Saharan Africa are projected to have growing populations through to 2054 (UN DESA, 2024).

Land-use change

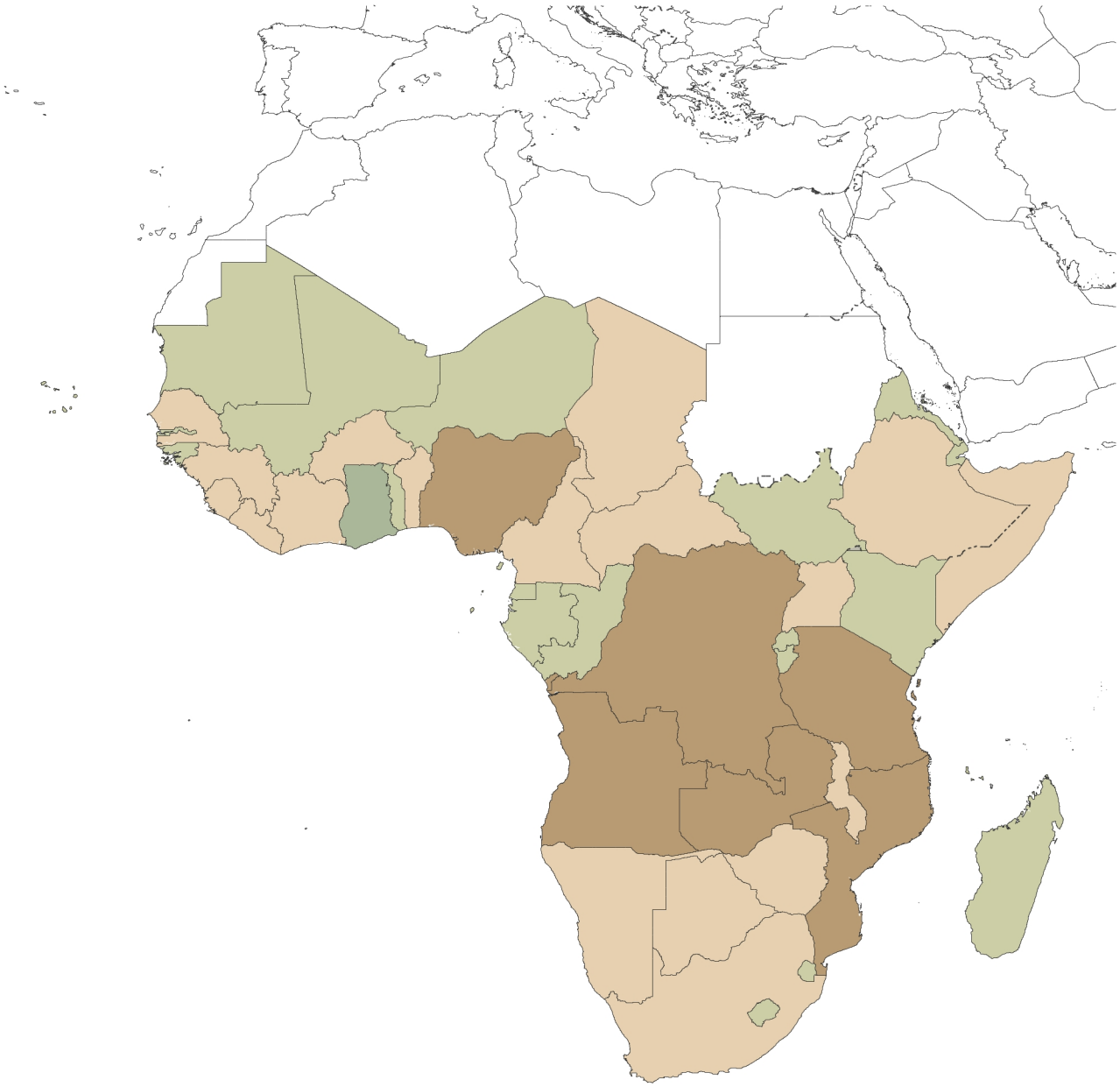
Deforestation and cropland expansion

At a broad level, the region has experienced a substantial increase in cropland area and a parallel decrease in forested area (or deforestation). Deforestation is a major driver of harmful soil changes, generally leading to decreases in SOM and soil biodiversity as well as increases in soil erosion. Deforestation also fundamentally changes the soil water balance for the deforested land, with negative consequences for surface water and groundwater.

Deforestation continues to be widespread in sub-Saharan Africa (Figure 14.4) (FAO, 2024a). Of the 50 sub-Saharan countries, 37 have experienced deforestation in the 2015–2022 period, with seven of these countries losing over 10 percent of their forestland (Benin, Chad, Côte d'Ivoire, the Gambia, Malawi, Mauritania and Uganda).

According to FAO (2024a), the countries with the greatest absolute loss of forestland in the 2015–2022 period include Angola (with a net loss of 3.9 million ha), the Democratic Republic of Congo (with a net loss of 7.7 million ha), Mozambique (with a net loss of 1.7 million ha), the United Republic of Tanzania (with a net loss of 3.3 million ha), and Zambia (with a net loss of 1.3 million ha), all of which rank in the ten highest globally. Not surprisingly, the 26.7 million ha net loss of forestland in the 2015–2022 period (FAO, 2024a) is the highest of any region. The loss of forests has environmental and social impacts, including soil erosion, loss of biodiversity and the displacement of Indigenous Peoples.

Figure 14.4. Country map of change in forestland area from 2015 to 2022

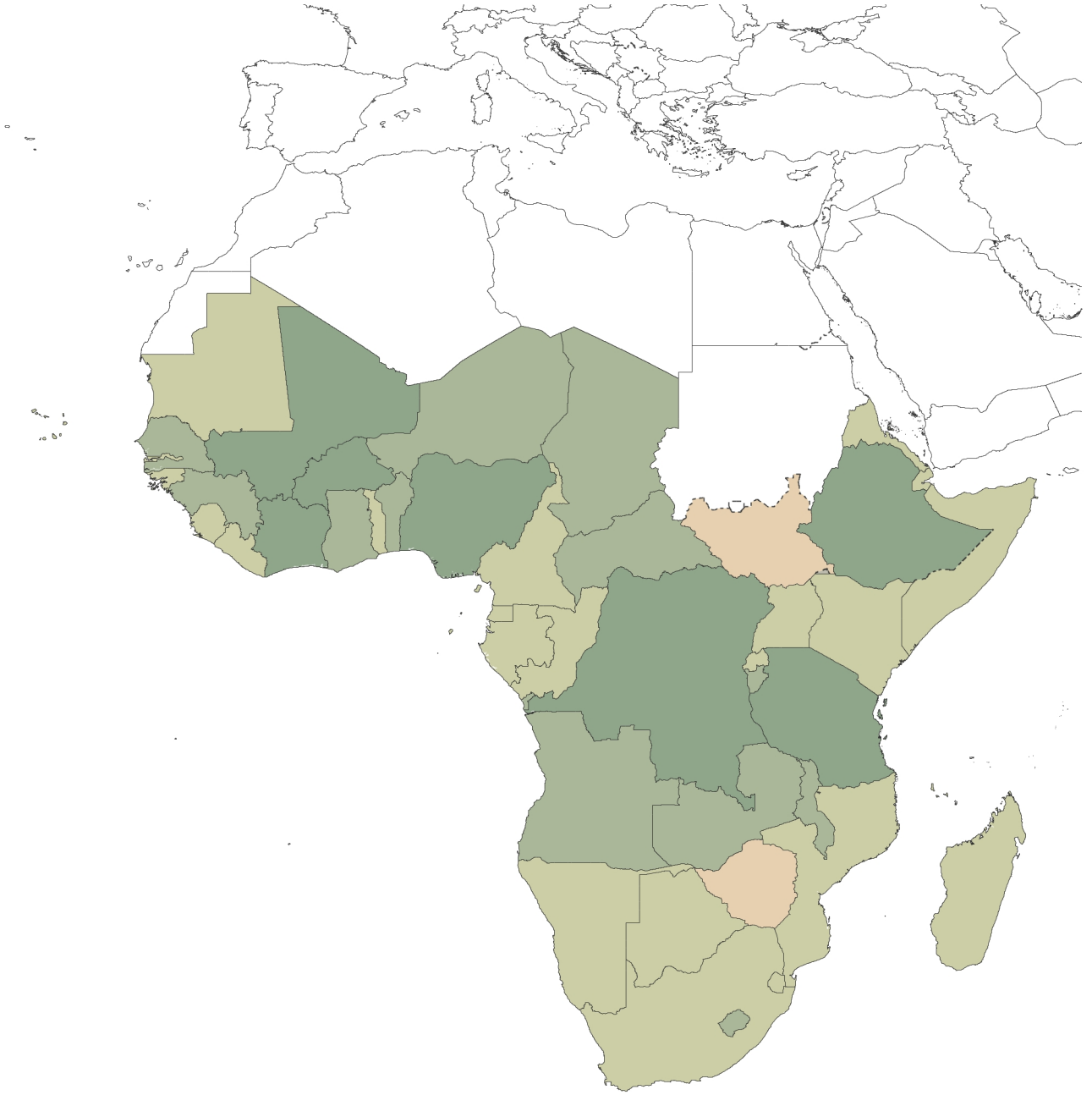


Refer to the disclaimer on page ii for the names and boundaries used in this map. Final boundary between the Republic of Sudan and the Republic of South Sudan has not yet been determined. Final status of the Abyei area is not yet determined.

◊ *Source:* **FAO**. 2024. *FAOSTAT: Land Use*. [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

The causes of deforestation are examined in greater detail in the subregional summaries, but, in many cases, the deforested land is converted to cropland. The absolute gain of cropland for the 2015–2022 has been greater than 1 million ha in Burkina Faso, Côte d'Ivoire, the Democratic Republic of the Congo, Ethiopia, Mali, Nigeria, and the United Republic of Tanzania (Figure 14.5).

Figure 14.5. Country map of change in cropland area from 2015 to 2022



Change in cropland from 2015 to 2022

- >1 million ha gain
- 100 000 ha gain to 1 million ha gain
- 100 000 ha loss to 100 000 ha gain
- 100 000 ha loss to 1 million ha loss

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◊ *Source:* **FAO**. 2024. *FAOSTAT: Land Use*. [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

Popatov *et al.* (2021) used time series of satellite data to track increases in cropland area between 2003 and 2019 and reported that Africa had the greatest net gain of cropland (an increase of 47 million ha, or 53.2 million ha, depending on the estimation method) of any region. It was estimated that 79 percent of the cropland gain in this period came from the conversion of natural vegetation or tree plantations, with a further 17 percent from the replacement of pastures, or the recultivation of abandoned agricultural fields. A cropland gain of 79 percent from natural vegetation is the highest of the seven global regions. In comparison, substantial losses of cropland occurred in the two island nations of Mayotte and Réunion. On the continent, losses of cropland occurred in South Sudan and Zimbabwe.

Permanent pastures and meadows

Many countries report no change in area of permanent pastures and meadows from 2015 to 2022 (Figure 14.6) (FAO, 2024a). Substantial losses of permanent pastures and meadows occurred in Ghana, Guinea-Bissau, Mayotte and Rwanda. Gains in permanent pastures and meadows occurred in Equatorial Guinea and Réunion. In all, Mayotte experienced a net loss of 71 percent of its agricultural land between 2015 and 2022 (FAO, 2024a).

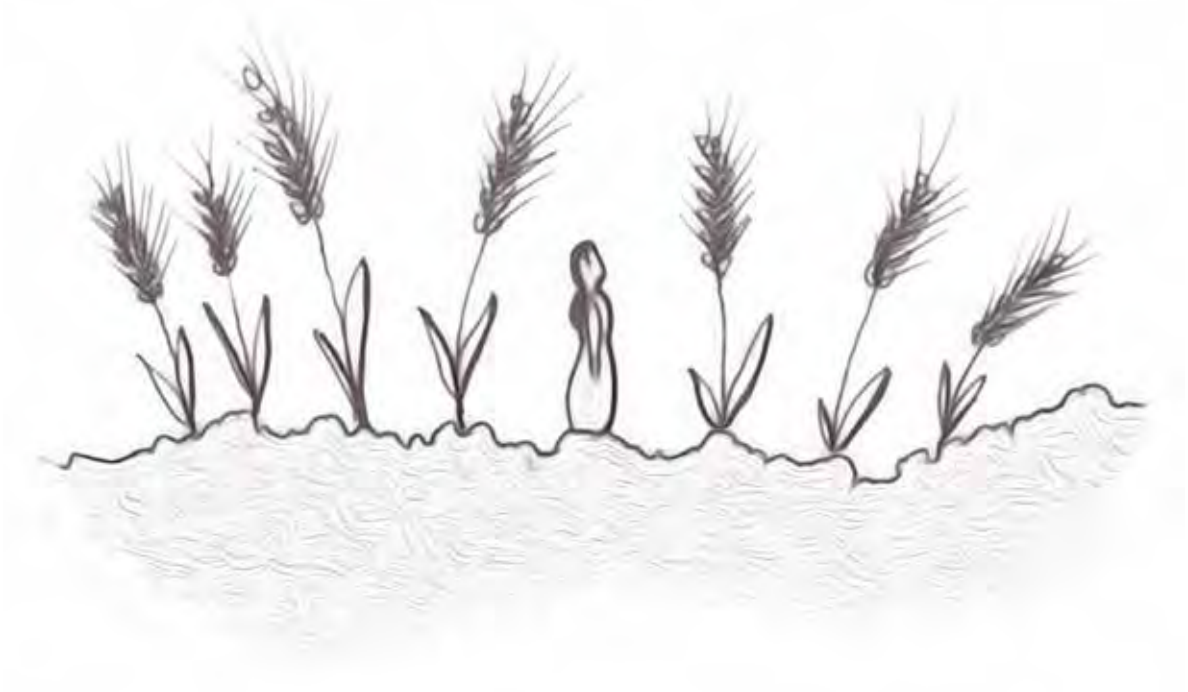
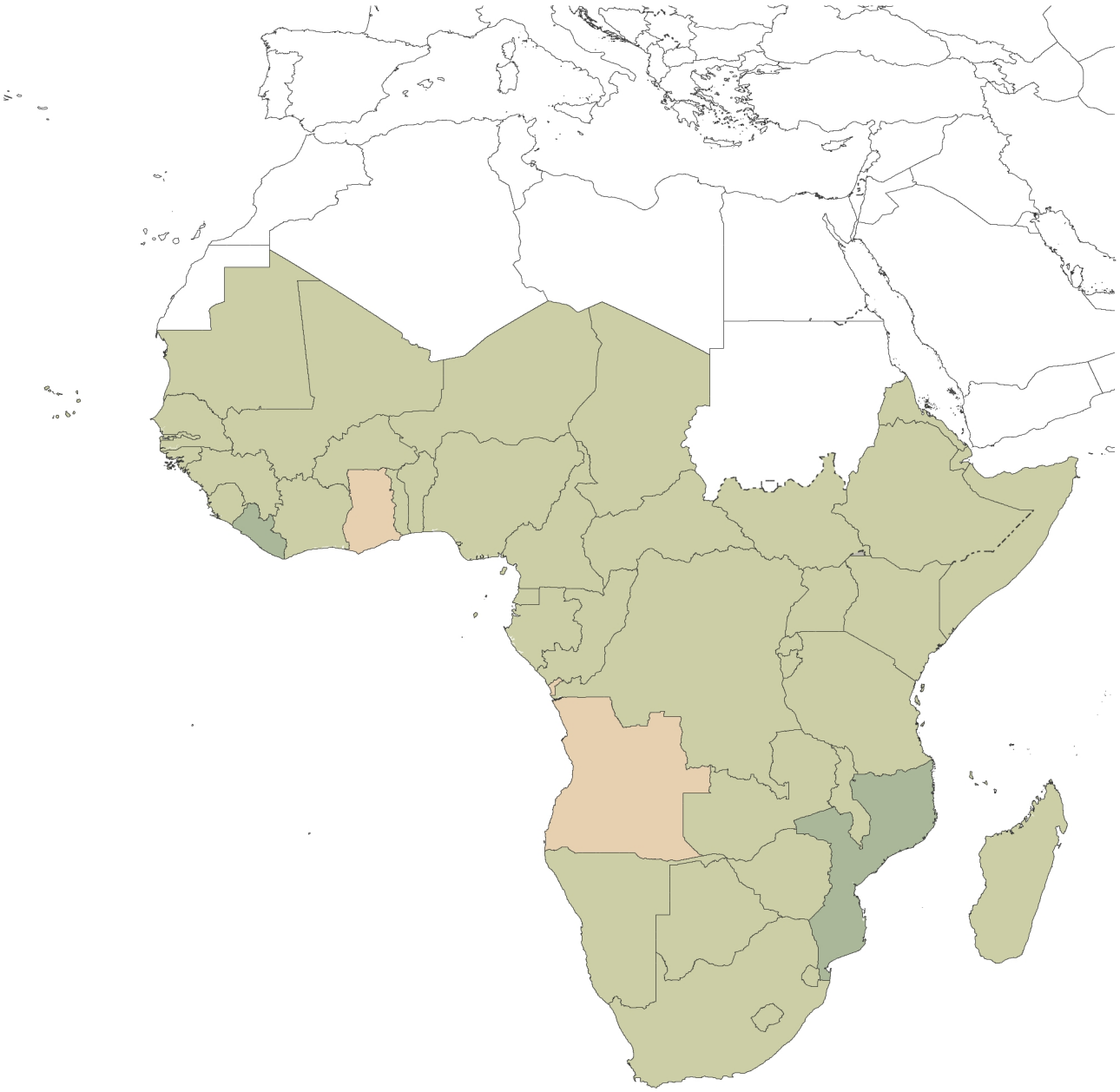


Figure 14.6. Country map of change in permanent pastures and meadows area from 2015 to 2022



Change in permanent meadows and pastures from 2015 to 2022

100 000 ha gain to 1 million ha gain | 100 000 ha loss to 100 000 ha gain | 100 000 ha loss to 1 million ha loss | No data

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◊ *Source:* **FAO**. 2024. *FAOSTAT: Land Use*. [Accessed on 5 November 2024]. https://www.fao.org/faostat/en/#data/RL. Licence: CC-BY-4.0.

In sub-Saharan Africa, natural grazing lands are important in the livelihoods of many communities for rearing livestock (Turner and Schlecht, 2019). However, grazing lands in sub-Saharan Africa are being lost for several reasons, including soil degradation due to overgrazing, fire and subsequent erosion, as well as conversion to agriculture and urban land. Many traditional grazing practices, which have been followed for hundreds of years to preserve grazing land resources in typically resource-poor locations, are under threat (Milne *et al*., 2016). As such, there is an urgent need for sustainable management of grazing lands in the region to ensure the continuing benefits of grazing lands.

Wetlands extent and change

Estimates of wetland area for countries in sub-Saharan Africa have been made by Li *et al.* (2022) and Fluet-Chouinard *et al.* (2022), but the area estimated for most countries is quite different between the two studies, in part because Li *et al.* (2022) included surface water in their estimates. The authors estimated that inland swamps had the largest area of the wetland classes in Africa with a total area estimated of 411 500 km^2, while the swamps of the Congo basin made up the largest wetland in sub-Saharan Africa. Overall, the Democratic Republic of the Congo had an estimated wetland area of 361 400 km^2, or nearly a quarter of the total wetland area in Africa (Li *et al*., 2022). Fluet-Chouinard *et al.* (2022) estimated the area of wetlands in the Democratic Republic of Congo as less (197 000 km^2), decreasing by 2 percent since 1700. The countries with the greatest wetland loss were Madagascar (a loss of 35 percent), Uganda (29 percent), and Sierra Leone (22 percent). The losses in Angola, Ghana, Guinea, Nigeria and Zimbabwe were between 10 and 20 percent (Fluet-Chouinard *et al.*, 2022).

Conservation agriculture

Kassam, Friedrich and Derpsch (2022) estimated that only 1.1 percent of cropland in Africa was under conservation agriculture management in 2018/2019. Of the countries of sub-Saharan Africa, the countries with the greatest area under conservation agriculture were South Africa (1 608 000 ha), Zambia (553 000 ha), Ghana (235 000 ha) and Malawi (211 000 ha). For South Africa, the area under conservation agriculture represents about 13 percent of the cropland area.

Agroforestry

In the *Global Forest Resources Assessment* (FAO, 2020), agroforestry is defined by lands with a tree canopy cover having more than 10 percent of trees able to reach a height of 5 m at maturity, together with temporary agricultural crops and/or pastures/animals. The six reporting countries in Western and Central Africa in 2020 had a total area in agroforestry of 11 234 000 ha, the second highest of any global region (FAO, 2020). The five reporting countries in Eastern and Southern Africa had the third-highest total (1 385 000 ha).

Nutrient management

Sub-Saharan Africa generally has low levels of nutrient inputs from all sources and many countries have nutrient imbalances for N (Figure 14.7), P (Figure 14.8) and K (Figure 14.9) indicative of insufficient nutrition for optimum plant growth or outright nutrient mining. Of the countries in sub-Saharan Africa, only Djibouti, Mauritius and South Africa show consistent evidence of sufficient nutrient inputs. The national fertilizer numbers may also actually underestimate the nutrient situation for smallholder farmers in sub-Saharan Africa as fertilizers are dominantly used on large commercial farms and for horticultural crops (Gatien *et al.,* 2023).

Although only seven sub-Saharan countries have negative N balances (Figure 14.7) a further seven (Botswana, Cabo Verde, the Central African Republic, Eritrea, the Gambia, Lesotho, Namibia, and Somalia) have average annual N outputs in agricultural products of less than 10 kg N/ha, which are even low by global standards (EU Nitrogen Expert Panel, 2015).

Figure 14.7. Country map of average values (2017 to 2021) for nitrogen use



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◊ *Source:* **FAO**. 2024. *FAOSTAT: Cropland Nutrient Balance*. [Accessed 10 October 2024] https://www.fao.org/faostat/en/#data/ESB. Licence: CC-BY-4.0.

Soil mining of P is very common in sub-Saharan Africa, with 26 countries showing both a negative average annual P balance (Figure 14.8) (FAO, 2024b) and long-term depletion of soil P reserves (1961 to 2019) (Zou, Zhang and Davidson, 2022) between 2017 to 2021.

Figure 14.8. Country map of average values (2017 to 2021) for phosphorus use



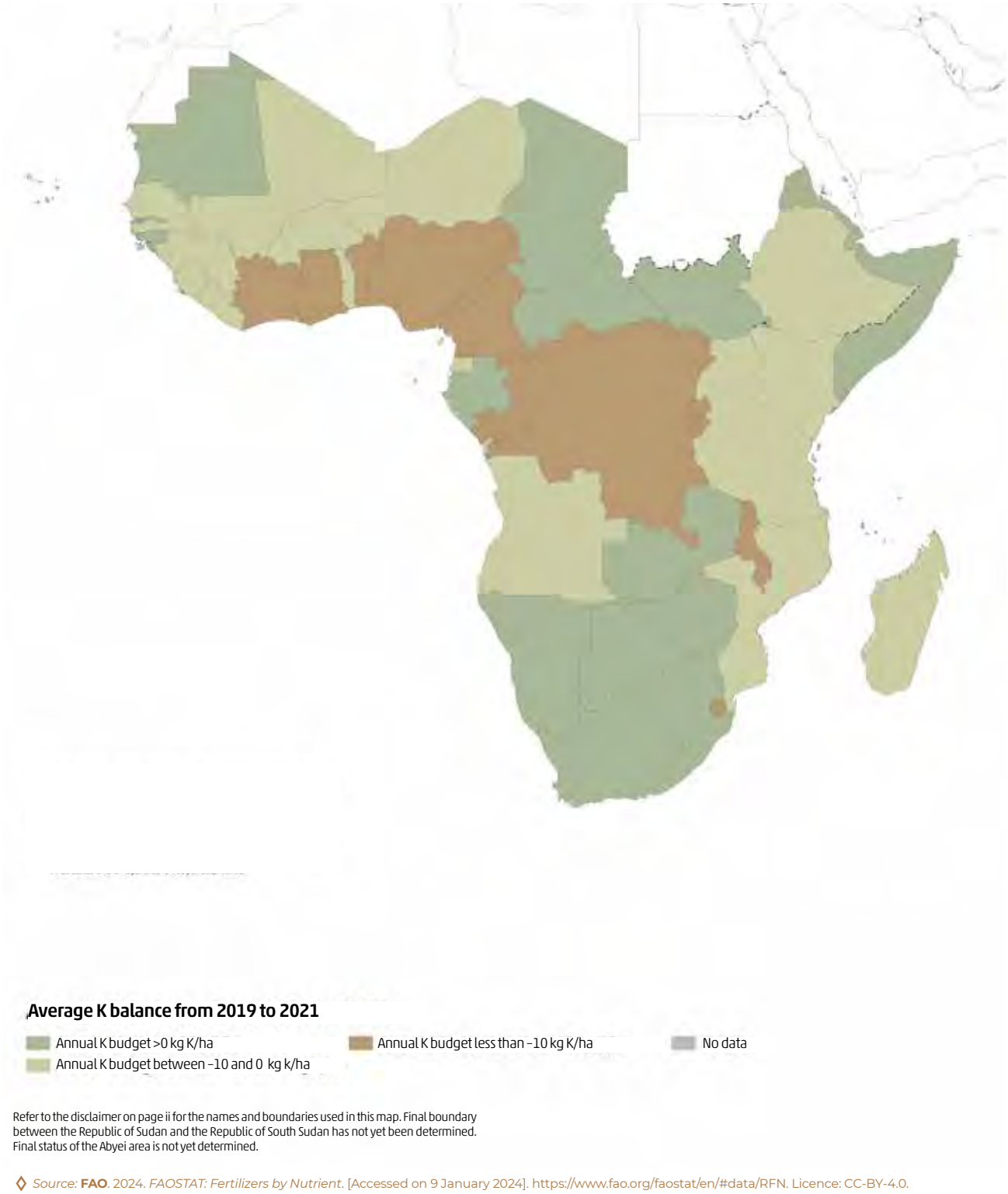
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◊ *Source:* **FAO**. 2024. *FAOSTAT: Fertilizers by Nutrient*. [Accessed on 9 January 2024]. https://www.fao.org/faostat/en/#data/RFN. Licence: CC-BY-4.0.

◊ **Zou, T., Zhang, X. & Davidson, E.A.** 2022. Global trends of cropland phosphorus use and sustainability challenges. *Nature*, 611(7934): 81–87. https://doi.org/10.1038/s41586-022-05220-z

Average annual K budgets also show moderate or high deficits for many countries (Figure 14.9).

Figure 14.9. Country map of average values (2017 to 2021) for potassium use



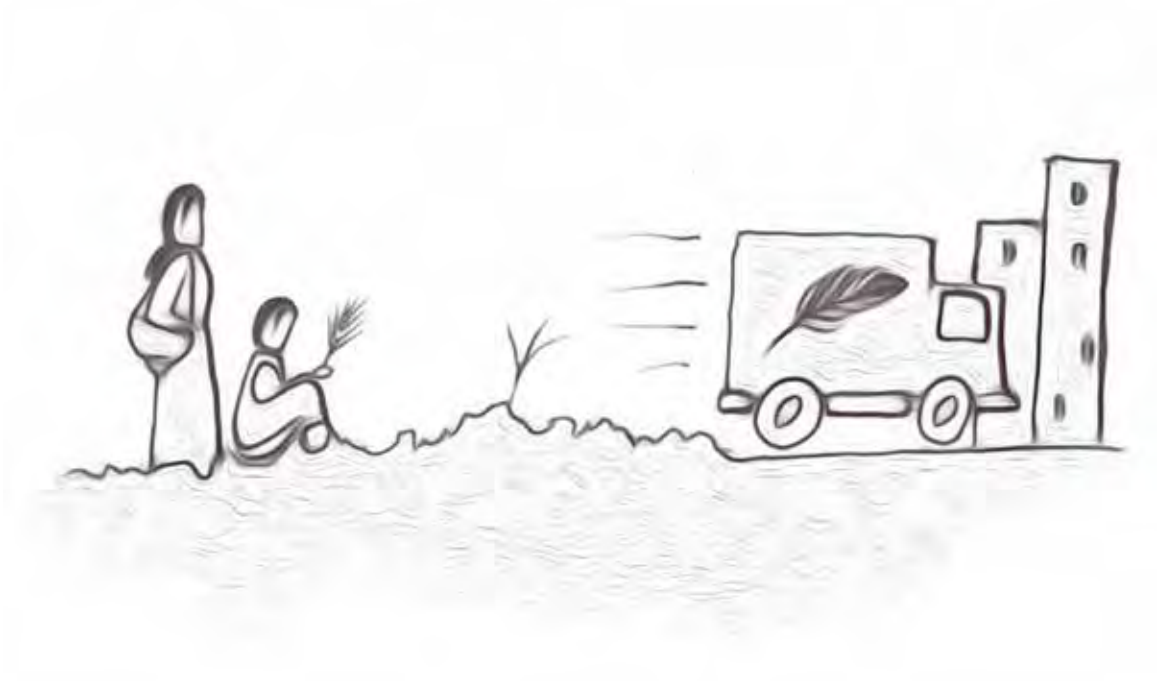
Source: **FAO**. 2024. *FAOSTAT: Fertilizers by Nutrient*. [Accessed on 9 January 2024]. https://www.fao.org/faostat/en/#data/RFN. Licence: CC-BY-4.0.

All seven countries of sub-Saharan Africa (Comoros, the Democratic Republic of the Congo, Côte d'Ivoire, Equatorial Guinea, Eswatini, Ghana, and the Niger) show evidence for mining of all three major nutrients (FAO, 2024a).

Many African soils are also deficient in secondary nutrients such as Ca, Mg and S as well as a range of micronutrients (Sileshi *et al.*, 2022). Many deficiencies are associated with the ancient, highly weathered soils or sandy soils in the region (Jones *et al.*, 2013).

Urban expansion

In 2018, sub-Saharan Africa had the lowest percentage of its population (42.5 percent) living in urban areas out of any of the global regions (UN DESA, 2019). As the population of sub-Saharan Africa grows, there will be corresponding growth in the urban population and the urban areas of the region are projected to experience the highest growth rates between 2018 and 2030. Expansion of urban land is projected to displace food production. Van Vliet, Eitelberg and Verburg (2017) predicted that the area of urban land in sub-Saharan Africa would increase from 11.3 million ha in 2000 to 68.4 million ha in 2040, with an estimated decrease of 3.6 percent in food production due to urbanization.



14.3 Regional state and trend of threats to soil functions for the equatorial forest and deciduous woodland subregions

14.3.1 Introduction

The equatorial forest and deciduous woodlands subregions are characterized by a high annual rainfall of above 1 200 mm per year and the soils exhibit features such as low pH and nutrient levels, including high amounts of exchangeable Al and H. Furthermore, the soils are highly weathered with a dominance of kaolinitic 1:1 clay minerals and presence of Fe/Al hydroxides (Mangalassery, Kalaivanan and Philip, 2019). The excess leaching of basic cations due to the abundant rainfall contributes to the specific characteristics of these tropical soils (Chapoto, Chabala and Lungu, 2016). Low natural nutrient levels coupled with decades of nutrient mining has rendered the soils marginally fertile, thereby posing a challenge for productive use (Masso *et al*., 2017).

Deforestation in the equatorial forest and deciduous woodland subregions of Africa has occurred at a substantial rate. FAO (2024) states that annual rates of deforestation in the 2015–2020 period were 4.41 million ha in Africa, with much of this occurring in those subregions. The main causes of deforestation in the equatorial forests subregion are agriculture, logging and mining activities, coupled with population growth and urbanization (FAO, 2020). More than 60 percent of forest disturbances in Gabon are due to selective logging and more than 90 percent of disturbances in the Democratic Republic of the Congo and the Central African Republic are due to small-scale agriculture (Tyukavina *et al*., 2018).

14.3.2 Soil erosion in the equatorial forest and deciduous woodland subregions

Soil erosion is a serious threat in the equatorial forest subregion of Africa (Labrière *et al.,* 2015). The main driving factors governing soil erosion in this area are changes in land use, vegetation cover and agricultural management. Soil erosion by water, wind, tillage, mass movement and root crop harvesting threatens food and fuel supplies and contributes to climate change (Wilken *et al*., 2020). Soil erosion causes the loss of topsoil by natural forces (water or wind), reducing the quantity and quality of arable land. Unmitigated, soil erosion can result in crop yield decline, negatively impact community resiliency and livelihoods, and even alter ecosystems by reducing biodiversity.

Soil erosion by both water and wind (under natural vegetation) is expected to be low. This is due to dense vegetation and forest litter that protects the surface from rainfall and wind impact and, due to aggregation, the erodibility of the often well-drained soils is relatively low. Nevertheless, the intensity and volume of rainfall are high, with the erosive force of rainfall is the strongest in the humid tropics (Ambassa-Kiki and Nill, 1999; Vrieling, Sterk and Jong, 2010).

Global mapping of water erosion by Borrelli *et al.* (2017) predicted that rates would be low (0 to 1 t/ha annually) in the Central African areas of the equatorial forest subregion, but that higher rates (5 to 20 t/ha annually) would occur in small areas of the equatorial forest in West Africa. Modelling by Carr *et al.* (2020) predicted much higher rates of water erosion (50 to 100 t/ha annually) under maize production in Liberia and Sierra Leone, but otherwise predicted similarly low rates for other parts of the equatorial forest subregion.

Soil erosion can be severe in West Africa, especially where the natural vegetation cover has been removed to introduce food crop farming. Borelli *et al.* (2017) predicted higher water erosion in the

deciduous woodland subregion in West Africa than in the equatorial forest subregion. Hotspots (>20 t/ha annually) of water erosion occur in Nigeria and Togo. Predicted rates were lower (0 to 3 t/ha annually) for areas of the deciduous woodland subregion in Central Africa and in Angola and Zambia.

Global mapping of dust emissions by Ginoux *et al*. (2012) and Chappell *et al.* (2019) did not show significant emissions from either the equatorial forest or the deciduous woodland subregions.

Harmattan dust deposition leads to a net accumulation of dust particles in the soils lying south of the 200 mm rainfall isohyets in temperate Africa. Convective storms cause a redistribution and homogeneity of the potential dust fraction across Sahelian soils at the onset of the rainy season (Herrmann,1996).

According to Herrmann (1996), annual dust deposition showed a gradient from 156 g/m^2 at the most northern site of Ouallam in Niger, to only 15 g/m^2 at the coastal site of Houéto in Benin. Dust proprieties showed seasonal variation but were constant over the year. The overall nutrient content of the dust increased with the distance to the source areas. Beside soil-borne minerals, ash generated by bush fires contribute to atmospheric deposition in the deciduous woodland zone.

Tillage is used to prepare the soil for seeding, improve soil physical properties, enhance rapid release of nutrients and control weeds (Cornelio and Rao, 2011). Tillage operations move soil mechanically in the tractive force direction and on the hillslope, with the movement enhanced by gravity during downslope tillage. Tillage erosion occurs by the action of agricultural tools (Zhang *et al.,* 2018, 2021). During animal-drawn plough and manual hoeing tillage, soil is typically turned downslope to minimize energy requirements, and thus only downslope soil displacement occurs. According to data summarized by Zhang *et al*. (2018), typical tillage transport coefficients (in kg/m for each tillage pass) for hoeing are approximately one-quarter of those for chisel ploughs and mouldboard ploughs. Erosion by tillage depends on soil physical parameters and variables such as speed, depth, direction and characteristics of soil tillage (Zhang *et al.,* 2018, 2021).

Tillage in the traditional farming systems of the West African semi-arid tropics (WASAT) is carried out by hand using a short-handled hoe. Following the initial shallow (5–7 cm) cultivation, seeding may take place either on level land, or on ridges constructed during or after tillage (Raison, 1989; Traore and Birhanu, 2019). The advantages and disadvantages of ridging have been reviewed by Kowal and Stockinger (1973). “Tying” of the ridges may take place as a soil and water conservation measure (Hulugalle, 1990). More recently, the increasing use of animal traction has resulted in more intensive tillage operations by small farmers (Starkey and Faye, 1990). Mechanized tillage systems (commonly disc ploughing to 20–30 cm depth) are reported to result in greater yields in comparison with either no tillage or hand tillage (Klaij and Hoogmoed, 1989). Yield increases are associated with increases in soil porosity and water infiltration at the beginning of the rainy season, following ploughing. The depth of water penetration and storage are greater with mechanical tillage to a depth of 20–30 cm than with either hand tillage or no tillage and with negligible residues (Ahmed, 1990). The wetter soil, both in the topsoil and subsoil, also results in a deeper and more extensive root system.

It has been estimated that 56 percent of the coastline in Benin, Côte d’Ivoire, Senegal and Togo is subject to an average coastal erosion of 1.8 m per year (Croitoru, Miranda and Sarraf, 2019). Erosion rates are especially high in Benin with an average loss of 4 m per year on 65 percent of the coast, thus land is being lost to the sea. Erosion is due to both natural and anthropogenic factors and has an annual cost of nearly USD 1 billion to the four countries. The coastline is dynamic, with some areas having no erosion at all, some experiencing land loss and others land gains. Beyond the economic cost, coastal degradation takes lives and destroys livelihoods (Croitoru, Miranda and Sarraf, 2019). Coastal communities are impacted by flooding due to extreme local rainfall and rivers overtopping their banks, causing damage to farmland, houses and infrastructure as well as to ecosystems such as beaches and mangroves.

14.3.3 Soil carbon loss in the equatorial forest and deciduous woodland subregions

A meta-analysis by Don, Schumacher and Freibauer (2011) for tropical areas found average relative and absolute losses of SOC due to deforestation ranging from 9 to 32 percent (Table 14.1). The highest losses were associated with conversion of primary forest to perennial crops and the lowest losses with conversion to secondary forest.

Table 14.1. Summary of soil organic carbon loss associated with land-use change for tropical forests

Land-use change type	Absolute SOC stock change (tonnes C/ha)	Relative SOC stock change (%)	Mean sampling depth (cm)	Mean time after LUC or time period (years)	Number of studies
Primary forest to grassland	–12.6	–12	36	25	93
Primary forest to cropland	–20.1	–25	36	28	56
Primary forest to perennial crops	–32.0	–30	48	49	20
Primary forest to secondary forest	–12.6	–9	39	28	71

◊ *Source:* **Don, A., Schumacher, J. & Freibauer, A.** 2011. Impact of tropical land-use change on soil organic carbon stocks - a meta-analysis. *Global Change Biology*, 17(4): 1658–1670. https://doi.org/10.1111/j.1365-2486.2010.02336.x

Some measures have been introduced to reduce soil SOC loss from land conversion. For instance, sustainability criteria for palm oil production guide new planting toward non-forestland cover on mineral soil, avoiding C debts caused by forest and peat conversion (Khasanah *et al.*, 2015). Effects on soil C stock of land-use change from forest and non-forest to oil palm on mineral soils include initial decline and subsequent recovery, but modelling efforts and life-cycle accounting are constrained by the lack of comprehensive data sets.

Many elements of SOC dynamics in tropical forests remain poorly understood. For example, the transfer of organic C from topsoil horizons to deeper horizons and the water table is still little documented in equatorial environments, despite the high primary productivity of the evergreen forest (Merdy *et al.,* 2021).

The vast peatlands of the Cuvette Centrale in the central Congo Basin span the Congo and the Democratic Republic of Congo (Crezee *et al.,* 2022). The extent of the peatlands has been recently estimated at approximately 167 600 km^2 with estimated C stores of approximately 29 billion tonnes C (Crezee *et al.,* 2022). The peatland complex accounts for 36 percent of the world's tropical peatland area. The area is famous for its rich animal life with substantial numbers of lowland gorilla (*Gorilla gorilla gorilla*), forest elephant (*Loxodonta cyclotis*), chimpanzee (*Pan troglodytes*) and bonobo (*Pan paniscus*) (Dargie *et al.,* 2019). Crezee *et al. (*2022) estimated that just 8 percent of the peat stocks lay within national level-protected areas and that logging, mining or palm oil concessions together overlaid 26 percent of peat stocks, while hydrocarbon concessions covered almost the whole peat complex. Given the vast C stores of this peatland area, further conservation measures are needed.

Extensive fires, slashing and burning, fuel extraction and drainage threaten peatlands (Lourenco, Woodborne and Fitchett, 2023). A total of 11.3 percent of land surface in Angola – including 46 percent of peatlands – was burned within a 20-year period (Lourenco, Woodborne and Fitchett, 2023). More than 64 percent of all wetlands in the eastern Democratic Republic of the Congo have been lost due to overexploitation (Chuma *et al*., 2021). If these rates continue, sub-Saharan African wetlands could soon lose their ecological significance (Kim *et al*., 2016).

The calcium carbonate content is very low in equatorial forest subregion soils (Jones *et al.*, 2013) and hence losses of soil inorganic carbon (SIC) are negligible for this subregion.

14.3.4 Soil biodiversity loss in the equatorial forest and deciduous woodland subregions

Soil biodiversity is crucial in maintaining ecosystem health, nutrient cycling, and overall sustainability (FAO and ITPS, 2020). The equatorial forest in sub-Saharan Africa is recognized as one of the most biodiverse areas on Earth, supporting a wide range of plant and animal species. For example, Myers *et al*. (2000) identified the West African coastal forest area of the equatorial forest as one of 25 hotspots for the conservation of biodiversity. The status of soil biodiversity in African equatorial forests varies across different regions and ecosystems. As these diverse ecosystems face various anthropogenic and natural pressures, understanding the dynamics of soil biodiversity change in this region is essential for conservation and sustainable land management practices.

The main drivers of soil biodiversity change in the region are land-use change and deforestation. Equatorial forest deforestation and conversion of forests for agricultural purposes in the Congo Basin led to a decline in soil microbial abundance and diversity, ultimately impacting soil fertility and ecosystem resilience (Kouamé *et al*., 2020). A meta-analysis of soil biodiversity patterns across equatorial Africa's diverse forestlands underscored the vulnerability of soil biodiversity to fragmentation, degradation, and land-use conversion (Winowiecki *et al*., 2021). Sustainable agroforestry practices, as an alternative to shifting agriculture, may offer a solution for preserving soil biodiversity while supporting agricultural productivity (Norgrove and Hauser, 2016). Although the factors contributing to the loss of soil biodiversity are known, there is a substantial lack of soil biodiversity data and datasets on taxa such as microarthropods and nematodes in the equatorial forest subregion (Bahram *et al*., 2018; Cameron *et al.,* 2018).

Modelling predicted that the soil microbial biodiversity of countries such as Kenya will be negatively affected by increased temperatures and decreased precipitation, while the fungal biodiversity of Benin will benefit from an increase in annual precipitation (Cowan *et al.,* 2022).

14.3.5 Nutrient mismanagement in the equatorial forest and deciduous woodland subregions

The nutrient balances for the countries of the equatorial forest subregion all indicate significant nutrient mining (Figure 14.7, Figure 14.8 and Figure 14.9) (FAO, 2024b; Zou, Zhang and Davidson, 2022; EU Nitrogen Expert Panel, 2015). Nitrogen outputs through crops are low (<20 kg/ha annually) in all countries except Cameroon. The N balance is negative in the Democratic Republic of the Congo and Equatorial Guinea. The P balance is below or only slightly above zero, and all the countries have been assessed to have negative P residuals, indicating mining of soil P. Potassium balances are also below zero or close to zero throughout the subregion. All these indicators suggest nutrient mining of soil and significant yield gaps for agricultural crops.

Although the nutrient balances for the countries grouped into the deciduous woodland subregion show evidence for nutrient mining, the nutrient status is generally better than in the equatorial forest subregion (Figure 14.7, Figure 14.8 and Figure 14.9). In the southern part of the deciduous woodland subregion, both Angola and Zambia have low or very low values for N export in crops but the indicators for N, P and K balance are positive or close to zero. In West Africa, evidence of nutrient mining is highest in Côte d'Ivoire and Ghana (with negative balances for N, P and K), but almost all the countries show evidence of mining of P and K.

The national statistics are supported by many site-specific studies. For example, under the semi-deciduous forest zone of Ghana, bare plots had the highest amounts nutrient loss, with 34 kg/ha for N, 12 kg/ha for P and 13 kg/ha for K, whereas control plots had deficits of 20 kg/ha for N, 8 kg/ha for P and 8 kg/ha for K. Plots covered by maize had the higher nutrient loss than soybean plots. The legume-based cropping systems under inorganic fertilizer and biochar amendments effectively reduced nutrient loss (Bashagaluke *et al*., 2018).

Agegnehu *et al*. (2021) indicated that soil acidity was present in Eastern, Central, Western and Southern Africa, where a wet tropical climate prevailed and about 35 percent of sub-Saharan Africa was affected by soil acidity.

Soil acidity in the equatorial forest subregion of Africa has been a subject of research and environmental concern. Highly acidic soils may hinder the adequate uptake of essential nutrients by plants and impact the diversity and abundance of soil organisms crucial for nutrient cycling, thus potentially impacting soil health, ecosystem resilience and agricultural productivity.

Soil acidification was attributed to several factors, including soil type, topography, land-use change, climate-related impacts and agricultural practices where the intensive use of chemical fertilizers reduced the soil pH (Asensio *et al.,* 2019). Land-use change and certain agricultural activities have been associated with alterations in soil pH levels. The conversion of forested areas to agricultural land or other uses may lead to changes in soil acidity due to shifts in organic matter content and nutrient cycling.

In the context of intensive N fertilizer use from 1980 to 2022, modelling by Uwiragiye *et al.* (2023) suggested that the soil pH in croplands with soil pH <6.5 had declined significantly, and predicted that soil acidification would become more severe by 2050 in sub-Saharan Africa. In response to soil acidification and its potential impact, there has been a growing focus on promoting sustainable land management practices and soil conservation strategies within the equatorial forest region. Research efforts have explored the potential of agroforestry systems and the positive effects of tree cover in regulating soil pH and sustaining soil health (Ekanem and Udosen, 2023).

14.3.6 Soil physical deterioration in the equatorial forest and deciduous woodland subregions

Highly mechanized, heavy machinery-dependent mining operations cause soil compaction, leading to reduced water infiltration, decreased plant growth and reduced nutrient availability. Mining also removes vegetation and topsoil that protect the soil from erosion and compaction. Mining activities occur in many countries in the equatorial forest subregion, including the Democratic Republic of the Congo, Liberia, Sierra Leone and Zambia (Cherlet *et al*., 2018). For example, in 2019, the extractive sector accounted for 46 percent of the Congolese government's revenues and 99 percent of the country's exports, and 19 percent of total Cameroonian government revenues and 32 percent of the country's exports by value (EITI, 2022, 2023). The countries of the African equatorial forest region intend to support mining with the aim of maintaining or accelerating the development of their countries. Thus, mining activities are set to accelerate, with potential adverse effects on agricultural soils.

In the deciduous woodland subregion, little is known about the effects of mining on soil. However, at the Bukuru mining site in Nigeria, 1 574 km^2 has been degraded by mining activities, indicating that the area allocated for agricultural production has declined, potentially leading to food insecurity, hunger, malnutrition, resource conflicts and ecosystem damage (Tambol, Vodounou and Moussa, 2023). In Burkina Faso, industrial mining has generally developed in rural areas where agriculture is the main subsistence activity. Decreased cereal yield and cereal production are jeopardizing food supply and exacerbating poverty in mining regions (Ouoba, 2018).

Given the economic importance of mining activities, ongoing research is needed to highlight the immediate impacts and pave the way for sustainable mining practices and effective conservation strategies in Africa's equatorial forests.

The expansion of agriculture in the equatorial forest subregion has led to soil compaction through agricultural practices such as ploughing, tilling and the use of heavy machinery (Modiba *et al*., 2023). These practices disrupt the soil structure, leading to increased bulk density and reduced pore space, and hinder water infiltration and root growth (Shaheb, Venkatesh and Shearer, 2021). This then reduces resilience to drought and disrupts soil microbial communities, affecting nutrient cycling and overall soil fertility and thus leads to decreased agricultural productivity.

In Ghana, the severity of soil compaction due to the use of heavy machinery has varied with land use and management in the semi-deciduous zone of the country. The bulk density of the subsoil has reached 1 800 kg/m^3 for conventionally tilled croplands and 1 760 kg/m^3 for grasslands, compared with 1 440 kg/m^3 for natural forestlands (Adesoyin *et al.,* 2023).

While the deciduous woodland in Africa may not face the same level of anthropogenic pressures and soil degradation as some other regions, it is still essential to understand and address potential issues related to soil compaction and degradation to ensure long-term health and ecosystem sustainability.

Organic amendments – including the application of poultry manure (Bawa *et al.,* 2019) or other organic wastes – have proved to be efficient in reducing compaction, increasing macroporosity and decreasing soil dry bulk density (MacCarthy *et al.,* 2020). To supply the increasing demand for food by the expanding population, the soils of the region will need to be exposed to additional intensive agriculture. Therefore, attention should be given to the promotion of sustainable agriculture practices including conservation tillage, agroforestry and cover cropping, which can help maintain soil structure and reduce compaction.

14.3.7 Urbanization and soil sealing in the equatorial forest and deciduous woodland subregions

Soil sealing has been observed in localized areas due to urban expansion and infrastructure development. The conversion of natural land into impervious surfaces has affected water infiltration and can lead to increased surface runoff during heavy rainfall. The progress of urbanization has led to the establishment and expansion of small- to medium-sized urban centres near forest zones. Population influx and economic developments have contributed to an increased urban footprint.

Soil sealing has been identified as a major threat to soil quality in the equatorial forest region of Africa, which can lead to soil infertility, erosion and decreased water infiltration, with consequential effects on the availability of clean water (Selige, Böhner and Schmidhalter, 2006). This can directly impact agriculture, food security and the health of both humans and natural ecosystems.

Urban expansion in sub-Saharan Africa has caused a loss of soil and vegetation cover and has led to a decrease in agricultural productivity (FAO, 2015; Sakketa, 2023). However, some governments in the region, including those of Cameroon, the Democratic Republic of Congo, Gabon and Nigeria have implemented land-use policies to promote sustainable urbanization and soil conservation practices (UN, 2019).

Efforts to mitigate soil sealing and urbanization have been made in the region with the adoption of Agenda 2063 by the African Union (AU) in 2015 (African Union Commission, 2019). This agenda promotes sustainable urbanization, including investment in green infrastructure, improved land-use planning and enhanced urban resilience in the face of climate change. While progress has been made to address these threats, there is a need to accelerate and scale up solutions to mitigate their impacts on soil and the broader ecosystem.

14.3.8 Soil pollution in the equatorial forest and deciduous woodland subregions

The equatorial rainforest of Africa includes the Congo Basin and the Gulf of Guinea coast has substantial mineral resources in the form of petroleum, natural gas, Au, tin (Sn), Co, and Pb. Human exploitation of the natural resources has led to environmental and soil pollution (Tindwa and Singh, 2023). Although there are currently no large-scale assessments of soil pollution, mining in the equatorial area is regarded as the most significant source of pollution, followed by waste and industrial activities, agriculture and auto mechanic workshops (FAO and UNEP, 2021). Soils at and around pollutant sites have been contaminated by trace elements (including Co, Cu, Cr, Pb, Hg, Ni, V, Zn and POPs). Low availability of micronutrients and soil acidity are among the main problems associated with the accumulation of heavy metals in soils.

The main contributors of soil pollution in agriculture are fertilizers, pesticides and organic waste used for production. A study on urban agricultural soil (Aboubakar *et al.,* 2021) reported very high levels of Pb contamination in 60 percent of soil samples. Ngole-Jeme (2016) reported moderate to high soil contamination and hazard quotients for Cr, Cd and Pb, exceeding the reference doses in soils of southwest Cameroon.

The main sources of soil pollutants in the deciduous woodland subregion include industrial activities, artisanal and small-scale mining, agricultural activities (particularly the mismanagement of agrochemicals), fertilizer misuse (especially on commercial farms), detergents, and poor management of solid waste, landfills and sewage (ECA, 2001; Rodriguez-Eugenio, McLaughlin and Pennock, 2018; Tindwa and Singh 2023).

The critical issue in the deciduous woodland subregion is the lack of sufficient data on soil pollution to inform policy decisions and good practices to reduce pollution. However, FAO and UNEP recently compiled information on soil pollution across regions, including a specific chapter on Africa (FAO and UNEP, 2021), specifically relevant for the deciduous woodland subregion. Soil pollution with heavy metals like Cd, Pb, As and Hg have been reported in countries in the region including Ghana, Nigeria and Togo, as well as pesticide contamination. While most of the pesticides may be less persistent in the environment than heavy metals, they still pose risks to soil pollution through leakage and poor handling.



14.4 Regional status and trend of threats to soil functions for the savannah and highlands and desert and semi-desert subregions

14.4.1 Introduction

Eighteen percent of sub-Saharan Africa is semi-arid and 36 percent is arid (Figure 13.1) (Montanarella *et al*., 2015). Arid and semi-arid soils of the savannah and highlands and desert and semi-desert subregions are characterized by water scarcity (due to low rainfall and high evapotranspiration), mostly sandy textures, low biological activity (and thus limited vegetation cover and low soil SOC contents) and nutrient deficiency, which limit crop production (Mulinge *et al*., 2016; Nyuma and Churu, 2022), meaning that incomes of the people in the area are extremely low (Fenta *et al*., 2019). Increasing aridity is a threat to plant productivity (Ayangbenro and Babalola, 2021) and soils are susceptible to wind erosion and salinization (WAD, 2018).

Mean annual rainfall is 500 to 1 000 mm in the semi-arid zone and 0 to 500 mm in the arid zone. The arid zones support less than 90 to 180 plant-growing days (Montanarella *et al*., 2015). The major soils include Fluvisols, Gleysols, Regosols, Vertisols, Calcisols, Gypsisols, Solonetz, and Solonchaks (Figure 14.3). Soils are mainly sandy and calcareous or siliceous, with salt and gypsum deposits occurring extensively. The soils generally have poor water retention, low nutrient contents and are devoid of vegetation cover throughout the year (Jones *et al*., 2013). The accumulation of soluble salts and Na lead to salinization and sodification that affect plant growth (Daba *et al*., 2021). The natural vegetation consists of scattered short trees, shrubs and grass.

14.4.2 Soil erosion in the savannah and highlands and desert and semi-desert subregions

Water erosion is a major threat to crop production (Wolka, Mulder and Biazin, 2018) in the savannah and highlands subregion, with an estimated 65 percent of agricultural land suffering from degradation (Tamene and Le, 2015). Soil losses are largely caused by human activities which accelerate water erosion (Haregeweyn *et al*., 2017). Most agricultural regions on hills and mountains have lost their fertile topsoil, and thus eroded gullies are common in farmlands (Zenebe *et al*., 2013). Fanta *et al*. (2020) estimated the mean annual gross soil loss by water erosion in East Africa at 6.3 t/ha, of which 50 percent originated from cropland. Yesuph and Dagnew (2019) and Gadisa and Midega (2021) estimated the annual topsoil loss in various parts of the Ethiopian highlands at 42 t/ha, with the rate reaching 300 t/ha in severe cases.

In most of the desert and semi-desert subregion, rainfall is often of high intensity and short duration, giving rise to severe soil erosion. The intensity of erosion is severe where the ground is bare (Watene *et al*., 2021). For example, in the East African Rift Valley, the mean annual erosion rate for the entire area was estimated at 6.26 t/ha, with a total soil loss of up to 132 million tonnes annually (Watene *et al*., 2021). However, sugarcane plant cover has reduced rates of water erosion in the semi-arid zone of the Rift Valley system (Asmamaw, Mohammed and Mathias, 2021). Water erosion increases on slopes where vegetation cover is reduced due to deforestation, overgrazing or overcultivation that leaves the soil surface bare.

Wind erosion is common in sandy and poorly vegetated semi-arid and arid regions of sub-Saharan Africa, as commonly observed in the Kalahari and the Horn of Africa deserts (WAD, 2018). While wind erosion is common in arid and semi-arid regions, it also occurs in the humid hot savannahs

of Central Africa between Cameroon and Gabon, along the Gulf of Guinea (Jones *et al*., 2013). Over 99 percent of wind erosion in East Africa regions occurs in the dry land zone, with less than 1 percent affecting the humid zones of Burundi, Kenya, Rwanda, the United Republic of Tanzania, and Uganda (Fenta *et al*., 2020). However, suspended clay particles and very fine sand particles, as well as organic matter, can be transported over a greater distance.

The main factor that limits wind erosion is the vegetation and residue cover remaining on the surface (Touré *et al*. 2019). Both natural and cultivated vegetation efficiently limit wind erosion, even at low covering rates. Grazing decreases vegetation cover and surface roughness, leading to decreased wind erosion thresholds. Pierre *et al*. (2017) suggested that wind erosion could be 1.5-times higher on cultivated surfaces compared to rangeland.

It is estimated that around 25 percent of the Horn of Africa deserts have moderate susceptibility to wind erosion risks, of which 90 percent is in Somalia and the Sudan, followed by Ethiopia, Eritrea, Kenya and Djibouti (Fenta *et al*., 2019). In South Africa, serious soil erosion occurs in the Namib desert – a narrow strip along the western coast – and the Kalahari Basin, which has a mean annual wind erosion of more than 90 t/ha (Zhao *et al*., 2021).

In a large part of the African highlands, soil tillage is undertaken by human labour or draft animals which disturbs the topsoil by breaking up soil aggregates and exposes the soil to erosion. However, minimum tillage – which would maintain soil structure and reduce soil erosion – is rarely practiced in the highlands of East African countries, as it allows more weeds to flourish (compared to conventional or intensive tillage), and hence is unpopular with farmers in subsistence agricultural farming systems throughout Africa (Kuyah *et al*., 2021).

In the desert and semi-desert subregion, subsistence draft animal farming is the main form of cultivation, and poor land management and tillage practices have led to severe soil erosion. Where intensive tillage is practiced, direct evaporation from the soil surface leads to soil desiccation and poor vegetation and crop growth, which accentuates erosion and runoff (Masvaya *et al*., 2017). In semi-arid rainfed farming in Ethiopia, Kebede *et al*., (2023) reported that surface runoff was 30 to 60 percent lower in a minimum tillage system than in a intensive tillage system, and soil loss was 49 to 76 percent lower. Minimum tillage practices have generally showed increases in SOC and agricultural productivity and a decrease in soil erosion in the semi-arid regions (Githongo *et al*., 2021).

Hillslope mass movement, in the form of rockslides and mudslides, is a concern on the hill slopes and in mountainous areas, including Ethiopia, Rwanda and Uganda (Broeckx *et al*., 2018). Deforestation, overcultivation and expansion of infrastructure (particularly cutting slopes for road and urban construction) reduce the shear strength of soils and upon wetting, soils are susceptible to mass movements (Broeckx *et al*., 2018). Hill slope lands are becoming more susceptible to mass wasting, mainly caused by overcultivation and deforestation such as on the foot slopes of Mount Kenya in Kenya.

Uprooting of crops is commonly practiced in parts of the East African highlands such as Ethiopia and Kenya. In these areas, sorghum and maize crop roots are uprooted for firewood purposes (Yitayh *et al.,* 2023). Root crop harvesting removes both soil and decomposable organic residues, aggravates soil structure deterioration and leaves the land prone to water erosion. Above ground mass is also collected for firewood, fencing, house construction and animal feed purposes. The loss of SOM due to uprooting and removal of above ground biomass lowers soil fertility and water storage capacity.

14.4.3 Soil carbon loss in the savannah and highlands and desert and semi-desert subregions

Agriculture is the dominant factor affecting SOC changes (Ewing *et al*., 2023). In the Ethiopian highlands, after 30 years of poor land management, SOC was depleted by 5.6 to 7.2 t/ha in croplands, while it was increased by 0.1 to 1.3 t/ha in shrublands (Abegaz *et al*., 2016). In Cameroon, SOC decreased from 22.3 to 15.7 million tonnes at a mean annual rate of 1.3 t/ha over three decades, which was attributed to land conversion to cropping (Nguemezi *et al*., 2021). Converting degraded lands to enclosures or afforestation improved SOC annually by 2.4 to 3.2 t/ha (Abegaz *et al*., 2020). However, conservation agriculture with integrated soil fertility management (Kihara *et al*., 2020), as well as minimum tillage (Githongo *et al*., 2021), had no influence on SOC.

Arid and semi-arid lands have little vegetation cover, meaning that there is little SOC accumulation. Land-cover changes, farming, high temperature and decomposition processes lead to rapid biodegradation of any available SOC. Consequently, the depletion of SOC lowers water storage and results in unstable aggregates which further increases both wind and water erosion (Mayer *et al*., 2020).

Significant losses in SOC stocks were observed following conversion of woodlands to farmlands in the Kalahari Basin (De Blécourt *et al*., 2019). The authors noted that in the top 40 cm of soils in the Kalahari Basin, woodland had the highest SOC stock (27.4 t/ha in northeast Namibia and 25.2 t/ha in southwest Zambia) followed by recently cultivated land (21.7 t/ha in northeast Namibia and 26.5 t/ha in southwest Zambia), while older agriculture areas had the lowest (19.2 t/ha in northeast Namibia and 19.9 t/ha in southwest Zambia.

The impacts of land-use change on SOC stock, such as the conversion from native vegetation (natural forest and grasslands) to cropland, averages 19.6 percent SOC losses in Namibia (De Blécourt *et al*. 2019) , 21 percent in Chad (Adoum *et al*., 2017), 25.8 percent in Ethiopia (Solomon *et al*., 2018) and 47 to 59 percent in South Africa (Loke *et al*., 2019). Dryland cropping systems in the Sahel of West Africa, East Africa deserts and the Kalahari Basin, have been particularly susceptible to the impacts of SOC erosion, suggesting a depletion of available SOC stock in less than a decade (Chappell *et al*., 2019). Salinization aggravates SOC depletion because of loss in Net Primary Productivity (NPP) and less input of biomass C into the soil (Patel *et al*., 2020).

Wetlands cover between 1 and 16 percent of Africa (Chuma *et al*., 2021). In the Ethiopian highlands, free grazing, grass harvesting and water extraction are the main pressures causing wetland degradation (Moges and Mebrate, 2022), and land-use change threatens the potential of wetlands to store C (Were *et al*., 2019). Unsustainable farming, aquaculture, settlement expansion and clay extraction for brick-making have degraded marshlands in the east of the Democratic Republic of the Congo (Chuma *et al*., 2021).

The conversion of wetlands to agriculture has intensified because of climate variability and the demand to feed a growing population (Mwita, 2013). Thus, wetlands in the semi-desert and desert of sub-Saharan Africa drain laterally towards watercourses, increasing salt levels in rivers and wetlands (Dube, Dube and Marambanyika, 2023). Grasslands in semi-arid and arid regions may have lower SOC storage potential than wetlands when expressed on a per-unit area basis (Minasny *et al*., 2017).

Soil inorganic carbon pools are likely to be the dominant form of soil C in arid to semi-arid regions (Loke *et al*., 2021) and almost all SIC exists in inert forms such as $CaCO_3$ and $MgCO_3$, where $CaCO_3$ is the most common. The exact amount of SIC gain or loss in sub-Saharan Africa remains unclear due to a lack of research, but it is evident that SIC accumulates as carbonate in flat and moderately concave drier landscapes (Pfeiffer, Padarian and Vega, 2023). Land management practices such as fertilization, irrigation and liming influence SIC, and thus affect soil C gain or loss (Sanderman, 2012). Long-term applications of organic and inorganic acid-forming fertilizers acidify soils, resulting in the

dissolution of carbonates and the emission of CO_2 (Zamanian, Zhou and Kuzyakov, 2021) equivalent to 12 to 18 percent of the initial C stock (Liu *et al*., 2023).

Land-use changes may reduce SIC stock within a few decades (Loke *et al*., 2019). Zamanian, Zhou and Kuzyakov (2021) estimated that about 40 percent of the dissolved SIC was reformed as carbonate minerals in the soil, with the remainder being lost as CO_2. Loke *et al*. (2019) estimated that the reduction of SIC due to cultivation in the three semi-arid regions of South Africa was 90 percent in Tweespruit, 82 percent in Kroonstad and 71 percent in Harrismith. In contrast to the cultivation of grass in the degraded cultivated soils in the Kroonstad agroecosystem, SIC increased from 1.13 to 8.11 g/kg in three semi-arid regions (Loke *et al*., 2019). In the Sudan, the SIC stock in topsoil (0–30 cm) averages 17 t/ha and the subsurface (30–100 cm) averages 24 t/ha in calcareous soils (Darwish, Atallah and Fadel, 2018).

14.4.4 Soil biodiversity loss in the savannah and highlands and desert and semi-desert subregions

The change of natural lands, forestlands and grasslands to other uses disturbs soil biodiversity and reduces organisms that are helpful in residue decomposition and soil quality maintenance. According to Leul *et al*. (2023), a change of forestland to agricultural use reduced SOM, whereas turning woodland and grassland to sugarcane farming lowered SOM and total N stocks in Ethiopia (Beza and Assen, 2016).

Soil biodiversity losses in the arid and semi-arid subregion are caused by the human-induced alteration of soil physical and chemical properties, which affects ecosystem services such as crop quality, the occurrence of soil-borne pests and diseases, N cycling, and water transfer (Laban, Metternicht and Davies, 2018). Intensive tillage also affects biodiversity: for example, in Zimbabwe's semi-arid region, under conventional tillage, the diversity of soil macroorganisms was 17.1 percent lower than under zero tillage and 26.1 percent lower than minimum tillage (Muoni *et al*., 2019). Under conventional tillage, soil microbial biomass C was 8.8 percent lower in Kenya (Nyamwange *et al*., 2021) and 8.16 percent lower in South Africa (Njaimwe *et al*., 2018) than minimum tillage. Studies of relative soil biodiversity and biomass changes across different land uses show the highest relative loss are due to conversions of forests and grasslands to agriculture land (Leul *et al*., 2023; CBD, 2019; Bach *et al*., 2020). Overgrazing of rangelands also affects soil microbial biomass C and N (Oduor *et al*., 2018b). However, the restoration of degraded lands improved soil biodiversity in the region (Dutta and Dutta, 2016; Oduor *et al*., 2018a, 2018b).

The effect of fertilizers on soil biodiversity was not consistent with the results of different research outputs of microbial diversity and fertilization in the region (Rughöf *et al*., 2016; Hamamoto *et al*., 2022), but organic fertilizer generally increased microbial diversity. In general, unsustainable soil management practices and climate changes have depleted SOM, promoted soil degradation and may have caused soil biota imbalances in the arid and semi-arid regions of sub-Saharan Africa (Cowan *et al*., 2022).

14.4.5 Nutrient mismanagement in the savannah and highlands and desert and semi-desert subregions

Soil nutrient depletion due to continuous removal of nutrients with harvests without replacement is a cause of high yield gaps (Bationo, Lamers and Lehmann, 2015). A nutrient surplus exists only in home gardens and newly cleared forestlands in the African highlands. Fields near farmers' homes, such as those used for home gardens and growing enset (*Ensete ventricosum welw*) in the Ethiopian highlands, receive a continuous supply of household waste that makes them rich in SOC and other

nutrients. The supply of house refuse (cattle manure, bedding materials and ash) decreases with distance from the homes and the soil fertility also declines in the same way. Enset farming is a key agricultural practice that can increase soil accumulation of C (Laekemariam, 2020). Nitrogen balance is more positive in the enset gardens primarily due to more continuous application of manure. Soil physicochemical properties such as bulk density, total porosity, organic C, C–N ratio, total N, pH, CEC, available P, and exchangeable bases (K^+, Ca^{2+} and Mg^{2+}), and extractable micronutrients (Fe^{2+}, Mn^{2+}, Zn^{2+}, Cu^{2+}) were affected by different land-use systems, whereby soil quality and health were relatively maintained under the forest system (Regassa *et al*., 2023). The availability of materials for composting also affects the nutrient contents of the soils, for example in the Jimma area of the Ethiopian highlands.

In the desert and semi-desert subregion, Djibouti stands out for its high inputs and positive nutrient balances (Figure 14.7, Figure 14.8 and Figure 14.9)). The cropland area of Djibouti is small (less than 1 000 km^2 of arable land out of a total land area of 23 200 km^2), only 0.2 percent of the total land area. Most nutrient input comes from organic sources: 97 percent of N (40.5 percent manure and 56.4 percent BNF), 95 percent of P and 99.4 percent of K from manure (FAO, 2023a).

Nutrient regimes differ across the 19 countries in the savannah and highlands subregion, but all but one have one or more indicators suggesting yield gaps or nutrient mining. Only South Africa has positive values of all the indicators for nutrient balances and a moderately high level of N export through crops (Figure 14.7, Figure 14.8 and Figure 14.9). Annual N outputs in crops are low (<20 kg/ha) in seven countries, indicating low yields and low nutrient content of crops. Botswana, Malawi, South Africa and South Sudan all have annual N nutrient budgets above 20 kg/ha, whereas Eswatini and Niger have slight negative N balances, indicating mining of N from the soil. Burkina Faso, Burundi, Chad, the Gambia and Lesotho had mean annual synthetic fertilizer inputs from 2017 to 2021 below 10 kg/ha (FAO, 2023a). Across all the countries, sources of N are relatively balanced, with 30 percent from fertilizer, 21 percent from manure and 32 percent from BNF (FAO, 2023a).

Phosphorus and K nutrient balances are more typically negative (an annual average of 0.0 kg P/ha and −5.0 kg K/ha respectively) across the savannah and highlands subregion, indicating a greater potential for nutrient mining (Figure 14.7, Figure 14.8 and Figure 14.9). Eleven countries have negative P balances but the predicted level of accumulated P residual from 1961 to 2019 is only very high in Lesotho, indicating high soil mining of P over time. Thirteen of the 19 countries have negative K balances in the 2017–2021 period. Inputs of K are primarily from manure (79 percent) rather than fertilizer (29 percent) whereas P inputs are more evenly balanced (51 percent fertilizer, and 49 percent manure) (FAO, 2023a).

The national level statistics are supported by many site-specific studies. For example, in the Rwandan highlands, the annual N and K depletions were 33.6 kg/ha and 71 kg/ha respectively, with a positive annual P balance of 2.3 kg/ha (Uwiragiye *et al*., 2022). In the highlands of Kenya, nutrient contents were below critical levels, ranging from 20 to 66 percent for N, 46 to 85 percent for P, 31 to 67 percent for S, 9 to 51 percent for Cu, and 80 to 87 percent for B (Mugo *et al*., 2020). The annual nutrient depletion rates in the southern Ethiopian highlands were −62 kg/ha for N, +9 kg/ha for P and −41 kg/ha for K (Abuye *et al*., 2021). The P balance was positive since P is less susceptible to leaching loss at lower pH conditions.

Nutrient depletions varied between crops. In the Ethiopian highlands under barley, the annual nutrient balances were −66 kg/ha for N, −5.9 kg/ha for P and −12.3 kg/ha for K. Under wheat, the annual nutrient balances were −50.7 for N, −2.6 for P and −5.4 for K, and under tef, the annual nutrient balances were −9.8 for N, +0.9 for P, and −3.2 for K (Esubalew, Amare and Molla, 2023). In the highlands of the United Republic of Tanzania, 90 percent of soils were deficient in N, P, K, and Mg. Copper and Zn were also deficient in many sites, although Fe and Mn were above the optimum requirements (Mng'ong'o *et al*., 2021). Van Beek *et al*. (2016) reported highly variable N, P and K balances in the diverse agroecological settings in Ethiopia. Nutrient depletion was generally

severe for N and depletion rates were highest in the intensively cultivated mountainous areas of the central and southern parts of Ethiopia, and increased over the 2012–2016 period.

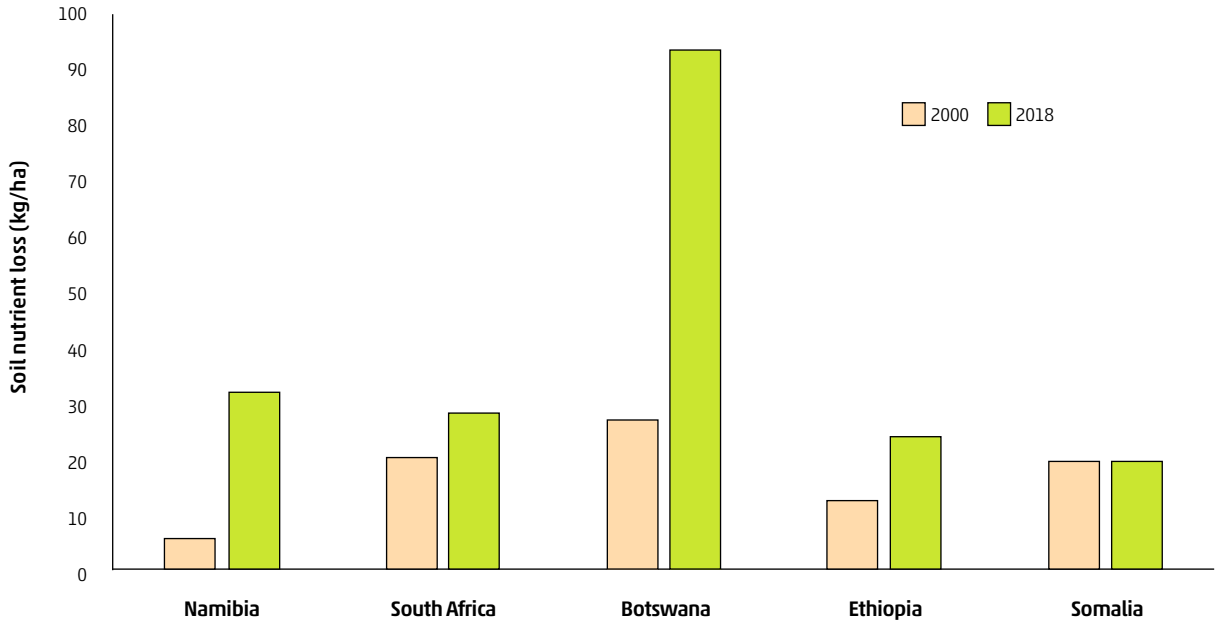
In the desert and semi-desert subregion, the natural replenishment of nutrients during fallow periods is now insufficient to maintain soil productivity over the long term due to increasing land pressure through practicing monocropping (Asmamaw, Mohammed and Mathias, 2021) and overgrazing of rangeland resources (Mekonnen and Hoekstra, 2018). According to Saito *et al*. (2019), the amount of N, P and K added as inputs was less than the amount removed as harvest or lost through erosion and leaching in arid and semi-arid areas. Inputs, crop withdrawal and agronomic budgets for N in semi-arid regions are low when compared to other regions (Bouwman *et al*., 2017), whereas the agronomic P budget in the region fluctuates around zero and the residual budget has small negative values. Phosphorus use efficiency fluctuates at around 60 percent (Magnone *et al*., 2022), but N use efficiency is less than 57 percent (Bouwman *et al*., 2017). Phosphorus nutrient budgets differ from N nutrient budgets in that there are no fixation or deposition components for P balances, but dissolved and particulate runoff losses must be considered.

Except for Djibouti, as discussed previously, the other countries of the desert and semi-desert subregion have similar nutrient regimes (Figure 14.7, Figure 14.8 and Figure 14.9). Annual N exports in crops are very low (average of 9.8 kg/ha) but since the annual N inputs only average 27 kg/ha, the average N balance is 20 kg/ha annually. The majority of N is from manure and biological N fixation except in Namibia, where 60 percent of N inputs are from synthetic fertilizers (FAO, 2023a).

For P, the overall annual input levels are low (averaging 3.3 kg/ha between 2017 and 2021) with a mean 87 percent of inputs from manure. The mean annual P balance is 1.0 kg/ha and the accumulated P residual between 1961 and 2019 was positive except for Namibia (Zou, Zhang and Davidson, 2022), which has a negative residual of −276 kg/ha, indicating nutrient mining of P. The mean annual K budget was positive (19.8 kg/ha), with an average of 95.4 percent from manure.

The estimated nutrient loss was the same or greater in 2018 than in 2000 with Botswana having the greatest difference (Figure 14.10).

Figure 14.10. Soil nutrient loss from 2000 to 2018 of semi-arid and arid regions of selected sub-Saharan African countries



◊ *Source:* **FAO**. 2023. *FAOSTAT: Fertilizers by Nutrient*. [Accessed on 5 February 2024]. https://www.fao.org/faostat/en/#data/RFN. Licence: CC-BY-4.0.

In the Ethiopian highlands, soil acidification is increasing, limiting crop production (Asmare and Markku, 2016). Desta *et al.* (2021) indicated that 45 percent of the rainfed agricultural areas of the Ethiopian highlands were being affected by soil acidity, with 15 percent of the area being strongly acidic (pH <5.5), 18 percent being moderately acidic (pH 5.6–6.0), and 12 percent being slightly acidic (pH 6.0–6.5).

Ferralsols, Alisols, Acrisols, Plinthisols and Nitisols are acidic soils (IUSS Working Group WRB, 2022) (Figure14.3). Acrisols cover 87.8 million ha and Ferralsols about 312.4 million ha of the total land area of Africa (Tully *et al*., 2015). In Ethiopia, the application of lime resulted in yield increases of 34 to 252 percent in wheat, barley and tef crops, 29 to 53 percent in broad bean (*Vicia faba*) and soybean crops, and 42 to 332 percent in potato crops. Liming gave increased yields of 111 to 182 percent for maize crops in Kenya and 45 to 103 percent for Mucuna (*Mucuna pruriens*) crops in Nigeria (Agegnehu *et al*., 2021).

14.4.6 Salt accumulation in the savannah and highlands and desert and semi-desert subregions

Irrigation has significantly increased crop production in the desert and semi-desert regions. However, if not managed well, irrigation can exacerbate soil salinization, threatening agricultural productivity (Meliyo *et al*., 2016; Kebede, 2023). In sub-Saharan Africa, estimates of the area impacted by soil salinization range from 19 million ha (Tully *et al*., 2015) to 69 million ha (Kebede, 2023).

Salt-affected soils are prevalent largely in the lowlands of the East African Rift Valley, along the coast of Western Africa, the countries of the Lake Chad Basin, and in pockets of Southern Africa (Kebede, 2023). Irrigated farmlands in four Ethiopian regions affected by salinity (>5 dS/m) were mapped using remote sensing and soil data, showing varying areas (30 percent in Afar, 6 percent in Oromia, 9 percent in Amhara and 3 percent in Tigray) and rated as having moderate to extreme salinity (Qureshi *et al*., 2020) (Table 14.2). Natural factors influence the formation of salt-affected soils, including salty parent materials, mineralogy, groundwater, windblown salt particles and floods (Omuto *et al*., 2020). Inappropriate irrigation water management (use of poor water quality, application of unmeasured irrigation water volume and insufficient drainage infrastructure for wastewater management) and poor agricultural practices can lead to salt accumulation. (Qureshi *et al*., 2020; Omar *et al.,* 2022).

Table 14.2. Percentage of area coverages by salinity soils classes of Afar, Amhara, Oromia and Tigray regions of Ethiopia

Soil salinity levels (dS/m)	Afar		Oromia		Amhara		Tigray	
	km²	%	km²	%	km²	%	km²	%
Non-saline (<2)	40 787	42	287 768.25	88.70	137 428	88	48 067	97.29
Low saline (2–5)	26 961	28	17 292.05	5.33	4 903	3	0	0
Medium saline (5–10)	9 798	10	17 152.54	5.29	11 892	8	1339	2.71
High saline (10–15)	5 618	5	1 576.72	0.49	1 230	0.8	0	0
Extremely saline (>15)	14 085	15	713.74	0.22	202	0.2	0	0
Total area	97 204	100	324 428.69	100	155 648	100	49 604	100

◊ *Source:* **Qureshi, A., Ertebo, T. & Melese, M.** 2020. *Extent, characterization and management strategies for the salt-affected soils in Ethiopia*. Project Report 2. Dubai, United Arab Emirates, ICBA. https://www.biosaline.org/sites/default/files/project-report-2-final-reduced.pdf

14.4.7 Soil pollution in the savannah and highlands and desert and semi-desert subregions

In the savannah and highlands subregion, at sites impacted by landfill leachate, Nyika *et al*. (2019) reported soil concentrations of 33.4 mg/kg of As, 481 mg/kg of Co, 219 mg/kg of Cu, 1 335 mg/kg of Cr, 8 991 mg/kg of Mn, 354 mg/kg of Ni, 46 mg/kg of Pb, 160 mg/kg of Zn, 435 mg/kg of V and 155 268 mg/kg of Pb in the Eastern Cape of South Africa.

Agricultural activities including application of fertilizers and pesticides have also been a source of soil pollution in sub-Saharan Africa (Fayiga, Ipinmoroti and Chirenje, 2018). Organic contaminants such as polybrominated diphenyl ethers (PBDEs), congeners found in technical octa-BDE mixtures, highly chlorinated PCBs and several heavy metals (Cu, Pb, Ni, Cd, Ag and Hg) showed elevated concentrations in the soils that are likely due to contamination by e-waste (Moeckel *et al*., 2020). Mining and mineral processing continues to be a source of lasting environmental problems in many developing economies (Lee *et al*., 2021).

Mining activities in the desert and semi-desert subregion include Au mining and smelting operations in South Africa, large Pb and Zn mining operations in Zambia, and coal mining in the United Republic of Tanzania (Semu, Tindwa and Singh, 2019). Industrial wastewater heavy metals (including Cr, Pb, Cd, Cu, and Zn) are discharged into rivers, which are then used for irrigation water in downstream lowland irrigated areas, resulting in soil pollution (Mufuruki *et al*., 2017; Abebe *et al*., 2023). Fertilizers, pesticides and organic wastes used in agricultural production are the main drivers of lowland irrigated soil pollution in the region (Tindwa and Singh, 2023). For example, commercial farmlands growing cotton, sugar cane and tobacco have been polluted over a long period of time with different types of organic pesticides in East Africa Rift zone countries, such as Ethiopia, Kenya, the United Republic of Tanzania and Uganda (Loha *et al.,* 2018).

In sub-Saharan Africa, medical wastes are mainly incinerated and dumped (UNEP, 2017). In Ghana, the ash generated in the process was identified to be a diffuse source of soil pollutants such as Cd, Ch, Pb, and Zn (Adama *et al*., 2016).

Sub-Saharan Africa had achieved only 30 percent sanitation coverage by 2015, with only a 4 percent increase from 1990 (WHO, 2016). For instance, soils are shown to be highly contaminated, with the near universal presence of *Ascaris lumbricoides* (a parasitic intestinal roundworm that causes human illness) around Addis Ababa (Debela *et al*., 2018). Concentrations of heavy metals (such as Zn, Cr, Ni, Co, and Pb) were reported in soil samples of the dumpsite of Addis Ababa and nearby open land and were higher than the internationally acceptable limit for soil (Beyene *et al*., 2023).

Natural sources of soil pollution also occur with tephra (volcanic ash, containing elements such as fluorine [F], I, Cu, Mo, Hg, Co, As, B, and Li) which contaminates root zone soils as well as surface and groundwater bodies, particularly in the East Africa Rift Valley (Davies, 2008).

14.4.8 Urbanization and soil sealing in the savannah and highlands and desert and semi-desert subregions

Urban expansion in the subregions leads to soil sealing through covering the soil surface with materials like clay-alkali materials, concrete and stone, due to new buildings, roads, parking places and the creation of other public and private spaces (FAO, 2022b). Sealed surfaces provide impedance to the infiltration of water, leading to increased runoff and soil erosion. On the other hand, concrete and stone-covered soils offer physical support for roadways and structures as well as protecting below ground infrastructure such as cables, pipes and tunnels. (Montanarella *et al*., 2016). Soil sealing in Africa has increased from 15 000 km^2 in 1990 to 38 000 km^2 in 2018 (Gong *et al*.,

2020), however, specific data for sub-Saharan Africa drylands was unavailable. Van Vliet, Eitelberg and Verburg (2017) assessed the impact of urban expansion on food production capacity from 2000 to 2040 and suggested that sub-Saharan Africa faced by far the greatest relocation of crop production.

14.4.9 Other threats in the savannah and highlands and desert and semi-desert subregions

The savannah and highlands and desert and semi-desert subregions have areas that are affected by soil physical deterioration, waterlogging and desertification.

Formation of a surface crust when soils are dry, the presence of a cultivation pan and soil compaction are indicators of soil structural deterioration that prevent seedling emergence, restrict root growth and development, reduce water and air infiltration, and increase the work required to cultivate the soil. Soil compaction is associated with use of heavy farm machinery (Agbeshie *et al*., 2020).

The plateaus of the Ethiopian, Kenyan and Rwandan highlands with slope gradients of less than 3 percent are prone to waterlogging (Figure 14.11). Under natural conditions, Vertisols and other heavy clayey soils are susceptible to waterlogging, leading to surface and subsurface drainage impedance.

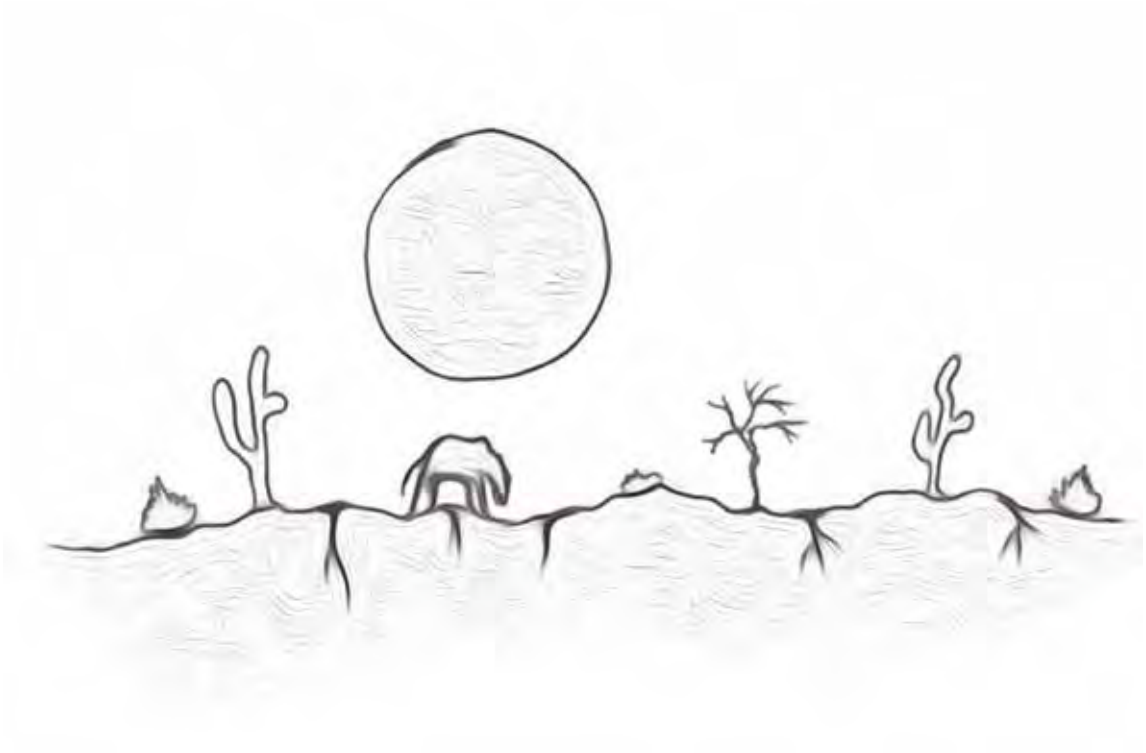
Vertisols, which cover about 12.6 million ha, with 7.6 million ha in the highlands (Figure 14.3), are major agricultural areas that have poor drainage and excess water that ponds on the land surface and saturates crop root zones, leading to plant nutrient deficiency in Ethiopia (Bedadi *et al*., 2023). Using a broad bed and furrow maker (Bedadi *et al*., 2023) and other tillage practices that incorporate conservation agriculture principles could effectively increase green water in the root zone available for crops, thus improving crop productivity and yields substantially on Vertisols in drylands without other inputs (Araya *et al*., 2015).

Figure 14.11. Waterlogged soil (Vertisol) in Fogera Plains, Ethiopia



Arid and semi-arid regions are being impacted by climate change, particularly from temperature and evaporation increases, and changes in rainfall patterns (Hoffmann *et al*., 2022). Drought-sensitive countries have had an increase in the prevalence of undernutrition from 17.7 percent of the population in 2010 to 21.8 percent in 2018 (FAO *et al*., 2019), attributed to increasing levels of water stress, desertification, land degradation and biodiversity loss (Ansah, Gardebroek and Ihle, 2020). Forest and shrub fires are problematic, as natural regeneration is hampered in the arid and semi-arid areas, which in turn impacts the development and productivity of soil (Nguyen Huy, Adjognon and Van Soest, 2023). *Prosopis juliflora*, a Latin American tree which is tolerant to high temperatures and salinity stress, has invaded semi-arid regions and producing organic substances that are toxic to native plants (Mekonnen *et al*., 2021; Mahdhi, Taieb and Habib, 2018).

The drier climatic conditions as well as the overexploitation of available resources have led to desertification and pressure on land and water resources, mainly in the Sahelian region, the Horn of Africa and the Kalahari basin of Africa (Ahsan, 2020). Desertification contributes to nearly 80 percent of soil and land degradation in Kenya, where 88 percent of the land is arid and semi-arid (Watene *et al*., 2021). Desertification and land degradation have been accelerated, resulting in chronic deficits in food and water throughout sub-Saharan Africa (Byaro, Nkonoki and Mafwolo, 2023).



14.5 Regional status and trend of threats to soil functions for the islands of Africa subregion

14.5.1 Introduction

The islands included in the sub-Saharan Africa subregion are Cabo Verde, Comoros, Madagascar, Mauritius, Reunion, Sao Tome and Principe, Seychelles and Zanzibar. Environmental conditions and land cover vary across the islands or, in the case of Madagascar, within the island. Little recent data on soil threats is available for the islands except for Madagascar. The two islands of Zanzibar are part of the United Republic of Tanzania and are not reported separately in the nation-level reporting system of FAO (e.g. FAO, 2024a).

Deforestation is a driver of harmful changes in soil functions. Historical levels of deforestation have been high in some of the African islands. In Mauritius, only 2 percent of the native forest remains, and deforestation has had a strong ecological impact along with increased erosion rates (Norder *et al*. 2017). In Madagascar in the 1953–2015 period, the island lost 44 percent of its natural forest (Vieilledent *et al*., 2018).

Current rates of deforestation in the remaining forest are relatively low except in Comoros and Sao Tome and Principe (Table 14.3). Cabo Verde, Mauritius and Reunion have experienced increases in forest in the 2010–2020 period.

14.5.2 Soil erosion in the islands of Africa subregion

African islands are particularly sensitive to water erosion due to the high amount of erosive rainfall, vulnerable soils, steep slopes and soil tillage. Annual soil losses by water erosion of 15 tonnes/ha in Madagascar (Razafindramanana, 2017) and 215 kg/ha in Mauritius (Sumner *et al*., 2016) have been recorded. Agricultural areas with no vegetation cover and low SOM contents are sensitive to water erosion. Soil erosion leads to downstream sedimentation which can affect floodplains, and sensitive estuarine and coastal environments (Razanamahandry *et al*., 2022; Sumner *et al*., 2016).

Coastal erosion is a threat to the smaller islands including Grands Comoros, Mauritius and the Seychelles, as well as Madagascar. Coastal erosion is a natural process but may be exacerbated by climate change, sea level rise and human activity, notably sand extraction such as in Comoros, Mauritius and Madagascar, changing vegetation cover and infrastructural developments. Coastal erosion includes beach erosion and also impacts reef–lagoon ecosystems (Sumner *et al*., 2016).

Tillage causes significant soil losses which in the Malagasy highlands, ranging from 1 to 15 t/ha annually depending on precipitation, runoff and slope on rice crops (Razafindramanana, 2017).

The main hillslope form of mass movement in Madagascar are “lavaka”. Lavaka are a type of gully that form via groundwater seeping through lateritic soils over saprolite, which create eroded headwalls on steep slopes. Recent studies have shown that the lavaka have existed long before humans came to Madagascar, but that human pressure bought about by the introduction of cattle and a growing population have triggered an increase in lavaka erosion (Brosens *et al*., 2022).

14.5.3 Soil carbon loss in the islands of Africa subregion

Deforestation has been a driver of SOC loss in the islands of Africa. The traditional farming practices in Madagascar are mainly characterized by a low amount of organic matter input in croplands,

which further contributes to the depletion of SOC (Razafimbelo *et al*., 2018; Rakotovao *et al.,* 2020). In formerly tree-covered areas, deforestation causes an average SOC loss up to 10.7 percent in the 30 cm soil layers in humid climates (Grinand *et al.* 2017. The regular practice of using fire in slash-and-burn cultivation contributes to the loss of SOC in the long term (Andriamananjara *et al*., 2019a).

The mangrove ecosystems in the southwest of Madagascar are characterized by a wide range of soil C concentrations, ranging from 0.31 to 12.71 percent. It is predicted that mangrove ecosystems will experience substantial soil C loss because of degradation and their conversion into aquaculture areas (Benson *et al*., 2017). In the northwest part of Madagascar, mangrove deforestation results in soil C loss, with an estimated loss rate that is four and a half times higher than the rate of C sequestration in undisturbed soils (Arias-Ortiz *et al*., 2021; Benson *et al*., 2017).

There are very few studies focused on SIC. The evidence of an active oxalate-carbonate pathway contributing to soil fertility and soil C fixation in mineral form was reported in tropical forest soils of the western part of Madagascar (Randevoson, 2019; Hervé *et al*., 2021).

14.5.4 Soil biodiversity loss in the islands of Africa subregion

The soils in Madagascar contain high biodiversity with high levels of endemicity, but taxonomic studies on soil organisms remain scarce. Earthworms have been studied over the last 15 years, enabling the description of numerous species (Razafindrakoto *et al*., 2017; Hong *et al*., 2019). Earthworms are subject to numerous threats such as deforestation and land cultivation. Numerous studies have recently aimed at increasing the abundance and diversity of soil organisms in the agrosystems of the highlands, in order to improve soil health, agrosystem productivity and sustainability. Soil organisms carry out major ecological functions of the soil which directly affect crop production, water retention and C sequestration (Blanchart *et al*., 2020; Ratsiatosika *et al*., 2021a; Trap *et al*., 2021a). However, these organisms are sensitive to agrosystem management practices (Rakotomanga *et al*., 2016; Raharijaona *et al*., 2023). These changes in land use or practices also affect the functions carried out by microorganisms (Razanamalala *et al*., 2018), free-living nematodes (Trap *et al*., 2021b) or earthworms (Ratsiatosika *et al*., 2021b).

14.5.5 Nutrient mismanagement in the islands of Africa subregion

The islands of Africa show very different patterns of macronutrient use (Figure 14.7, Figure 14.8 and Figure 14.9). Both Mauritius and Seychelles have relatively high levels of N export in crops and adequate but not excessive levels of N, P and K. Comoros and Madagascar have low levels of N export in crops but there is evidence of insufficient N and K inputs and possible mining of P. The Atlantic islands of Cabo Verde and Sao Tome and Principe have very low levels of N export in crops, indicating possible N mining. Sao Tome and Principe may also be experiencing mining of P.

In Madagascar, nutrient mining results mainly from conventional and traditional practices, slash and burn practices and forest clearing, which consequently promote or accentuate nutrient leaching and erosion. Under conventional tillage, C and nutrient losses were greater compared to direct seeding mulch-based cropping systems due to water erosion, dropping from 336 to 6.4 kg/ha for C, from 25.7 to 0.53 kg/ha for total N, and from 6.63 to 0.15 kg/ha for P (Razafindramanana *et al*., 2017). Low fertilizer application does not cover the nutrients in exported rice, resulting in continuous nutrient mining (Fanjaniana *et al*., 2021). A decreasing trend of soil mineral N content was observed from transplantation to harvest during the two cropping seasons of rice, from 200 to 50 kg/ha in the 0–90 cm soil layer, attributed mainly to increased plant N uptake and N leaching (Ranaivoson *et al*., 2018).

Decreasing topsoil C content of between 133 and 3.2 t/ha was observed by Andriamananjara *et al*. (2016) where land use changed from closed canopy forest to degraded land due to slash and burn practices. In the rainforest of southeast Madagascar, soil total N decreased following field abandonment (Randrianarison *et al*., 2016), although there was little change in total C.

Improved management practices can increase soil nutrients. In upland soil in Madagascar, three years of a rice–stylo (*Stylosanthes hamata*) rotation system with residues incorporation resulted in the accumulation of N in soil with N recovery of 64 percent in 0–30 cm soil depth, while three years of continuous P mineral application in a rice cropping system increased the available P in topsoil with a proportion of P recovery of 20 to 79 percent (Zemek *et al.,* 2018; Andriamananjara *et al*., 2019b). Further studies showed increasing soil N contents after two to four years of conservation agriculture (Randrianjafizanaka *et al*., 2018, Rodenburg *et al*., 2020).

Unsustainable soil management has led to soil acidification in Madagascar. Decreasing soil pH was observed in Bambara groundnut crop soils amended with triple superphosphate (TSP) after three years of application, compared to farmyard manure application (Andriamananjara *et al.,* 2019b). Biomass removal increased soil acidity in the Madagascar highlands (Fanjaniana *et al*., 2021). Remedial measures have been shown to be effective in decreasing acidity. The initial acidic soil pH (pH in water of 4.42) can be increased by the addition of dolomite, biochar and organic fertilization, including manure treatments in the Madagascar highlands (Raboin *et al*., 2016).



14.6 Trends in sustainable soil management

14.6.1 Summary of state of and trend in threats to soil functions

The information presented in this chapter was used to assess the state and trend of threats to soil functions through a management-based approach. Instead of directly measuring or modelling the physical severity of each soil threat, as was done in the 2015 SWSR report (FAO and ITPS, 2015), the current report assesses the level of adoption of SSM practices known to mitigate each soil threat.

The state reflects the effectiveness of management practices in place as of 2015 in preventing or reducing a specific soil threat. At the regional or subregional scale, changes in management occur at decadal or longer time scales and hence the state as of 2015 is relevant for 2015–2024 assessment period.

The trend indicates the direction of change in the adoption of SSM practices between 2015 and the finalization of this report (late 2024). In short, trend reflects the how SSM adoption has progressed over time.

There is little cause for optimism in the assessment of the state and trend of the threats to soil functions in sub-Saharan Africa (Figure 14.12 and Table 14.3). Poor management conditions, such as high rates of deforestation and the removal of crop residues due to competing demands and overgrazing have resulted in severe threats to soil functions from soil erosion, SOC change and the loss of soil biodiversity. Nutrient mining is widespread throughout sub-Saharan Africa, with significant socioeconomic impediments to improved nutrient balances. Soil pollution from mining and urban growth is widespread and the major expansion of urban areas predicted by 2050 will cause increased conflicts between urbanization and other uses of the soil. In conclusion, SSM has progressed little in sub-Saharan Africa since 2015.

Figure 14.12. Assessment of state and trend of threats to soil functions in sub-Saharan Africa through a management-based approach



Note: For further details, see Table 14.4.

Table 14.3. Summary of threats to soil functions in sub-Saharan Africa and the confidence in the assessment of their state and trend

Threat to soil functions	Summary/comments	Confidence	
		Condition	Trend
Erosion	There is severe water erosion in highland, tropical forest and temperate forest biomes, while there is high soil loss due to wind erosion in savannahh, semi-desert and desert biomes. Deforestation in forested regions is a major driver. Socioeconomic constraints such as the competing uses for crop residues throughout region limits the adoption of SSM.	High	High
Nutrient mismanagement	There is substantial nutrient mining of N and P in Chad, Comoros, Côte d'Ivoire, the Democratic Republic of Congo, Equatorial Guinea, Eswatini, the Gambia, Ghana, Niger, and Togo. Major socioeconomic impediments exist due to increased synthetic fertilization inputs and to nutrient inputs from plant residues and manures.	High	High
SOC change	There is a mismanagement-induced decline in SOC due to deforestation, the complete removal of residues for competing uses and the high rate of oxidation due to tillage.	High	High
Soil biodiversity change	There is a loss of soil biodiversity due to mismanagement practices, including deforestation, overgrazing and burning	،Low	Medium
Salt accumulation	Salt accumulation is an issue in the desert and semi-desert subregion. Inappropriate irrigation water management (use of poor water quality, application of unmeasured irrigation water volume, and insufficient drainage infrastructure for wastewater management), and poor agricultural practices all lead to salt accumulation.	Low	Medium
Soil pollution	Soil pollution associated with metropolitan areas and mining was found. Substantial heavy metal contamination was found associated with poorly regulated mining sites.	Low	Low
Urbanization and soil sealing	Urban areas of sub-Saharan Africa are predicted to experience the highest growth rate in the world, which will put pressure on surrounding lands.	Low	Low
Soil acidification	Soil acidification problems are common in heavily weathered soils and can be intensified through the application of some forms of synthetic fertilizer. Acidification limits the use of legumes for N fixation in many soils and limits the availability of P.	High	High

Note: The threats are presented in the order of most severe threat (erosion) to least severe (soil acidification).

14.6.2 Progress and impediments in the adoption of sustainable soil management

Sub-Saharan Africa faces challenges in terms of SSM due to a range of environmental and socioeconomic factors. At the same time, there have been encouraging developments in recent years in terms of adopting more SSM practices in the region.

One promising development is the growing recognition of the importance of healthy soils for agricultural productivity (African Union Commission, 2024). In the past, many farmers in the region have relied on practices that degrade soil quality, such as slash and burn agriculture, monoculture cropping and heavy tillage. However, in recent years there has been increasing awareness of the need to adopt more sustainable practices that maintain or even improve soil health (African Union Commission, 2024).

One example of progress towards SSM is the adoption of conservation agriculture (Kuyah *et al.*, 2021). This approach involves minimal soil disturbance, maintaining soil cover with crops or other vegetation and rotating crops to improve soil fertility. Conservation agriculture has been shown to improve crop yields, reduce erosion and increase resilience to droughts and other environmental stresses. Competing uses for crop residues and weed pressures are a major limiting factor in the

adoption of conservation agriculture (Kuyah *et al.*, 2021; Corbeels *et al.*, 2020)).

Given the current trend of urbanization, development and population growth, soil pollution in Africa is expected to witness an exponential growth in waste production in the next 20 to 50 years, which may reach a peak in the years beyond 2100 (Hoornweg *et al.*, 2015).

Many countries in sub-Saharan Africa have launched initiatives to promote SSM practices at the national or regional level. For example, the AU launched the Soil Health and Fertility Initiative (SHFI) to promote sustainable land and soil management. This can be evidenced by the recent 2024 Africa Fertilizer and Soil Health summit organized by the AU to deliberate on Africa's recent widespread decades-long decline in soil quality of farmland (African Union Commission, 2024).

Sub-Saharan Africa plays a significant role in the global C budget (Olorunfemi *et al.,* 2022; Ntinyari and Gweyi-Onyango, 2021) with about 5.7 Pg C stored in the top 100 cm soil depth, of which 50 percent is accumulated in the top 30 cm (Silatsa *et al.*, 2020). The region contributes the least to GHG emissions, accounting for 7 percent of the total global emissions (Ntinyari and Gweyi-Onyango, 2021; Pelster *et al.*, 2017), whereby land-use changes and agricultural activities, especially deforestation, accounts for approximately one-third of the emissions (Olorunfemi *et al.*, 2022). According to Van Loon *et al.* (2019), agricultural emissions will increase by more than 50 percent in 2050 compared to 2015.

Despite encouraging developments, there are still significant impediments to the widespread adoption of SSM practices in sub-Saharan Africa with negative nutrient balances a key issue in many countries (Falconnier *et al.*, 2023; Sileshi *et al.*, 2022). One major challenge is the lack of access to agricultural inputs and technology, including fertilizers, seeds and equipment. Small-scale farmers often struggle to access these resources, which can limit their ability to adopt sustainable practices that require initial investment.

Another challenge to SSM is the lack of knowledge and awareness among farmers about the benefits of healthy soil and the availability of SSM practices. Many farmers may not be aware of the best ways to maintain soil health or the risks associated with practices that degrade soil quality, particularly those that lead to soil erosion and nutrient mining.

In addition, some of the cultural and social factors that have traditionally shaped agricultural practices in sub-Saharan Africa can be obstacles to adopting SSM practices. For example, the preference for intercropping and mixed cropping may be at odds with the need to maintain soil cover with single crop systems. Similarly, traditional beliefs about the use of manure and other organic fertilizers may need to be modified or adapted to more sustainable practices.

Finally, there are economic and institutional factors that may impede the adoption of SSM practices in sub-Saharan Africa and these are examined in more detail in Chapter 6 of this report. These include access to credit and financing, land tenure issues, and inadequate policy frameworks and regulatory environments to support SSM initiatives. Additionally, political instability and weak governance can also hinder progress in SSM. In some African countries, lack of regulatory enforcement has resulted in unsustainable land-use practices and soil degradation.

The adoption of SSM practices is essential for improving agricultural productivity and promoting food security in sub-Saharan Africa. While there have been encouraging developments in this area, there are still significant challenges that need to be addressed to achieve widespread adoption of SSM practices. Challenges range from access to resources and technology to cultural and social factors, and require a coordinated effort to address them at the regional or national level.

14.6.3 Recommendations for future priorities for action

Based on the progress made and impediments to SSM practices in sub-Saharan Africa, the following recommendations should be considered as future priorities for action:

- *Increase access to agricultural inputs and technology:* Governments should prioritize investments in infrastructure and programmes that increase access to and sustainable use of affordable and high-quality agricultural inputs, such as fertilizers, seeds and equipment. Additionally, programmes focusing on access to credit and financing should also be implemented to support small-scale farmers.
- *Improve knowledge and awareness about SSM practices:* Education and extension services should be developed to provide farmers with knowledge and skills to maintain soil health. These programmes should be designed to reach a diverse range of farmers and promote the benefits of SSM practices.
- *Promote policy frameworks and regulatory environments:* Governments should create policies and regulatory frameworks that support SSM practices. These frameworks should address issues such as land tenure and ensure that sustainable land and soil management practices are incentivized and included in national agricultural plans. This should also include laws governing agricultural land-use, environmental protection and natural resource management. Governments should also increase transparency and accountability in regulatory enforcement.
- *Address cultural and social factors:* Cultural and social factors that influence agricultural practices should be addressed through education and outreach programmes. These programmes should focus on promoting the key benefits of SSM practices, while also incorporating existing beliefs and cultural practices.
- *Increase investment in research and development:* Investment in research and development should be prioritized to develop new or improved SSM practices that are more context-specific and relevant to sub-Saharan African farmers.
- *Foster partnerships among stakeholders:* Partnerships among governments, farmers, research institutions and private sector entities should be facilitated. This collaboration will help to develop and implement effective and SSM practices.
- *Promote regional cooperation:* Regional cooperation among neighbouring countries can help to address cross-border issues related to soil management. This includes harmonizing policies and regulations across borders, sharing knowledge and technical expertise, and collaboration on research and development.



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Chapter 15 | Global assessment of state and trend in the adoption of sustainable soil management practices

15. 1 Overview of the methodology for assessing the state and trend in the adoption of sustainable soil management practices

This chapter summarizes the assessment of the state and trends in the adoption of SSM practices aimed at addressing threats to soil functions across the seven regions of the GSP, over the period 2015–2024 (Figure 15.1). It synthesizes the findings from earlier chapters into a unified global overview.

As indicated in Chapter 1, the SWSR 2026 report evaluates the effectiveness of soil management practices that were in place as of 2015 to address soil-related threats, while also tracking the direction of change in SSM adoption between 2015 and 2024. Unlike previous assessments, it does not present a global biophysical status map or a direct update of the 2015 condition ratings but instead focuses on management practices and their evolution over the past decade.

Figure 15.1. Regions used in the Status of the world's soil resources report



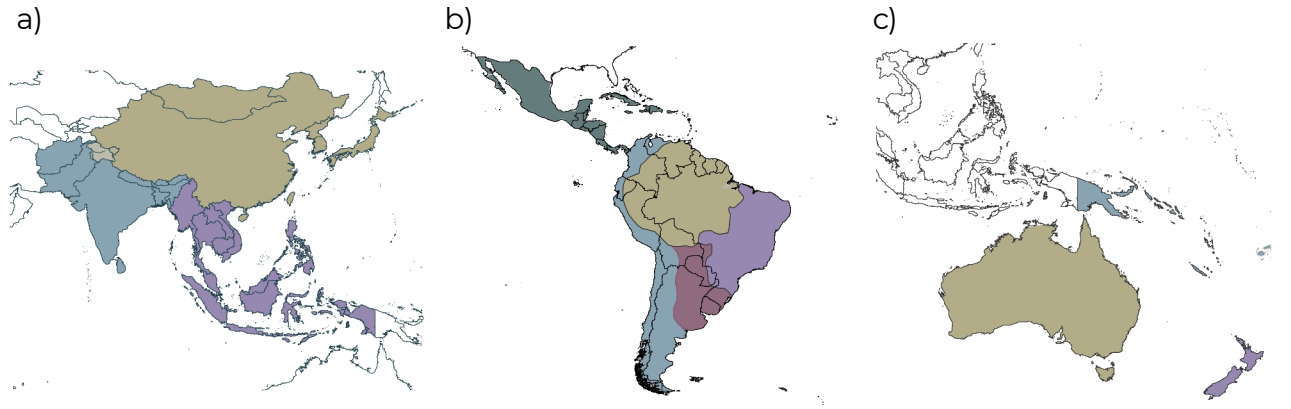
Refer to the disclaimer on page ii for the names and boundaries used in this map.

The threats considered in the report are soil erosion, nutrient mismanagement, the loss of SOC, salt accumulation, soil pollution, the loss of soil biodiversity, and urbanization and soil sealing. The state reflects the effectiveness of management practices in place as of 2015 in preventing or reducing a specific soil threat. At the regional or subregional scale, changes in management occur at decadal or longer time scales and hence the state as of 2015 is relevant for the 2015 to 2024 assessment period. A *very poor* state indicates that existing practices have allowed negative impacts to occur, whereas a *very good* state indicates that the practices have minimized the threat to the lowest possible level for that region. The **trend** indicates the direction of change in the adoption of SSM practices between 2015 and the finalization of this report (late 2024). In short, trend reflects how SSM adoption has progressed over time. An *improving* trend indicates that the adoption of SSM practices is becoming more widespread, and the associated threat is being reduced. A *deteriorating* trend indicates that SSM practices have not been widely adopted, or that the area under SSM has declined, resulting in an increased level of threat.

The assessments focused on managed lands, including arable land, managed forests and areas affected by urban or infrastructure development.

In this report, ITPS members from three regions presented their assessments at the subregional level rather than for the entire region (Figure 15.2). This approach was chosen due to large disparities between subregions, making a meaningful regional-level assessment unfeasible, while the information proved to be sufficient for reliable subregional assessments.

Figure 15.2. Subregional divisions: a) Asian region, b) Latin America and the Caribbean region and c) Southwest Pacific region



Note: The boundaries of the subregions included in the Latin America and the Caribbean map have been established for illustrative purposes only and do not correspond to well-defined biomes or watersheds. Some subregions include multiple biomes but are named after the predominant biome. For more details, please refer to the descriptive text for each subregion.

Refer to the disclaimer on page ii for the names and boundaries used in this map. Dotted line represents approximately the Line of Control in Jammu and Kashmir agreed upon by India and Pakistan. The final status of Jammu and Kashmir has not yet been agreed upon by the parties. Final boundary between the Republic of Sudan and the Republic of South Sudan has not yet been determined. A dispute exists between the Governments of Argentina and the United Kingdom of Great Britain and Northern Ireland concerning sovereignty over the Falkland Islands (Malvinas).

15.1.1 Assessment of confidence

The assessors have expressed their confidence in state and trends evaluations using high, moderate and low ratings. The confidence level is dependent on the type, amount, quality and consistency of available evidence and the degree of scientific agreement among assessors and within the literature. The ratings follow the IPCC confidence terms (Kause *et al.*, 2022) (see Section 7.2 for more details).

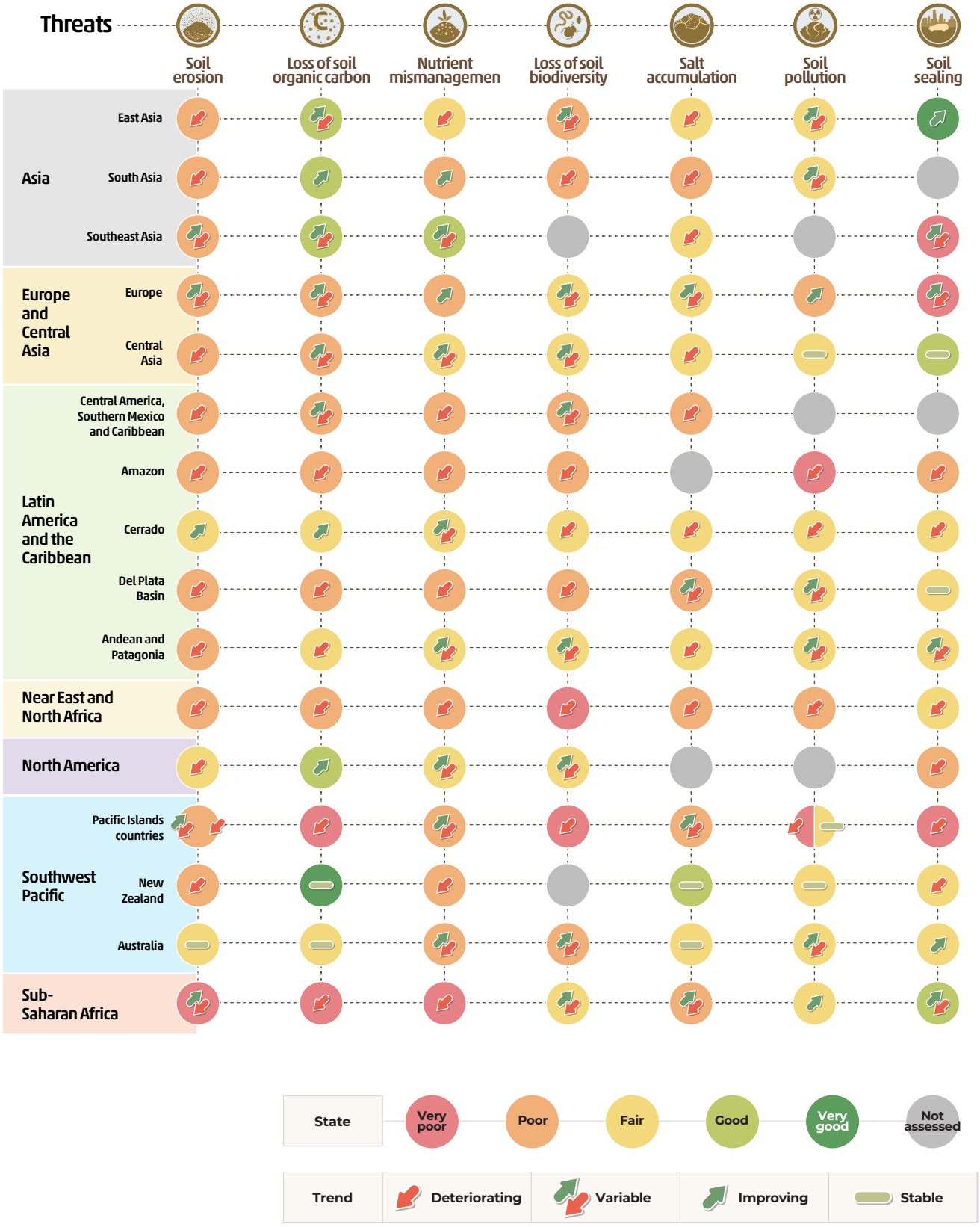
Confidence ratings are generally consistent across the regions and subregions (Figure 15.3). Confidence is highest for soil erosion, with most regions being assigned a high rate. High confidence is also reported for SOC loss, nutrient mismanagement, and urbanization and soil sealing. Soil

pollution carries a medium level of confidence while confidence in assessments of salinization and sodification varies by region. Confidence is generally low for the loss of soil biodiversity. The 2015 report (FAO and ITPS, 2015) also noted limited regional and national data on changes to soil biodiversity.

Figure 15.3 presents the state and trend assessments in the adoption of SSM practices for the seven global soil threats by region. The following sections summarize the state and trend for each individual threat.



Figure 15.3. Summary of global assessment of state and trend in the adoption of sustainable soil management practices to address threats to soil functions



Note: See Chapter 8 to Chapter 14 for details.

15.2 Global assessment of state and trend in the adoption of sustainable soil management practices to address threats to soil functions

Overall, the current management of global soil resources remains unsustainable. More than half (52.5 percent) of the regional and subregional state assessments rate the soil management as poor or very poor, 33 percent rate the soil management as fair and only 14 percent consider it good or very good (Figure 15.3). The adoption of SSM is reported to be deteriorating in 58 percent of the regional and subregional assessments, stable or variable in 32 percent and improving in just 10 percent.

These findings indicate that current soil management practices are undermining soil functions and the delivery of ecosystem goods and services they support. As such, they do not comply with the definition of SSM outlined in the *Revised World Soil Charter* and therefore are not sustainable (FAO, 2015).

The current assessments show a slight decline compared to 2015. In 2015, 54 percent of regional assessments rated the state as fair, good or very good and 13 percent of trend assessments showed improvement. These figures drop to 49 percent and 10 percent, respectively in the current assessments. Regional assessment chapters (Chapter 8 to Chapter 13) note that the current assessments have been able to draw on longer term data and more structured analyses across many studies that were not as widely available in 2015.

The top three threats – soil erosion, the loss of SOC and nutrient mismanagement – maintain the same ranking (level of severity of the threat) in both the 2015 and 2025 reports. However, the loss of soil biodiversity rises to fourth place in the 2025 report compared to sixth in the 2015 report, likely reflecting growing recognition of its importance to soil health.

The assessed state and trend of the current management of global soil resources align with the major drivers of soil change discussed in Chapter 4. Between 2015 and 2024, the global population increased by 0.8 billion people (UN DESA, Population Division, 2024), raising the demand for food, wood and other commodities. The increased demand, along with rising needs for fuel and fibre have led to the expansion of harvested area of main crops (FAO, 2024b). From 2000 to 2022, the total harvested area rose by 24 percent, reaching 1.47 billion ha. Between 2015 and 2022 alone, it grew by 80 million ha (FAO, 2025).

Expanding harvested areas and shifting diets are bringing more land into agricultural use. Converting forests to cropland increases erosion and depletes SOC, alters microbial communities and disrupts natural nutrient cycles. FAO (2024a) estimated a net annual forest loss of 4.7 million ha per year between 2010 and 2020. Between 2010 and 2020, forest lost was greatest in sub-Saharan Africa, the NENA region (driven largely by deforestation in the Sudan) and Latin America and the Caribbean (FAO, 2024b).

Growing demand for food and other commodities has also driven increased use of inorganic fertilizers. Usage rose between 2015 and 2020 but declined across all three major types (N, P and K) between 2020 and 2022. FAO (2024a) attributed this decline to higher fertilizer prices linked to rising natural gas costs and supply chain disruptions from conflicts.

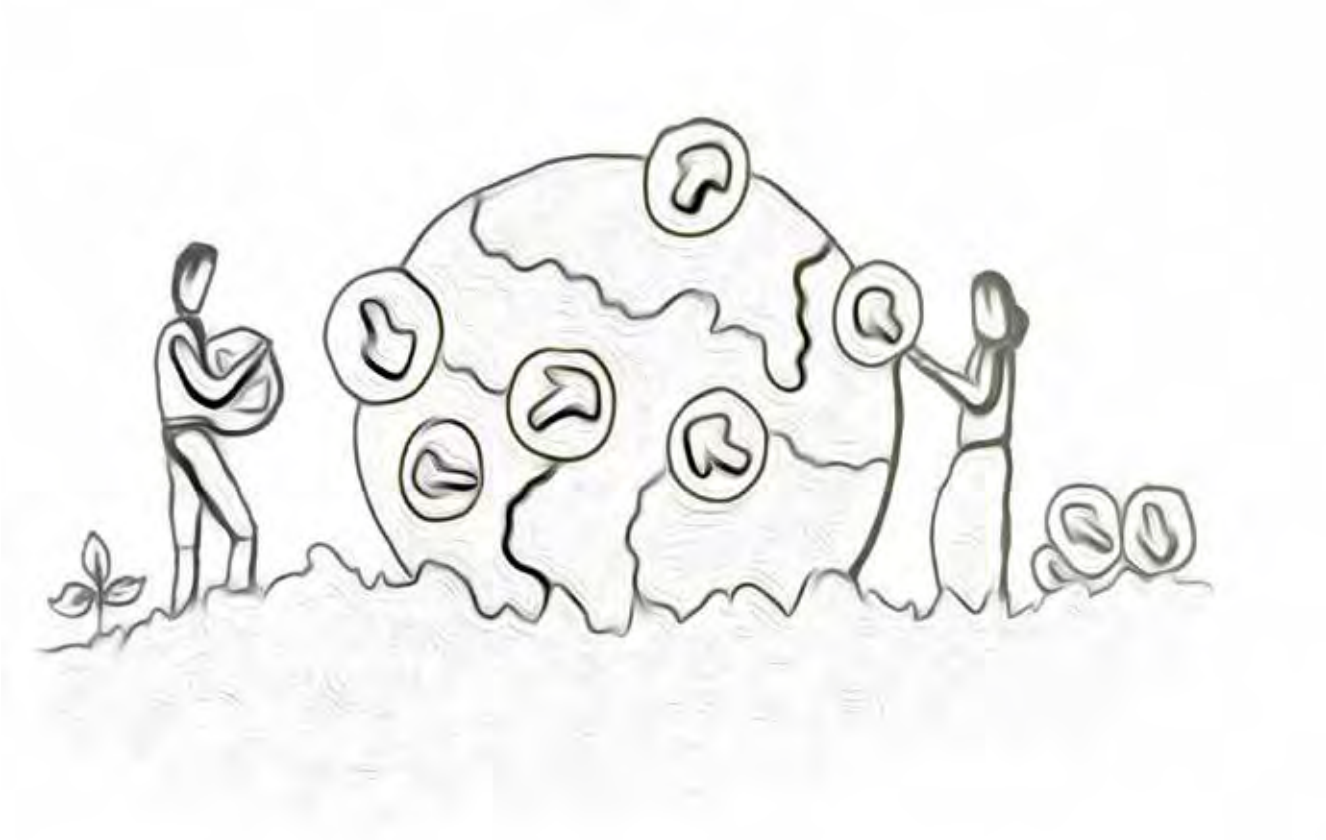
Adopting SSM practices reduces threats to soil functions (FAO and ITPS, 2017). More accurate data on SSM adoption would help quantify reductions in those threats but such information remains limited, even at the regional level.

Kassam, Friedrich, and Derpsch (2022) estimated that approximately 205 million ha were under conservation agriculture, including no-till or reduced tillage, in 2018/19, accounting for about 14.7 percent of global cropland. Cover crops offer similar benefits but data on their adoption is scarce (Feng *et al.* 2024).

Overall, the rate of urbanization is projected to slow in many areas through 2030. Predictions of urban growth between 2018 and 2030 suggest lower rates of urban expansion than those observed from 1990 to 2018 (UN DESA, Population Division, 2019). The highest growth is expected in sub-Saharan Africa, the Near East and a few cities in India and China. Van Vliet, Eitelberg and Verburg (2017) estimated that by the year 2000, urbanization had already claimed 166 million ha of land previously available for food production (about 4.7 percent of cultivable land worldwide). By 2040, the loss of cultivable land due to urbanization could rise to 8.8 percent globally, with the highest projected losses in China (16.4 percent) and Europe (18.0 percent).

There is increasing recognition that armed conflicts severely damage soils (e.g. Certini and Scalenghe, 2024) (see Section 4.4). Violent conflicts have reached their highest levels since the Second World War (UN, 2023). Even after hostilities cease, activities such as the movement of heavy vehicles, digging, and munitions explosions continue to compact soil, disturb soil layers and degrade soil structure, leading to erosion, furrowing and mud formation. These processes reduce available arable land and soil productivity (Broomandi *et al*., 2020; Certini, Scalenghe and Woods, 2013).

Conflicts in the Gaza Strip and Ukraine illustrate these impacts. FAO, UNITAR and UNOSAT (2025) reported that as of 28 September 2025, 86.9 percent of cropland in Gaza had been damaged. In Ukraine, by 2024, the armed conflict had affected 9.2 million ha of black soils (Baliuk, Kucher and Plisco, eds, 2024).



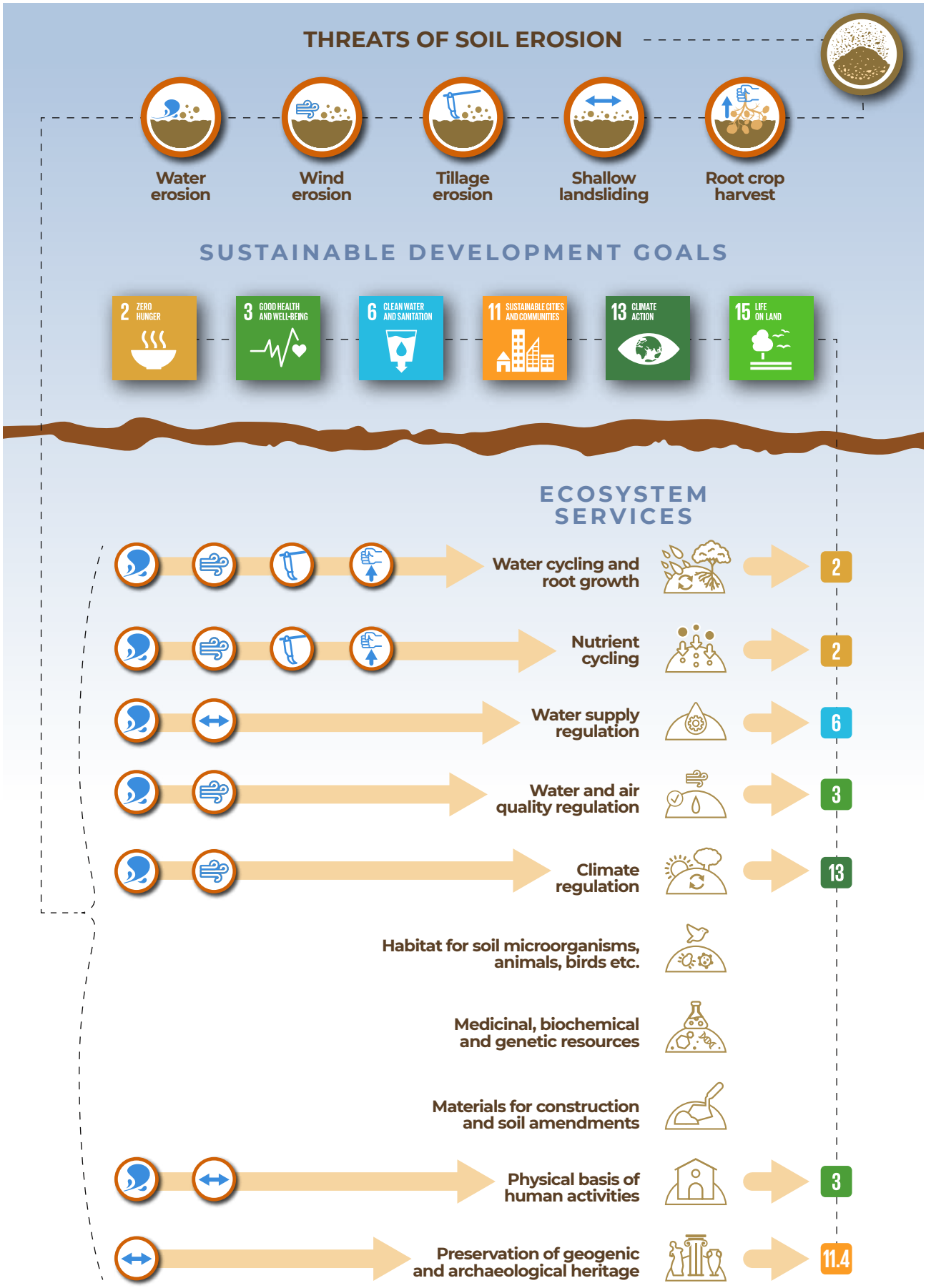
15.3 Global state and trend for individual soil threats

15.3.1 Soil erosion

Soil erosion directly affects human well-being by degrading topsoil, which is essential for food security, water quality, climate resilience and ecosystem health. The primary drivers of soil erosion include water and wind, along with human-induced factors such as tillage, shallow landsliding and the harvesting of root crops. Erosion reduces soil health and productivity, thereby impacting critical ecosystem services offered by soil, like water regulation, nutrient cycling and carbon storage. As a result, soil erosion poses a direct threat to progress on multiple SDGs that depend on the maintenance of functional soils (Figure 15.4). Specifically, soil erosion affects SDG 2: Zero Hunger, SDG 3: Good Health and Well-being, SDG 6: Clean Water and Sanitation, SDG 13: Climate Action and SDG 15: Life on Land, as discussed in Chapter 2 of this report. Section 3.3.2 also highlights the link between erosion and the loss of SOC which further affects SDG 13.



Figure 15.4. Major adverse effects of soil erosion on ecosystem services and soil-related Sustainable Development Goals



Across the seven regions, 81 percent of assessments judge the state of soil management to be poor or very poor, with 65 percent reporting a deteriorating trend since 2015 (Figure 15.5 gives an overview and Table 15.1 gives the summaries in more detail). The state of soil erosion has worsened since 2015 in Europe and Central Asia, sub-Saharan Africa, the Pacific Island countries, and New Zealand. Deforestation and other land-use conversions are cited as major drivers of high erosion rates in several regions, as is the lack of soil cover associated with current management practices.

Figure 15.5. Regional and subregional summaries for state and trend in the adoption of sustainable soil management practices to address soil erosion

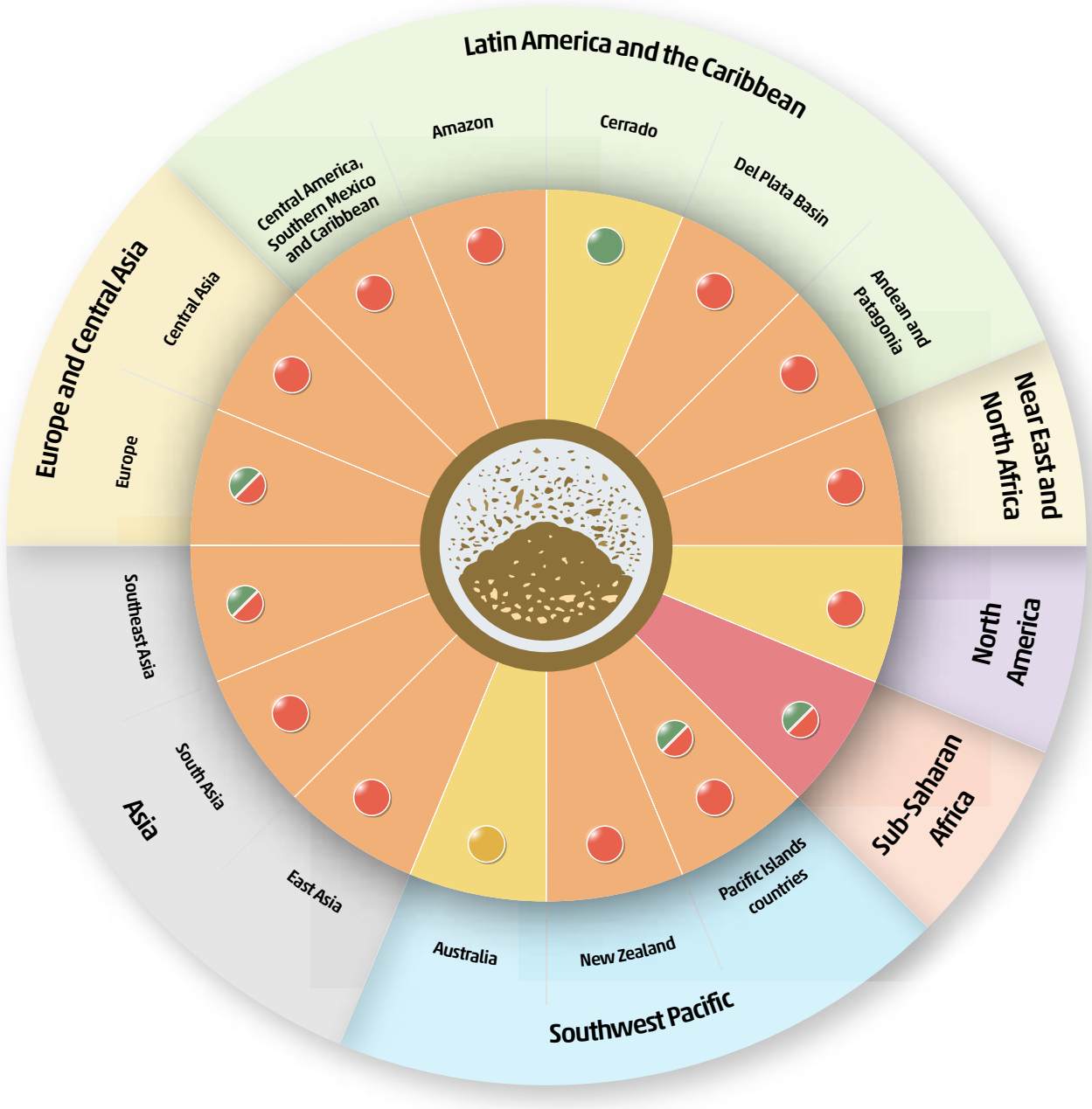


Table 15.1. Regional and subregional summaries for state and trend in the adoption of sustainable soil management practices to address soil erosion

Region	Summary
Asia	**East Asia**: Complex topography, heavy rainfall, lack of soil cover, and predation damage of the understorey of vegetation by wild animals also promote soil erosion.
	South Asia: The increase in soil erosion is mainly attributed by the conversion of forest and other natural vegetation into agricultural land.
	Southeast Asia: Intensive agriculture, shifting cultivation and heavy rain in sloping areas increase the soil erosion in most of the Southeast Asia countries.
Europe and Central Asia	**Europe**: Soil erosion affects half of European Union agricultural land and a substantial part of agricultural lands in other parts of the region. Common Agricultural Policy measures have been positive to reduce erosion in some areas of the European Union. **Central Asia**: Despite soil conservation measures, erosion of black soils under intensive farming is still increasing in Central Asia.
Latin America and the Caribbean	**Central America, southern Mexico and the Caribbean**: Areas of bare soil, deforestation and coastal erosion by rising sea levels have increased wind and water erosion.
	Amazon: The expansion of pasture and cropland has led to deforestation and higher erosion with low clay soils being especially vulnerable (landsliding common).
	Cerrado: Deforestation, overgrazing and poor pasture management have increased erosion, while improved management (including no-till) has decreased erosion.
	Del Plata Basin: There is water erosion in steep humid landscapes and wind erosion in semi-arid areas. The limited adoption of SSM techniques and extreme weather is increasing erosion.
	Andean countries and Patagonia: There is increasing water erosion due to intensive cropping in previously forested land and more frequent fires, and wind erosion due to poor management in western Argentina and Patagonia.
Near East and North Africa	Water erosion is severe in the mountains of coastal regions throughout the region, in inland wadis and in croplands. Wind erosion is high in approximately 70% of the NENA region (especially southern areas of North Africa) with low risk only in the coastal regions. There is limited adoption of improved soil cover practices.
North America	Soil erosion remains a significant threat. Of significant concern is that tillage is increasing after decades of decline. A growth in herbicide resistance and crop residue production may explain the increase in tillage. Anthropogenic wind erosion is high in the Sonoran Desert of northern Mexico and the high plains of the United States.
Sub-Saharan Africa	There is severe water erosion in highland, tropical forest and temperate forest biomes, while there is high soil loss due to wind erosion in savannah, semi-desert and desert biomes. Deforestation in forested regions is a major driver. Socioeconomic constraints such as the competing uses for crop residues throughout region limit the adoption of SSM.
Southwest Pacific	**Pacific Island countries**: There is severe erosion associated with overgrazing and high rainfall, and with coastal erosion due to rising sea levels. The trend is deteriorating for higher islands and variable for atolls.
	Australia: There is a high proportion of cropland in no-till and reduced tillage, which has substantially decreased erosion.
	New Zealand: There is high water erosion in areas with erodible geology. Landsliding is a significant threat due to terrain and geology.

Three areas are judged to be in a fair state for soil erosion: the Cerrado subregion in Latin America and the Caribbean, Australia in the Southwest Pacific, and North America. These areas report widespread adoption of no-till or reduced tillage with residue retention. However, North America shows a deteriorating trend, partly because tillage appears to have increased in some areas due to herbicide-resistant weeds.

Global modelling studies suggest that water erosion rates on cropland exceeded acceptable thresholds (i.e. greater than 6 tonnes per hectare [t/ha] annually). Borrelli *et al.* (2017) estimated an average annual soil loss of 13 t/ha per year in 2012. Carr *et al.* (2020) estimated average water erosion rates of 11 t/ha per year for maize-producing land and 6 t/ha annually for wheat-producing land. While

both studies highlight high erosion levels, there is limited agreement on which areas experience the most severe erosion. A synthesis of field-based observations concluded that agricultural lands have the highest water erosion rates, whereas forest and shrublands exhibit the lowest, although variability remains high within each land-use class (García-Ruiz *et al.*, 2015).

FAO and ITPS (2015) estimated that water erosion reduced crop yields by 0.3 percent annually on affected agricultural land. Global annual reduction in food production due to erosion ranged from 0.4 percent (Sartori *et al*., 2019) to less than 1 percent of average global maize and wheat production between 2013 and 2018 (Carr *et al*., 2021). Although erosion-related yield losses are substantial, overall economic losses (considering on- and offsite effects) due to water erosion are likely to exceed those from yield decline alone (e.g. Graves *et al*., 2015). However, no regional or global estimates are currently available (see Section 3.2.3).

Modelling studies broadly agree on the locations of high dust emissions from wind erosion, although per hectare estimates vary significantly. Most dust emissions originate from natural sources, particularly in North Africa, Saudi Arabia, the Taklamakan Desert of East Asia, the southwest of the United States of America and northern Mexico, the deserts of Chile and Peru in South America and the Namib Desert in southeast Africa (Ginoux *et al*., 2012; Chappell *et al.,* 2023). Anthropogenic dust emissions are also significant in the Sahel in sub-Saharan Africa, the highlands of Saudi Arabia, northern China and Mongolia, the high plains of the United States, the Sonoran Desert in Mexico, river valleys in Patagonia, southeast Africa, and areas in Australia (Ginoux *et al*., 2012; Chappell *et al.,* 2023). Despite the known extent, no global studies have estimated the economic impacts of wind erosion.

Tillage erosion is recognized as a major contributor to soil loss. However, global and regional models still lag behind those for water and wind erosion. A synthesis of field studies reported annual average tillage erosion rates of 30.1 t/ha, compared to 6.3 t/ha for wind erosion and 7.4 t/ha for water erosion (mainly sheet and rill erosion) (Borrelli *et al*., 2021). No global economic assessments of tillage erosion are currently available.

Meta-analyses since 2015 show that vegetation cover – either from living plants or crop residues – is the most effective way to control and limit water and wind erosion in agricultural soils (FAO, 2019). Practices such as residue retention (common in no-till systems), cover crops and mulching protect the soil surface and reduce erosion. Du *et al.* (2022) found that cover crops reduced water erosion by 92 percent in cropland; Liu *et al.* (2021) reported a 73 percent reduction in tree–crop systems. Water erosion control is more effective when crop residues remain on the soil surface in no-till systems than if residue is removed (Mhazo, Chivenge and Chaplot, 2016; Xiao *et al*., 2021). Other effective strategies include vegetative measures (e.g. agroforestry and grass strips along the slope contour) and structural interventions (e.g. terraces, trench-ridge systems and stone bunds) (Xiong, Sun and Chen, 2018). Traditional terraces are especially effective soil conservation measures, and their abandonment or removal often results in longer, steeper slopes that worsen soil erosion (Wei *et al*., 2015), which is rarely offset by cover crops.

Less common but highly destructive erosion processes such as gullying and shallow landsliding require specialized interventions. Gullying can be mitigated by preventing runoff from reaching gully heads and by installing check dams to reduce water's erosive force (Bartley *et al*., 2020; Frankl *et al*., 2021). Shallow landsliding is frequently triggered by forest removal, so maintaining forest cover in vulnerable, landslide-prone areas is an effective preventative strategy (Dewi and Wijayanti, 2023).

Adopting no-till systems inherently reduces tillage erosion. However, its full effectiveness depends on proper implementation, such as controlling implement speed and depth (Öttl *et al*., 2022).

Erosion prevention and control practices are most successful when applied through a landscape-level approach to soil and water conservation. These approaches target high-risk land units (such as catchments and river basins), by establishing forest or grass cover and conserving (traditional)

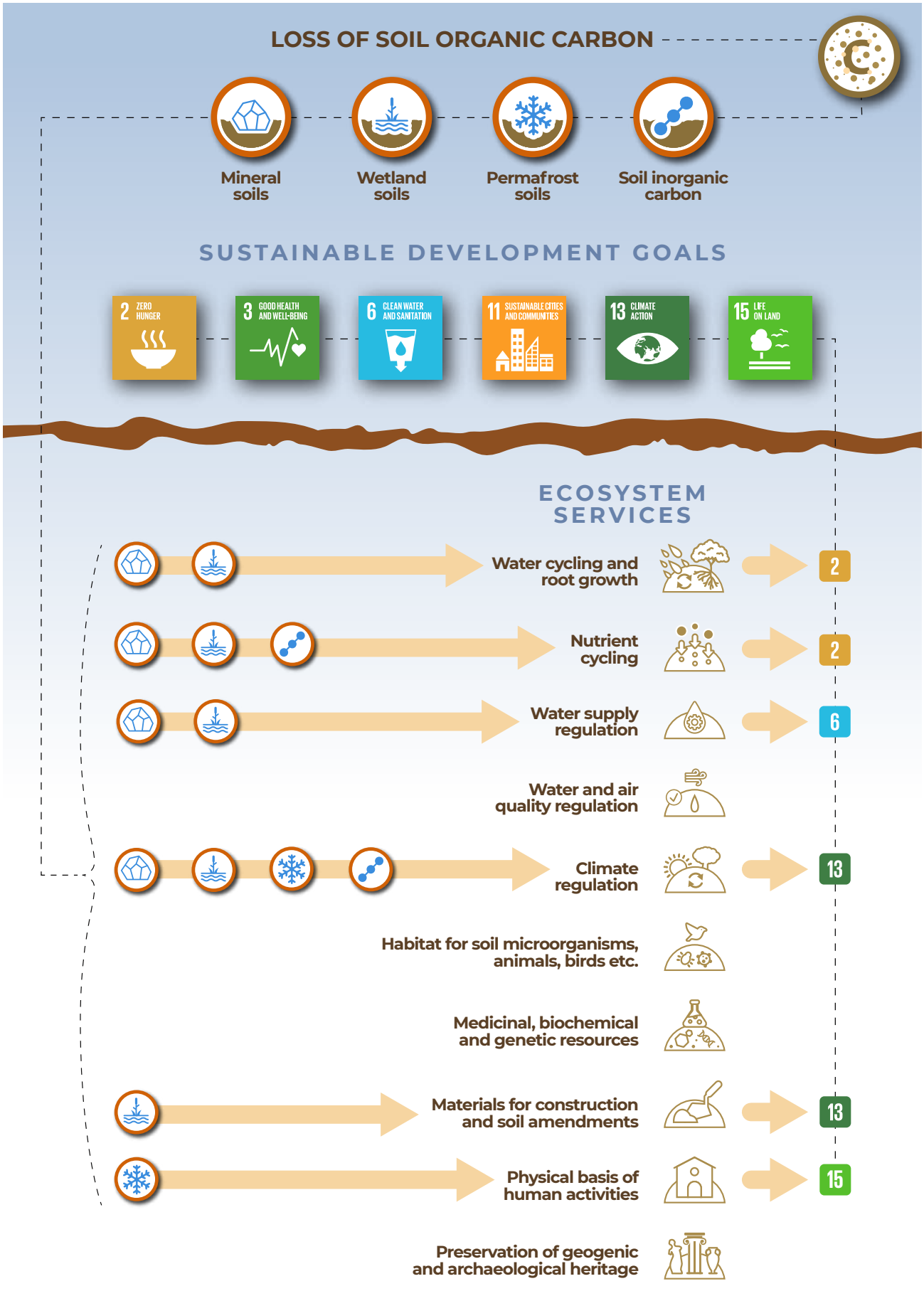
terraces. On less vulnerable areas, agronomic and vegetative practices can be implemented to complement structural measures (Shames, Heiner and Scherr, 2017).

15.3.2 Loss of soil organic carbon

Soil organic carbon is essential for human well-being, because it supports soil fertility, food security, water retention and climate regulation. The loss of SOC is both an environmental and a socioeconomic concern, affecting vulnerable communities around the world. Soil organic carbon-rich (black) soils are the most productive soils contributing to global food security. The loss of SOC threatens progress towards several SDGs, including SDG 2: Zero Hunger, SDG 13: Climate Action and SDG 15: Life on Land (Figure 15.6). Soil organic carbon is a key component of soil organic matter, which serves as an important nutrient source and plays a critical role in the formation and maintenance of good soil structure, facilitating optimum water movement. Soils are the largest terrestrial carbon store. According to the IPCC Sixth Assessment Report, permafrost soils contain approximately 1 200 billion tonnes of carbon while mineral and wetland soils outside of the permafrost region store another 1 700 billion tonnes. By comparison, terrestrial vegetation holds about 450 billion tonnes of carbon (Canadell *et al.*, 2021).



Figure 15.6. Major adverse effects of loss of soil organic carbon on ecosystem services and soil-related Sustainable Development Goals



The state of soil management aimed at reducing SOC losses is judged to be very poor in sub-Saharan Africa and the Pacific Island countries and poor in Europe and Central Asia, the NENA region and several subregions of Latin America and the Caribbean, including Central America, southern Mexico and the Caribbean, the Amazon and the Del Plata Basin (Figure 15.7 gives an overview and Table 15.2 gives the summaries in more detail). The trend is deteriorating in three subregions in Latin America and the Caribbean, the NENA region, sub-Saharan Africa and the Pacific Island countries. Across many of these regions, deforestation and agricultural practices that leave the soil uncovered are identified as the main drivers of SOC loss.

Figure 15.7. Regional and subregional summaries for state and trend in the adoption of sustainable soil management practices to address loss of soil organic carbon

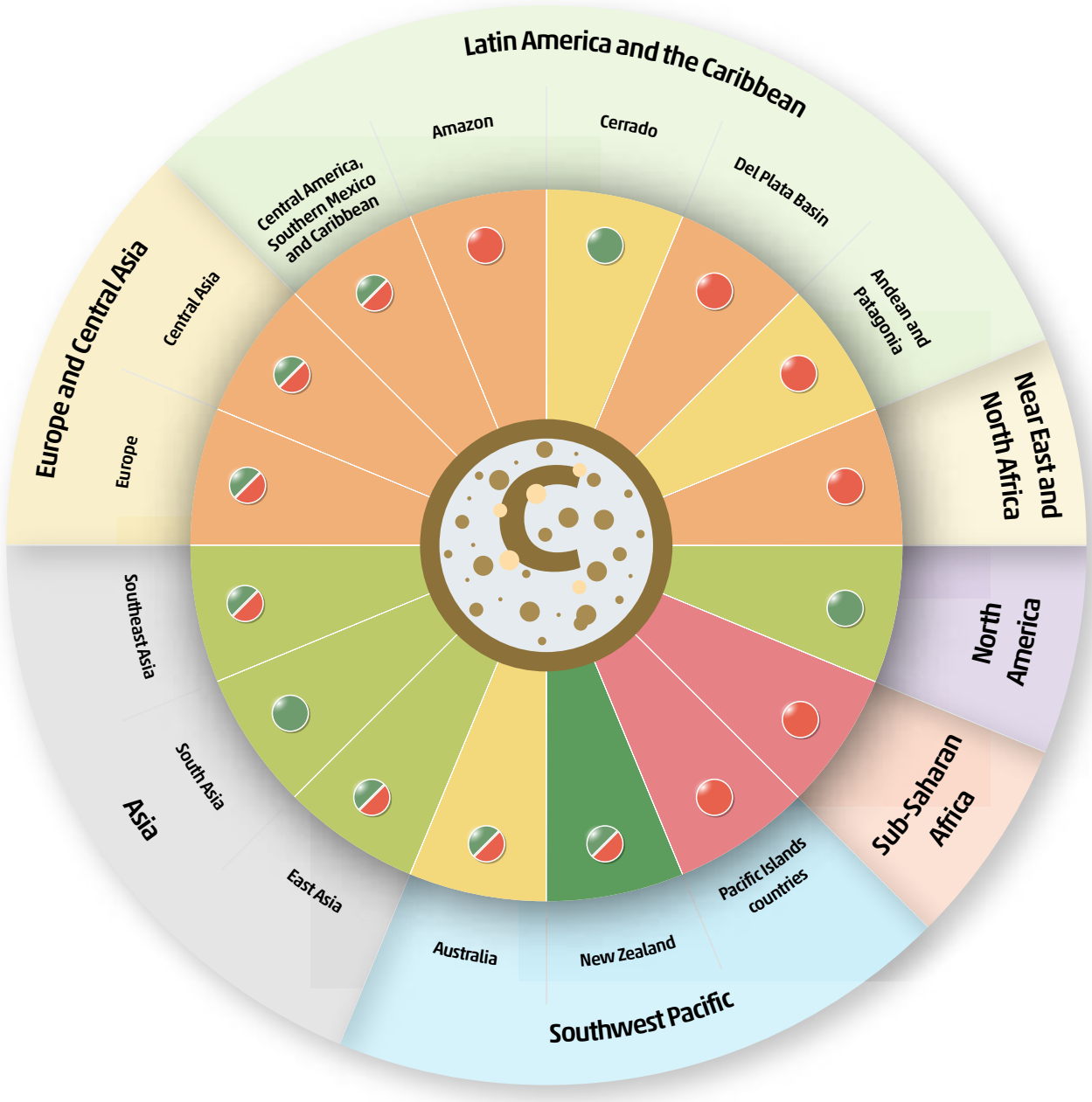


Table 15.2. Regional and subregional summaries for the state and trend in the adoption of sustainable soil management practices to address loss of soil organic carbon

Region	Summary
Asia	**East Asia**: Complex topography, heavy rainfall, lack of soil cover, and predation damage of understory of vegetation by wild animals also promote soil erosion. Reforestation in China has stabilized or increased SOC stores.
	South Asia: Change of land use and soil erosion has been a major driver for SOC decline. Reforestation in India has stabilized or increased SOC stores.
	Southeast Asia: Change of land use, high rates of deforestation, soil erosion and conversion of mangroves has been a major driver for SOC decline.
Europe and Central Asia	**Europe**: Mineral soils are improving due to reforestation in spite of carbon losses in agricultural soils; but there are still high losses in drained peatlands. Boreal soils have a good condition, although poor in more southern latitudes.
	Central Asia: The low content of organic carbon is observed in a large portion of agricultural lands of the region, especially in arid conditions.
Latin America and the Caribbean	**Central America, southern Mexico and the Caribbean**: Increases of SOC are likely in young volcanic soils but losses are found elsewhere.
	Amazon: A major driver of SOC loss is deforestation and conversion to pasture/cropland, No-till is ineffective at mitigating loss due to the rapid decomposition of organic matter. Overexploited areas suffer severe SOC depletion.
	Cerrado: Conservation practices (no-till, agroforestry, integration of crops, livestock, and forestry), play a significant role in mitigating erosive processes, with improvements in some areas of the Cerrado.
	Del Plata Basin: There are losses or no improvement of SOC content, mainly due to monocropping, too simple rotations and unsustainable grazing systems.
	Andean countries and Patagonia: There are losses due to megafires and an increase in soil temperature due to climate change, lower crop productivity, erosion and non-adoption of conservation practices, expansion of agriculture and livestock production in arid and semi-arid environments.
Near East and North Africa	There are SOC losses due to wind and water erosion and substantial expansion of urban lands, along with the high decomposition of SOC under mediterranean climates. There is very limited adoption of reduced tillage systems.
North America	Overall, SOC is increasing due to a growth in crop residue production, agricultural land abandonment and reforestation. Climate change is a major threat, as a substantial proportion of SOC is found in permafrost and peatlands.
Sub-Saharan Africa	There is mismanagement-induced decline of SOC due to deforestation, the complete removal of residues for competing uses and a high rate of oxidation due to tillage.
Southwest Pacific	**Pacific Island countries**: There is high SOC loss due to inappropriate management practices (e.g. burning fallow land, mechanization and agriculture intensifications resulting in shortened fallow).
	Australia: Many managed soils have low SOC, particularly cropping and rangeland soils in drier regions. Maintaining ground cover of >50% is essential to reduce SOC loss by erosion.
	New Zealand: Many managed soils have high SOC, particularly pastoral (grassland) soils. Soil carbon losses occur under intensive vegetable growing.

North America, South Asia and the Cerrado subregion report an improving SOC trend during the assessment period. Increased organic inputs and reforestation in some countries in South Asia and the widespread adoption of no-till in North America and the dominantly pastoral systems of New Zealand have helped maintain SOC levels.

Conversion of natural vegetation to cropland is the major driver of SOC losses (Karstens *et al.*, 2022). Meta-analyses show that, on average, converting forest to cropland results in a 25 percent SOC loss from upper 0–30 cm of soil (Smith *et al.,* 2016; Sanderman, Hengl and Fiske, 2017; Beillouin *et al.*, 2023), while converting grasslands to cropland causes about 16 percent loss (Beillouin *et al.*, 2023). By contrast, conversion of forest land to grassland does not significantly affect SOC levels (Beillouin *et al.,* 2023). Total SOC loss has been estimated at between 133 billion tonnes of carbon since the initial adoption of agriculture (Sanderman, Hengl and Fiske, 2017) to 227 billion tonnes between 1860 and 2010 (Smith *et al.*, 2016).

Human-induced climate change also contributes to carbon loss from soil to the atmosphere. Since the release of the IPCC Sixth Assessment Report, new research has strengthened the evidence that climate warming increases soil respiration and SOC loss (Hashimoto, Ito and Nishina, 2023; Ren *et al.*, 2024; Nissan *et al.*, 2023).

Preserving or enhancing existing SOC stores is critical for preventing further emissions of carbon from soils to the atmosphere and for maintaining soil functions. Land-use changes such as converting cropland to forest or grassland can significantly increase SOC stocks. Beillouin *et al.* (2023) estimated average SOC increases of 57 percent after conversion to forest and 26 percent after conversion to grassland. The conversion of cropland to perennial crops, orchards or agroforestry was also estimated to increase SOC by about 20 percent (Beillouin *et al.*, 2023).

Meta-analyses consistently find that adopting SSM practices in croplands, grasslands and forests can significantly increase SOC stocks. Among these, biochar application produces the greatest gain (Beillouin *et al.*, 2023), followed by manure applications. However, both management practices involve a transfer of carbon from production and processing to application sites, hence full life-cycle assessments are needed to evaluate their long-term carbon sequestration benefits. Concerns also remain about the scalability of biochar and whether it provides additional benefits beyond carbon sequestration in different soils (e.g. Bayeve, 2023).

Cover crops have also been shown to increase SOC (Beillouin *et al.*, 2023; Fohrafellner *et al.*, 2024; Bolinder *et al.*, 2020). Cover crops contribute to reduce wind and water soil erosion, helping protect SOC stocks. While SOC benefits are evident across climate zones, they are more variable in water-limited environments where cover crops may reduce available soil moisture, especially in drier years. In cold climates, cover crops and residue retention can increase snowmelt runoff and contribute to higher dissolved phosphorus (P) inputs to surface waters.

Reduced or no-till practices generally lead to higher SOC stocks in upper soil layers, although gains across the whole soil profile tend to be minimal (Haddaway *et al.*, 2017; Chenu *et al.,* 2019). Still, when combined with continuous soil cover, these practices also prevent soil erosion and further SOC loss.

Residue retention, a key component of no-till and reduced tillage systems in many regions can significantly impact SOC stocks. Crop residues may be left in the field (or retained), removed for other uses – such as animal feed, bedding or energy production – or burned. Physical removal is common in sub-Saharan Africa (e.g. Castellanos-Navarrete *et al.*, 2015) while burning is prevalent in Asian rice systems (Somasundaram *et al.*, 2020). Beillouin *et al.* (2023) found that residue retention increased SOC by an average of 12.9 percent, a gain similar to that from cover crops (12.8 percent). Li *et al.* (2023) noted the largest SOC increases in maize and rice systems and the smallest in wheat.

Agricultural carbon sequestration measures can meaningfully reduce emissions to the atmosphere. Roe *et al.* (2021) estimated that cost-effective sequestration measures in agriculture could rival reductions from decreasing deforestation and forest degradation.

Wetland soils – including organic soils in peatlands and mangrove forests and mineral soils in fens and swamps – store more SOC than other soil types. However, land-use change is a major source of carbon loss in these areas. Beillouin *et al.* (2023) estimated that converting wetlands to cropland results in an average SOC loss of 25 percent, comparable to losses from forest to cropland conversion. Preventing wetland conversion and reducing or eliminating peat extraction are central to preserving SOC stocks. Coastal mangrove forests also face multiple threats, including erosion, agriculture and aquaculture expansion, extreme climate events, clearing and urban expansion. Common protection strategies include policy interventions aimed at higher protection levels and economic and financial incentives to support conservation (Convention on Wetlands, 2021).

Finally, permafrost organic soils remain highly vulnerable to warming. Only reducing anthropogenic warming can prevent large-scale carbon release from these massive soil carbon reservoirs.

15.3.3 Nutrient mismanagement

This report examines three main forms of nutrient mismanagement:

1. **Nutrient mining** refers to the chronic undersupply of plant nutrients relative to the amount removed in harvested products. Over time, this depletes soil nutrient reserves and reduces organic matter.

2. **Nutrient excess** involves the chronic oversupply of nutrients beyond what crops absorb. Excess P and N pollute waterways, while NO_3^- poses risks to human health. Excess N also leads to the emission of N_2O, a GHG, and NH_3, which is harmful to human health.

3. **Soil acidification** is primarily driven by the application of acidic fertilizers. Increased acidity lowers soil fertility and damages both soil microbiota and larger soil organisms.

Nutrient mismanagement threatens progress towards several SDGs, including SDG 2: Zero Hunger, SDG 3: Good Health and Well-being, SDG 6: Clean Water and Sanitation, SDG 13: Climate Action and SDG 15: Life on Land (Figure 15.8).

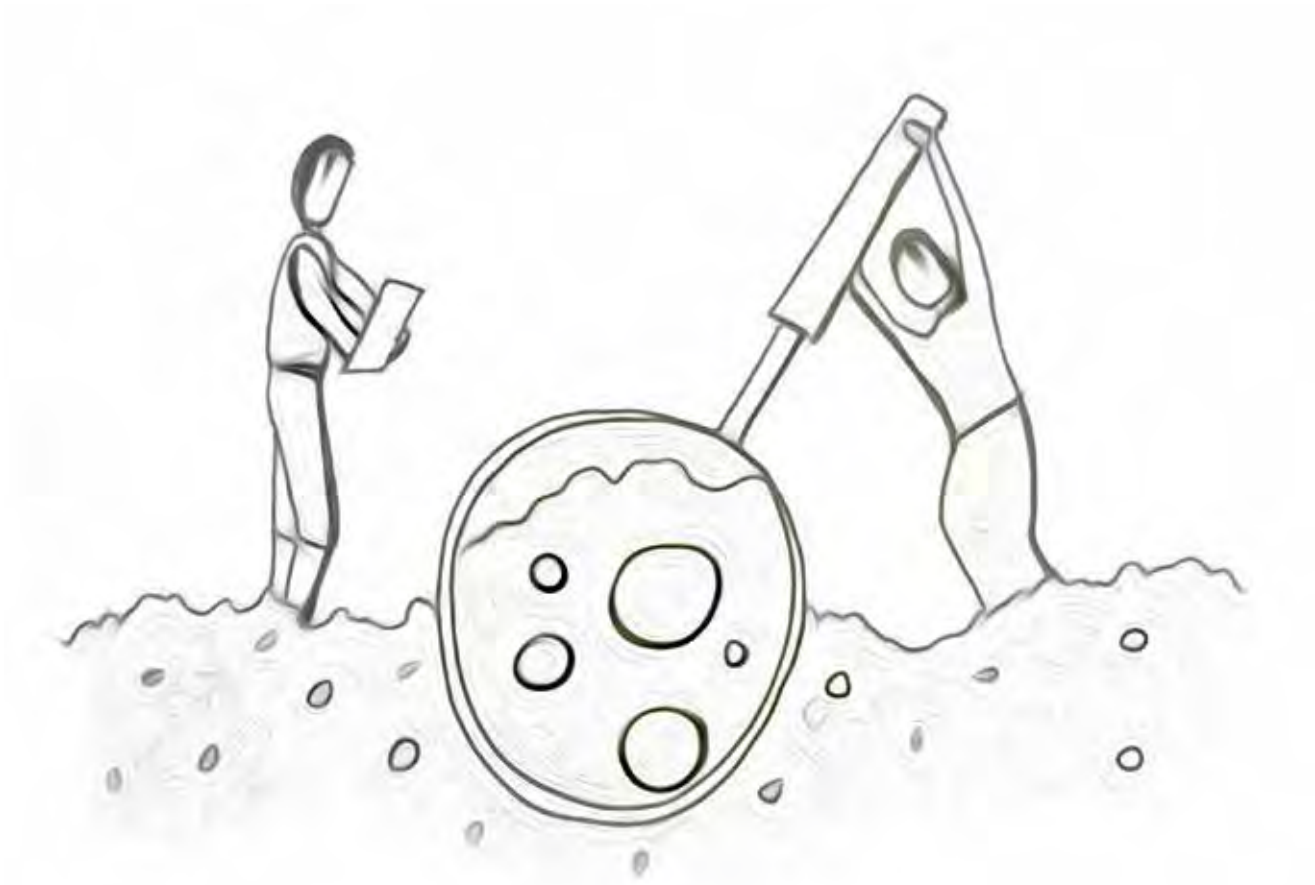
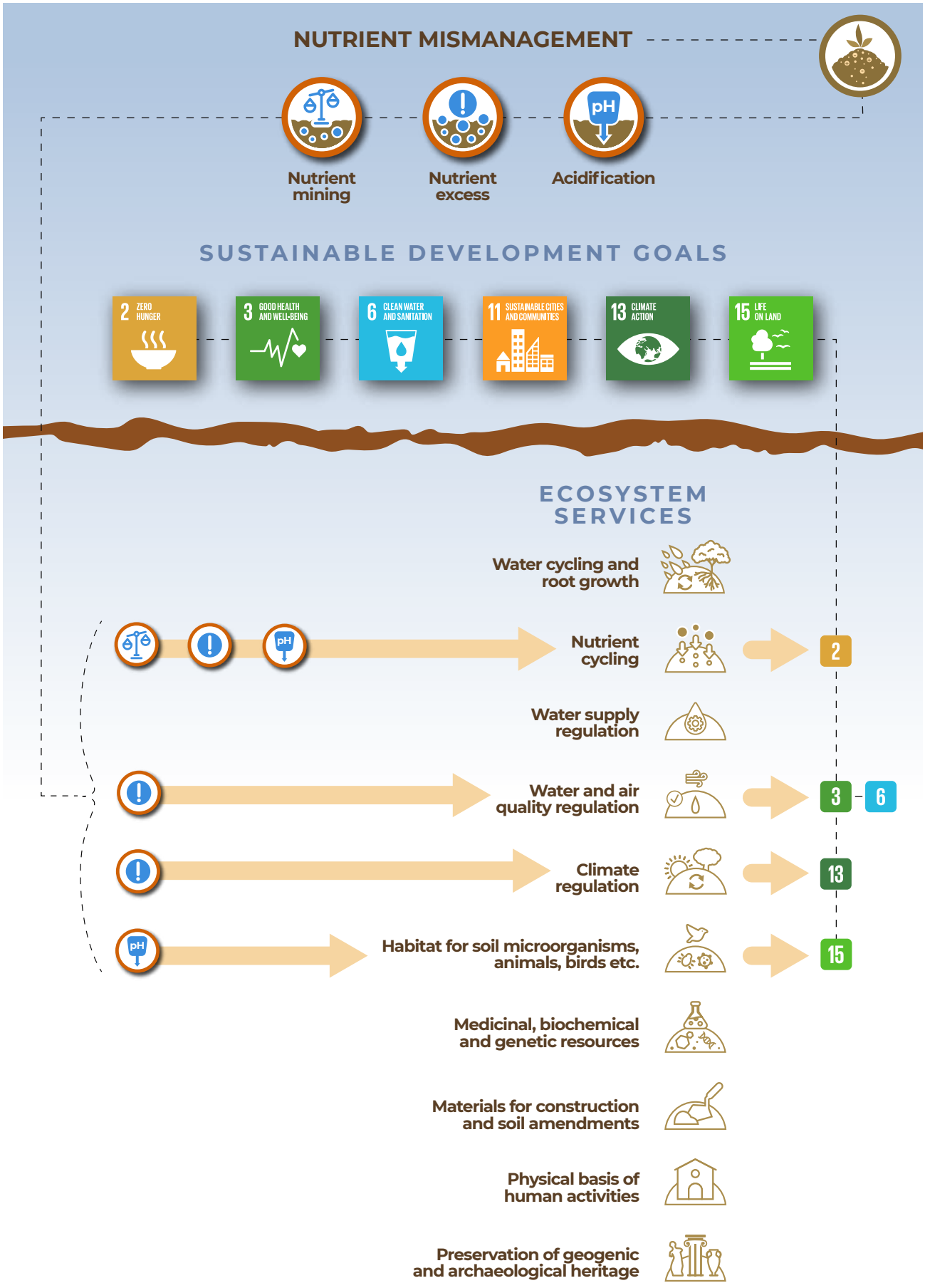


Figure 15.8. Major adverse effects of nutrient mismanagement on ecosystem services and soil-related Sustainable Development Goals



Nutrient mismanagement is ranked as the third most severe threat across the regions and subregions (Table 15.3). It ranks second in North America, the NENA region, sub-Saharan Africa, New Zealand and the Del Plata Basin subregion. In North America and New Zealand, the dominant issue is nutrient excess while in the NENA region, sub-Saharan Africa and the Del Plata Basin subregion, the main concern is nutrient mining. Armed conflicts have contributed to inadequate nutrient inputs in the NENA region and sub-Saharan Africa through disruptions to global fertilizer supply chains. In other regions and subregions, nutrient mismanagement typically ranks fourth or fifth. Several countries in Asia, Europe and Latin America and the Caribbean report significant levels of nutrient excess.

Across all assessments, 63 percent of regions and subregions rate the adoption of SSM practices to address nutrient mismanagement as very poor or poor and 50 percent report a deteriorating trend (Figure 15.9 gives an overview and Table 15.3 gives the summaries in more detail). North America is the only region judged to have a good state and an improving trend.



Figure 15.9. Regional and subregional summaries for state and trend in the adoption of sustainable soil management practices to address nutrient mismanagement

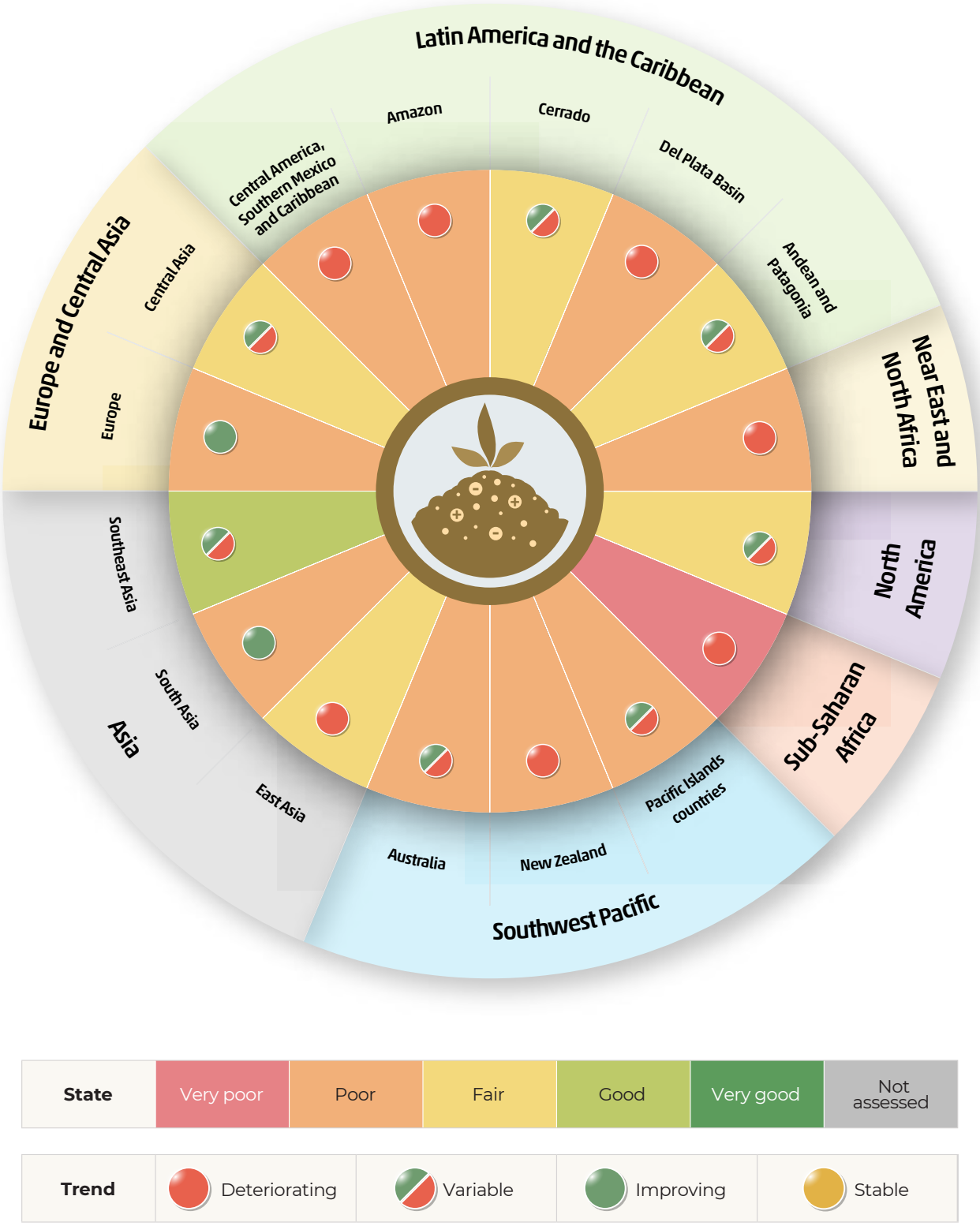


Table 15.3. Regional and subregional summaries for state and trend in the adoption of sustainable soil management practices to address nutrient mismanagement

Region	Summary
Asia	**East Asia**: There are excess N and P inputs in China, the Republic of Korea and Mongolia and excess P in Japan. Government policies aimed at reducing the use of inorganic fertilizers are expected to lower nutrient surplus and its environmental effects.
	South Asia: There are excess N and P inputs in Bangladesh, India and Pakistan and probable P mining in Afghanistan. Removing crop residues from fields has accelerated nutrient depletion, especially K depletion.
	Southeast Asia: Most countries show acceptable N and P balances but there are excess P inputs in Viet Nam and P mining in the Philippines. Most countries are implementing site-specific nutrient management to improve the nutrient imbalance situation.
Europe and Central Asia	**Europe**: Excess N and P inputs remain a problem in Western Europe due to over-fertilizing, while nutrient deficit affects some other areas of the region. This is improving thanks to precision fertilization and increased circularity.
	Central Asia: No clear trends are observed in relation to this threat, with nutrient surplus or deficit depending on the region.
Latin America and the Caribbean	**Central America, southern Mexico and the Caribbean**: There are excess N and P inputs in the Bahamas, Barbados, Belize, Costa Rica, Panama and Trinidad and Tobago, together with possible N mining in Antigua and Barbuda, Haiti, Saint Kitts and Nevis and Trinidad and Tobago. Preserving soil biodiversity and addressing acidification and pollution are crucial for maintaining soil fertility.
	Amazon: Amazon soils naturally lack nutrients (P, K, Ca and Mg) and have high acidity and Al content, together with a high loss of nutrients from agricultural land and major agronomic and logistical limits to fertilization. Agroforestry and pasture management offer solutions, with N deficiencies mitigated by natural legume cover crops.
	Cerrado: There are major beneficial N inputs from biological N fixation and from the adoption of suitable crops but there is an uneven adoption of SSM generally.
	Del Plata Basin: There are persistent N,P and K deficits and more recent deficits of base elements and micronutrients. There is evidence of an acidification decrease trend in many locations.
	Andean countries and Patagonia: There is evidence of both surplus and deficits in nutrient levels, as well as some evidence of acidification in volcanic soil, due to excessive use of N fertilizers and successive cultivation.
Near East and North Africa	There is excess nutrient application of N and P in Kuwait, Qatar and the United Arab Emirates, and N in Egypt. There is significant nutrient mining due (in part) to conflicts in the Sudan, the Syrian Arab Republic and Yemen.
North America	Advances in crop genetics and management have continued to lower nutrient surpluses, however, major hotspots of nutrient loss remain. Manure-based nutrient management is a particular challenge.
Sub-Saharan Africa	There is substantial nutrient mining of N and P in Chad, Comoros, Côte d'Ivoire the Democratic Republic of the Congo, Eswatini, Equatorial Guinea, the Gambia, Ghana, the Niger and Togo. There are major socioeconomic impediments to increased inputs of synthetic fertilizers and competing uses for plant residues and manures.
Southwest Pacific	**Pacific Island countries**: There are significant nutrient decline in the islands due to intensive agriculture, reduced fallow systems and soil organic matter decline.
	Australia: Large areas under agriculture have soil acidity issues, which affects the nutrient balance. The problem is likely to get worse, especially for vulnerable soil types with low buffering capacity due to low levels of liming.
	New Zealand: Various degrees of high N and P inputs and losses to the wider environment are experienced due to intensive pastoral and vegetable growing land uses.

Acidification is identified as a separate threat in Asia, sub-Saharan Africa, Australia and the Pacific Island countries. At the national level, the severity of this threat can be high. For example, Desta *et al.* (2021) estimated that 45 percent of rainfed agricultural soil in Ethiopia were slightly to strongly acidic. In the Pacific Island countries, acidification is considered the most severe threat. The trend is also deteriorating in Australia, as well as in the Andean countries and Patagonia (Figure 15.10 gives an overview and Table 15.4 gives the summaries in more detail).

Figure 15.10. Regional and subregional summaries for state and trend in the adoption of sustainable soil management practices to address acidification

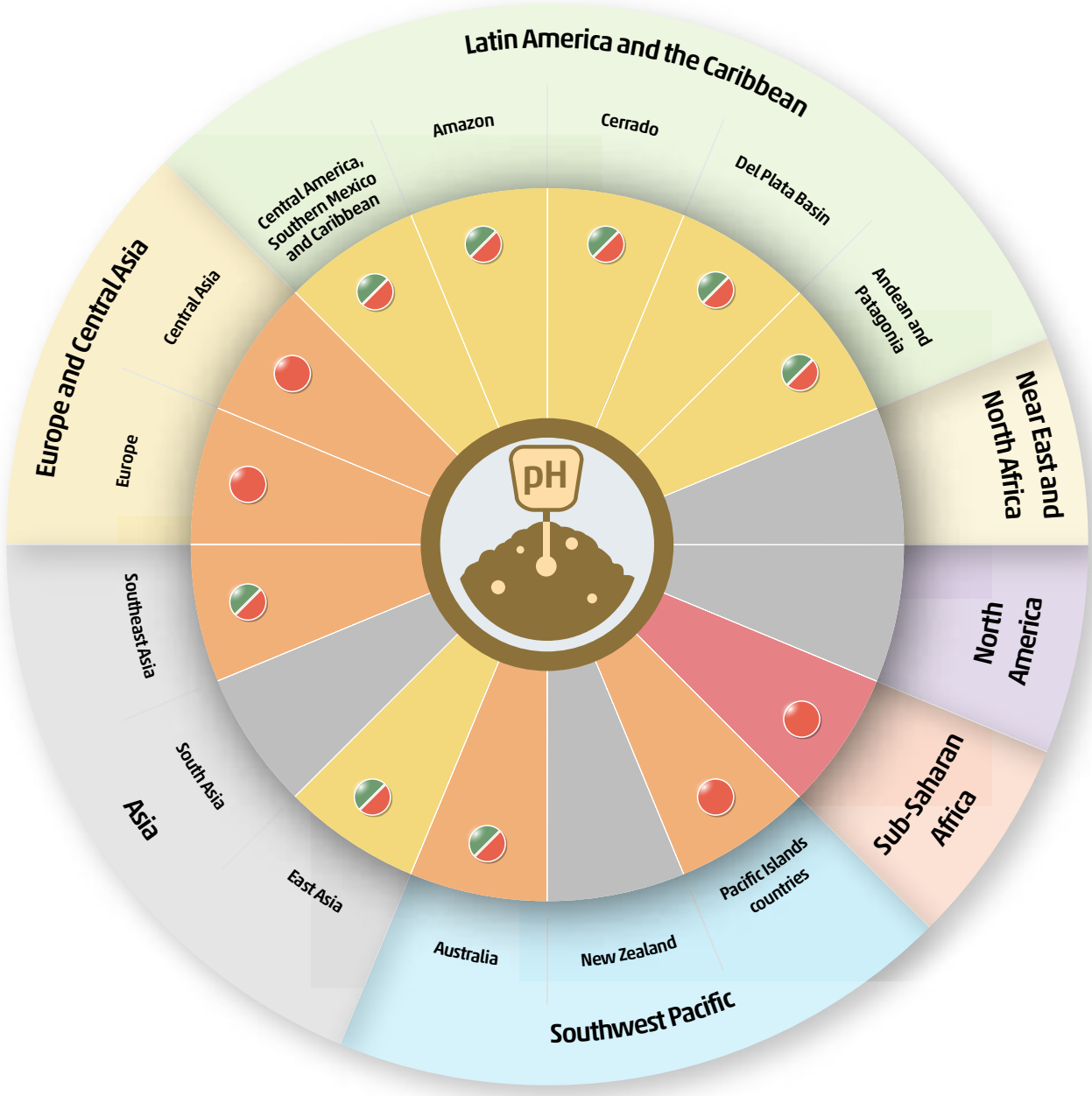


Table 15.4. Regional and subregional summaries for state and trend in the adoption of sustainable soil management practices to address acidification

Region	Summary
Asia	**Southeast Asia:** Soil acidification is reported throughout in Southeast Asia due to highly weathered soils, intensive agrochemical usage and improper management of water resources.
Asia	**East Asia:** Soil acidity has decreased in the Republic of Korea due to liming, but acidity has increased in arable land in China.
Europe and Central Asia	Areas have been degraded locally by acidification.
Latin America and the Caribbean	**Andean countries and Patagonia**: Soil acidification is limited mainly to those originated from volcanic materials, under excessive use of N fertilizers and successive cultivation. There is leaching of basic cations substituted by acid cations, metal toxicity, impact on nutrient mobility, availability, and absorption.
Sub-Saharan Africa	Soil acidification problems are common in heavily weathered soils and can be intensified through the application of some forms of synthetic fertilizer. Acidification limits the use of legumes for N fixation in many soils and also limits availability of P.
Southwest Pacific	**Pacific Island countries**: There is high fertilizer use in some islands (e.g. Fiji) on acidic soils.
	Australia: There are widespread issues due to soil acidification by fertilizers, especially for vulnerable soil types with a low buffering capacity. Low levels of liming and poor nutrient balances intensify the issue.

Nutrient excess and nutrient mining are identified through nutrient balances, which compare inputs from all major sources with outputs in harvested crops. National data are available from FAOSTAT (FAO, n.d.) and IFASTAT (2025). Many studies use spatial crop maps to disaggregate national data to specific cropping areas (e.g. Kaltenegger and Winiwarter, 2020). Nutrient mining may also involve micronutrients, while insufficient micronutrient supply contributes to yield gaps.

Both nutrient excess and mining can be reduced through fertilizer best management practices, commonly referred to as the 4Rs in some countries: right source, right rate, right timing and right placement (Johnson and Bruulsema, 2014; Davidson *et al.,* 2016). Mineral, synthetic and organic fertilizers provide N and P. In addition, biological N fixation by legumes such as soybean and pulse crops plays a crucial role in nutrient supply in many countries.

Implementation of the 4Rs requires accurate, site-specific information on nutrient rates, timing, sources and placement. However, in countries affected by nutrient mining, several socioeconomic constraints limit adoption. These include high fertilizer costs (Davidson *et al.*, 2016), soil-related limitations to biological N fixation (Graham and Vance, 2003) and competing uses for crops residues and manure (Castellanos-Navarrete *et al.*, 2015).

Enhanced efficiency fertilizers (EEFs) offer potential to better align plant N demands with N supply and to increase yields (Gu *et al.*, 2023). Several studies and meta-analyses show that EEFs can significantly reduce emissions of NH_3 and N_2O as well as N losses to water through leaching and runoff. Gu *et al.* (2023) estimated that EEFs could improve N use efficiency by 21 percent. However, yield gains are often insufficient to offset the higher costs of EEFs compared to conventional fertilizer, especially for maize and wheat. Only rice shows sufficient yield increases to compensate for the cost (Li *et al.*, 2018; Lam *et al.*, 2022).

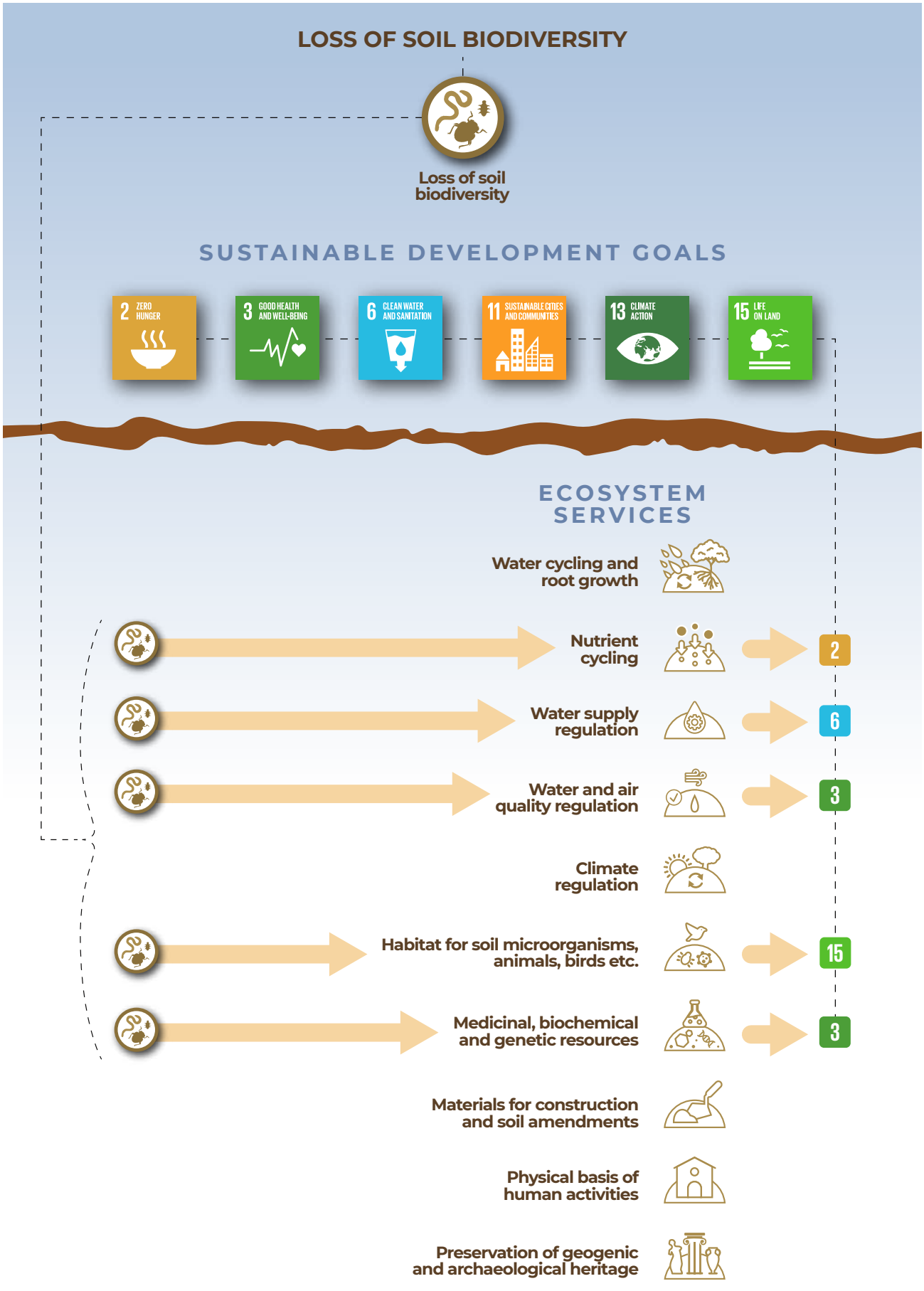
Human-induced soil acidification can be addressed by applying lime, which adds Ca- and Mg-rich materials. However, several barriers limit the adoption of liming, including large application quantities needed, high transport costs, low farmer awareness, potential CO_2 emissions and weak or non-existent lime supply chains (Warner *et al.*, 2023). Moreover, the benefits of lime additions often take time to appear, which can slow adoption, even when research shows clear benefits and decision-support systems are in place to guide application.

15.3.4 Loss of soil biodiversity

Soil is the most biodiverse ecosystem on Earth, hosting an estimated 59 percent of all species (Anthony, Bender and Van de Heijen, 2023). The *Revised World Soil Charter* (FAO, 2015) identifies the preservation of soil biodiversity as a core principle. Soil organisms play a vital role in nutrient cycling, soil structure formation, contaminant breakdown and the functioning of the soil food web. They underpin essential ecosystem services and contribute directly to human health and well-being. Soil biodiversity loss threatens progress towards several SDGs, including SDG 2: Zero Hunger, SDG 3: Good Health and Well-being, SDG 6: Clean Water and Sanitation and SDG 15: Life on Land (Figure 15.11). Maintaining soil biodiversity is therefore central to sustaining soil health and delivering of soil functions.



Figure 15.11. Major adverse effects of loss of soil biodiversity on ecosystem services and soil-related Sustainable Development Goals



Our collective knowledge of soil life has grown substantially since the publication of the 2015 SWSR (FAO and ITPS, 2015). Recent research now considers the entire soil microbiome, including archaea, bacteria, viruses, fungi and protists (Fierer, 2017). Global distribution maps have been developed for larger soil organisms such as nematodes (Van den Hoogen *et al.*, 2019, 2020) and earthworms (Phillips *et al.* (2019). In addition, Orgiazzi *et al.* (2016) produced a global map of threats to soil biodiversity as part of the *Global Soil Biodiversity Atlas*. The *State of knowledge of soil biodiversity* report (FAO *et al.*, 2020) synthesized this expanded knowledge base.

In general, the trend in the adoption of SSM practices to address soil biodiversity loss is judged to be deteriorating or variable across all regions and subregions (Figure 15.12 gives an overview and Table 15.5 gives the summaries in more detail). No region or subregion is currently considered to be in a favourable condition. Habitat modification, particularly deforestation, has been cited in several regional assessments as the most common cause of biodiversity decline, reflecting broader research findings (e.g. Bach *et al.*, 2020). The conversion of forests and grasslands to agriculture tends to support smaller and less diverse soil communities compared to those found in grasslands and forests (FAO *et al.*, 2020).

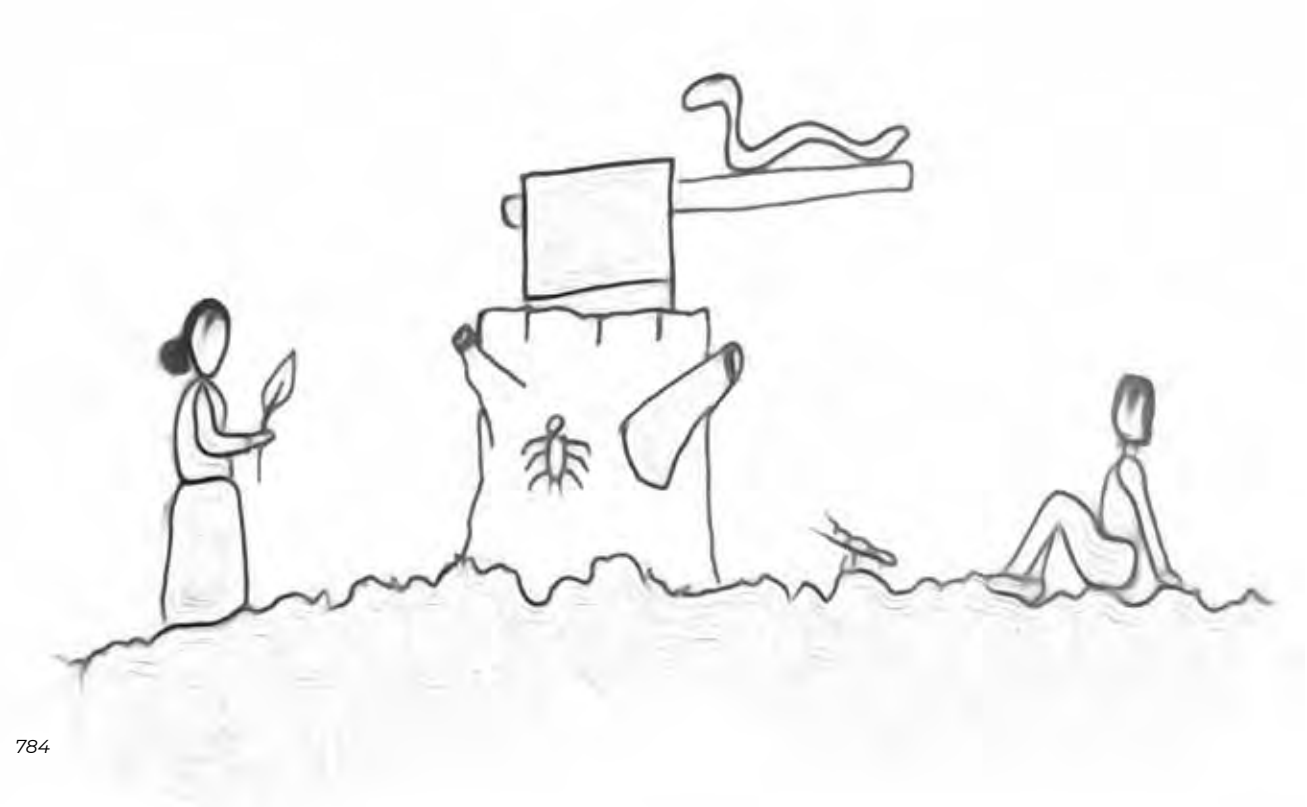


Figure 15.12. Regional and subregional summaries for state and trend summaries in the adoption of sustainable soil management practices to address loss of soil biodiversity

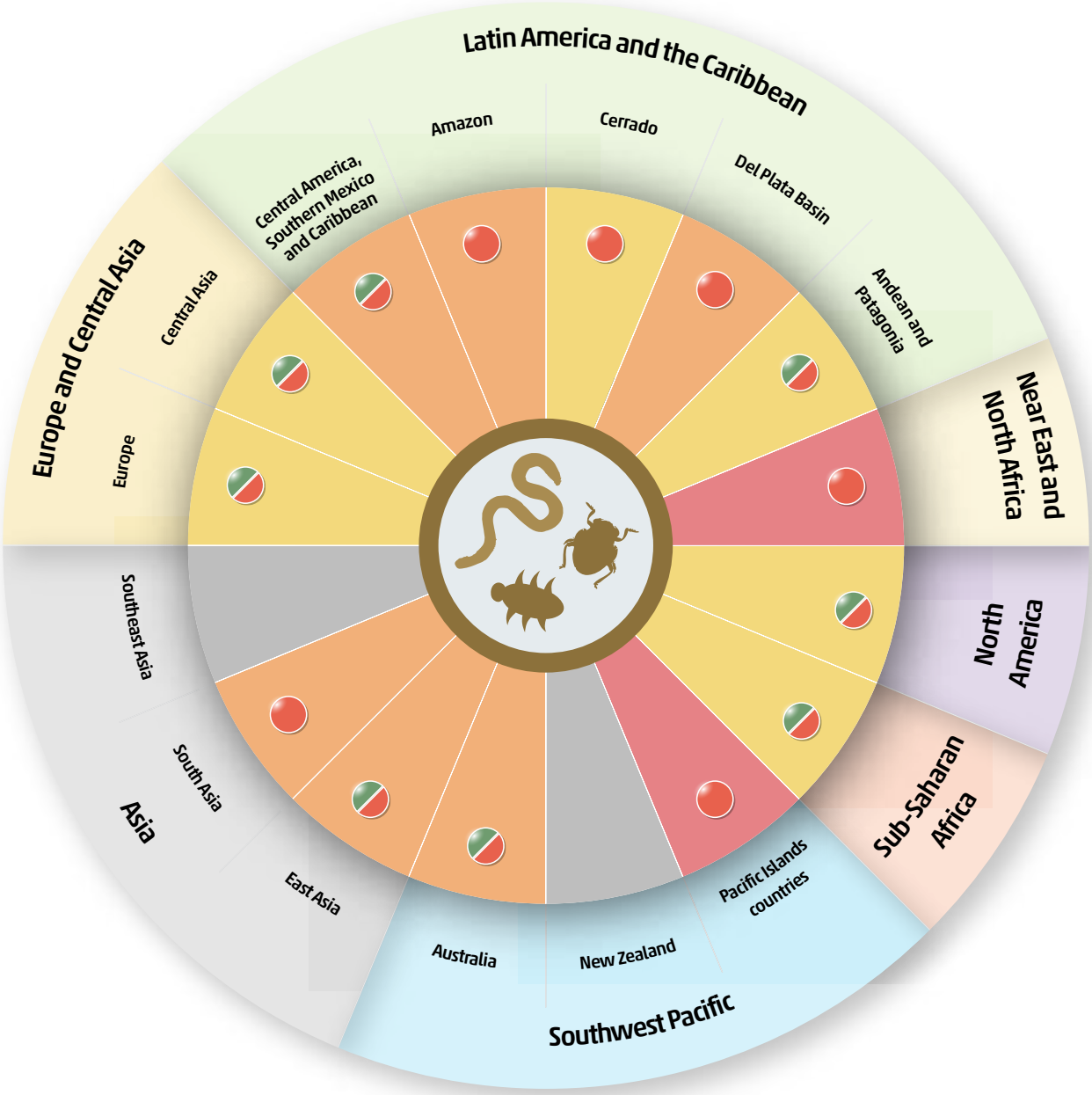


Table 15.5. Regional and subregional summaries for state and trend in the adoption of sustainable soil management practices to address loss of soil biodiversity

Region	Summary
Asia	**East Asia**: The diversity of soil microorganisms and larger soil organisms are being generally decreased by conventional tillage. No-till and cover crop practices improve microbial diversity.
	South Asia: There are threats from loss of above-ground diversity, pollution and nutrient over-loading, over-grazing, intensive agriculture, soil erosion, desertification and climate change.
	Southeast Asia: Not assessed.
Europe and Central Asia	**Europe**: It is decreasing in many places due to intensive agriculture, but increasing due to land abandonment, and follows the same trend as SOC change.
	Central Asia: It is decreasing in many places due to intensive agriculture, but increasing due to land abandonment, following the same trend as SOC change.
Latin America and the Caribbean	**Central America, southern Mexico and the Caribbean**: Data on soil biodiversity in tropical and subtropical regions, including in this subregion is limited. Soil respiration rates are highest in coastal regions with negative trends in savannah and evergreen broadleaf forests.
	Amazon: Over-exploited areas face severe biodiversity loss. Edaphic macroinvertebrates suffer declines in species richness and diversity, although some bacterial groups and root symbionts show resilience. Solutions like low-intensity grazing, forest-pasture integration and Indigenous Peoples' knowledge application are gaining recognition.
	Cerrado: Biodiversity is significantly impacted by agricultural expansion. Areas subjected to over-exploitation experience a severe decline in biodiversity. Deforestation and fires, mainly in the pasture lands and expansion areas, lead to soil degradation.
	Del Plata Basin: Studies on soil biodiversity are less encompassing and based on localized data. Monocropping or extremely simple rotations are leading to an increasing decline of soil biodiversity, worsened by the generalized decrease in SOC.
	Andean countries and Patagonia: The change of land use and soil cover, the megafires of natural forests, water use and pollution by mining activities that affect wetlands and agriculture have threatened and are still present as a threat to soil health and soil biodiversity.
Near East and North Africa	Soil biodiversity threats are classed as very high in some areas of North Africa, specifically the coastal areas from Casablanca to Benghazi and the Nile Delta. Desertification is a major threat to soil biodiversity. In the Near East, the threats to soil biodiversity are classed as very high along the Mediterranean coastline of Lebanon and the Syrian Arab Republic as well as large areas of Iraq and western parts of the Islamic Republic of Iran.
North America	There is some adoption of SSM, but insufficient, due to a lack of monitoring. In managed lands, trends are variable and associated with the implementation of conservation agriculture practices.
Sub-Saharan Africa	There is a loss of soil biodiversity due to mismanagement practices including deforestation, overgrazing and burning practices.
Southwest Pacific	**Pacific Island countries**: Although knowledge on current trends in soil biodiversity is very limited in the islands, it is commonly agreed that it is reduced by agriculture intensification (e.g. monocropping, mechanisation and the increased use of chemicals).
	Australia: There is limited data but a high percentage of minimum and no-till cropland should lead to improved biodiversity.
	New Zealand: Very limited data prevent any assessment.

A meta-analysis by Qu *et al.* (2024) found that converting forests to cropland, grassland or plantation generally resulted in ecosystems dominated by bacteria rather than fungi, with fungal communities becoming more homogenous than in native soils. Changes in fungal diversity can significantly affect soil functions such as organic matter decomposition and nutrient availability, especially in low N ecosystems (Dong *et al.*, 2024).

In agricultural landscapes, production intensity can further reduce soil biodiversity. Intensive agriculture, which relies on irrigation, heavy machinery and the increasing use of fertilizer and pesticides, significantly alters soil habitats (Geisen, Wall and Van der Putten, 2019). Mechanical disturbance reduces the biodiversity of larger soil organisms such as earthworms (de Graaf *et al.*,

2019; Tsiafouli *et al.,* 2015). Fertilization can negatively affect soil biodiversity by increasing soil acidity (de Graaf *et al.,* 2019; Hu *et al*., 2024). A meta-analysis by Gunstone *et al.* (2021) found that pesticides mainly had negative effects on non-target invertebrates.

Since 2015, recent studies have strengthened the connection between improved soil biodiversity and specific SSM practices. Changes in soil pH are a major driver in shifts in soil biodiversity (Zhou, Wang and Luo, 2020) and the addition of lime (or similar soil amendments) is the most common way to manage soil pH. Applying organic fertilizers such as manure generally has positive effects on soil biodiversity (Dang *et al*., 2022; Morugán-Coronado *et al*., 2022). This is especially relevant in farming systems that depend on organic inputs, such as organic farming and regenerative agriculture. Reduced tillage or no-till agriculture also promote soil biodiversity and increase the biomass of organisms, with earthworms benefiting the most (Briones and Schmidt, 2017).

15.3.5 Salt accumulation

Soils with high levels of salts (saline soils) and Na (sodic soils) are collectively referred to as salt-affected soils. While they occur naturally in semi-arid and arid regions and in some marine coastal areas, this report focuses specifically on human-induced salt accumulation in soils, which can limit root growth and reduce the nutrient supply via the impact of salts on the soil organisms that drive nutrient cycling. Additionally, flushing salts from the soil can increase salinity of surface waters, posing risks to aquatic ecosystems. Human-induced salt accumulation in soils threatens progress towards SDG 2: Zero Hunger, SDG 6: Clean Water and Sanitation and SDG 15: Life on Land (Figure 15.13).

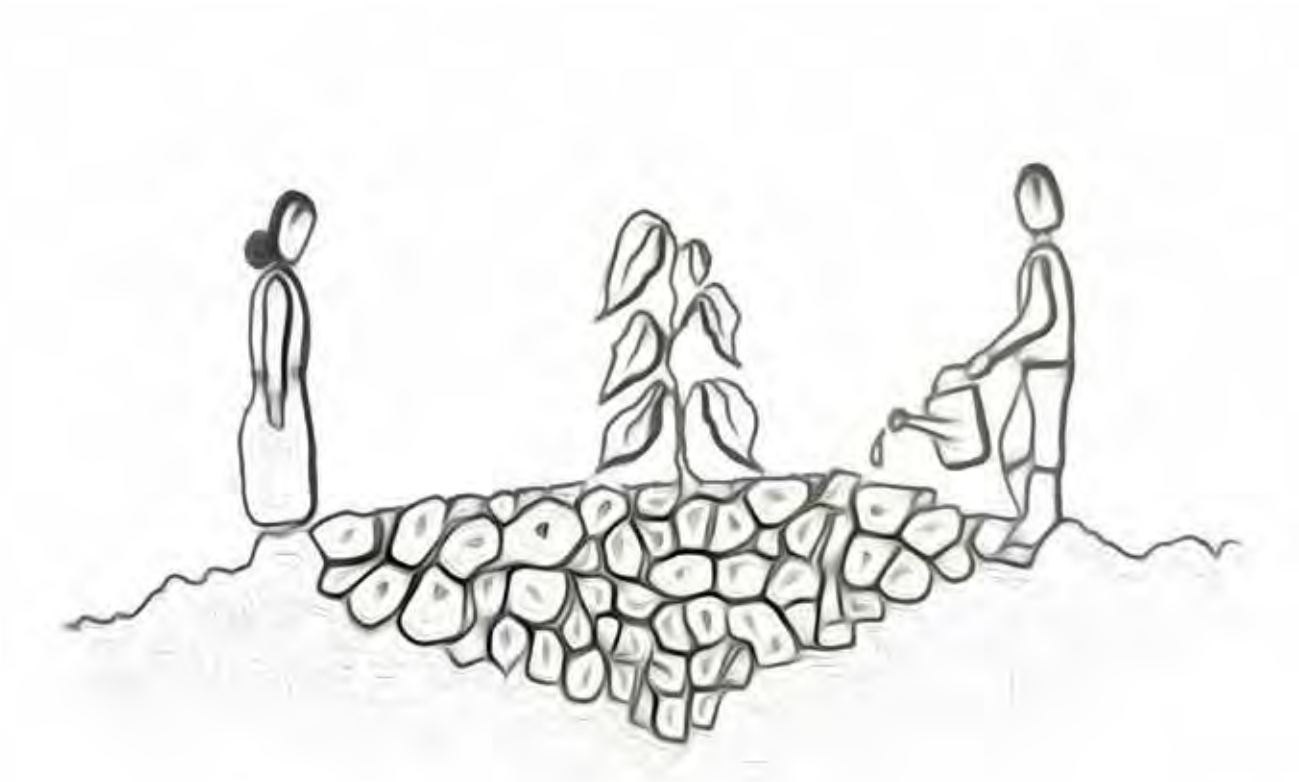
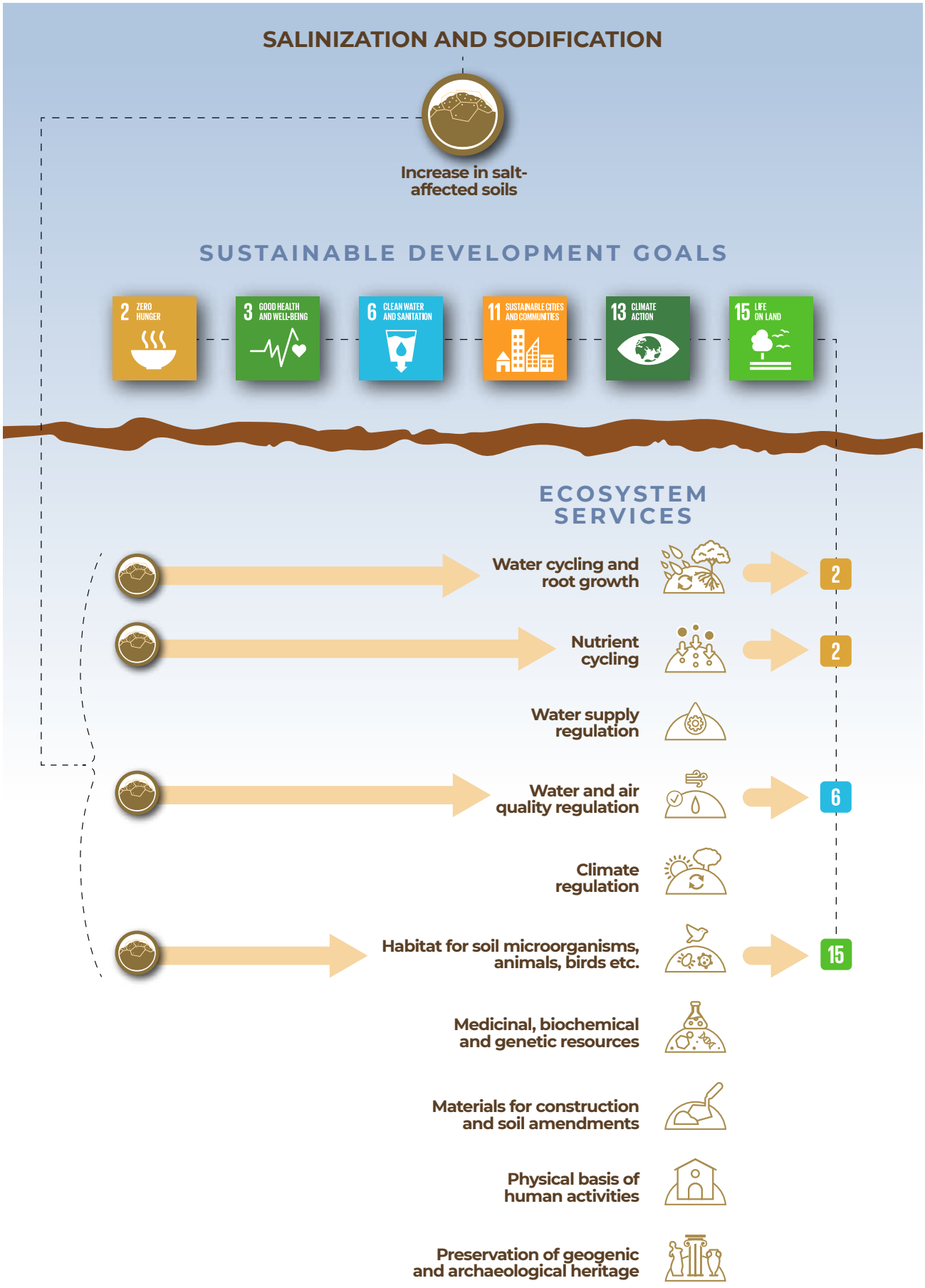


Figure 15.13. Major adverse effects of salinization on ecosystem services and soil-related Sustainable Development Goals



Major studies, such as the *Global status of salt-affected soils* report (FAO, 2024d), have improved our understanding of the extent of salt-affected soils. However, data on changes in human-induced salt-affected soils remains limited.

Nine regions or subregions report a deteriorating trend and six report a poor state for salt accumulation (Figure 15.14 gives an overview and Table 15.6 gives the summaries in more detail). The NENA region has the highest percentage of salt-affected soils, with both the state worsening since 2015 and a continuing deteriorating trend. Similar conditions are found in arid and semi-arid areas of sub-Saharan Africa, South Asia, and Central America, southern Mexico and the Caribbean. Coastal areas in Asia and the Pacific Island countries also face a high risk of increased salinization. In the Del Plata Basin subregion and Australia, increases in salt-affected soils are primarily driven by improper land clearance and changes to vegetation.

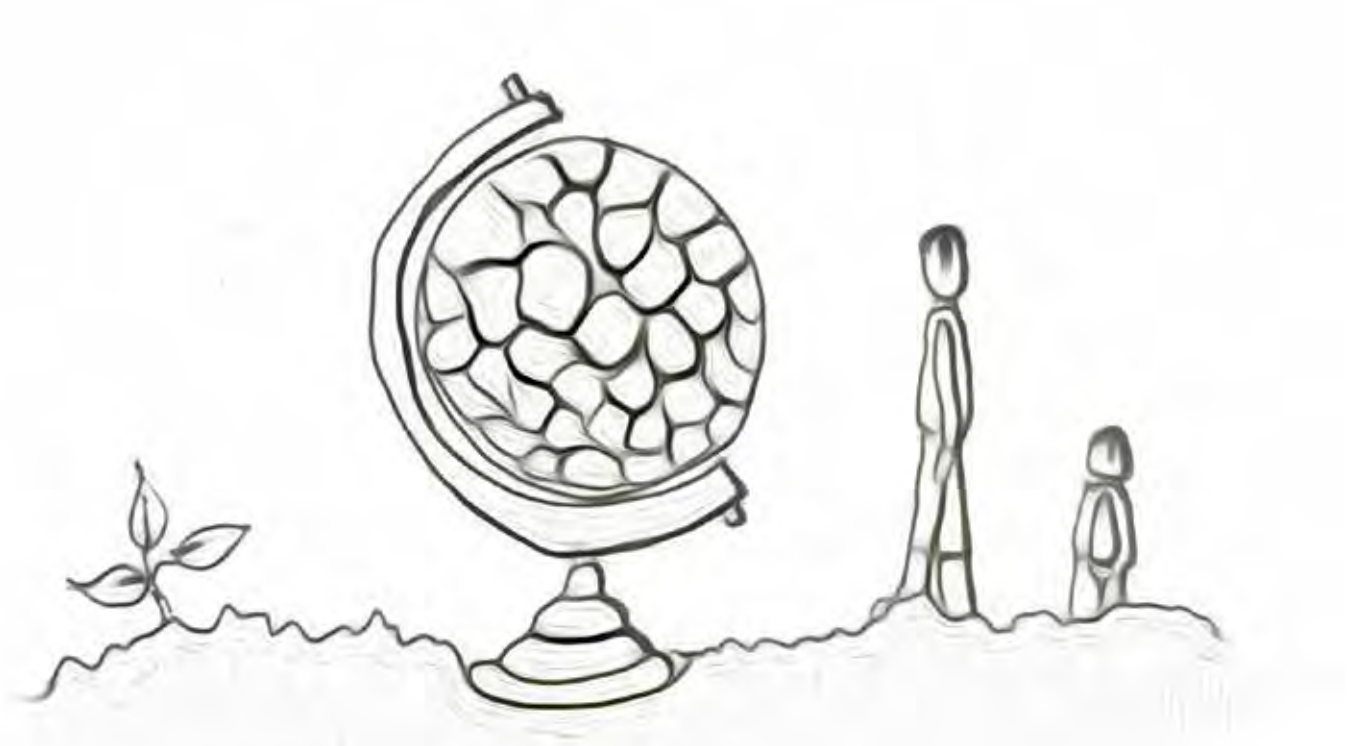


Figure 15.14. Regional and subregional summaries for state and trend in the adoption of sustainable soil management practices to address salt accumulation

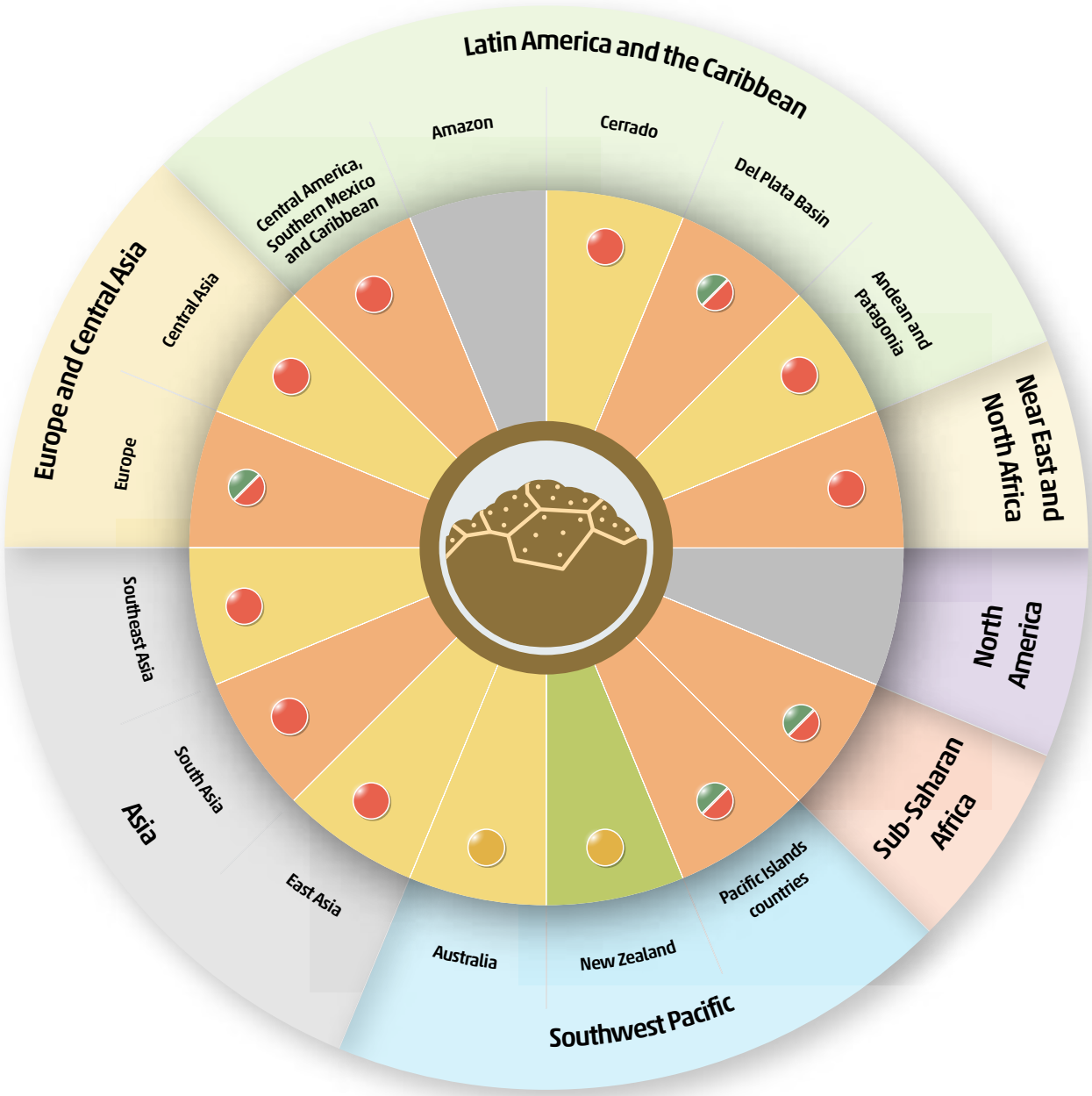


Table 15.6. Regional and subregional summaries for state and trend in the adoption of sustainable soil management practices to address salt accumulation

Region	Summary
Asia	**East Asia**: There are local specific threats from hydrological conditions such as enclosed relief, Na-rich parent materials and slow groundwater flow or waterlogging, and supplementary irrigation with saline water.
	South Asia: There is an increase in salinity particularly in uncultivable land due to a rising water table. Abandoned paddy lands are more prone to developing soil salinity compared to active paddy lands.
	Southeast Asia: Salt-affected soils are being decreased by the management of the ground water table, the application of salt resistant crops, soil moisture conservation and organic matter application
Europe and Central Asia	**Europe**: There are locally highly affected soils in southern Europe, and a possible threat due to climate change in coastal areas in the future.
	Central Asia: In spite of better irrigation technology preventing salinisation, climate aridification is causing salt accumulation in some soils.
Latin America and the Caribbean	**Central America, southern Mexico and the Caribbean**: Coastal areas, particularly in the Caribbean, face elevated soil salinity due to seawater intrusion, exacerbated by rising sea levels. Human activities such as improper irrigation and deforestation are further contributing to soil salinity.
	Amazon: Not assessed.
	Cerrado: As climate change progresses toward drier conditions in the Cerrado and as the use of central pivot irrigation expands, this threat is increasing.
	Del Plata Basin: Many soils in the plains have high water tables, saline groundwater and sodic soils. They may be being salinized by upward soluble salt rising, promoted by land clearing, overgrazing and use of improper water (high sodium bicarbonate content) in irrigation.
	Andean countries and Patagonia: There are increases in salinization in response to climate change conditions, given the lower availability and quality of irrigation water, lower salt leaching and the concentration of surface salts due to capillarity in the central-northern area of Chile, and the coastal areas of Colombia and Patagonia.
Near East and North Africa	There is a high proportion of salinized land. A loss of vegetation cover is linked to changes in salinity in global modelling. There have been increases in irrigated land since 2015 in most NENA countries (especially Near East) which increases risks due to improper water use.
North America	Not assessed.
Sub-Saharan Africa	Salinization is an issue in the subregion. Inappropriate irrigation water management (the use of poor water quality, the application of unmeasured irrigation water volume and insufficient drainage infrastructure for wastewater management) and poor agricultural practices have led to salt accumulation.
Southwest Pacific	**Pacific Island countries**: Soil salinization is a serious threat to food security, especially in low-lying atolls through sea water intrusion, sea sprays and inundation by tsunamis.
	Australia: It has been a problem in the southern agricultural zone historically, especially the southwest of Western Australia, with a high effect on productive lower landscape positions.
	New Zealand: Not a substantial issue.

Improper water management in irrigation is the most cited cause of increased salt-affected soils, although many estimates are based on outdated data. Sustainable soil management practices, such as adequate drainage and carefully regulated water application during the design and operation of irrigation projects, can mitigate this threat, but adoption remains uneven. Currently, only 22 percent of irrigated lands are drained (Ritzema, 2016). In addition, improper flushing of salts into waterways can lead to water quality issues.

Remediation of sodic soils typically requires deep ploughing to break up unfavourable soil structure in the rooting zone and to incorporate gypsum and organic amendments into the upper soil layers. In some cases, simply adding gypsum can help improve soil chemistry (Vargas Rojas *et al.,* 2018). While the benefits of these interventions can last for several decades, results vary across different conditions and studies (Vargas Rojas *et al.,* 2018). In areas with saline-sodic soils, such as the inland

Pampas of Argentina, extending drainage systems and improving grazing management help stabilize the groundwater table at deeper levels and prevent salinization from saline groundwater raising (FAO, 2024).

15.3.6 Soil pollution

Soil pollution occurs when chemicals or substances are present at a higher-than-normal concentration and cause harm to non-targeted organisms. Pollution may originate from point sources with limited distribution, such as fuel spills, or from diffuse sources, such as industrial emissions fallout (Rodriguez-Eugenio, McLaughlin and Pennock, 2018). The global assessment by FAO and UNEP in 2021 significantly advanced our understanding of soil pollution (FAO and UNEP, 2021).

Contaminants enter the soil through various pathways, including mining, waste dumping, atmospheric deposition, organic material and agrochemicals, armed conflict, irrigation with contaminated water, and chemical spills. The links between potentially toxic elements and human health are well established. For example, Cd can negatively affect the thyroid, lungs, kidneys, bones and joints (FAO and UNEP, 2021). These toxic elements also harm soil organisms, impairing microbial traits such as stress resistance, nutrient cycling and pathogenesis (Liu *et al*., 2023). In extreme cases, access to land is prohibited due to severe pollution. For instance, following the Fukushima Daiichi Power Plant nuclear accident in 2011, 3 900 km^2 of soil was contaminated with the radionuclide caesium (^{137}Cs) and 254 km^2 classified as severely contaminated. By 2019, only 2 percent of the contaminated forest land had been remediated (Onda *et al*., 2020; Evrard, Laceby and Nakao, 2019).

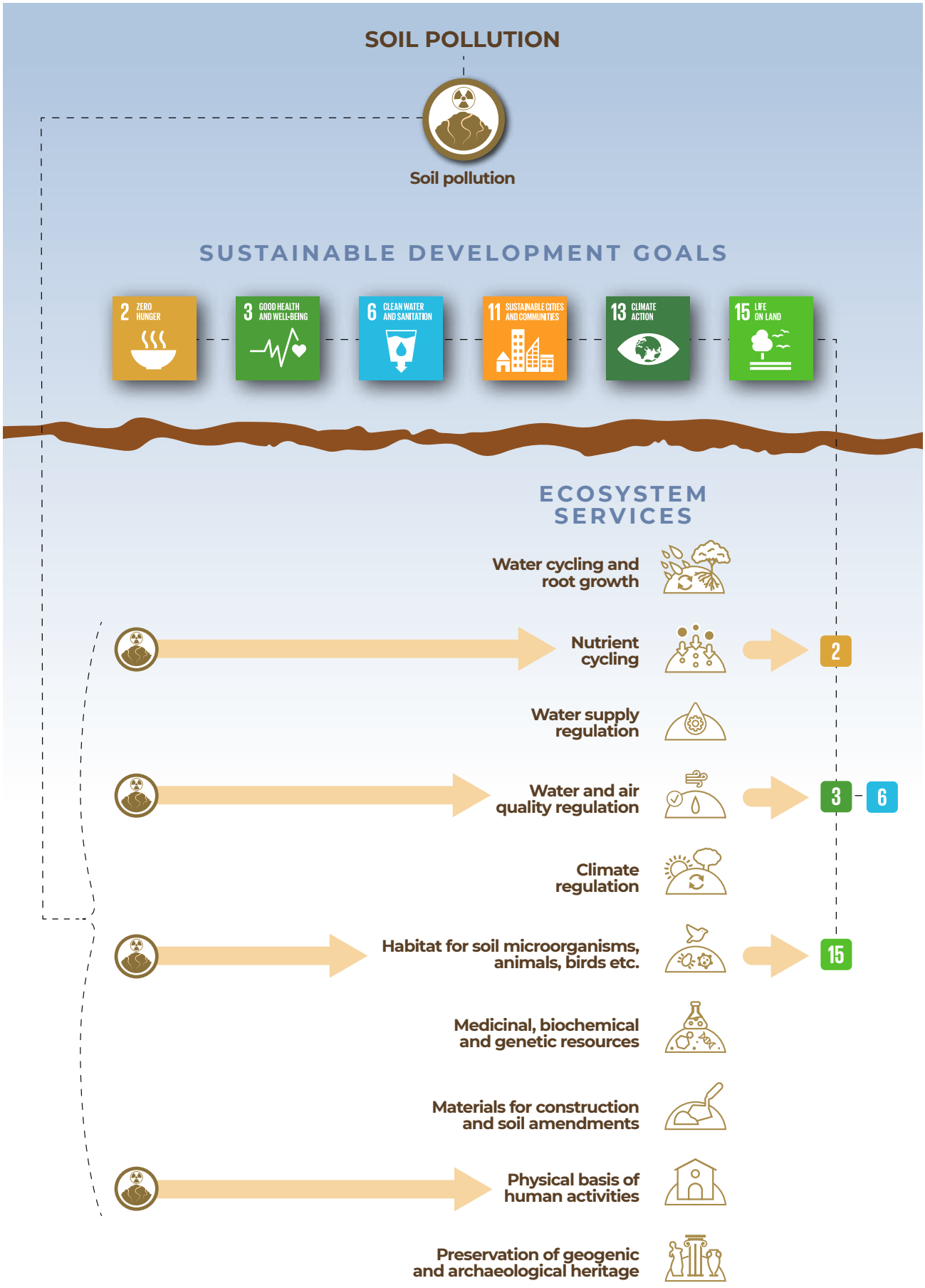
Since 2015, there has been growing concern about organic and inorganic compounds used in health, beauty, clothing, packaging and other consumer products, as well as in agriculture, energy, pharmaceutical and related industries, and their roles as environmental contaminants. These so-called “emerging contaminants” were not previously monitored (FAO and UNEP, 2021). Thousands of potential pollutants fall under this category, including plant and animal hormones, endocrine disruptors, antibiotics, antibiotic-resistant microbes, and microplastics.

Among these, microplastics and associated chemicals are particularly important emerging soil pollutants (FAO and UNEP, 2021). Common sources include the application of sewage sludge as fertilizer, irrigation with non-treated wastewater, plastic mulching, littering, tyre wear and atmospheric deposition (Büks and Kaupenjohann, 2020). Identifying contaminated areas and developing effective remediation methods remain key research priorities.

Soil pollution poses a serious risk to achieving SDG 2: Zero Hunger, SDG 3: Good Health and Well-being, SDG 6: Clean Water and Sanitation and SDG 15: Life on Land (Figure 15.15. Major adverse effects of soil pollution on ecosystem services and soil-related Sustainable Development Goals).



Figure 15.15. Major adverse effects of soil pollution on ecosystem services and soil-related Sustainable Development Goals



Soil management to prevent, control and mitigate soil pollution is generally rated as fair.However, in Europe, it is assessed as poor, and very poor in the Amazon subregion due to localized mining. The Pacific Island countries assessment also highlights a poor state due to multiple sources of contaminants (Figure 15.16 gives an overview and Table 15.7 gives the summaries in more detail). In many regions, the trend is stable or variable. However, ongoing mining activities in the Amazon and legacy contamination issues in the Pacific Island countries contribute to deteriorating trends in these areas.

Figure 15.16. Regional and subregional summaries for state and trend in the adoption of sustainable soil management practices to address soil pollution

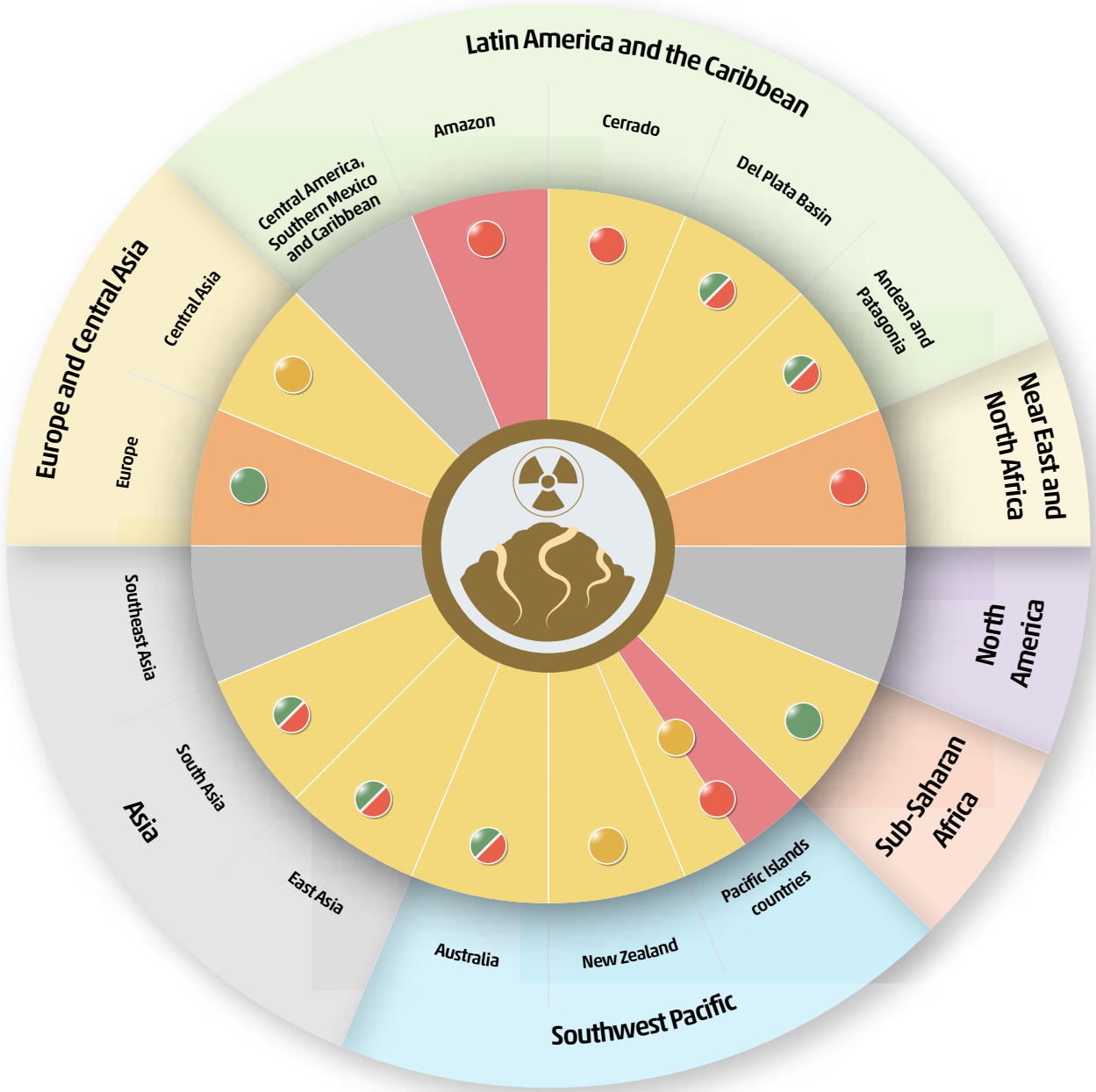


Table 15.7. Regional and subregional summaries for state and trend in the adoption of sustainable soil management practices to address soil pollution

Region	Summary
Asia	**East Asia**: Excessive use of agrochemicals, urbanization and wastewater from industries affect China, the Republic of Korea and Japan. Emerging pollutants such as antibiotics, microplastics and polycyclic aromatic hydrocarbons are being detected in East Asia.
	South Asia: Agrochemicals and industries pose the threat of soil pollution, especially with trace elements (heavy metals). Wastewater irrigation, waste disposal, vehicular exhaust, wastewater sludge and construction are the key sources of local soil contamination.
	Southeast Asia: Not assessed.
Europe and Central Asia	**Europe:** There is widespread heavy metal and biocide pollution but this is improving due to less inputs. Microplastics and emerging organic compounds are being revealed as common in soils. In general, there are better conditions in forests than in agricultural soils. Radionuclide pollution is still a severe threat in some regions of Belarus and Ukraine.
	Central Asia: Heavy metal pollution by different sources is threatening some areas.
Latin America and the Caribbean	**Central America, southern Mexico and the Caribbean**: Not assessed.
	Amazon: Mining operations in the Amazon are rapidly expanding. Numerous new operations are illegal, leading to soil and water contamination with mercury (Hg). The mining area is small compared to the whole region but there is a lack of data on Hg usage, pollution extent, and its impact on soil and water. However, the risks to human health are well documented.
	Cerrado: Agrochemicals used in monocropping farms are the main contributors to contamination. As agricultural activities expand in the Cerrado, the risks of soil pollution escalate, especially pollution of aquifers.
	Del Plata Basin: Non-point pollution by residues or byproducts of herbicides (e.g. AMPA from glyphosate) is an emerging threat. More assessments are required to confirm the trend.
	Andean countries and Patagonia: Mining, oil extraction, industrial processes, livestock production and agricultural activities (fertilization and the use of agrichemicals) are causing the accumulation of inorganic and organic contaminants in soil, tailings and surrounding river networks.
Near East and North Africa	Hotspots of pollution are increasingly found, resulting from either agricultural mismanagement practices, armed conflicts, oil refineries and cement companies. In conflict areas, unusual toxic metals (Rb, Sr and Zr) and high levels of other heavy metals are evident.
North America	Not assessed
Sub-Saharan Africa	Soil pollution is present, associated with metropolitan areas and mining. Substantial heavy metal contamination is associated with poorly regulated mining sites.
Southwest Pacific	**Pacific Island countries**: There is a lack of data on pollution in mountainous islands. Diffuse source pollution is believed to be the main issue but there is also significant point contamination from mining sites, waste dumps, chemical stores, septic tank discharges, old industrial sites, etc. Atolls are stable in a fair state with a stable trend due to legislative controls (rating shown for high islands), while atolls have a fair state and a stable trend.
	Australia: Legacy sites have been identified and many known contaminated areas have been remediated. There are concerns remaining about emerging contaminants (e.g. PFAS and microplastics)
	New Zealand: Legacy areas exist, especially in industrial and urban centres, and mapping of these areas is in progress. Many known contaminated areas have been remediated. Information is needed on emerging pollutants.

Remediation techniques for polluted soils are well established. Since 2015, the literature has increasingly emphasized nature-based solutions, such as phytoremediation, microbial remediation and immobilization using biochar. However, pollutants, such as radionuclides or the persistent organic pollutants (POPs), which are regulated under the Stockholm Convention (Secretariat of the Stockholm Convention, 2025), still require more intensive methods, including excavation, containment, incineration and other physical and chemical remediation techniques. Continued efforts to identify pollutants and conduct risk assessments at contaminated sites worldwide are essential for selecting the most appropriate remediation approaches and ensuring the success of remediation programmes (FAO and UNEP, 2021).

15.3.7 Urbanization and soil sealing

Urbanization involves the conversion of agricultural, natural or seminatural areas to urban uses, including settlements, urban fringes, industrial estates and transport infrastructure (FAO, 2022). Soil sealing refers to the permanent covering of the soil with impervious materials, such as concrete, asphalt, tar, buildings and other structures that cannot be easily removed (FAO, 2022). Since 2015, growing urban populations, particularly in Africa and Asia, have increased pressure for land conversion, with cropland projected to be the main source of new urban land (Chen *et al.*, 2020). In contrast, urban population growth has been lower in highly urbanized regions, such as Europe.

Poor urbanization management and soil sealing continue to represent the most severe threat in Europe (Figure 15.17 gives an overview and Table 15.8 gives the summaries in more detail), as was also reported in 2015 (FAO and ITPS, 2015). The Pacific Island countries and Southeast Asia also report a very poor state in the adoption of SSM practices to address this threat. While urbanization has been lower in most other regions and subregions, many still report challenges associated with rapid urban expansion. Overall, seven regions or subregions are assessed as having a deteriorating trend in the adoption of SSM practices to address this threat.



Figure 15.17. Regional and subregional summaries for state and trend in the adoption of sustainable soil management practices to address urban expansion and soil sealing

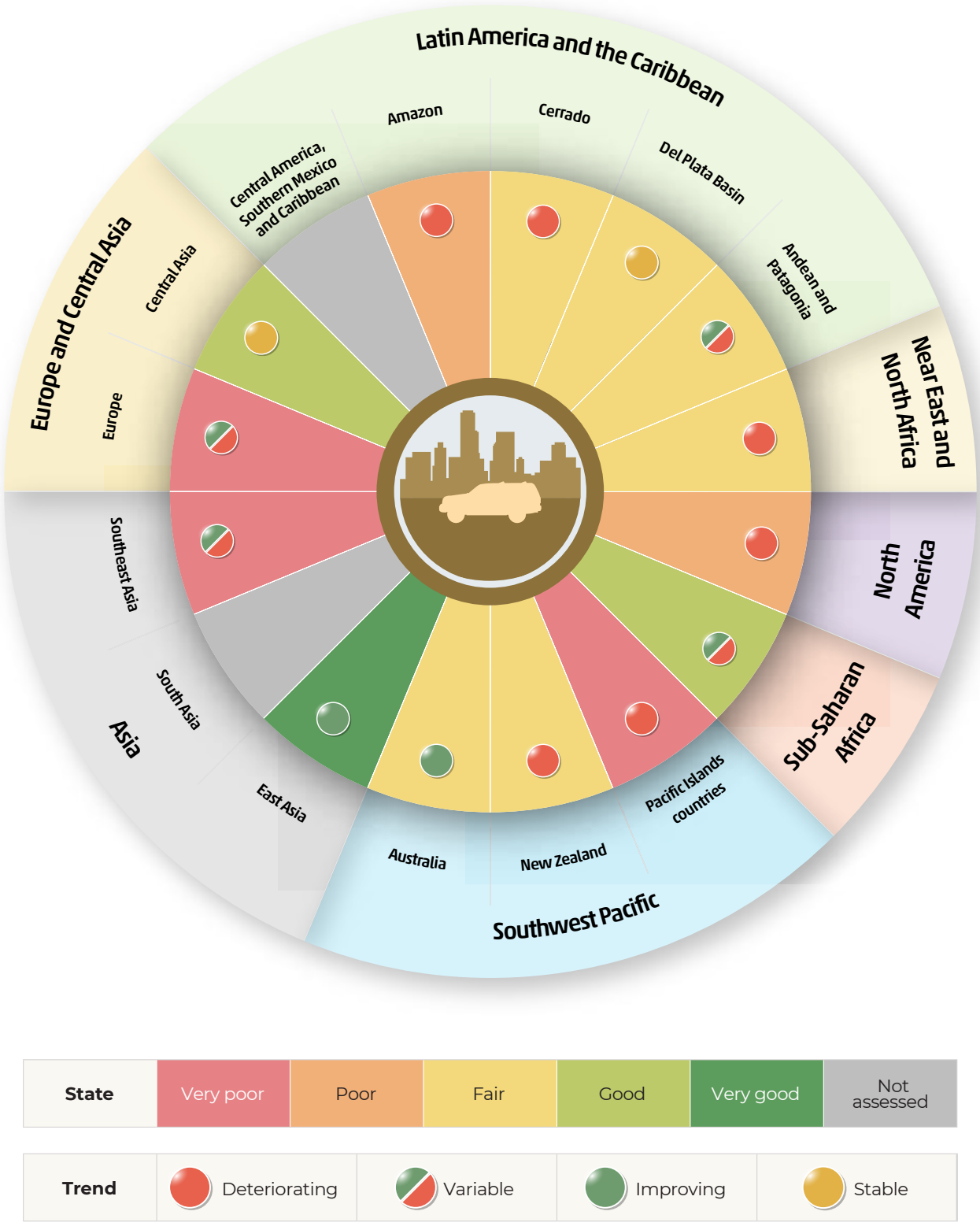


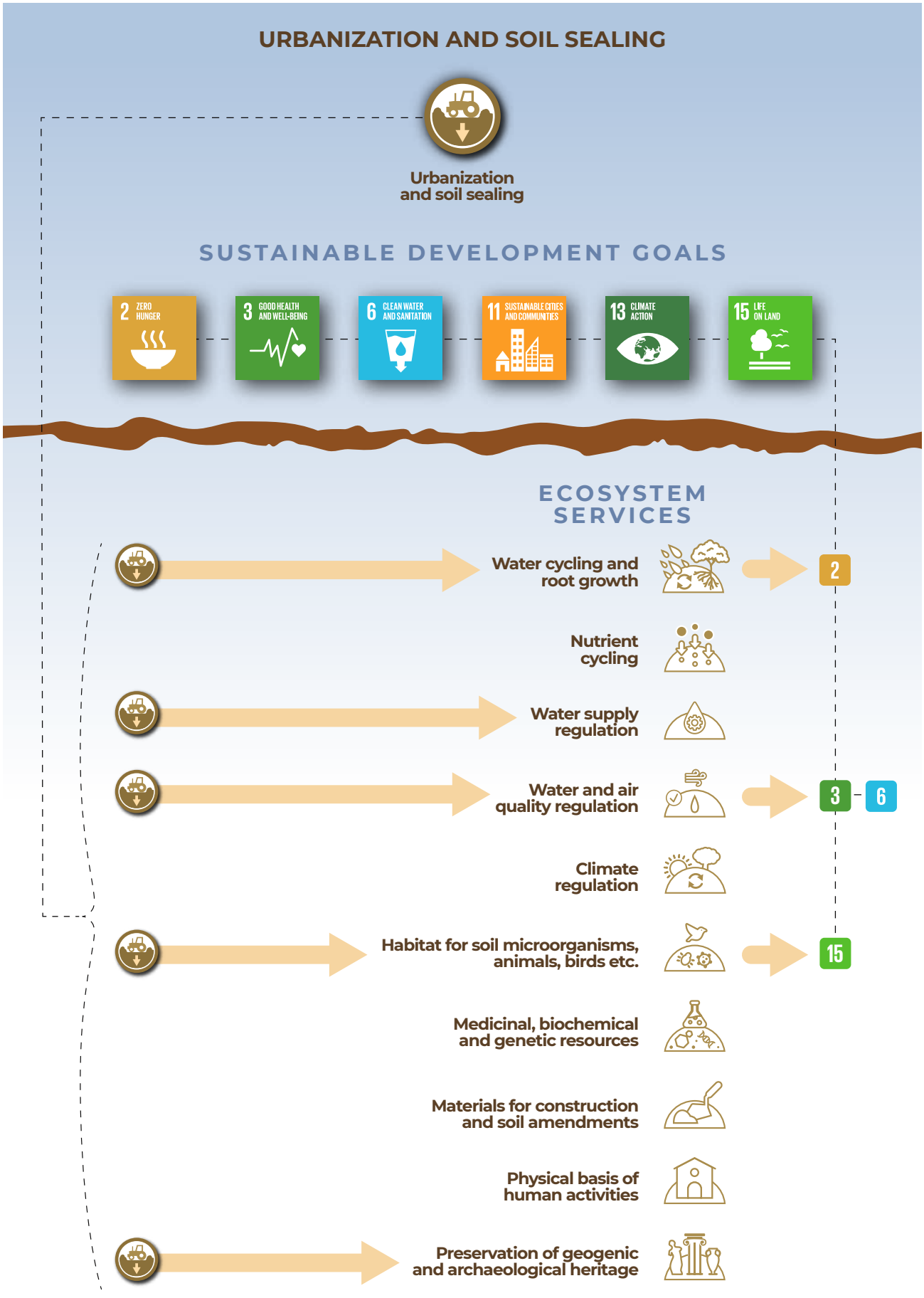
Table 15.8. Regional and subregional summaries for state and trend in the adoption of sustainable soil management practices to address urban expansion and soil sealing

Region	Summary
Asia	**East Asia**: To alleviate environmental issues caused by impervious surfaces, governments have set up specific plans to improve the situation and include appropriate data in urban planning and disaster prevention.
	South Asia: Not assessed.
	Southeast Asia: Competition for prime agricultural land for non-agricultural development is common throughout the region.
Europe and Central Asia	**Europe**: There are contrasting state and trends. Sealed soils occupy 4.2% of European Union territory with a declining trend, while in other parts of the region, soil sealing by urbanisation mostly affects suburban areas and has not experienced strong changes.
	Central Asia: Soil sealing by urbanisation has not experienced many changes at the national level since the 2015 report.
Latin America and the Caribbean	**Central America, southern Mexico and the Caribbean**: Not assessed.
	Amazon: The urbanized area remains relatively small compared to the size of the region. Population growth persists, driving urbanization that replaces lands once covered by natural forest or used for agriculture and other purposes. Soil sealing is a consequence of urban expansion, leading to the loss of nearby agricultural fields and impacts natural cycles of the fluvial systems in the Amazon.
	Cerrado: Population growth persists, driving urbanization that is replacing lands once covered by natural forest or used for agriculture and other purposes.
	Del Plata Basin: Not an issue except around large cities.
	Andean countries and Patagonia: In some valleys and flood plains near the major cities and capitals, urban development advances on agricultural fertile soils and on peri-urban forest soils.
Near East and North Africa	The region is already highly urbanized and high continuing growth in the urban population puts surrounding land at risk.
North America	Urbanization continues to increase at the expense of prime farmland. Urban sprawl continues to encroach upon unmanaged lands with little or no mitigation efforts.
Sub-Saharan Africa	Urban areas of sub-Saharan Africa are predicted to experience the highest growth rate in the world, which will put pressure on surrounding lands.
Southwest Pacific	**Pacific Island countries**: Historical settlement patterns and rapid growth of settlements are leading to the high loss of cropland and horticulture
	Australia: Urbanization and soil sealing are continuing to encroach on agricultural land. The effects of the expansion of renewable energy (e.g. by large-scale solar farms) are largely unknown.
	New Zealand: Urbanization and soil sealing are continuing to affect highly productive agricultural land.

Converting cropland to urban land reduces the production of agricultural commodities (Gardi, Florczyk and Scalenghe, 2021). The impact on other ecosystem goods and services depends on the degree of surface modifications. Semi-natural sites such as urban forests and urban gardens and vegetated pseudo-natural engineered soils – where topsoil overlays disturbed subsoil – can still provide many ecosystem services. These include high food and non-food biomass production, water storage, runoff and flood protection, and biodiversity support (Morel, Chenu and Lorenz, 2015).

In contrast, sealed soils offer very limited services. Soil sealing most severely affects provisioning services (e.g. food and biomass production), as well as the regulation of hydrological flows, carbon sequestration and biodiversity (FAO, 2022) (Figure 15.18). Gong *et al.* (2020) estimated that the global area of artificial impervious surfaces reached 79.8 million ha in 2018, representing a 1.5-fold increase from the 1990 level. The largest increases since 1990 have occurred in Asia and North America, followed by Europe, South America and Africa. Solar photovoltaic farms have emerged as a new form of land sealing, competing with agriculture or forestry in some regions. Ongoing research is needed to help balance ecosystem services across these competing land uses (e.g. Hilker *et al.*, 2024).

Figure 15.18. Major adverse effects of urbanization and soil sealing on ecosystem services and soil-related Sustainable Development Goals



Urban planning and design should account for the impacts of land use conversion on soil functions and ecosystem services. Good soil governance depends on sound urban planning that minimizes the percentage of sealed soil and promotes the use pervious materials over impermeable material in both urban areas and peri-urban areas (Naumann *et al.*, 2018). Incorporating green infrastructure – such as parks, forests, gardens, urban and peri-urban agriculture, green water and storm water infrastructure, low-impact development, urban drainage systems, bioswales, rain gardens and permeable surfaces – can substantially reduce runoff, delay peak flows and flatten hydrographs (Minixhofer and Stangl, 2021).

15.3.8 Other threats to soil functions

Two additional threats highlighted in regional assessments are acidification and soil compaction. Acidification, primarily driven by non-judicious use of synthetic fertilizer (including inappropriate use, underuse and overuse), was previously discussed in Section 15.3.3, as synthetic fertilizers are one of the major drivers of soil acidification.

Soil compaction is the loss of pore space caused by pressure on the soil surface. It reduces water infiltration and increases the risk of waterlogging. It typically occurs when machinery operates on soil that cannot bear the increased pressure without structural disruption. A soil's workability, or its ability to withstand pressure, depends on highly dynamic factors such as moisture, structure, temperature, drainage and mineral content. Poor workability is frequently cited by farmers and other land managers as a key concern in assessing soil conditions (e.g. Klein *et al.,* 2024). However, assessing soil compaction at the regional scale remains difficult due to inconsistent measurement methods and varying definitions (Sonderegger and Pfister, 2021).

Unsustainable management that leads to compaction is considered a serious threat in East Asia, Southeast Asia, Europe and Central Asia, and in the Pacific Island countries (Figure 15.19 gives an overview and Table 15.9 gives the summaries in more detail). The issue is closely linked to increasing mechanization and the use of heavier farm equipment as well as in a lower extent to overgrazing.

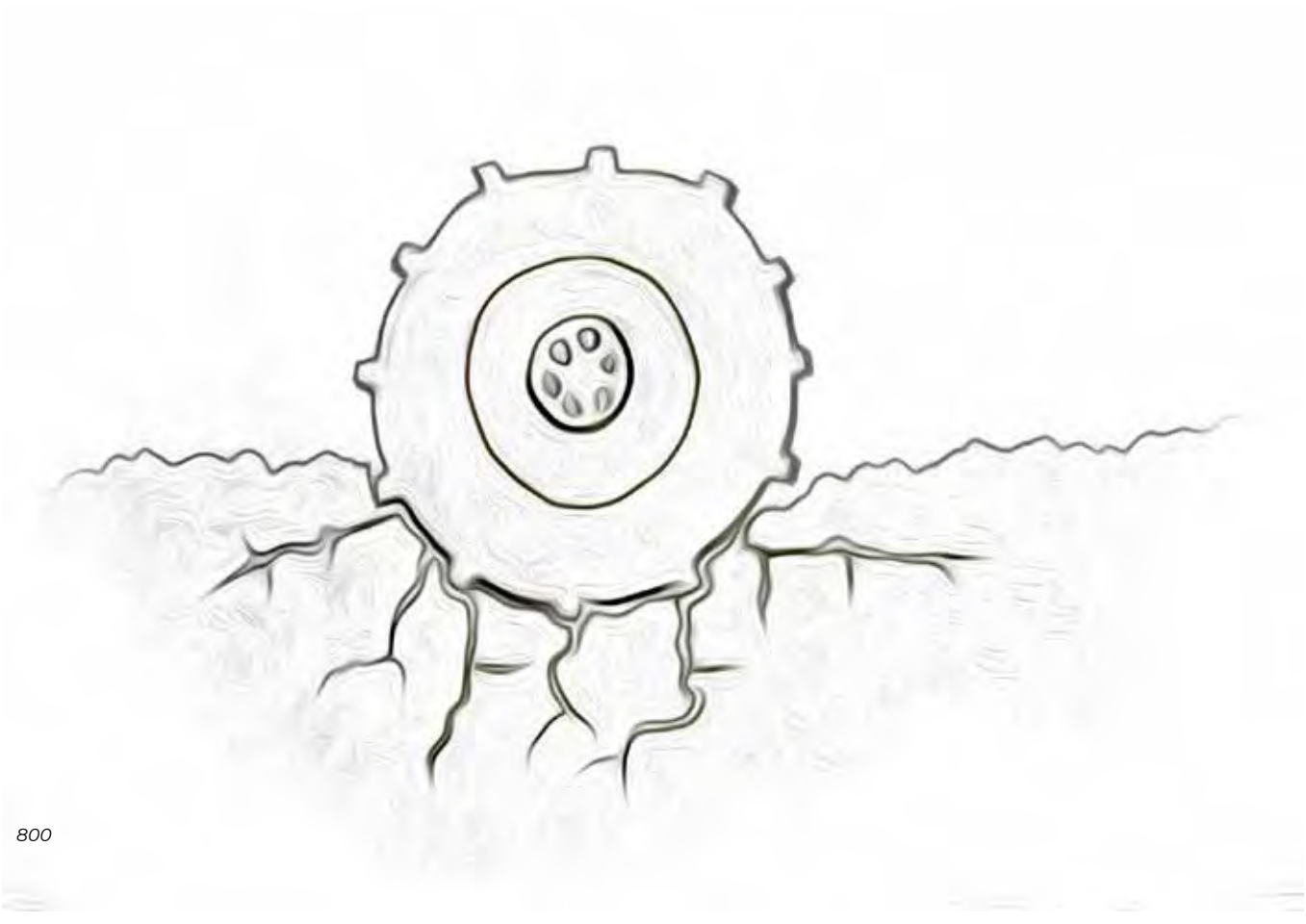


Figure 15.19. Regional and subregional summaries in the adoption of sustainable soil management practices to address soil compaction

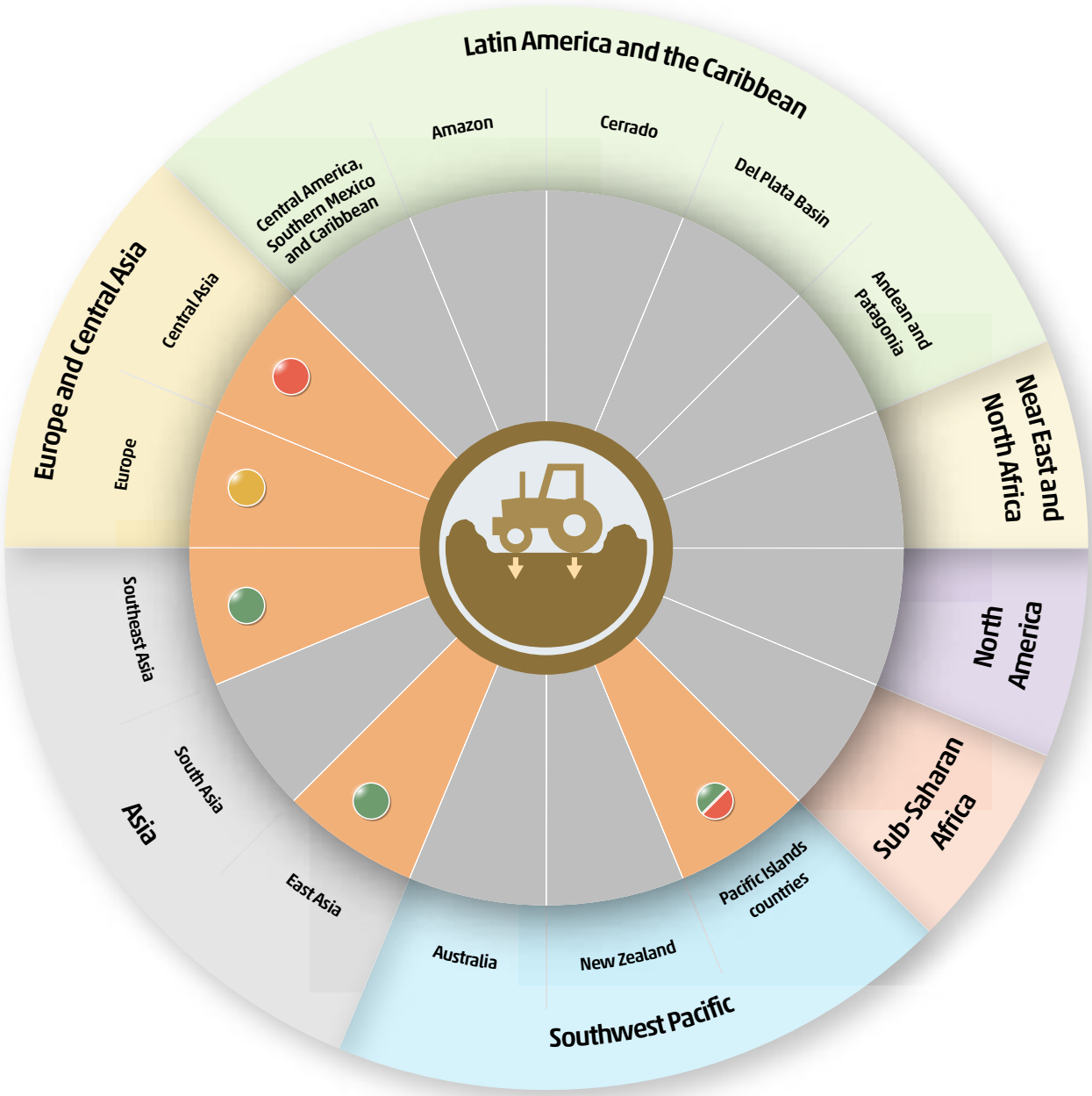
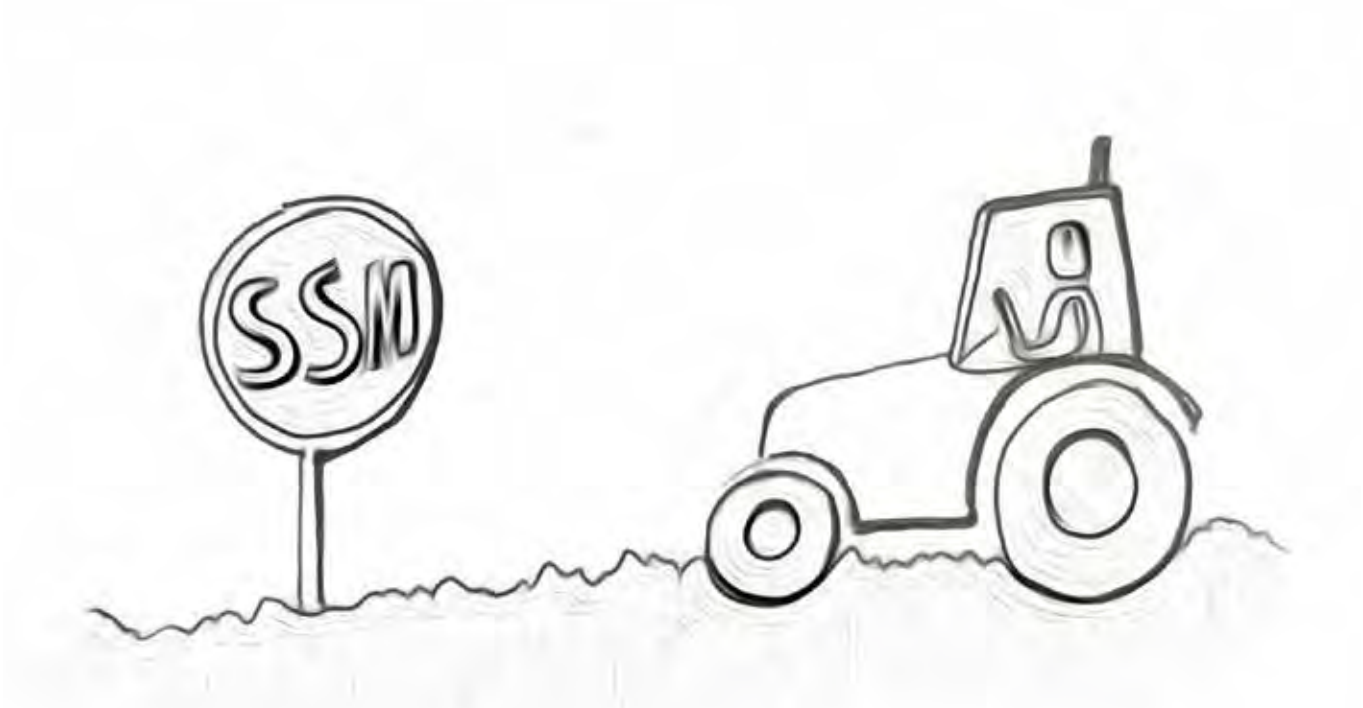


Table 15.9. Regional and subregional summaries in the adoption of sustainable soil management practices to address soil compaction

Asia	**East Asia:** Mechanization of land management has caused soil compaction in agricultural land in China, leading to waterlogging issues.
	Southeast Asia: The mechanization of land management has caused soil compaction in agricultural land in the Philippines, leading to waterlogging issues.
Europe and Central Asia	**Europe**: Farm machinery continues to get heavier. The lack of objective data prevents a precise assessment.
	Central Asia: The general perception is that compaction is severe in some places, but there is a lack of objective data.
Southwest Pacific	**Pacific Island countries**: Compaction is associated with machine traffic and mechanization.



15.4 Limitations of current report and suggestions for future improvements

There are gaps and deficiencies in the report due to limited global and regional data on soil management over the past decade. Improved data, ranging from field to national scales, are essential to support policy development and promote wider adoption of SSM.

Potential data sources to fill the gap and provide better information should include farmers, industry data (e.g. fertilizer sales and grain production) and remotely sensed information on specific management practices. Advances in remote sensing, artificial intelligence and machine learning could significantly enhance soil and land data collection and management.

To facilitate soil management data collection and use, it is necessary to:

- provide more detailed descriptions of management practices, including regional variations;
- determine the area under specific management practices;
- develop high-resolution topographic and soil datasets (e.g. 5 to 10 m scale) for monitoring and modelling erosion, SOC dynamics, GHG emissions and other indicators; and
- support the establishment and consolidation of national soil information systems.

Ideally, any assessment of the state and trend of soils threats should be based on actual measured data and field observations combined with accurate modelling of processes such as erosion and SOC cycling. However, it was judged that such modelling was not feasible for the current report due to data constraints. Advances in modelling require higher quality management and topographic data. Ground-truthing model output also requires improved field measurement at scales comparable to model output. For example, model results at a scale of 10 000 m^2 cannot be reliably compared with field data from 60 m^2 plots (García-Ruiz *et al.*, 2015).

It is also important to develop locally relevant thresholds, standards or benchmarks to assess soil conditions and land degradation. Improving the assessment of the cumulative effects of past, present and future threats – such as soil erosion and SOC loss – will support evidence-based decision-making for SSM and restoration of degraded soils and lands.

Identifying and promoting context-specific SSM practices, tailored to regional and local condition, down to individual fields, is essential. These practices must account for soil, environmental, socioeconomic and cultural factors. In addition, much of the existing soil science is based on temperate and continental environments and requires adaptation for tropical and island context. Furthermore, the development of locally appropriate SSM practices should integrate the knowledge of Indigenous Peoples, as well as traditional and local knowledge, including gender roles and perspectives.

The information presented in this report shows that soil science research is essential for achieving the transition to a more sustainable and resilient agriculture envisioned in the 2030 Agenda (UN, 2015). However, the adoption and implementation of SSM will require broader collaboration among scientists, policymakers, extension workers, soil and land managers, Indigenous Peoples, and other population groups.

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Chapter 16 | Turning knowledge into action: breaking barriers to sustainable soil management

16.1 Introduction

The purpose of this report is to assess the progress made worldwide since the preparation of the 2015 SWSR in addressing the threats to soil functions through the adoption of sustainable soil management (SSM) practices. The promotion of SSM practices is central to the work of the Intergovernmental Technical Panel on Soils (ITPS) and its parent organization, the Global Soil Partnership (GSP). The focus on SSM has sharpened with the publication of foundational documents such as the *Revised World Soil Charter* (FAO, 2015) and the *Voluntary Guidelines for Sustainable Soil Management* (FAO, 2017). A key difference between this report and its 2015 predecessor is that while the 2015 report focused primarily on identifying and analysing status and trends of soil threats, in the current report, there is a stronger emphasis on SSM.

The United Nations (UN) Sustainable Development Goals (SDGs) were also released in 2015 as part of the *Transforming Our World: The 2030 Agenda for Sustainable Development (2030 Agenda)*. Soils are explicitly mentioned in four SDGs: SDG 2 (Zero Hunger), SDG 3 (Good Health and Wellbeing), SDG 12 (Responsible Consumption and Protection) and SDG 15 (Life on Land). Well-functioning soils are also critical for the achievement of other SDGs such as SDG 6 (Clean Water and Sanitation), and SDG 13 (Climate Action). Connections between soils and the SDGs are explored in Chapter 2 of this report.

This final chapter distills detailed technical findings into key messages and provides targeted recommendations aimed at overcoming the remaining barriers to the widespread adoption of SSM.



16.2 The global adoption of sustainable soil management and the challenges hindering progress

The current edition of the SWSR builds on a decade of progress in soil science and the collection of soil data, policy and practices. Since the release of the first report in 2015, global understanding of soil threats has deepened, data availability and quality have improved, and assessment methodologies have become more harmonized.

The ITPS has identified five key messages presented here that synthesize the main findings from research conducted since 2015. These findings are drawn from the global assessment of threats to soil functions (Chapter 3), the direct and indirect drivers of these threats (Chapter 4), the analysis of widely adopted SSM practices (Chapter 5), and the policy, institutional and technical measures needed to support broader adoption of SSM (Chapter 6).

In defining SSM, the *Revised World Soil Charter* (FAO, 2015) explains that:

> *Soil management is sustainable if the supporting, provisioning, regulating, and cultural services provided by soil are maintained or enhanced without significantly impairing the soil functions that enable those services or biodiversity.* (FAO, 2015, p. 4).

Guided by this definition and the specific principles outlined in the *Voluntary Guidelines for Sustainable Soil Management* (FAO, 2017), the regional assessments evaluated whether the soil management in place as of 2015 has remained effective in preventing or reducing a specific soil threat (state), and how the adoption of SSM has changed since 2015 (trend), as detailed in Chapter 8 through to Chapter 14. A global synthesis of these regional findings is provided in Chapter 15. The literature and scientific evidence supporting each key message can be found in the relevant chapter(s) referenced at the end of each statement.

Based on this comprehensive body of regional and global evidence, the ITPS concludes the following key messages.

Key message 1: Soil management remains largely unsustainable worldwide, with the adoption of sustainable soil management practices still at alarmingly low levels

Since 2015, there has been an extraordinary effort by the soil science community to assess how specific soil management practices can mitigate global threats to soil functions. Reducing the harm caused by these threats remains central to SSM (see Chapter 5).

However, despite some localized progress, the current management of the world's soil resources is generally unsustainable. Approximately half of the regional assessments show a low or very low adoption of SSM practices, alongside worsening threats to soil functions (Table 16.1). Only 10 percent of the regional assessments reported improving trends in adoption of SSM practices (for a detailed description of the methodology used in measuring the progress on SSM, see Chapter 7). These trends mean that soils are not on track to sustainably provide services in the future, undermining efforts to achieve land degradation neutrality (LDN) and related SDGs (see Chapter 8 through to Chapter 14).

Table 16.1. Global status and trend of the adoption of sustainable soil management to minimize the global threats to soil functions

Main soil threats	Status of adoption of SSM	Level of confidence (state)**	Trend in adoption of SSM	Level of confidence (trend)**
Erosion	Mostly poor	High	Mostly deteriorating	High
SOC loss	Poor with some regions with very poor status and other in good condition	Medium to high	Deteriorating with some regional improvements	Medium to high
Nutrient mismanagement	Poor	Medium to high	Mostly deteriorating with some regional improvements	Medium to high
Biodiversity loss	All fair to poor	Medium to low	Deteriorating or variable	Medium
Salinization	Mostly fair to poor	Medium	Mostly deteriorating	Medium
Soil pollution	Mostly fair with some regions with poor or very poor status	Medium	Deteriorating or variable	Medium

Note: SSP1 = sustainable development, SSP2 = middle-of-the-road development, SSP3 = regional rivalry, SSP4 = inequality (SSP4), and SSP5 = fossil-fuelled development.
* Each threat assessment is accompanied by a confidence rating based on evidence quality and scientific agreement, following IPCC methodology.

Key message 2: Interlinked soil threats drive a degradation cascade, demanding integrated solutions

This report focuses on measuring the adoption of SSM in addressing seven key threats: soil erosion, loss of soil organic carbon (SOC), nutrient mismanagement (including acidification), loss of soil biodiversity, salt accumulation, soil pollution, and soil sealing due to urban expansion. While soil compaction and waterlogging were addressed in the 2015 report, they are examined here only at the subregional level.

As in 2015, soil erosion remains the most severe threat to soil functions across most regions. Current efforts to manage and control erosion are largely inadequate, with the state of soil management efforts to control soil erosion judged to be poor or very poor. Alarmingly, since 2015, 65 percent of the regional and subregional assessments report a deteriorating trend. Soil erosion is also the most interconnected threat, stripping away topsoil, reducing rooting depth, depleting nutrients and carbon, limiting water infiltration, and increasing runoff, thereby heightening the risk of floods and droughts (see Chapter 3).

Other major threats to soil functions, ranked by severity, include the loss of SOC, nutrient mismanagement (both excessive use of nutrients and nutrient mining), loss of soil biodiversity, salt accumulation, soil pollution and urban expansion (Table 16.1). The assessment indicates that the adoption of SSM practices to address most of these soil threats remains largely fair, poor or very poor. Nonetheless, there have also been some positive and notable developments in Asia, North America and New Zealand, which have made progress in reducing SOC loss. In these regions, the trend is stable or improving (see Chapter 15).

Soil biodiversity loss is the most deteriorating threat globally, although confidence in trend is low. Loss of soil biological diversity diminishes nutrient cycling, gas exchange, pest regulation and resilience to stress.

Practices such as deforestation, frequent or deep tillage, insufficient or no soil cover, poor irrigation and drainage management, inappropriate use of agrochemicals, overgrazing and traffic on wet soils remove protective vegetation and destabilize soil structure. The negative effects of these unsustainable management practices are exacerbated by climate change, conflicts and other human activities that compete for land use (see Chapter 4).

Because these threats are interlinked, tackling a single threat (such as erosion) can generate cascading benefits, with improved SOC retention, better nutrient cycling, enhanced biodiversity and greater resilience to climate stress. Integrated approaches that combine complementary SSM practices are particularly effective. For example, the joint adoption of reduced or no-till farming with residue retention and cover crops has been shown to simultaneously reduce erosion and runoff, while enhancing soil structure, improving nutrient use efficiency and stabilizing yields over time. Similarly, integrated soil fertility management – combining mineral fertilizers with organic amendments, legumes and context-specific nutrient management – can address nutrient mining while reducing environmental losses. Agroforestry systems that integrate trees with crops and livestock, often combine permanent soil cover, diversified rooting systems and organic inputs, thereby improving carbon sequestration, water regulation and biodiversity. Such integrated approaches tackle multiple degradation processes concurrently, help counteract the cumulative impacts of land-use change, climate variability and management pressures, and are more likely to deliver durable, system-wide improvements in soil health than isolated, single-practice interventions.

Key message 3: Scientific knowledge and proven practices for sustainable soil management have expanded greatly, but this knowledge is not being applied at scale

Many appropriate SSM practices exist and there is awareness of the importance of sustainably managing soils to ensure the provision of functions and ecosystem services. However, in many regions, the mechanisms and support systems needed to implement these practices at scale are ineffective or absent.

A major focus of research in soil science since 2015 has been the assessment of the effectiveness of existing SSM practices. Hundreds of individual studies have assessed the effectiveness of multiple SSM practices, including addition of organic amendments such as manure, reduced tillage or no-till, cover crops, the application of enhanced efficiency fertilizers, residue retention, biochar additions and structural measures such as terracing. Extensive, long-term studies conducted across diverse geographic regions, soil types, climate conditions and cropping systems have produced valuable insights into the performance of these practices under varying conditions. The findings of these studies, summarized in Chapter 5, show that SSM practices can indeed mitigate soil threats effectively, although it is consistently highlighted that there is no one-size-fits-all solution. The success and applicability of specific practices depend heavily on local biophysical factors (e.g. soil texture and rainfall patterns) and socioeconomic conditions (e.g. access to inputs and land tenure security). Such knowledge allows for more tailored recommendations, guiding regional and local decisions on which SSM techniques are most appropriate.

However, despite their proven benefits, there is a gap between knowing and doing. Farmers and land managers may understand the benefits of SSM practices, but they often face practical, economic or institutional barriers to their adoption. Furthermore, current initiatives, while locally important, are not yet having a regional or global impact on slowing or stopping soil and land degradation. Thus, it is important to redouble efforts to establish an enabling environment (e.g. policies, incentives and tenure security) to ensure that the implementation of SSM practices is accelerated.

Key message 4: When sustainability delivers quick results, farmers act, but to reap long-term benefits that benefit society, policy support becomes essential

For farmers and land managers, SSM practices are more widely adopted when they deliver clear short-term economic benefits, such as improved soil fertility, increased yields, reduced input costs or greater resilience to droughts. In contrast, adoption remains limited for practices whose benefits accrue primarily to society – such as improved water quality, reduced nutrient losses, lower greenhouse gas (GHG) emissions, carbon sequestration and biodiversity conservation – and where

private economic gains are delayed or less immediately visible (or if there is even a perception of potential yield reduction). Often, the benefits of SSM take several years or longer to materialize and may initially involve added costs or even short-term yield reductions (see Chapter 5).

This "implementation gap" causes an imbalance that discourages the adoption of practices that, while beneficial over the long term for both farmers and society, may appear less profitable in the short term. Thus, policy measures, financial incentives and other enabling conditions – including market-based instruments – are needed to offset early costs, reduce risks during the transition period and support farmers in realizing the full long-term value of SSM (see Chapter 6).

Key message 5: Even the best sustainable soil management strategies falter without the right combination of information, engagement and the backing of governance

Initiatives promoting the adoption of SSM often fail to achieve their goals when an adequate enabling environment to support their implementation is not present. This enabling environment refers to the provision of timely, relevant and accessible soil information, extension services and technical advice along with strong and inclusive governance structures (see Chapter 6).

To be useful, information should be findable, accessible, interoperable and reusable and most importantly be turned into actionable guidance and decision-support tools for soil and land management. For this to happen, it is necessary to establish strong education and extension services to inform stakeholders about locally relevant and tailored SSM and other sustainable agricultural practices and methods and support those services in their adoption.

A major point of agreement across studies, as summarized in Chapter 6, is that initiatives which do not respect, traditional and Indigenous Peoples' knowledge often fail. Respect in this context means actively involving local communities in the design of SSM programmes, valuing and blending their knowledge with scientific insights, and ensuring that Indigenous Peoples and community representatives have a voice in decision-making. It also means maintaining dialogue during implementation, so that any issues can be addressed in a culturally sensitive way as they emerge. Successful implementation can also be hindered if gender roles and disparities are not addressed. In many regions, women are heavily engaged in farming and land care, yet may lack formal land rights or decision-making power to change soil management practices.

The adoption of certain SSM practices may offer limited short-term benefits in terms of crop yields or require significant upfront investment in specific technologies or inputs. In such cases, the availability of targeted incentives becomes critical to overcoming financial and behavioural barriers to adoption. Market-based incentives, supportive subsidies, regulations and cross-compliance are types of incentive mechanisms that can support the uptake of SSM. However, the effectiveness of these incentives depends heavily on their economic viability. When payments or benefits fail to offset the costs – whether they are direct or indirect, or opportunity costs associated with transitioning to new practices – farmers are unlikely to adopt them. Conversely, when incentive structures are designed to adequately compensate for income losses and additional costs, adoption rates tend to increase significantly.



16.3 New challenges, familiar solutions: fostering collaborative commitment and shared responsibilities to promote sustainable soil management

Globally, according to the evidence compiled in the report, soil resources remain under severe pressure. Sustainable soil management is effective in reducing threats to soil functions but the adoption of SSM practices since 2015 has been limited. The main barrier to wider adoption of SSM is the lack of effective incentives and soil governance structures. The enabling environment for the adoption of SSM is also often weak, further limiting implementation. Inclusive governance, knowledge co-production and targeted policies are critical to achieving widespread SSM adoption.

Urgent, coordinated and sustained action is needed at all levels, from local to national, regional and international, and by multiple actors involved in soil management (including international organizations, regional, national and local governments, policymakers, technicians, land managers, the private sector, scientists and civil society). The following recommendations proposed by the ITPS are intended to target the removal of barriers that currently limit both the assessment and adoption of SSM:

1. Strengthen soil governance and legal frameworks by:

    a. promoting greater integration of soil in multilateral environmental agreements and in development policies and agendas related to agrifood systems, the environment, climate, biodiversity, land and water through the involvement of international organizations, regional economic communities and governments;

    b. strengthening national soil-related policies and legislation to protect soils, promote sustainable management and integrate soil objectives into agrifood systems, environment, climate, biodiversity, urban and infrastructure development, and land and water agendas, with a focus on empowering smallholders, vulnerable groups and rural communities through the involvement of governments, with the support of international organizations and other partners;

    c. enhancing soil governance through inclusive, transparent and coordinated decision-making processes, following a coherent, participatory and cross-sectoral approach through the involvement of governments;

    d. promoting land tenure security and land-use rights to incentivize long-term soil stewardship through the involvement of governments and international organizations;

    e. integrating SSM into national and local land–use plans, zoning decisions and infrastructure development to prevent soil degradation, through the involvement of governments.

2. Promote education, capacity development, and knowledge exchange by:

    a. strengthening international cooperation and science–policy dialogue through existing intergovernmental and multistakeholder platforms, such as the Global Soil Partnership (GSP), through the involvement of international organizations and governments;

    b. promoting peer-to-peer knowledge sharing and training on SSM, drawing

on local champions or innovators to share traditional and local knowledge through the involvement of international organizations, governments, extension services and land managers;

c. addressing gender-based barriers to SSM adoption by applying gender- responsive and transformative analysis and delivering targeted support to women and marginalized groups through the involvement of international organizations, governments and extension services;

d. investing in education, extension and capacity-building programmes that deliver context-specific recommendations to land managers through the involvement of governments and international organizations

3. Improving soil and SSM monitoring systems through harmonized soil data and science-based models by:

a. measuring and reporting the area of land under specific SSM practices (e.g. cover crops and conservation tillage) to support agrifood systems transformation through the involvement of international organizations, governments and scientists;

b. improving the availability, accessibility and usability of soil information for decision-making at all levels through the involvement of international organizations, governments and scientists;

c. improving soil monitoring and modelling by investing in higher-resolution data, better field validation and cross-model comparisons through the involvement of international organizations and scientists;

d. calibrating specific SSM practices to local conditions through the involvement of governments, extension services, scientists and international organizations;

e. developing regionally relevant thresholds for the threats to soil functions through the involvement of scientists;

f. cooperating in the development of multidisciplinary research projects with a system-wide vision that provide scientific information and evidence for policy development and adoption of SSM through the involvement of scientists;

g. providing accessible information on implementation of practices through the involvement of governments, extension services and land managers; and

h. monitoring the impact of SSM practices on the soil and keeping simple farm records on inputs, yields, and observed changes over time through the involvement of extension services and land managers.

4. Enhancing incentive mechanisms to support adoption of SSM by:

a. promoting the integration and recognition of soil in international financing mechanisms and institutions such as investment and multilateral development banks, private sector and national sectoral budgets that favour the adoption of SSM and supporting governments in accessing them through integrated and systemic actions through the involvement of governments, the private sector, investment and development banks, financing institutions and international organizations; and

b. developing and deploying targeted economic incentives, grants and subsidies for adopting SSM, such as payments for ecosystem services that compensate land users for practices that deliver societal benefits through the involvement of governments, the private sector, investment banks and financing institutions, and international organizations).

5. Foster inclusive, multistakeholder and multidisciplinary decision-making that promotes the adoption of the SSM by:

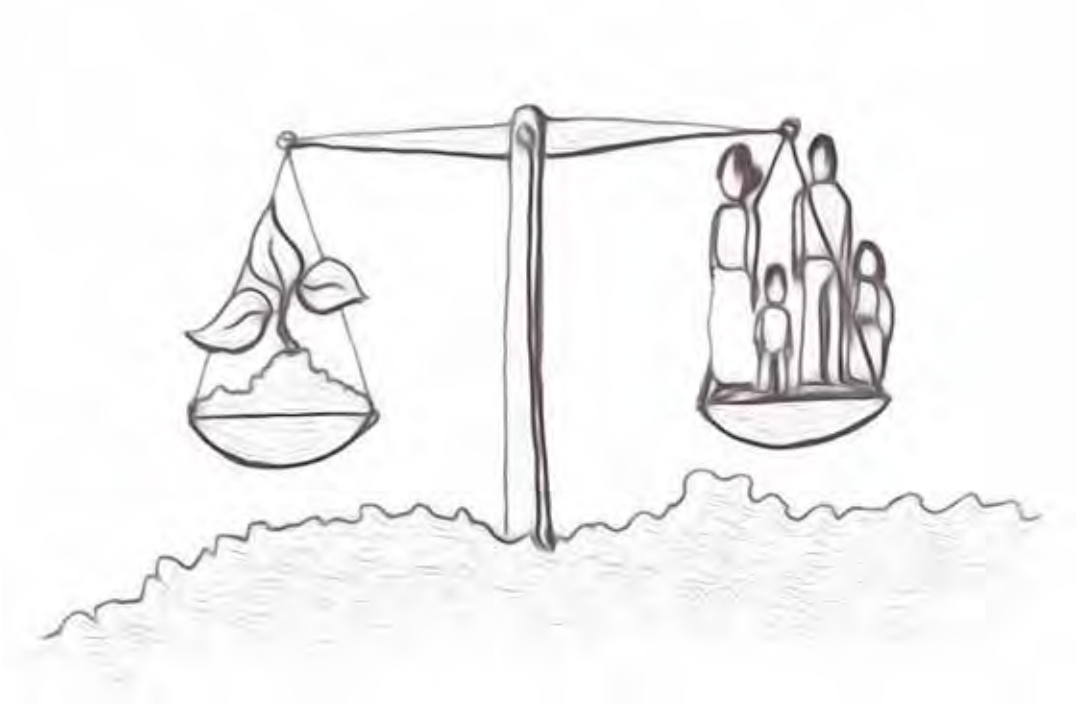
   a. promoting the establishment and strengthening of national soil partnerships and other comparable multisectoral and multistakeholder coordination mechanisms to facilitate knowledge and technology transfer, targeted advisory services and access to financing mechanisms through the involvement of national and local governments and their different agencies and departments, extension services, scientist, the private sector, non-governmental organizations, civil society organizations and land managers;

   b. promoting the transition to sustainable agrifood systems through the development, promotion and application of integrated solutions that address human–environment interactions and the rural–urban interface through the involvement of international organizations, governments, extension services, scientists, the private sector and land managers;

   c. engaging local communities and respecting traditional and Indigenous Peoples' knowledge in designing and implementing SSM initiatives through the involvement of international organizations, governments, extension, services scientists, the private sector and land managers;

   d. encouraging the restoration and rehabilitation of degraded soils, using context-appropriate SSM through the involvement of international organizations, governments, extension services, scientists, the private sector and land managers; and

   e. acting as stewards of the soil to ensure that this essential natural resource is safeguarded for future generations through the involvement of land managers.




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The Global Soil Partnership (GSP) is a globally recognized mechanism established in 2012. Our mission is to position soils in the Global Agenda through collective action. Our key objectives are to promote Sustainable Soil Management (SSM) and improve soil governance to guarantee healthy and productive soils, and support the provision of essential ecosystem services towards food security and improved nutrition, climate change adaptation and mitigation, and sustainable development.

The Intergovernmental Technical Panel on Soils (ITPS) provides scientific and technical advice on global soil issues. Members of the ITPS are nominated by the GSP Plenary Assembly for an initial term of 2 years, renewable for one additional term (with agreement of the GSP Plenary Assembly). These experts act in their personal capacity, not receive instructions from any government or institution, and provide the best possible scientific and technical knowledge available. The ITPS advise the GSP and FAO itself as well provide an authoritative technical voice on global soil issues.

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