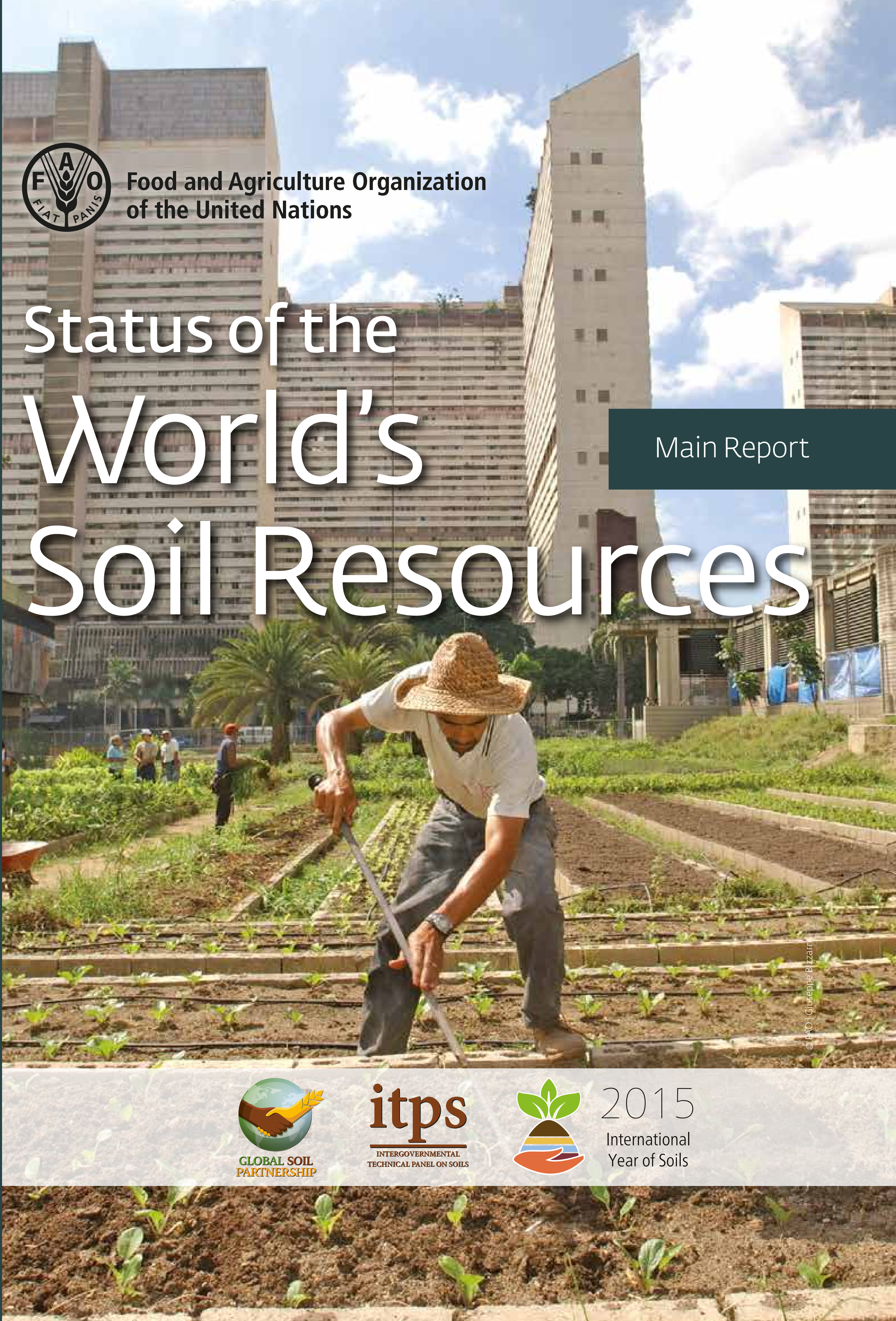
Food and Agriculture Organization
of the United Nations

Status of the World's Soil Resources

Main Report



© FAO | Giuseppe Bizzarri

GLOBAL SOIL PARTNERSHIP

itps
INTERGOVERNMENTAL TECHNICAL PANEL ON SOILS

2015
International Year of Soils

Status of the World's Soil Resources
Main report

Prepared by
Intergovernmental Technical Panel on Soils (ITPS)

Luca Montanarella (Chair), Mohamed Badraoui, Victor Chude, Isaurinda Dos Santos Baptista Costa, Tekalign Mamo, Martin Yemefack, Milkha Singh Aulang, Kazuyuki Yagi, Suk Young Hong, Pisoot Vijarnsorn, Gan Lin Zhang, Dominique Arrouays, Helaina Black, Pavel Krasilnikov, Jaroslava Sobocá, Julio Alegre, Carlos Roberto Henriquez, Maria de Lourdes Mendonça-Santos, Miguel Taboada, David Espinosa Victoria, Abdullah Alshankiti, Sayed Kazem Alavi Panah, Elsiddig Ahmed El Mustafa El Sheikh, Jon Hempel, Dan Pennock, Marta Camps Arbestain, Neil McKenzie.

Editorial board: Luca Montanarella (Chair), Victor Chude (Africa), Kazuyuki Yagi (Asia), Pavel Krasilnikov (Europe), Seyed Kazem Alavi Panah (Near East and North Africa), Maria de Lourdes Mendonça-Santos (Latin America and the Caribbean), Dan Pennock (North America), Neil McKenzie (SW Pacific).

Managing editor: Freddy Nachtergaele

Coordinating Lead Authors and Regional Coordinators/Authors:
Mubarak Abdelrahman Abdalla, Seyed Kazem Alavipanah, André Bationo, Victor Chude, Juan Comerma, Maria Gerasimova, Jon Hempel, Srimathie Indraratne, Pavel Krasilnikov, Neil McKenzie, Maria de Lourdes Mendonça-Santos, Chencho Norbu, Ayo Ogunkunle, Dan Pennock, Thomas Reinsch, David Robinson, Pete Smith, Miguel Taboada and Kazuyuki Yagi.

Reviewing Authors: Dominique Arrouays, Richard Bardgett, Marta Camps Arbestain, Tandra Fraser, Ciro Gardi, Neil McKenzie, Luca Montanarella, Dan Pennock and Diana Wall.

Other contributing Authors:

Adams, Mary Beth
Adhya Tapan, Kumar
Agus, Fahmuddin
Al Shankithi, Abdullah
Alegre, Julio
Aleman, Garcia
Alfaro, Marta
Alyabina, Irina
Anderson, Chris
Anjos, Lucia
Arao, Tomohito
Asakawa, Susumu
Aulakh, Milkha
Ayuke, Frederick
Bai, Zhaohai
Baldock, Jeff
Balks, Megan
Balyuk, Svyatoslav
Bardgett, Richard
Basiliko, Nathan
Batkhishig, Ochirbat
Bedard-Haughn, Angela
Bielders, Charles
Bock, Michael
Bockheim, James
Bondeau, Alberte
Brinkman, Robert
Bristow, Keith
Broll, Gabrielle
Bruulsma, Tom
Bunning, Sally
Bustamante, Mercedes
Caon, Lucrezia
Carating, Rodel
Cerkowniak, Darrel
Charzynski, Przemyslaw
Clark, Joanna
Clothier, Brent
Coelho, Maurício Rizzato
Colditz, Roland René
Collins, Alison
Compton, Jana
Condron, Leo
Corso, Maria Laura
Cotrufo, Francesca
Critchley, William
Cruse, Richard
da Silva, Manuela
Dabney, Seth
Daniels, Lee
de Souza Dias, Moacir
Dick, Warren
Dos Santos Baptista, Isaurinda
Drury, Craig
El Mustafa El Sheikh, Ahmed Elsiddig
Elder-Ratutokarua, Maria
Elliott, Jane
Espinosa, David
Fendorf, Scott

Ferreira, Gustavo
Flanagan, Dennis
Gafurova, Laziza
Gaistardo, Carlos Cruz
Govers, Gerard
Grayson, Sue
Griffiths, Robert
Grundy, Mike
Hakki Emrah, Erdogan
Hamrouni, Heidi
Hanly, James
Harper, Richard
Harrison, Rob
Havlicek, Elena
Hempel, Jon
Henriquez, Carlos Roberto
Hewitt, Allan
Hiederer, Roland
House, Jo
Huising, Jeroen
Ibánez, Juan José
Jain, Atul
Jefwa, Joyce
Jung, Kangho
Kadono, Atsunobu
Kawahigashi, Masayuki
Kelliher, Frank
Kihara, Job
Konyushkova, Maria
Kuikman, Peter
Kuziev, Ramazan
Lai, Shawntine
Lal, Rattan
Lamers, John
Lee, Dar-Yuan
Lee, Seung Heon
Lehmann, Johannes
Leys, John
Lobb, David
Ma, Lin
Macias, Felipe
Maina, Fredah
Mamo, Tekalign
Mantel, Stephan
McDowell, Richard
Medvedev, Vitaliy
Miyazaki, Tsuyushi
Moore, John
Morrison, John
Mung'atu, Joseph
Muniz, Olegario
Nachtergaele, Freddy
Nanzyo, Masami
Ndiaye, Déthié
Neall, Vince
Noroozi, Ali Akbar
Obst, Carl
Ogle, Stephen
Okoth, Peter
Omutu, Christian
Or, Dani
Owens, Phil
Pan, Genxing
Panagos, Panos
Parikh, Sanjai
Pasos Mabel, Susan (†)
Paterson, Garry
Paustian, Keith
Petri, Monica
Pietragalla, Vanina
Pla Sentis, Ildefonso
Polizzotto, Matthew
Pugh, Thomas
Qureshi, Asad
Reddy, Obi
Reid, D. Keith
Richter, Dan
Rivera-Ferre, Marta
Rodriguez Lado, Luis
Roskruge, Rick
Rumpel, Cornelia
Rys, Gerald
Schipper, Louis
Schoknecht, Noel
Seneviratne, Sonia
Shahid, Shabbir
Sheffield, Justin
Sheppard, Steve
Sidhu, Gurjant
Sigbert, Huber
Smith, Scott
Sobocká, Jaroslava
Sönmez, Bülent
Spicer, Anne
Sposito, Garrison
Stolt, Mark
Suarez, Don
Takata, Yusuke
Tarnocai, Charles
Tassinari, Diego
Tien, Tran Minh
Toth, Tibor
Trumbore, Susan
Tuller, Markus
Urquiaga Caballero, Segundo
Urquiza Rodrigues, Nery
Van Liedekerke, Marc
Van Oost, Kristof
Vargas, Rodrigo
Vargas, Ronald
Vela, Sebastian
Vitaliy, Medvedev
Vrscaj, Boris
Waswa, Boaz
Watanabe, Kazuhiko
Watmough, Shaun
Webb, Mike
Weerahewa, Jeevika
West, Paul
Wiese, Liesl
Wilding, Larry
Xu, Renkou
Yan, Xiaoyuan
Yemefack, Martin
Yokoyama, Kazunari
Zhang, Fusuo
Zhou, Dongmei
Zobeck, Ted

Editorial team: Lucrezia Caon, Nicoletta Forlano, Cori Keene, Matteo Sala, Alexey Sorokin, Isabelle Verbeke, Christopher Ward.

GSP Secretariat: Moujahed Achouri, Maryse Finka and Ronald Vargas.

FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS
Rome, 2015

Disclaimer and copyright

Recommended citation:
FAO and ITPS. 2015.
Status of the World's Soil Resources (SWSR) – Main Report.
Food and Agriculture Organization of the United Nations
and Intergovernmental Technical Panel on Soils, Rome, Italy

ISBN 978-92-5-109004-6

Table of contents

Foreword

This document presents the first major global assessment ever on soils and related issues.

Why was such an assessment not carried out before? We have taken soils for granted for a long time. Nevertheless, soils are the foundation of food production and food security, supplying plants with nutrients, water, and support for their roots. Soils function as Earth's largest water filter and storage tank; they contain more carbon than all above-ground vegetation, hence regulating emissions of carbon dioxide and other greenhouse gases; and they host a tremendous diversity of organisms of key importance to ecosystem processes.

However, we have been witnessing a reversal in attitudes, especially in light of serious concerns expressed by soil practitioners in all regions about the severe threats to this natural resource. In this more auspicious context, when the international community is fully recognizing the need for concerted action , the Intergovernmental Technical Panel on Soils (ITPS), the main scientific advisory body to the Global Soil Partnership (GSP) hosted by the Food and Agriculture Organization of the United Nations (FAO), took the initiative to prepare this much needed assessment.

The issuance of this first "Status of the World's Soil Resources" report was most appropriately timed with the occasion of the International Year of Soils (2015) declared by the General Assembly of the United Nations. It was made possible by the commitment and contributions of hosts of reputed soil scientists and their institutions. Our gratitude goes to the Lead Authors, Contributing Authors, Editors and Reviewers who have participated in this effort, and in particular to the Chairperson of the ITPS, for his dedicated guidance and close follow up.

Many governments have supported the participation of their resident scientists in the process and contributed resources, thus also assuring the participation of experts from developing countries and countries with economies in transition. In addition, a Technical Summary was acknowledged by representatives of governments assembled in the Plenary Assembly of the GSP, signaling their appreciation of the many potential uses of the underlying report. Even more comprehensive and inclusive arrangements will be sought in the preparations of further, updated versions.

The report is aimed at scientists, laymen and policy makers alike. It provides in particular an essential benchmark against periodical assessment and reporting of soil functions and overall soil health at global and regional levels. This is of particular relevance to the Sustainable Development Goals (SDGs) that the international community pledged to achieve. Indeed, these goals can only be achieved if the crucial natural resources – of which soils is one – are sustainably managed.

The main message of this first edition is that, while there is cause for optimism in some regions, the majority of the world's soil resources are in only fair, poor or very poor condition. Today, 33 percent of land is moderately to highly degraded due to the erosion, salinization, compaction, acidification and chemical pollution of soils. Further loss of productive soils would severely damage food production and food security, amplify food-price volatility, and potentially plunge millions of people into hunger and poverty. But the report also offers evidence that this loss of soil resources and functions can be avoided. Sustainable soil management, using scientific and local knowledge and evidence-based, proven approaches and technologies, can increase nutritious food supply, provide a valuable lever for climate regulation and safeguarding ecosystem services.

We can expect that the extensive analytical contents of this report will greatly assist in galvanizing action at all levels towards sustainable soil management, also in line with the recommendations contained in the updated World Soil Charter and as a firm contribution to achieve the Sustainable Development Goals.

We are proud to make this very first edition of the Status of the World's Soil Resources report available for the international community, and reiterate once again our commitment to a world free of poverty, hunger and malnutrition.

JOSÉ GRAZIANO DA SILVA
FAO Director-General

Acknowledgments

The Status of the World's Soil Resources report was made possible by the commitment and voluntary work of the world's leading soil scientists and the institutions they are affiliated with. We would like to express our gratitude to all the Coordinating Lead Authors, Lead Authors, Contributing Authors, Review Editors and Reviewers. We would also like to thank the editorial staff and the GSP Secretariat for their dedication in coordinating the production of this first seminal report.

Appreciation is expressed to many Governments who have supported the participation of their resident scientists in this major enterprise. In particular, our gratitude to the European Commission who financially supported the development and publication of this report.

List of abbreviations

AAFC	Agriculture and Agri-Food Canada
ACSAD	Arab Centre for the Study of Arid Zones and Dry Lands
AD	Anno Domini
AEZ	Agro-Ecological Zones
AFES	Association Française Pour L'étude Du Sol
AFSIS	African Soil Information Service
AGES	Austrian Agency for Health and Food Safety
AGRA	Alliance for a Green Revolution in Africa
AKST	Agricultural Knowledge Science and Technology
ALOS	Advanced Land Observation Satellite
AMA	Agencia De Medio Ambiente
AMF	Arbuscular Mycorrhizal Fungi
ANC	Acid-Neutralising Capacity
AOAD	Arab Organization for Agricultural Development
AOT	Aerosol Optical Thickness
APO-FFTC	Asian Productivity Organization- Food & Fertilizer Technology Center
ARC	Agricultural Research Council
ASGM	Artisanal and Small-Scale Gold Mining
ASI	Advanced Science Institutesseries
ASP	Asia Soil Partnership
ASSOD	Assessment of Human-Induced Soil Degradation in South and Southeast Asia
AU	African Union
BASE	Biome of Australia Soil Environments
BC	(1) Black Carbon; (2) Before Christ
BD	Biodiversity
BDP	Bureau for Development Policy
BIH	Bosnia And Herzegovina
BMLFUW	Austrian Federal Ministry of Agriculture, Forestry, Environment and Water Management
BNF	Biological Nitrogen Fixing
BOM	Bureau of Meteorology
BP	Before Present (1 January 1950)
C:N	Carbon To Nitrogen Ratio
CA	Conservation Agriculture
CAAA	Clean Air Act Amendments
CAADP	Comprehensive Africa Agriculture Development Programme

CACILM	Central Asian Countries Initiative for Land Management
CAMRE	Council of Arab Ministers Responsible For the Environment
CAZRI	Central Arid Zone Research Institute
CBD	Convention on Biological Diversity
CBM-CFS	Carbon Budget Model of the Canadian Forest Sector
CCAFS	Climate Change, Agriculture and Food Security
CCME	Canadian Council Of Ministers of the Environment
CE	Common Era (Also Current era or Christian era)
CEC	(1) Cation Exchange Capacity; (2) Commission of the European Communities
CECS	Chemicals of Emerging Concern
CEPAL	Comisión Económica Para América Latina Y El Caribe
CF	Commercial Farming
CGIAR	Global Agricultural Research Partnership
CIAT	International Center for Tropical Agriculture
CIFOR	Center for International Forestry Research
CITMA	Ministerio De Ciencia, Tecnologia Y Medio Ambiente
CLIMSOIL	Review of Existing Information on the Interrelations between Soil and Climate Change
CLM	Contaminated Land Management
CMIP 5	Coupled Model Intercomparison Project Phase 5
COM	Commission Working Documents
CONABIO	Comision Nacional Para El Conocimiento Y Uso De La Biodiversidad
CONAFOR	Comisión Nacional Forestal
COSMOS	Cosmic-Ray Soil Moisture Observing System
CRC	Risk of Colorectal Cancer
CRP	Conservation Reserve Program
CSA	Climate-Smart Agriculture
CSIF-SLM	Country Strategic Investment Framework for Sustainable Land Management
CSIRO	Commonwealth Scientific and Industrial Research Organisation
CSM-BGBD	Conservation and Sustainable Management of Below-Ground Biodiversity
CSSRI	The Central Soil Salinity Research Institute
CSWCR&TI	Central Soil & Water Conservation Research & Training Institute (India)
DAFWA	Department Of Agriculture and Food, Western Australia
DBC	Dissolved Black Carbon
DDT	Dichlorodiphenyltrichloroethane
DEA	Deliberate Evacuation Area
DECA	Department Of Environment and Conservation, Australia
DED	Dust Event Days
DENR	Department Of Environment and Natural Resources

DEST	Australian Government Department of Education, Science and Training
DGVMS	Dynamic Global Vegetation Models
DIC	Dissolved Inorganic Carbon
DLDD	Desertification, Land Degradation and Drought
DNA	Deoxyribonucleic Acid
DOC	Dissolved Organic Carbon
DOI	Digital Object Identifier
DPYC	Dissolved Pyrogenic Carbon
DSEWPAC	Department Of Sustainability, Environment, Water, Population and Communities
DSI	Dust Storm Index
DSMW	Digital Soil Map of the World
EA-20km	Twenty Km Evacuation Area
EAD	Environment Agency Abu Dhabi
EC DG ENV	European Commission Directorate-General for Environment
EC	European Commission
EEA	European Environment Agency
EEAA	Egyptian Environmental Affairs Agency
EEZ	Exclusive Economic Zone
ELD	Economics of Land Degradation
EM-DAT	Emergency Events Database
ENSO	El Niño Southern Oscillation
EOLSS	Encyclopedia of Life Support Systems
EPA CERCLIS	United States Environmental Protection Agency, Comprehensive Environmental Response, Contamination and Liability Information System
EPA	United States Environmental Protection Agency
ERW	Explosive Remnants of War
ES	Ecosystem Services
ESA	United Nations Economic and Social Affairs Department
ESAFS	East and Southeast Asia Federation of Soil Science Societies
ESCWA	United Nations Economic and Social Commission for Western Asia
ESDB	European Soil Database
ESP	Exchangeable Sodium Percentage
ESRI	Environmental Systems Research Institute
EU SCAR	European Standing Committee on Agricultural Research
EU	European Union
FAO	Food and Agriculture Organization of the United Nations
FAOSTAT	Food and Agriculture Organization Corporate Statistical Database
FAO-WRB	Food and Agriculture Organization World Reference Base

FDNPS	Fukushima Dai-Ichi Nuclear Power Station
FFS	Farmer Field School
FIA	Forest Inventory and Analysis
FSI	Forest Survey in India
FSR	Fund-Service-Resources
GAP	Southeast Anatolia Development Project Region
GDP	Gross Domestic Product
GEF	Global Environment Facility
GEO	Global Environmental Outlook
GHG	Greenhouse Gases
GIS	Geographic Information System
GIZ	Deutsche Gesellschaft Für Internationale Zusammenarbeit (GIZ) Gmbh
GLADA	Global Land Degradation Assessment
GLADIS	Global Land Degradation Information System
GLASOD	Global Assessment of Human-Induced Soil Degradation
GLC 2000	Global Land Cover 2000 Project
GLC-SHARE	Global Land Cover SHARE
GLRD	Gender and Land Rights Database
GRACE	Gravity Recover and Climate Experiment
GRID	Global Resource Information Database
GSBI	Global Soil Biodiversity Initiative
GSM	Global Soil Map
GSP	Global Soil Partnership
HORTNZ	Horticulture New Zealand
HTAP	Hemispheric Transport of Air Pollution
HWSD	Harmonized World Soil Database
HYDE	History Database of the Global Environment
IAASTD	International Assessment of Agricultural Knowledge, Science and Technology for Development
IAATO	International Association of Antarctic Tour Operators
ICAR	Indian Council of Agricultural Research
ICARDA	International Center for Agriculture Research In The Dry Areas
ICBA	International Center for Biosaline Agriculture
ICBL	International Campaign to Ban Landmines
ICRAF	International Center for Research in Agroforestry
IDP	Internally Displaced Peoples
IFA	International Fertilizers Association
IFAD	International Fund for Agricultural Development

IFADATA	International Fertilizer Industry Association Database
IFPRI	International Food Policy Research Institute
IGT-AMA	Instituto De Geografía Tropical Y La Agencia De Medio Ambiente
IIASA	International Institute for Applied Systems Analysis
ILCA	International Livestock Centre for Africa
IMAGE	Integrated Modelling Of Global Environmental Change
IMBE	Mediterranean Istitute of Biodiversity and Ecology
IMF	International Monetary Fund
IMK-IFU	Institute of Meteorology and Climate Research Atmospheric Environmental Research
INIA	Instituto De Investigaciones Agropecuarias (Chile)
IPBES	Intergovernmental Panel on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
IRENA	International Renewable Energy Agency
IROWC-N	The Indicator of Risk of Water Contamination by Nitrogen
IROWC-P	Indicator of Risk of Water Contamination by Phosphorus
ISA	Impervious Surface Area
ISAM	Integrated Impacts of Climate Change Model
ISBN	International Standard Book Number
ISCO	International Soil Conservation Organization
ISCW	Institute for Soil, Climate and Water
ISFM	Integrated Soil Fertility Management
ISO	International Standards Organization
ISRIC	International Soil Reference and Information Centre
ISS-CAS	Institute of Soil Science – Chinese Academy of Sciences
ISSS	International Society for the Systems Sciences
ITPS	Intergovernmental Technical Panel on Soils
IUSS	International Union of Soil Sciences
IW	International Waters
JRC	Joint Research Centre (European Commission)
LAC	Latin America and the Caribbean
LADA	Land Degradation Assessment in Drylands
LCCS	Land Cover Classification System
LD	Land Degradation
LDCS	Least Developed Countries
LPFN	the Landscapes for People, Food and Nature
LPJ-GUESS	Lund-Potsdam-Jena General Ecosystem Simulator
LRTAP	Long-Range Transboundary Air Pollution
LS	Topographic Factors

LU	Land Use
MA	Millennium Ecosystem Assessment
MADRPM	Ministère De l'Agriculture Du Développement Rural et Des Pêches Maritimes
MAF	New Zealand Ministry of Agriculture and Forestry
MAFF	Ministry of Agriculture, Forestry and Fishery of Japan
MDBA	Murray–Darling Basin Authority (Australia)
MDGS	Millennium Development Goals
MENARID	Integrated Natural Resources Management in the Middle East And North Africa
MGAP	Ministry of Livestock, Agriculture and Fisheries
MNP	Netherlands Environmental Assessment Agency
MODIS	Moderate Resolution Imaging Spectroradiometer
NAAS	National Academy of Agricultural Sciences of India
NAIP	National Agricultural Investment Plan
NAMA	Nationally Appropriate Mitigation Action
NAP	(1) National Action Programme; (2) National Action Plan
NAPA	National Adaptation Programme of Action
NBSAP	National Biodiversity Strategy and Action Plan
NBSS&LUP	National Bureau Of Soil Survey And Land Use Planning
NDVI	Normalized Difference Vegetation Index
NENA	Near East And North Africa Region
NEPAD	The New Partnership for Africa's Development
NEST	Nigerian Environmental Study Action Team
NGO	Non-Governmental Organization
NISF	National Institute for Soils And Fertilizers
NLWRA	National Land and Water Resources Audit
NOAA AVHRR	National Oceanic and Atmospheric Administration - Advanced Very High Resolution Radiometer
NPK	Nitrogen (N), Phosphorus (P) and Potassium (K)
NPL	National Priorities List
NRC	National Research Council USA
NRCAN	Natural Resources Canada
NREL	National Resource Ecology Laboratory
NRI	National Resources Inventory Program
NRM	Natural Resources Management
NRSA	National Remote Sensing Agency (India)
NSW	New South Wales
NT	No-Tillage
NUE	Nitrogen Use Efficiency

OECD	Organization for Economic Co-Operation And Development
OM	Organic Matter
ÖNORM	National Standard Published By the Austrian Standards Institute
ÖPUL	Austrian Environment Programme for Agriculture
ORNL-CDIAC	Oak Ridge National Laboratory-Carbon Dioxide Information Analysis Center
OSWER	Office of Solid Waste and Emergency Response
PAH	Polycyclic Aromatic Hydrocarbon
PAM	Polyacrylamide
PCB	Polychlorinated Biphenyl
PCM	Pyrogenic Carbonaceous Matter
PEA	Participatory Expert Assessment
PHC	Petroleum Hydrocarbon
PL	Plastic Limit
PLAR	Participatory Learning-Action-Research
PMID	Pubmed Identifier
PNUD	Programa De Las Naciones Unidas Para El Desarrollo
POC	Particulate Organic Carbon
POP	Persistent Organic Pollutant
PVC	Polyvinyl Chloride
Radar-AMEDAS	Radar-Automated Meteorological Data Acquisition System
RAPA	Regional Office for Asia and the Pacific
RELMA	Sida's Regional Land Management Unit
ROTAP	Review Of Transboundary Air Pollution
RSN	Residual Soil Nitrogen
RUSLE	Revised Universal Soil Loss Equation
SAGYP-CFA	Secretaría De Agricultura, Ganadería Y Pesca – Consejo Federal Agropecuario
SAV	Submerged Aquatic Vegetation
SCAN	Soil Climate Analysis Network
SCARPS	Salinity Control and Reclamation Projects
SCWMRI	Soil Conservation and Watershed Management Research Institute
SD	Soil Degradation
SDGS	Sustainable Development Goals
SEC	Staff Working Documents of European Commission
SEEA	System of Environmental Economic Accounting
SEED	Sustainable Energy and Environment Division
SF	Subsistence Farming
SFR	Stock-Flow-Resources

SKM	Sinclair Knight Merz
SLAM	Sustainable Land and Agro-Ecosystem Management
SLC	Soil Landscapes of Canada
SLM	Sustainable Land Management
SMAP	Soil Moisture Active Passive
SMOS	Soil Moisture Ocean Salinity
SOC	Soil Organic Carbon
SOE	State of the Environment
SOER	European Environment State and Outlook Report
SOLAW	State Of Land and Water
SOM	Soil Organic Matter
SOTER	Soil and Terrain Database
SOW-VU	Centre for World Food Studies of the University Of Amsterdam
SPARROW	Spatially Referenced Regressions on Watershed Attributes
SPC	Secretariat of the Pacific Community
SPI	Science-Policy Interface
SRI	Salinity Risk Index
SSA	Sub-Saharan Africa
SSM	Sustainable Soil Management
SSR	Shift Soil Remediation
SSSA	Soil Science Society of America
ST	Soil Taxonomy
STATSGO 2	Digital General Soil Map of the United States
STEP-AWBH	Soil, Topography, Ecology, Parent Material – Atmosphere, Water, Biotic, Human Model
SWC	Soil and Water Conservation
SWSR	Status of the World's Soil Resources
TEEB	Economics of Ecosystems and Biodiversity
TEOM	Tapered Element Oscillating Microbalances
TOC	Total Organic Carbon
TOMS	Total Ozone Mapping Spectrometer
TOT	Transfer of Technology
TSBF	Tropical Soil Biology and Fertility
UN	United Nations
UNCCD	United Nation Convention to Combat Desertification
UNCED	United Nations Conference on Environment And Development
UNDCPAC	United Nations Desertification Control Program Activity Center
UNDESA	United Nations Department of Economic And Social Affairs
UNDP	United Nations Development Program

UNEP DEWA	United Nations Environment Programme and Department of Early Warning and Assessment
UNEP	United Nations Environment Programme
UNESCO	United Nations Educational, Scientific and Cultural Organization
UNFCCC	United Nations Framework Convention on Climate Change
UNFPA	United Nations Population Fund (Formerly the United Nations Fund for Population Activities)
UNISDR	United Nations Office for Disaster Risk Reduction
UNSO	United Nations Development Programme - Office to Combat Desertification and Drought
USDA	United States Department Of Agriculture
USEPA	United States Environmental Protection Agency
USGS	United States Geological Survey
USLE	Universal Soil Loss Equation
UXO	Unexploded Ordnance
WANA	West Asia-North Africa
WCED	World Commission on Environment and Development
WFP	United Nations World Food Programme
WMO	World Meteorological Organization
WOCAT	World Overview of Conservation Approaches and Technologies
WOTR	Watershed Organization Trust
WRB	World Reference Base for Soil Resources
WRI	World Resources Institute
WWF	World Wildlife Fund

List of tables

List of boxes

List of figures

Preface

The main objectives of **The Status of the World's Soil Resources** are: (a) to provide a global scientific assessment of current and projected soil conditions built on regional data analysis and expertise; (b) to explore the implications of these soil conditions for food security, climate change, water quality and quantity, biodiversity, and human health and wellbeing; and (c) to conclude with a series of recommendations for action by policymakers and other stakeholders.

The book is divided into two parts. The first part deals with global soil issues (Chapters 1 to 8). This is followed by a more specific assessment of regional soil change, covering in turn Africa South of the Sahara, Asia, Europe, Latin America and the Caribbean, the Near East and North Africa, North America, the Southwest Pacific and Antarctica. (Chapters 9 to 16). The technical and executive summaries are published separately.

In Chapter 1 the principles of the World Soil Charter are discussed, including guidelines for stakeholders to ensure that soils are managed sustainably and that degraded soils are rehabilitated or restored. For long, soil was considered almost exclusively in the context of food production. However, with the increasing impact of humans on the environment, the connections between soil and broader environmental concerns have been made and new and innovative ways of relating soils to people have begun to emerge in the past two decades. Societal issues such as food security, sustainability, climate change, carbon sequestration, greenhouse gas emissions, and degradation through erosion and loss of organic matter and nutrients are all closely related to the soil resource. These ecosystem services provided by the soil and the soil functions that support these services are central to the discussion in the report.

In Chapter 2 synergies and trade-offs are reviewed, together with the role of soils in supporting ecosystem services, and their role in underpinning natural capital. The discussion then covers knowledge - and knowledge gaps - on the role of soils in the carbon, nitrogen and water cycles, and on the role of soils as a habitat for organisms and as a genetic pool. This is followed in Chapter 3 by an overview of the diversity of global soil resources and of the way they have been assessed in the past. Chapter 4 reviews the various anthropogenic and natural pressures - in particular, land use and soil management – which cause chemical, physical and biological variations in soils and the consequent changes in environmental services assured by those soils.

Land use and soil management are in turn largely determined by socio-economic conditions. These conditions are the subject of Chapter 5, which discusses in particular the role of population dynamics, market access, education and cultural values as well as the wealth or poverty of the land users. Climate change and its anticipated effects on soils are also discussed in this chapter.

Chapter 6 discusses the current global status and trends of the major soil processes threatening ecosystem services. These include soil erosion, soil organic carbon loss, soil contamination, soil acidification, soil salinization, soil biodiversity loss, soil surface effects, soil nutrient status, soil compaction and soil moisture conditions.

Chapter 7 undertakes an assessment of the ways in which soil change is likely to impact on soil functions and the likely consequences for ecosystem service delivery. Each subsection in this chapter outlines key soil processes involved with the delivery of goods and services and how these are changing. The subsections then review how these changes affect soil function and the soil's contribution to ecosystem service delivery. The discussion is organized according to the reporting categories of the Millennium Ecosystem Assessment, including provisioning, supporting, regulating and cultural services.

Chapter 8 of the report explores policy, institutional and land use management options and responses to soil changes that are available to governments and land users.

The regional assessments in Chapters 9 to 16 follow a standard outline: after a brief description of the main biophysical features of each region, the status and trends of each major soil threat are discussed. Each chapter ends with one or more national case studies of soil change and a table summarizing the results, including the status and trends of soil changes in the region and related uncertainties.

Global soil resources

Coordinating Lead Authors:
Maria Gerasimova (Russia), Thomas Reinsch (United States), Pete Smith (United Kingdom)

Contributing Authors:
Lucia Anjos (Brazil), Susumu Asakawa (Japan), Ochirbat Batkhishig (Mongolia), James Bockheim (United States), Robert Brinkman (Netherlands), Gabrielle Broll (Germany), Mercedes Bustamante (Brazil), Marta Camps Arbestain (ITPS/New Zealand), Przemyslaw Charzynski (Poland), Joanna Clark (United Kingdom), Francesca Cotrufo (United States), Maur'cio Rizzato Coelho (Brazil), Jane Elliott (Canada), Maria Gerasimova (Russia), Robert I. Griffiths (United Kingdom), Richard Harper (Australia), Jo House (United Kingdom), Peter Kuikman (Netherlands), Tapan Kumar Adhya (India), Richard McDowell (New Zealand), Freddy Nachtergaele (Belgium), Masami Nanzyo (Japan), Christian Omutu (Kenya), Genxing Pan (China), Keith Paustian (United States), Dan Pennock (ITPS/Canada), Cornelia Rumpel (France), Jaroslava Sobocká (Slovakia), Mark Stolt (United States), Mabel Susna Pasos (Argentina), Charles Tarnocai (Canada), Tibor Toth (Hungary), Ronald Vargas (Bolivia), Paul West (United States), Larry P. Wilding (United States), Ganlin Zhang (ITPS/China), Juan JosŽ Ibánez (Spain), Felipe Macias (Spain).

Reviewing Authors:
Dominique Arrouays (ITPS/France), Richard Bardgett (United Kingdom), Marta Camps Arbestain (ITPS/New Zealand), Tandra Fraser (Canada), Ciro Gardi (Italy), Neil McKenzie (ITPS/Australia), Luca Montanarella (ITPS/EC), Dan Pennock (ITPS/Canada) and Diana Wall (United States).

1 | Introduction

1.1 | The World Soil Charter

"Soils are fundamental to life on earth."

We know more about soil than ever before, yet perhaps a smaller percentage of people than at any point in human history would understand the truth of this statement. The proportion of human labour devoted to working the soil has steadily decreased through the past century, and hence the experience of direct contact with the soil has lessened in most regions. Soil is very different in this regard from food, energy, water and air, to which each of us requires constant and secure access. Yet human society as a whole depends more than ever before on products from the soil as well as on the more intangible services it provides for maintenance of the biosphere.

Our goal in this report is to make clear these essential connections between human well-being and the soil, and to provide a benchmark against which our collective progress to conserve this essential resource can be measured.

The statement that begins this section is drawn from the opening sentence of the preamble of the revised World Soil Charter (FAO, 2015):

> 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 also provides a valuable lever for climate regulation and a pathway for safeguarding ecosystem services and biodiversity.

The World Soil Charter presents a series of nine principles that summarize our current understanding of the soil, the multi-faceted role it plays, and the threats to its ability to continue to serve these roles. As such, the nine principles form a succinct and comprehensive introduction to this report.

Principles from the World Soil Charter:

Principle 1: Soils are a key enabling resource, central to the creation of a host of goods and services integral to ecosystems and human well-being. The maintenance or enhancement of global soil resources is essential if humanity's overarching need for food, water, and energy security is to be met in accordance with the sovereign rights of each state over their natural resources. In particular, the projected increases in food, fibre, and fuel production required to achieve food and energy security will place increased pressure on the soil.

Principle 2: Soils result from complex actions and interactions of processes in time and space and hence are themselves diverse in form and properties and the level of ecosystems services they provide. Good soil governance requires that these differing soil capabilities be understood and that land use that respects the range of capabilities be encouraged with a view to eradicating poverty and achieving food security.

Principle 3: 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.

Principle 4: The implementation of soil management decisions is typically made locally and occurs within widely differing socio-economic contexts. The development of specific measures appropriate for adoption by local decision-makers often requires multi-level, interdisciplinary initiatives by many stakeholders. A strong commitment to including local and indigenous knowledge is critical.

Principle 5: The specific functions provided by a soil are governed, in large part, by the suite of chemical, biological, and physical properties present in that soil. Knowledge of the actual state of those properties, their role in soil functions, and the effect of change – both natural and human-induced – on them is essential to achieve sustainability.

Principle 6: Soils are a key reservoir of global biodiversity, which ranges from micro-organisms to flora and fauna. This biodiversity has a fundamental role in supporting soil functions and therefore ecosystem goods and services associated with soils. Therefore it is necessary to maintain soil biodiversity to safeguard these functions.

Principle 7: All soils – whether actively managed or not – provide ecosystem services relevant to global climate regulation and multi-scale water regulation. Land use conversion can reduce these global common-good services provided by soils. The impact of local or regional land-use conversions can be reliably evaluated only in the context of global evaluations of the contribution of soils to essential ecosystem services.

Principle 8: Soil degradation inherently reduces or eliminates soil functions and their ability to support ecosystem services essential for human well-being. Minimizing or eliminating significant soil degradation is essential to maintain the services provided by all soils and is substantially more cost-effective than rehabilitating soils after degradation has occurred.

Principle 9: Soils that have experienced degradation can, in some cases, have their core functions and their contributions to ecosystem services restored through the application of appropriate rehabilitation techniques. This increases the area available for the provision of services without necessitating land use conversion.

These nine principles lead to guidelines for action by society (Box 1.1). The guidelines are introduced with a clear statement of our collective goal: *'The overarching goal for all parties is to ensure that soils are managed sustainably and that degraded soils are rehabilitated or restored.'* This opening statement is followed by a series of specific guidelines for different segments of human society. Future updates of this report will document our success in implementation of these guidelines, and in achieving the goal set by the signatories of the World Soil Charter.

Box 1.1 | Guidelines for Action

(from the World Soil Charter)

The overarching goal for all parties is to ensure that soils are managed sustainably and that degraded soils are rehabilitated or restored. Good soil governance requires that actions at all levels – from states, and, to the extent that they are able, other public authorities, international organizations, individuals, groups, and corporations – be informed by the principles of sustainable soil management and contribute to the achievement of a land-degradation neutral world in the context of sustainable development. All actors and, specifically, each of the following stakeholder groups are encouraged to consider the following actions:

Actions by Individuals and the Private Sector

1. All individuals using or managing soil must act as stewards of the soil to ensure that this essential natural resource is managed sustainably to safeguard it for future generations.
2. Undertake sustainable soil management in the production of goods and services.

Actions by Groups and the Science Community

1. Disseminate information and knowledge on soils.
2. Emphasize the importance of sustainable soil management to avoid impairing key soil functions.

Actions by Governments

1. Promote sustainable soil management that is relevant to the range of soils present and the needs of the country.
2. Strive to create socio-economic and institutional conditions favourable to sustainable soil management by removal of obstacles. Ways and means should be pursued to overcome obstacles to the adoption of sustainable soil management associated with land tenure, the rights of users, access to financial services and educational programmes. Reference is made to the Voluntary Guidelines on the Responsible Governance of Tenure of Land, Forests and Fisheries in the Context of National Food Security adopted by the Committee on World Food Security in May 2012.
3. Participate in the development of multi-level, interdisciplinary educational and capacity-building initiatives that promote the adoption of sustainable soil management by land users.
4. Support research programs that will provide sound scientific backing for development and implementation of sustainable soil management relevant to end users.
5. Incorporate the principles and practices of sustainable soil management into policy guidance and legislation at all levels of government, ideally leading to the development of a national soil policy.
6. Explicitly consider the role of soil management practices in planning for adaptation to and mitigation of climate change and maintaining biodiversity.
7. Establish and implement regulations to limit the accumulation of contaminants beyond established levels to safeguard human health and wellbeing and facilitate remediation of contaminated soils that exceed these levels where they pose a threat to humans, plants, and animals.
8. Develop and maintain a national soil information system and contribute to the development of a global soil information system.
9. Develop a national institutional framework for monitoring implementation of sustainable soil management and overall state of soil resources.

Actions by International Organizations

10. Facilitate the compilation and dissemination of authoritative reports on the state of the global soil resources and sustainable soil management protocols.
11. Coordinate efforts to develop an accurate, high-resolution global soil information system and ensure its integration with other global earth observing systems.
12. Assist governments, on request, to establish appropriate legislation, institutions, and processes to enable them to mount, implement, and monitor appropriate sustainable soil management practices.

1.2 | Basic concepts

Prior to the 20th century, soil was considered almost exclusively in the context of agriculture and food production. As the global impact of humanity on natural resources has increased over the past 150 years, the connections between soil and broader environmental concerns began to be made. The recognition of these connections has accelerated through time, and new and innovative ways of relating soils to people have begun to emerge in past the two decades. The rise in complexity of soil knowledge and application was synthesized by Bockheim *et al*. (2005) (Table 1.1) in their summary of milestones in pedology; concepts introduced since 2005 have been added by the authors of this chapter. We can see that the number and breadth of concepts have been expanding rapidly over the past two decades.

Table 1.1. | Chronology of introduction of major concepts in pedology and holistic soil management (after Bockheim et al., 2005).

Period	Pedology	Soil management
Pre 1880	Concept of soil as a medium for plant growth and as a weathered rock layer.	
1880–1900	Appearance of fundamental pedology concepts: soil as a natural body; soil horizons/profiles; soil-forming factors; early ideas of soil geography.	
1900–1940	Global acceptance of concepts of soil as a natural body and soil-forming factors; development of first regional soil classification systems; soil surveys initiated; identification of key soil-forming processes.	Soil conservation
1940– 1960	Factors of soil formation and genesis of soils clarified; development of global soil taxonomic systems; intensified soil mapping.	
1960–1985	Refinement of global soil taxonomic systems; identification of pedon concept; development of early soil models and soil cover pattern concept; recognition of co-evolution of soils and landforms.	World Soil Charter (1981) Land capability/ suitability assessment Assessment of human-induced degradation (GLASOD)
1985– 2000	Increased understanding of soil processes; refinement of global soil models; further refinement of global soil taxonomic systems; development of statistical and computer-based soil information systems.	Sustainable soil management Soil quality Soil health
2000-2015	Earth System Science, pedosphere, digital soil mapping, pedodiversity, ethnopedology, pedometrics, proximal sensing, soil systems. hydropedology, critical zone.	Soil security Carbon sequestration

The connections between soils and societal issues – such as food security, sustainability, climate change, carbon sequestration, greenhouse gas emissions, and degradation through erosion and loss of organic matter and nutrients – are central to the recently developed concept of soil security (McBratney, Field and Koch, 2014). Soil security has been defined as the maintenance or improvement of the world's soil resources so that they can provide sufficient food, fibre, and fresh water, contribute to energy sustainability and climate stability, maintain biodiversity, and deliver overall environmental protection and ecosystem services (Bouma and McBratney, 2013).

There have been major developments over the past three decades in our broader understanding of human impact on the earth and of frameworks to assess this impact. The structure and content of this report comprise a synthesis of themes and concepts from many major initiatives in environmental science and pedology. The most important of these themes and concepts are discussed in the following paragraphs.

Sustainable soil management

The concept of sustainable development is most closely associated with the 1987 report of the United Nations World Commission on Environment and Development, better known as the Brundtland Commission after its chairperson, Gro Harlem Brundtland of Norway (World Commission on Environment and Development, 1987). The report popularized a compelling definition of sustainability: development that meets the needs of the present without compromising the ability of future generations to meet their own needs. The concept of sustainability has since been widely applied to many aspects of human society, including wide application in soil science and land management generally. As defined in the World Soil Charter, sustainable soil management comprises activities that maintain or enhance the supporting, provisioning, regulating and cultural services provided by soils without significantly impairing either the soil functions that enable those services or biodiversity. The concept of sustainable soil management is central to pillar one of the Global Soil Partnership[1]: "Promote sustainable management of soil resources for soil protection, conservation, and sustainable production".

Soil degradation and threats to soil functions

The concept of soil degradation and its assessment have been developed as part of more holistic assessments of human-induced degradation carried out by FAO, UNEP and other UN agencies.

An early initiative was the Global Assessment of Soil Degradation (GLASOD) project undertaken in the late 1980s to inventory soil degradation. GLASOD evaluated 13 types of soil degradation: water erosion (topsoil loss and mass movement, including rill and gully formation), wind erosion (topsoil loss, terrain deformation – primarily dune activity), and overblowing (surface burial from aeolian deposition), loss of nutrients and/or organic matter, salinization, acidification, pollution, compaction and physical degradation, waterlogging, and subsidence of organic soils. GLASOD has not been updated (see Chapter 3 for more details).

The Soil Thematic Strategy of the European Union (CEC, 2006) formalized the concept of threats to soil and its many functions. Five specific threats are identified under Article 6 of the draft Soil Framework Directive proposed in the Strategy: (1) erosion by wind and water; (2) organic matter decline; (3) compaction; (4) salinization; and (5) landslides of soil and rock material. Elsewhere in the proposed Directive, soil sealing ('the permanent covering of the soil with an impermeable surface' p.15) and soil contamination ('the intentional or unintentional introduction of dangerous substances on or in the soil' p. 18) are also identified as threats.

1 The Global Soil Partnership was initiated by FAO and the EU in 2011. For a description of the five pillars, see Table 8.1 in Chapter 8. For a full description of the Partnership, see www.fao.org/globalsoilpartnership).

Soil functions and ecosystem services

The assessments of threats to soil functions leads to a need to formally identify the functions that the soil performs. The proposed Soil Framework Directive (CEC, 2006) of the European Union recognizes seven soil functions that are vulnerable to soil threats:

1. biomass production, including agriculture and forestry
2. storing, filtering and transforming nutrients, substances and water
3. biodiversity pool, such as habitats, species and genes
4. physical and cultural environment for humans and human activities
5. source of raw materials
6. acting as a carbon pool
7. archive of geological and archaeological heritage.

The EU Soil Thematic Strategy was developed at the same time as the Millennium Ecosystem Assessment (MA, 2005) initiated by the United Nations in 2000. The goal of the MA was to assess the consequences of ecosystem change for human well-being and to lay the scientific basis for actions that would promote conservation and sustainable use of ecosystems. The MA was built on the framework for ecosystem services developed by Daily, Matson and Vitousek (1997) and Costanza *et al*. (1997).

The categories of ecosystem services were formalized by the Millennium Ecosystem Assessment into four broad classes: provisioning, regulating, supporting, and cultural services. The range of major ecosystem services provided by soil, and the specific soil functions that enable those services, are summarized in Table 1.2.

Soils and natural capital

The services provided by soils are primarily determined by the three core soil properties (texture, mineralogy, and organic matter), which together form the natural capital of soils (Palm *et al*. 2007). Soil texture and mineralogy are inherent properties of soil that are initially inherited from the parent materials and which change only very slowly over time. In a natural state, soil organic matter (SOM) reaches equilibrium with the environment in which the soil forms, but SOM responds quickly to human-induced changes. Management of SOM is central to sustainable soil management because of its rapid response to change and our ability to manipulate it.

Planetary boundaries and safe operating space for humanity

Specific soil processes are central to Earth-system processes that provide the safe operating space for humanity – the concept of 'planetary boundaries' that cannot be exceeded without causing potentially disastrous consequences for humanity (Röckstrom *et al*. 2009; Steffen *et al*., 2015). Currently stresses in the nitrogen cycle, climate change, and biodiversity loss are suggested to be beyond safe operating boundaries. Human impact on the natural reservoir of soil biodiversity and on the rate of N and C cycling in soils is a significant aspect of this stress. Whereas GLASOD had highlighted nutrient depletion through crop production without the application of sufficient manure and fertilizer to replenish nutrient loss, the concept of planetary boundaries also focuses our attention on over-application of nutrients in some regions and its consequences for atmospheric and hydrological systems. Addressing the nutrient deficit in regions such as Sub-Saharan Africa while remaining within the safe operating space for humanity requires a significant reduction of nutrient additions in area of excess inputs (Steffen *et al*., 2015).

Biodiversity

Biodiversity cuts across most of the concepts presented above, and loss of biodiversity is identified by Röckstrom *et al.* (2009) as one of three components currently operating beyond safe planetary boundaries. Biodiversity is more than simply an ecosystem service, even though specific benefits can be identified from the biodiversity pool. This cross-cutting importance of biodiversity was formalized in the Convention on Biological Diversity signed in 1992 at the United Nations Conference on Environment and Development in Brazil. Soils are widely recognized as a major reservoir of global biodiversity, and preservation of this (largely unknown) pool of biodiversity is essential.

Table 1.2 | Ecosystem services provided by the soil and the soil functions that support these services.

Ecosystem service	Soil functions
Supporting services: Services that are necessary for the production of all other ecosystem services; their impacts on people are often indirect or occur over a very long time	
Soil formation	∑ Weathering of primary minerals and release of nutrients ∑ Transformation and accumulation of organic matter ∑ Creation of structures (aggregates, horizons) for gas and water flow and root growth ∑ Creation of charged surfaces for ion retention and exchange
Primary production	∑ Medium for seed germination and root growth ∑ Supply of nutrients and water for plants
Nutrient cycling	∑ Transformation of organic materials by soil organisms ∑ Retention and release of nutrients on charged surfaces
Regulating services: benefits obtained from the regulation of ecosystem processes	
Water quality regulation	∑ Filtering and buffering of substances in soil water ∑ Transformation of contaminants
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

Climate regulation	∑ Regulation of CO_2, N_2O, and CH_4 emissions
Erosion regulation	∑ Retention of soil on the land surface
Provisioning Services: products ('goods') obtained from ecosystems of direct benefit to people	
Food supply	∑ Providing water, nutrients, and physical support for growth of plants for human and animal consumption
Water supply	∑ Retention and purification of water
Fibre and fuel supply	∑ Providing water, nutrients, and physical support for growth of plant growth for bioenergy and fibre
Raw earth material supply	∑ Provision of topsoil, aggregates, peat etc.
Surface stability	∑ Supporting human habitations and related infrastructure
Refugia	∑ Providing habitat for soil animals, birds etc.
Genetic resources	∑ Source of unique biological materials
Cultural services: nonmaterial benefits which people obtain from ecosystems through spiritual enrichment, aesthetic experiences, heritage preservation and recreation	
Aesthetic and spiritual	∑ Preservation of natural and cultural landscape diversity ∑ Source of pigments and dyes
Heritage	∑ Preservation of archaeological records

The contribution of many of the concepts outlined above is apparent throughout the World Soil Charter. This synthesis of concepts is perhaps most evident in the definition of sustainable soil management used in the World Soil Charter:

> 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 concepts of soil functions, the threats to functions, and the ecosystem services provided by soils are central both to the structure of this book and to the content of each chapter.

References

Bockheim, J.G., Gennadiyev, A.N., Hammer, R.D. & Tandarich, J.P. 2005. Historical development of key concepts in pedology. *Geoderma*, 124: 23-36.

Bouma, J. & McBratney, A.B. 2013. Framing soils as an actor when dealing with wicked environmental problems. *Geoderma*, 200: 130-139.

Commission of the European Communities (CEC). 2006. Communication from the Commission to the Council, the European Parliament, the European Economic and Social Committee and the Committee of the Regions. Thematic Strategy for Soil Protection. COM 231 Final, Brussels.

Costanza R., d'Arge R., de Groot R., Farber S., Grasso M., Hannon B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P. & van den Belt, M. 1997. The value of the world's ecosystem services and natural capital. *Nature*, 387: 253-260.

Daily, G.C., P.A. Matson & Vitousek P.M. 1997. Ecosystem services supplied by soil. *In* G. Daily, ed. pp. 113-132. *Nature's Services: Societal Dependence on Natural Ecosystems*. Washington, DC, Island Press. 412 pp.

FAO. 2015. World Soil Charter (also available at http://www.fao.org/3/a-mn442e.pdf)

Hole, F.D. & Campbell, J.B. 1985. Soil Landscape Analysis. London, Routledge & Kegan Paul. 196 pp.

McBratney, A. B., Field, D. J., & Koch, A. 2014. The dimensions of soil security. *Geoderma*, 213: 203-213.

Millennium Ecosystem Assessment. 2005. Ecosystems and Human Well-Being: Synthesis. Washington, DC, Island Press. 800 pp.

NRC 1998. *The Canadian System of Soil Classification*. *3rd ed.* Canada, Ottawa. 187 pp.

Palm, C., Sanchez, P., Ahamed, S. & Awiti, A. 2007. Soils: a contemporary perspective. *Annu. Rev. Environ. Resour.*, 32: 99-129.

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin III, F.S., Lambin, E.F., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., de Wit, C.A., Hughes, T., van der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P.K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R.W., Fabry, V.J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P. & Foley, J.A. 2009. A safe operating space for humanity. *Nature*, 461: 472-475.

Steffen, W., Richardson, K., Rockström, J., Cornell, S.E., Fetzer, I., Bennett, E.M., Biggs, R. , Carpenter, S.R., de Vries, W., de Wit, C.A., Folke, C., Gerten, D., Heinke, J., Mace, G.M., Persson, L. M., Ramanathan, V., Reyers, B., & Sörlin, S. 2015. Planetary boundaries: Guiding human development on a changing planet. *Science*, 347 (6223): 1259855. 10 pp.

World Commission on Environment and Development. 1987. *Report of the World Commission on Environment and Development: Our Common Future.* UK, Oxford, Oxford University Press. 383 pp.

2 | The role of soils in ecosystem processes

Soils play a critical role in delivering ecosystem services. Management to change an ecosystem process in support of one regulating ecosystem service can either provide co-benefits to other services or require trade-offs (Robinson *et al*., 2013; Dominati, Patterson, and Mackay, 2010). Recent reviews have provided examples of some of these synergies and trade-offs (Smith *et al*., 2013) and illustrated the role of soils in supporting ecosystem services and underpinning natural capital (Robinson, Lebron and Vereecken, 2009, Robinson *et al*., 2014, Dominati, Patterson and Mackay, 2010). In this chapter, we present current knowledge – and knowledge gaps – on the role of soils in the carbon, nitrogen and water cycles, and on their role as a habitat for organisms and as a genetic pool.

2.1 | Soils and the carbon cycle

Carbon (C) storage is an important ecosystem function of soils that has gained increasing attention in recent years due to its interactions with the earth's climate system. Soil is a major C reservoir that holds more carbon than is contained in the atmosphere and terrestrial vegetation combined. All three of these reservoirs are in constant exchange. In many soils, soil organic matter (SOM), which contains roughly 55–60 percent C by mass, comprises most or all of the C stock – referred to as soil organic carbon (SOC). In arid and semi-arid soils, significant inorganic C (IC) can be present as pedogenic carbonate minerals or 'caliche' (typically Ca/ $MgCO_3$), formed from the reaction of biocarbonate (derived from CO_2 in the soil) with free base cations, which can then be precipitated in subsoil layers (Nordt, Wilding and Drees, 2000). Also soils derived from carbonate-containing parent material (e.g. limestone) can have significant amounts of inorganic carbon. However, in most cases changes in inorganic C stocks are slow and not amenable to traditional soil management practices. Hence inorganic carbon does not play a significant role in terms of management of ecosystem services. For this reason, the further discussion of soil C in this chapter will focus on soil organic carbon.

A general overview of the ecosystem C cycle as it interacts with soils is given in Figure 2.1. The major input of organic C to soils is provided by the uptake and fixation of CO_2 by plants (the net result of photosynthesis and above- and below-ground plant respiration), and by the subsequent incorporation of plant residue C (both above- and below-ground) into soil. Some of the fixed plant C may be removed by harvest before entering the soil. Conversely, C additions from offsite sources (e.g. compost, manure) may occur. Organic matter on and in the soil is subject to comminution and mixing by soil fauna and to enzymatic breakdown and metabolism by microorganisms, resulting in release of CO_2 via microbial respiration (also referred to as organic carbon mineralization). Microbial transformations as well as interactions of organic matter with soil minerals greatly influence the stabilization of organic C and its rate of mineralization. In flooded soils, emissions of methane (CH_4) from microbial metabolism can represent a significant gaseous C efflux. Erosion can also directly affect the soil C balance through the removal and/or deposition of the C contained in the transported soil. Leaching of dissolved organic (DOC) and dissolved inorganic carbon (DIC) through the soil profile and out into groundwater and surface water represents an additional loss pathway that can be significant in some soils.

Maintaining and increasing SOC stocks through improved land use and management practices can help to counteract increasing atmospheric CO_2 concentrations (Paustian *et al.*, 1998, Smith *et al.*, 2007; Whitmore, Kirk and Rawlins, 2014). Increasing soil C content also improves other chemical and physical soil properties, such as nutrient storage, water holding capacity, aggregation and sorption of organic and/or inorganic pollutants (Kibblewhite, Ritz and Swift, 2008). Carbon sequestration in soils may therefore be a cost-effective and environmentally friendly way to store C. It can also enhance other ecosystem services derived from soil, such as agricultural production, clean water supply, and biodiversity by increasing SOM content and thereby improving soil quality (Lal, 2004).

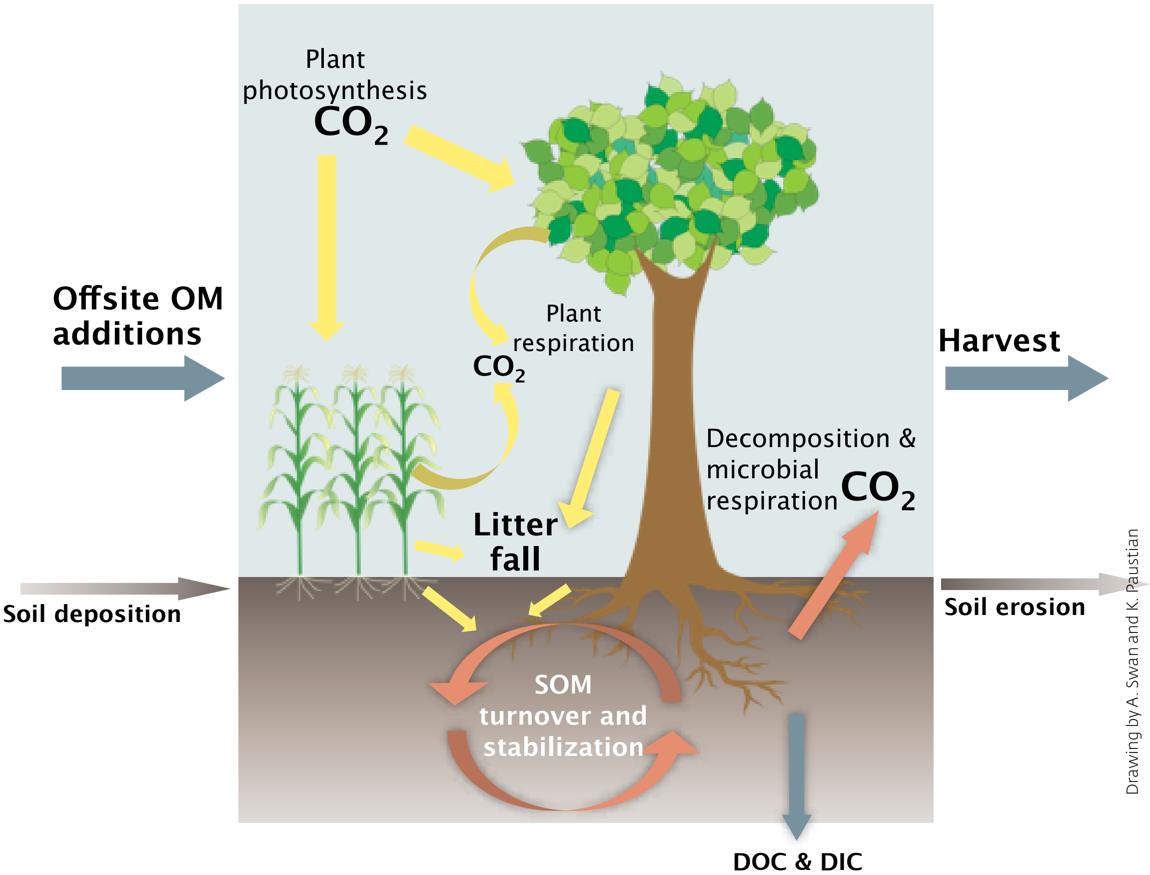


Figure 2.1 | Overview of ecosystem processes involved in determining the soil C balance.

2.1.1 | Quantitative amounts of organic C stored in soil

Organic C stocks in the world's soils have been estimated to comprise 1 500 Pg of C down to 1 m depth and 2 500 Pg down to 2 m (Batjes, 1996). Recent studies, based on newer estimates for the C stored in boreal soils under permafrost conditions, suggest that soil C storage may be even greater, accounting for as much as 2000 Pg to 1 m depth (Tarnocai *et al.*, 2009). Although the highest C concentrations are found in the top 30 cm of soil, the major proportion of total C stock in many soils is present below 30 cm depth (Batjes, 1996). In the northern circumpolar permafrost region, at least 61 percent of the total soil C is stored below 30 cm depth (Tarnocai *et al.*, 2009).

2.1.2 | Nature and formation of soil organic C

Soil organic matter (SOM) is composed of plant litter compounds as well as of microbial decomposition products. SOM is thus a complex biogeochemical mixture derived from organic material in all stages of decomposition (von Lützow *et al*., 2006; Paul, 2014). Due to microbial degradation and mineralization to CO_2 (and CH_4 in anaerobic environments), the majority of plant litter compounds added to soil remain for a relatively short time (from a few days to a few years). This is particularly the case if the organic compounds are added on the soil surface. However, some organic matter compounds may persist in the soil for decades or centuries or even for millennia (Paul *et al*., 1997; von Lützow *et al*., 2006). It is increasingly accepted that, despite their recalcitrant nature, plant litter compounds (e.g. lignin) themselves do not substantially contribute to SOM persistence in soil (Thévenot, Dignac and Rumpel, 2010). Longer term stabilization is generally conferred through interactions with soil minerals (e.g. through surface binding or occlusion within microaggregates), which reduce SOM exposure to enzymatic degradation (Sollins, Homann and Caldwell, 1996; Six, Elliott and Paustian, 2000; Schmidt *et al*., 2011). Thus, the location of SOM within the soil matrix has a much stronger influence on its turnover than its chemical composition (Chabbi, Kögel-Knabner and Rumpel, 2009; Dungait *et al*., 2012)

One consequence of the role of reactive mineral surfaces in SOC stabilization is that the surface area of the soil mineral fraction, which is finite and a function of soil texture (e.g. clay, silt or sand content) and of mineralogy, may set an upper limit for the amount of SOM that a particular soil can hold (Six, Elliott and Paustian, 2002). A recent conceptual model (Figure 2.2) by Cotrufo *et al*. (2013), based on studies showing that microbially-derived decomposition products make up most of the mineral-stabilized organic matter, postulates that relatively labile litter compounds with higher microbial growth yield efficiency contribute proportionally more to the stable mineral-associated SOM pool than do more recalcitrant plant compounds with low microbial growth yield efficiency. This concept is in agreement with the current understanding that microbial material is building up much of the stabilised SOM pool (Miltner *et al*., 2012).

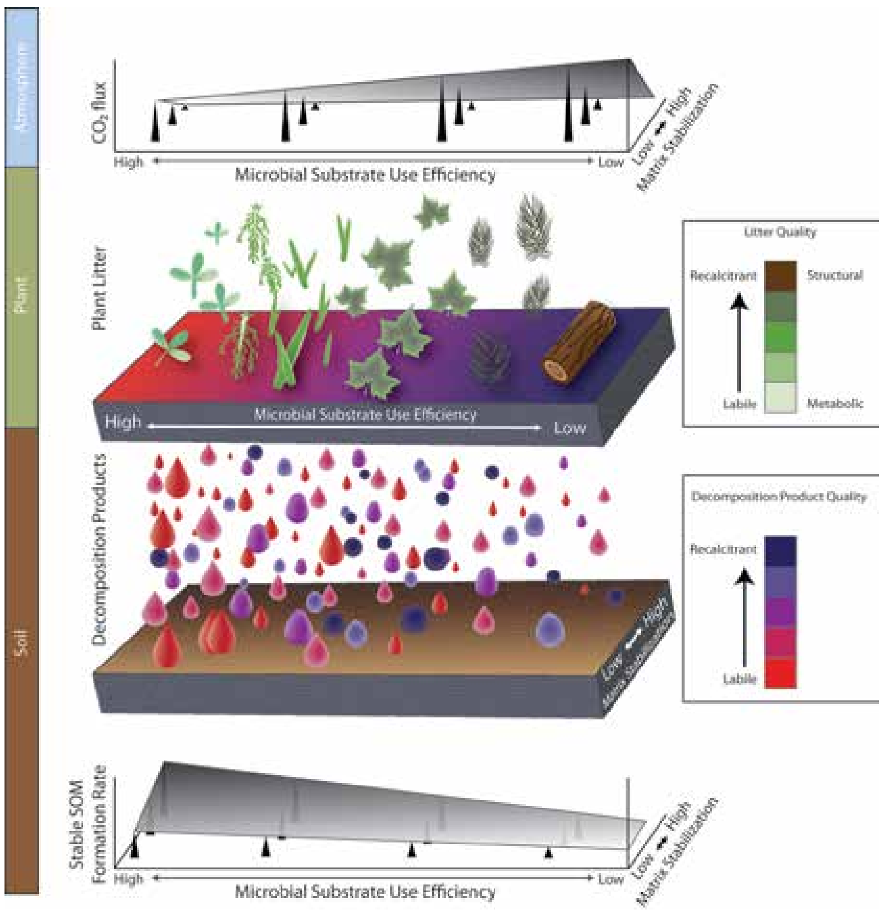


Figure 2.2 | Conceptual model of interactions between litter quality, microbial products and soil mineral interactions affecting the formation and stabilization of organic matter.
Source: *Cotrufo et al.*, 2013.

With this emphasis on the importance of SOM location within the soil, microbial accessibility of organic material at very small scales has become a focus of research in recent years (Lehmann, Kinyangi and Solomon, 2007). The development of powerful new tools like X-ray spectroscopy and secondary ion mass spectrometry now allows the visualisation of organo-mineral interactions at nanoscale. As a result, the location and distribution of organic matter within the soil mineral matrix may now be assessed in more detail (Lehmann *et al.*, 2008). However, before the results obtained with these tools yield information concerning soil C formation at macroscale, upscaling and integration of spatial heterogeneity is necessary (Mueller *et al.*, 2013).

As well as exhibiting tremendous heterogeneity in terms of its composition, the distribution of SOC within the soil is also very heterogeneous, particularly with respect to depth within the soil profile. Whereas upper soil layers receive greater amounts of aboveground litter ('shoot C' from leaves and stems), subsoil C originates primarily from root-derived C as well as from plant- and microbial-derived dissolved organic carbon (DOC) transported down the soil profile. Root C has a greater likelihood of being preserved in soil compared to shoot C (Balesdant and Balabane, 1996) and studies suggest that root C therefore accounts for a larger proportion of SOM (Rasse, Rumpel and Dignac, 2005). In general, C cycling and C formation is most active in topsoil horizons, whereas stabilised C with longer turnover times makes up a greater proportion of the total SOC found in deep soil horizons (Scharpenseel and Becker-Heidmann, 1989; Trumbore, 2009). The accumulation of stabilised C with long residence times in deep soil horizons may be due to continuous transport, temporary immobilisation and microbial processing of DOC within the soil profile (Kalbitz and Kaiser, 2012) and/or efficient stabilisation of root-derived organic matter within the soil matrix (Rasse, Rumpel and Dignac, 2005).

An additional long-term C pool in many soils is pyrogenic carbonaceous matter, formed from partially carbonised (e.g., pyrolysed) biomass during wildfires (Schmidt and Noack, 2000). A portion of this material has a highly condensed aromatic chemical structure (often referred to as pyrogenic carbon or black carbon) that resists microbial degradation and can persist in soils for long periods (Lehmann *et al.*, 2015).

2.1.3 | Soil C pools

For modelling purposes, soil C is usually divided into a number of pools (typically from two to five) in order to represent the heterogeneity in residence time of the vast mixture of different organic compounds in soil (Smith *et al.*, 1997). A useful three pool split of soil C (excluding litter) – into a labile pool, an intermediate pool, and a refractory (stable) pool – is employed in several soil C models, including the Century model (Parton *et al.*, 1987). The *labile pool* represents easily degradable plant material, microbial biomass and labile metabolites, and may turn over within a few months or years. Conceptually, the *intermediate* pool comprises microbially-processed organic matter that is partially stabilized on mineral surfaces and/or protected within aggregates, with turnover times in the range of decades. The *refractory pool*, including highly stabilized organic matter-mineral complexes and pyrogenic C, may remain in soils for centuries or millennia.

Individual model pools (as opposed to the total C stock) are typically not defined as measureable pools per se. The kinetics of the model conceptual pools are instead inferred from C dating and tracer studies, laboratory incubations and total SOC dynamics in long-term field experiments (McGill, 1996; Paustian, 1994). Many carbon cycle, ecosystem and crop growth models successfully employ this type of functional representation of SOM (Krull, Baldock and Skjemstad, 2003; Stockman *et al.*, 2013). Nonetheless, ways to reconcile 'measurable' and 'modelable' pools have been under discussion for a number of years (Elliott, Paustian and Frey, 1996; Smith *et al.*, 2002; Dungait *et al.*, 2012). This reconciliation remains a desirable goal for improving understanding of SOC dynamics (Schmidt *et al.*, 2011).

2.1.4 | Factors influencing soil C storage

Fundamentally, the amount of SOC stored in a given soil is determined by the balance of C entering the soil, mainly via plant residues and exudates, and C leaving the soil through mineralization (as CO_2), driven by microbial processes, and to a lesser extent leaching out of the soil as DOC. Locally C can also be lost or gained through soil erosion or deposition (Figure 2.1), leading to a redistribution of soil C at local, landscape and regional scales.

Consequently, a main control on SOC storage is the amount and type of residues that are produced by plants as the primary producers in the ecosystem. Plant productivity and subsequent senescence and death lead, through plant necromass breakdown, to the input of organic C to the soil system. Thus, broadly speaking for a given pedoclimatic condition, higher levels of plant residue inputs will tend to support higher SOC stocks, and vice versa. C levels of many soils are also influenced by fertiliser additions, which are indispensable for sustaining plant productivity in agricultural systems.

In addition to productivity and plant C inputs, climatic factors, such as soil temperature and water content greatly influence soil C storage through their effect on microbial activity. In general, higher soil temperatures increase microbial decomposition of organic matter. Temperature is, therefore, taken as major control of SOM storage in soil C cycle models, although the temperature sensitivity of decomposition for different SOM fractions remains an area of uncertainty (Conant *et al.*, 2011).

Water also influences soil C storage through several processes. Moist but well-aerated soils are optimal for microbial activity. Decomposition rates consequently decrease as soils become drier. However, flooded soils have lower rates of organic matter decay due to restricted aeration (e.g. O_2 depletion due to limited O_2 diffusion in water) and thus may often yield soils with very high amounts of soil C (e.g. peat and muck soils). High precipitation may also lead to C transport down the soil profile as dissolved and/or particulate organic matter. During extreme events, such as drought, SOM decomposition may initially decrease but may subsequently increase after rewetting (Borken and Matzner, 2008). Fire may decrease soil C storage at first, but over the longer term may increase C storage through positive effects on plant growth and through input of very stable pyrogenic C (Knicker, 2007).

The quantity and composition of SOC in mineral soils is also strongly dependent on soil type, with clay content influencing not only the amount but also the composition of soil C. In clay rich soils, higher organic matter content and a higher concentration of O-alkyl C derived from polysaccharides may be expected, compared to sandy soils which are characterised by lower C contents and high concentrations of alkyl C (Rumpel and Kögel-Knabner, 2011). Aliphatic material may contribute to the hydrophobicity of soils, which could lead to reduced microbial accessibility and therefore increased C storage.

Bioturbation (the reworking of soils by animals or plants) may further influence the amount as well as the chemical nature of soil C. It may greatly influence the heterogeneity of soils by creating hotspots. On biologically active sites, incorporation and transformation of organic compounds into soil is usually enhanced by bioturbation, leading to organo-mineral interactions and increase of C storage (Wilkinson, Richards and Humphreys, 2009).

Microbial decomposition of SOM may be stimulated (or reduced) by labile organic matter input through the 'priming effect' (Jenkinson, 1971; Kuzyakov, 2002). Positive priming refers to mineralisation of otherwise stable C through shifts in microbial community composition (Fontaine, Mariotti and Abbadie *et al.*, 2003). However, in some cases, the addition of organic matter to soil may also cause changes in the soil microbial communities with regard to the preferentially degraded substrate and therefore impede mineralisation of native SOM (Sparling, Cheschire and Mundie, 1982; Kuzyakov, 2002).

Plant communities are main controlling factors of these processes because they influence organic matter input and microbial activity by their effects on soil water, labile C input, pH and nutrient cycling.

2.1.5 | Carbon cycle: knowledge gaps and research needs

Substantial progress has been made in recent years towards a deeper understanding of the processes controlling soil C storage. There has been progress also in improving and deploying predictive models of soil C dynamics that can guide decision makers and inform policy. However, it is equally true that many new (and some old) gaps in our knowledge have been identified and the need for further research has been assessed. Recent research on soil C dynamics has been driven in part by increasing awareness of: (1) the importance of small scale variability for microbial C turnover (Vogel *et al.*, 2014); (2) interactions between the C cycle and other biogeochemical cycles (Gärdenäs *et al.*, 2011); and (3) the importance of soil C not only at the field scale but at regional to global scales (Todd-Brown *et al.*, 2013).

The most cited knowledge gaps and research needs include:

Basic understanding

- Controls on microbial efficiency of organic matter processing, including biodiversity
- The degree of association or separation of organic matter and microbial decomposer communities in the mineral soil matrix
- Role of soil fauna in controlling carbon storage and cycling
- Dynamics of dissolved organic carbon and its role in determining C storage and decomposition
- Pyrogenic C stabilization and interactions of pyrogenic C with native soil C and mineral nutrients
- Role of soil erosion in the global C cycle

Predictive modelling and assessment

- Reconciliation of measured and modelled SOM fractions
- More explicit representation of microbial controls
- Improved modelling of C in subsurface soil layers
- Distributed soil C observational and monitoring networks for model validation
- More realistic and spatially-resolved representation of soil C in global-scale models

2.1.6 | Concluding remarks

Both biotic and abiotic factors control soil C content and dynamics through their effect on plant litter inputs and microbial decomposer communities. The understanding of the C cycle and the role of soils as a sink or source of CO_2 depends on our ability to integrate knowledge of physical, chemical and biological processes operating at small scales (nm, µm, soil profile) and of the spatial heterogeneity of SOM distribution and decomposition processes at increasing scales (field, region, globe). At the global scale, soils are a major component of the planet's C cycle and can have a strong influence on the concentration of CO_2 in the atmosphere. Thus, land management needs to be based on an understanding of the controls on SOM distribution, stabilisation and turnover in order to safeguard and increase the organic matter content of our soils. This will be an important contribution to both food security and the mitigation of greenhouse gases.

2.2 | Soils and the nutrient cycle

Soils support plant growth and so are vital to humanity. They provide nutrients such as nitrogen (N), phosphorous (P), potassium (K), Calcium (Ca), Magnesium (Mg), Sulphur (S) and many trace elements that support biomass production. Biomass is important for food supply, for energy and fibre production and as a (future) source for the chemical industry. Since the 1950s, higher biomass production and yield increases have been supported through mineral/synthetic fertilization (Figure 2.3). However, intensification of agricultural practices and of land use has in many regions resulted in a decline in the content of organic matter content in agricultural soils. In some areas, extensive use of mineral fertilizers has resulted in atmospheric pollution, greenhouse gas emissions (e.g. CO_2 and N_2O), water eutrophication and human health risks (Galloway *et al.*, 2008).

In coming years, human population and demand for food, feed and energy will continue to rise. In order to sustain biomass production in the future and to mitigate negative environmental impacts, fertile soils need to be preserved. Where soil fertility has declined, it needs to be restored by maintaining sufficient amounts of organic matter in soils (Janzen, 2006). This can be achieved by measures of sustainable management (see Chapter 8 of this volume), including by targeted additions of mineral and organic amendments to soils.

The soil function 'fertility' refers to the ability of soil to support and sustain plant growth, including through making N, P and other nutrients available for plant uptake. This process is facilitated by: (i) nutrient storage in soil organic matter; (ii) nutrient recycling from organic to plant-available mineral forms; and (iii) physical and chemical processes that control nutrient sorption, availability, displacement and eventual losses to the atmosphere and water.

Managed soils represent a highly dynamic system and it is this very dynamism that makes soils function and supply ecosystem services. Overall, the fertility and functioning of soils depend on interactions between the soil mineral matrix, plants and microbes. These are responsible for both building and decomposing SOM and therefore for the preservation and availability of nutrients in soils. To sustain soil functions, the balanced cycling of nutrients in soils must be maintained.

After carbon (Section 2.1), N is the most abundant nutrient in all forms of life, since it is contained in proteins, nucleic acids and other compounds. Humans and animals ultimately acquire their N from plants, which in terrestrial ecosystems occurs mostly in mineral form (e.g. NH_4^+ and NO_3^-) in soils. The parent material of soils does not contain significant amounts of N (as opposed to P and other nutrients). New N enters the soil through the fixation of atmospheric N_2 by a specialized group of soil biota. However, the largest flux of N

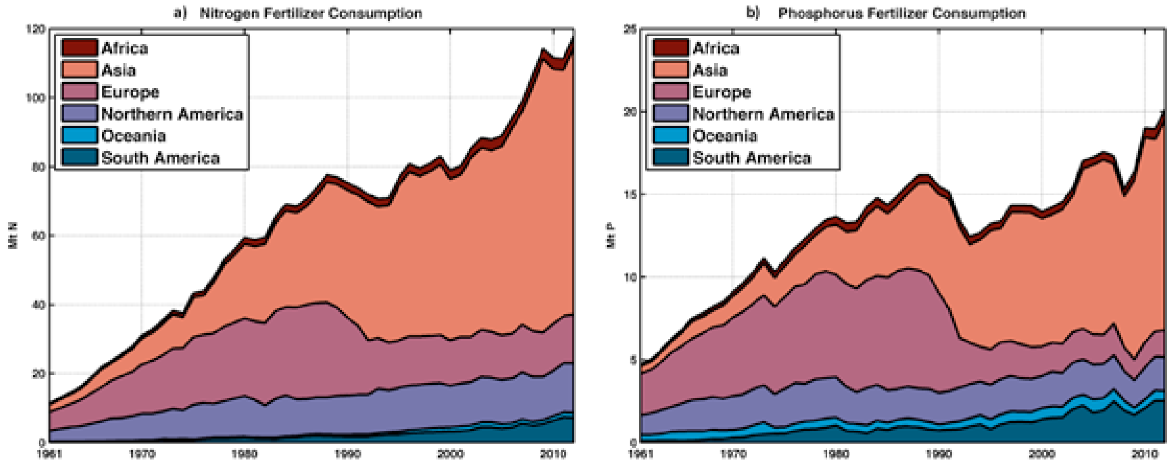


Figure 2.3 | Global (a) nitrogen (N) and (b) phosphorus (P) fertilizer use between 1961 and 2012 split for the different continents in Mt P per year. Source: FAO, 2015.

in soils is generated through the continuous recycling of N internal to the plant-soil system: soil mineral N is taken up by the plant, it is fixed into biomass, and eventually N returns in the form of plant debris to the soil. Here soil biota decompose it, mineralizing part of the N and making it newly available for plant growth, while transforming the other part into SOM, which ultimately is the largest stock of stable N in soil. Nitrogen is lost from the soil to the water system by leaching and to the atmosphere by gas efflux (NH_3, N_2O and N_2).

In most natural ecosystems, N availability is a limiting factor to productivity and N cycles tightly in the system with minimal losses. Through the cultivation of N_2 fixing crops, the production and application of synthetic N fertilizer, and the deposition of atmospheric N, humans have applied twice as much reactive N to soils as the N introduced by natural processes, thereby significantly increasing biomass production on land (Vitousek and Matson, 1993). However, since mineral fertilizer use efficiency is generally low and far more fertilizer is often used than plants actually need, a high percentage of N fertilizer is lost from the soil. This is generating a myriad of deleterious cascade effects on the environment and on human health (Galloway *et al.*, 2008). This phenomenon is spread over most of the globe. However, in some regions of the world, in particular Sub-Saharan Africa, which are characterized by eroded soils and where economic constraints limit the use of fertilizers, productivity is still strongly constrained by low levels of soil-available N and other nutrients, notably P (Figure 2.2).

Phosphorus is an essential element for all living organisms. It cycles internally in the plant-soil system, moving from the parent material through weathering to biochemical molecules (e.g. nucleic acid, phospholipids) and back to mineral forms after decomposition (e.g. H_3PO_4). In natural soils P is among the most limiting nutrients, since it is present in small amounts and only available in its soluble forms, which promptly react with calcium, iron and aluminum cations to precipitate as highly insoluble compounds. Adsorbed on those compounds, P can be lost from soils, entering the aquatic system through erosion and surface runoff. To correct this lack of available P, 'primary' P is mined and added to soils in the form of mineral fertilizer. This external input has led to positive agronomic P balances (McDonald *et al.*, 2011). There are, however, large variations in the world, with large surpluses in the United States, Europe and Asia, and deficits in Russia, Africa and South-America (Figure 2.4). Additionally, since plant P uptake is a relatively inefficient process with roughly 60 percent of the total P input to soils not taken up, it has been estimated that the amount of P exported from terrestrial to aquatic systems has tripled, with significant impacts on the environment (Bennett, Carpenter and Caraco, 2001).

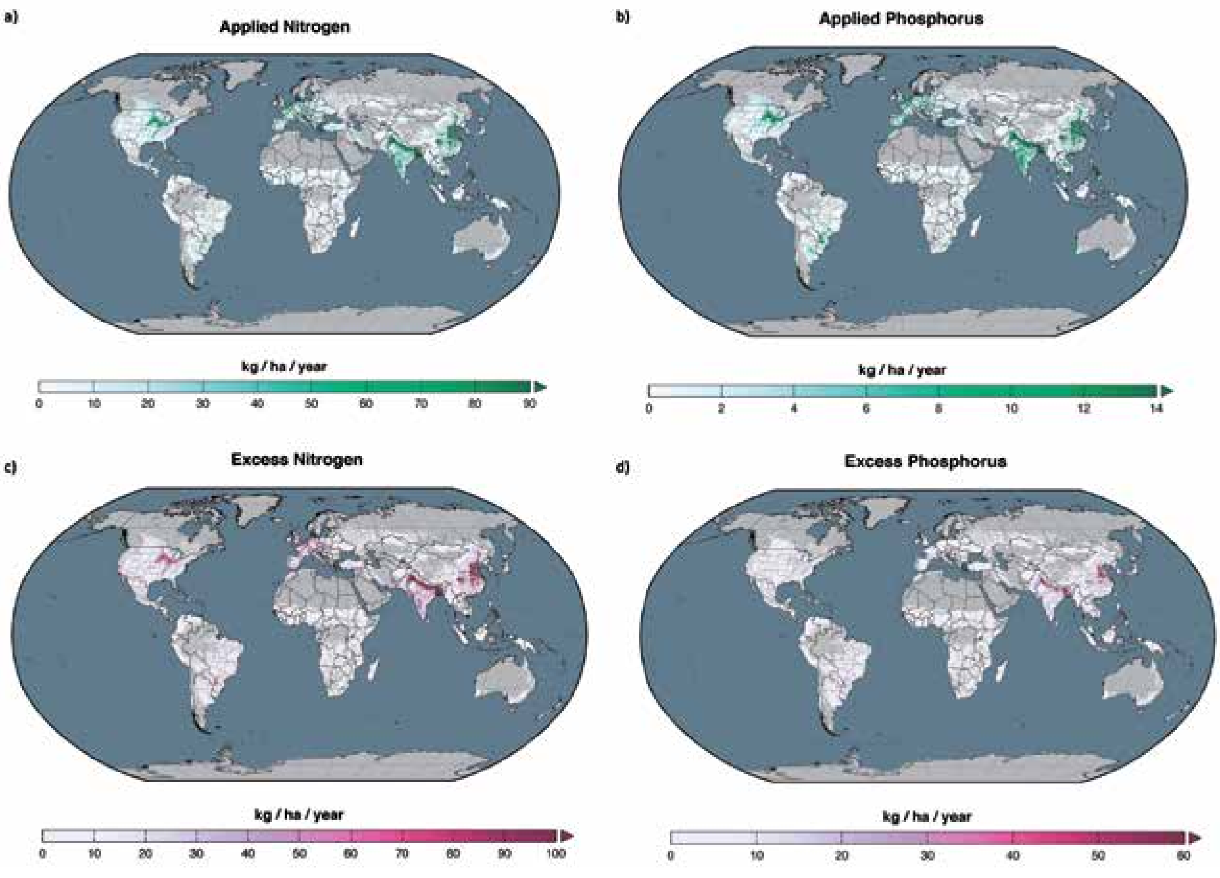


Figure 2.4 | Applied and excess nitrogen and phosphorus in croplands. Nitrogen and phosphorus inputs and excess were calculated using a simple mass balance model, extended to include 175 crops. To account for both the rate and spatial extent of croplands, the data are presented as kg per ha of the landscape: (a) applied nitrogen, including N deposition; (b) applied phosphorus; (c) excess nitrogen; and (d) excess phosphorus.
Source: West *et al.*, 2014.

Management practices need to be implemented that sustain, restore or increase soil fertility and biomass production while limiting associated negative impacts. This can be achieved by promoting the accrual of soil organic matter and nutrient recycling, applying balanced C amendments and fertilization of N, P and other nutrients to meet plant and soil requirements, while limiting overuse of fertilizer. Carbon, N and P cycling in soils is coupled by tight stoichiometric relationships (e.g. relatively fixed C:N:P in plants and microorganisms). This means that an enduring increase or decrease of carbon in soils cannot be achieved without a proportional change in nitrogen and phosphorous (and several other nutrients). This is a fundamental consideration in any programs for carbon sequestration and land restoration because of the significant costs. Therefore, their management needs to be planned in concert.

Nutrient management has been extensively studied, with the aim of identifying and proposing management practices (e.g. precision agriculture) that improve nutrient use efficiency and productivity while reducing potentially harmful losses to the environment (van Groenigen *et al.*, 2010; Venterea, Maharjan and Dolan, 2011). However, our ability to predict the ecosystem response to balanced fertilization is still limited and the relationship requires continued monitoring. Further benefits are anticipated from improved plant varieties with root morphologies that have better capacity to extract P from soils or use it more efficiently.

More generally, further research is needed into organic matter responses to agricultural C inputs and into the potential for restoring and increasing soil organic matter to promote long term soil fertility (e.g. Lugato, Berti and Giardini, 2006). Hence, we stress the importance of an integrated approach to nutrient management which supports plant productivity while preserving or enhancing soil organic matter stocks and reducing nutrient losses to the atmosphere or aquatic systems. Prediction and optimization of performance would benefit from continued data acquisition across the whole range of climate and environmental and agro-ecological conditions.

2.2.1 | The nutrient cycle: knowledge gaps and research needs

In the second half of the 20th century, higher biomass yields were supported by higher use of fertilizer (N, P) inputs. This is now considered unsustainable in many situations. Alternatives are required that make better use of inherent soil fertility, improve resource use efficiency, and prevent losses of N and P. Examples in agriculture include sustainable intensification and new crop varieties that have root systems with improved extraction capability or which have higher internal P use efficiency. At the food system level, more effective nutrient management would benefit from a focus on a '5R strategy': (1) realign P and N inputs; (2) reduce P and N losses to water, thereby minimizing eutrophication impacts; (3) recycle the P and N in bio-resources; (4) recover P and N from wastes to use as fertilizer; and (5) redefine use and use-efficiency of P in the food chain (Withers *et al.*, 2015).

In addition, a better understanding of biogeochemical processes at the molecular level is needed. This should include: (i) research into the role of plant symbionts on the weathering of minerals and support of nutrient uptake, and (ii) development of target-specific 'smart' agrochemical agents that enhance nutrient uptake.

2.3 | Soils and the water cycle

Soils provide important ecosystem services through their function within the water cycle. These services include provisioning services of food and water security, regulating services associated with moderation and purification of water flows, and cultural services such as landscapes and water bodies that meet recreation and aesthetic values (Dymond, 2014). Water stored in soil is used for the evapotranspiration and plant growth that supply food and fibre. Soil water also stabilizes the land surface to prevent erosion and regulates nutrient and contaminant flow. At a catchment and basin scale, the capacity of the soil to infiltrate water attenuates stream and river flows and can prevent flooding, while water that percolates through soil can replenish groundwater and related streamflow and surface water ecosystems.

The soil functions of accepting, storing, transmitting and cleaning of water shown in Table 2.1 are inter-related. Soil water storage depends on the rate of infiltration into the soil and on soil hydraulic conductivity that redistributes water within and through the soil profile. Similarly, infiltration and hydraulic conductivity are dependent on the water stored in the soil. The initially high rate of infiltration into dry soil declines as the soil water content increases and water replaces air in the pore space. Conversely, hydraulic conductivity increases with soil moisture content as a greater proportion of the pores are transmitting water. Water content and transmission times are also important to the filtering function of soil because contact with soil surfaces and residence time in soil are controls on contaminant supply and removal.

Optimum growth of most plants occurs when roots can access both oxygen and water in the soil. The soil must therefore infiltrate water, drain quickly when saturated to allow air to reach plant roots, and retain and redistribute water for plant use. The ideal soil for plant production depends on climatic conditions and on the soil requirements of the crop. For instance, in dry regions it can be an advantage to have soils with a high clay content to retain water, while sandier soils that drain quickly are better suited to wetter regions.

Soil structural stability and porosity are also important for the infiltration of water into soil. Organic matter improves soil aggregate stability. While plant growth and surface mulches can help protect the soil surface, a stable, well-aggregated soil structure that resists surface sealing and continues to infiltrate water during intense rainfall events will decrease the potential for downstream flooding. Porosity determines the capacity of the soil to retain water and controls transmission of water through the soil. In addition to total porosity, the continuity and structure of the pore network are important to these functions and also to the further function of filtering out contaminants in flow.

Table 2 | Soil functions related to the water cycle and ecosystem services

Soil Function	Mechanism	Consequence	Ecosystem service
Stores (Storage)	Water held in soil pores supports plant and microbial communities	Biomass production Surface protection	Food Aesthetics Erosion control
Accepts (Sorptivity)	Incident water infiltrates into soil with excess lost as runoff	Storm runoff reduction	Erosion control Flood protection
Transmits (Hydraulic conductivity)	Water entering the soil is redistributed and excess is transmitted as deep percolation	Percolation to groundwater	Groundwater recharge Stream flow maintenance
Cleans (Filtering)	Water passing through the soil matrix interacts with soil particles and biota	Contaminants removed by biological degradation/retention on sorption sites	Water quality

Another important role of soil water is its support of biota that can degrade compounds into beneficial forms that may also retain nutrients. Soil organic matter is important to this role - together with mineral soil (especially the clay fraction), SOM provides sorption sites, but sorption capacity is finite. Flow through macropores that bypass the soil matrix where biota and sorption sites are generally located can quickly transmit water and contaminants through the soil to groundwater or artificial drains. However, for filtering purposes a longer, slower route through the soil matrix is more effective.

Soil management alters the ecosystem services provided by water (Table 2.2). Soil conservation practices and sustainable management help to retain regulating ecosystem services such as soil organic matter and structural stability. Similarly, the promotion of soil as a C-sink to offset greenhouse gas emissions helps to maintain or improve soil functions. On the other hand, deforestation, overgrazing and excessive tillage of fragile lands lead to deterioration of the soil structure and to loss of soil function and surface water quality (Steinfield *et al.*, 2006).

Anthropogenic modifications to the water cycle can aid soil function. In dry regimes, inadequate soil moisture can be mitigated through supplementary irrigation, and where excessive precipitation causes problems, waterlogging can be relieved by land drainage. However, irrigation and drainage can have consequences for water regulation services. Irrigation that enables a shift to intensive land use can increase the contaminant load of runoff and drainage water (McDowell *et al.*, 2014). Furthermore, drainage of wetland soils has been shown to reduce water and contaminant storage capacity in the landscape and can increase the potential for downstream flooding. The abstraction of surface or groundwater for irrigation disrupts the natural water cycle and may stress downstream ecosystems and communities. Irrigation of agricultural lands accounts for about 70 percent of ground and surface water withdrawals, and in some regions competition for water resources is forcing irrigators to tap unsustainable sources. Irrigation with wastewater may conserve fresh water resources but brings the risk of water-borne contaminants in soil and crops (Sato *et al.*, 2013) and the accumulation of salts in some environments.

Table 2.2 | Examples of global trends in soil management (Steinfield et al., 2006; Setälä et al., 2014) and their effects on the ecosystem services mediated by water.

Management (global trend)	Provisioning	Regulating	Cultural
Land use change (agricultural to urban)	Decreased biomass, decreased availability of water for agricultural use	Increased impervious surface, decreased infiltration, storage, soil-mediated water regulation	Decreased natural environment
Land use change (increase in change of arable to intensive grassland)	Land use change (increase in change of arable to intensive grassland)	Increased C sequestration, greater requirement for water, stress on ecosystem health of downstream waterways	
Irrigation (increase)	Increased biomass over dryland agriculture, decreased availability of water for urban use	Increased C sequestration, but decreased filtration potential	Infrastructure alters landscape
Drainage (increase in marginal land)	Decreased soil saturation, increased biomass, reduction in wetlands	Decreased C sequestration, denitrification and flood attenuation	Decreased recreational potential (e.g. ecotourism)

The soil management practices to maintain the ecosystem services of food and water security and flow regulation within the soil and water cycle are reasonably well established. However, their application is not universal and poor management leads to a loss of function. Under climate change scenarios of increased climatic variability with more extremes of precipitation, soil functions will be stressed and better soil management will be required (Walthall *et al.*, 2012).

2.4 | Soil as a habitat for organisms and a genetic pool

Soils represent a physically and chemically complex and heterogeneous habitat supporting a high diversity of microbial and faunal taxa. For example, 10 g of soil contains about 1010 bacterial cells of more than 106 species (Gans *et al.*, 2005), and an estimated 360 000 species of animals are dwellers in soil (Decaëns *et al.*, 2006). These complex communities of organisms play critical roles in sustaining soil and wider ecosystem functioning, thus conferring a multitude of benefits to global cycles and human sustainability. Specifically, soil biodiversity is critical to food and fibre production. It is also an important regulator of other vital soil services including nutrient cycling, moderation of greenhouse gas emissions, and water purification (Wall *et al.*, 2012). It is also recognized that the stocks of soil biodiversity represent an important biological and genetic resource for biotechnological exploitation (Brevik and Sauer, 2015). Previous methodological challenges in characterizing soil biodiversity are now being overcome through the use of molecular technologies. As a result significant progress is currently being made in opening the 'black box' of soil biodiversity (Allison and Martiny, 2008), particularly in assessing the normal operating ranges of soil biodiversity under different soil, climatic and land use scenarios. Addressing these knowledge gaps is of fundamental importance, both as an entry point to understanding wider soil processes and as a way to gauge the likely consequences of land use or climatic change on both biodiversity and soil ecosystem services.

The development of molecular technologies has aided morphological characterisations and allowed quantification of stocks and changes in soil biodiversity. This has led to a surge in studies characterizing soil biodiversity at different scales – from large landscape-scale surveys to locally focused studies. The large-scale surveys yield the broader picture, and conclusions are emerging identifying the importance of soil parameters in shaping the biodiversity of soil communities (Fierer and Jackson, 2006). In essence, the same geological, climatic and biotic parameters that ultimately dictate pedogenesis are also involved in shaping the communities of soil biota and thus in regulating the spatial structure of soil communities observed over large areas (Griffiths *et al.*, 2011). Locally focused experimentation then typically reveals more specific changes in broad taxonomic features with respect, for example, to local changes in land use or climate. Many studies have focused on assessing one component of soil diversity, but even greater advances utilizing next-generation high throughput sequencing now allow the analysis of 'whole soil foodwebs'. This permits a thorough interrogation of trophic and co-occurrence interaction networks. The challenge is to consolidate both approaches at different scales to understand the differing susceptibility of global soil biomes to change.

Alongside these new developments in assessing biodiversity, it is essential to link the biodiversity characteristics measured to specific soil functions. This helps understanding the pivotal roles of soil organisms in mediating soil services. The development of stable isotope tracer methodologies (e.g. Radajewski *et al.*, 2000) to link substrate utilization to the identified active members in situ serves to clarify the physiological activity of these soil organisms. Additionally, improved sequencing techniques are now becoming an increasingly cost-effective for assessing the biodiversity of functional genes in soils for both eukaryotes and prokaryotes (Fierer *et al.*, 2013). This potentially allows a more trait-based approach to understanding soil biodiversity, akin to recent approaches applied to larger and more readily functionally understood organisms above-ground. It is becoming increasingly apparent that often, as is typical in natural ecosystems, functionality and biodiversity co-vary with other environmental parameters. Further manipulative experimentation is required to determine the fundamental roles of soil biodiversity versus other co-varying factors in driving soil functionality.

Clearly, we are learning more and more about how global change affects soil biodiversity and functioning. Global-scale syntheses on soil biodiversity are still lacking, but projects such as the Global Soil Biodiversity Atlas (European Commission, 2015) are combining information from across the globe and making it publicly available. However, much remains to be done. More than 20 years ago, many of these issues were raised (for example, in Furusaka, 1993), and to date many of the factors involved have yet to be unravelled. A key barrier to achieving syntheses is the lack of concerted soil surveys that address multiple functions using standardized methodologies.

New technologies for soil biodiversity assessment generate large sequence datasets that are typically archived in publicly accessible databases. However, morphological datasets remain largely unpublished. The best approach to addressing the gaps would be to adopt agreed standard operating procedures for soil function measurements (e.g. as developed in the recent EU-funded EcoFINDERS project) and to ensure that results are widely accessible.

Ultimately the new methods are revealing the high sensitivity of changes in soil biological and genetic resources to threats such as poor management. We now need to recognize the distinct types of organisms found in different soils globally, and to understand their functional roles in order to predict vulnerability of these resources to future change.

References

Allison, S.D. & Martiny, J.B.H. 2008. Resistance, resilience, and redundancy in microbial communities. *Proc. Natl. Acad. Sci. US*, 105: 11512-11519.

Balesdant, J. & Balabane M. 1996. Major contribution of roots to soil carbon storage inferred from maize cultivated soils. *Soil Biol. Biochem*., 28:1261-1263.

Batjes, N.H. 1996. Total carbon and nitrogen in the soils of the world. *European Journalof Soil Science*, 47: 151-163.

Bennett, E., Carpenter, S.R. & Caraco, N.F. 2001. Human Impact on Erodable Phosphorus and Eutrophication: A Global Perspective. *BioScience*, 51: 227-234.

Borken, W. & Matzner, E. 2008. Reappraisal of drying and wetting effects on C and N mineralization and fluxes in soils. *Global Change Biology*, 15: 808-824.

Brevik, E.C. & Sauer, T.J. 2015. The Past, Present, and Future of Soils and Human Health Studies. *Soil*, 1: 35-46.

Chabbi, A., Kögel-Knabner, I. & Rumpel, C. 2009. Stabilised carbon in subsoil horizons is located in spatially distinct parts of the soil profile. *Soil Biology and Biochemistry*, 41: 256-271.

Conant, R.T., Ryan, M.G., Agren, G.I., Birge, J.H.E., Davidson, E.A., Eliasson, P.E., Evans, S.E., Frey, S.D., Giardina, C.P., Hopkins, F.M., Hyvonen, R., Kirschbaum, M., Lavallee, J.M., Leifeld, J., Parton, W.J., Steinweg, J.M., Wallenstein, M.D., Wetterstedt, J.A.M. & Bradford, M.A. 2011. Temperature and soil organic matter decomposition rates – synthesis of current knowledge and a way forward. *Global Change Biology*, 17: 3392-3404.

Cotrufo, M.F., Wallenstein, M.D., Boot, C., Denef, K. & Paul, E. 2013. The microbial efficiency-matrix stabilisation (MEMS) framework integrates plant litter decomposition with soil organic matter stabilization: do labile plant inputs form stable organic matter? *Global Change Biology*, 19: 988-995.

Decaëns, T., Jiménez, J.J., Gioia, C., Measey, G.J. & Lavelle, P. 2006. The values of soil animals for conservation biology. *Eur. J. Soil. Biol*., 60: 807–819.

Dominati, E., Patterson, M. & Mackay, A. 2010. A framework for classifying and quantifying the natural capital and ecosystem services of soils. *Ecological Economics*, 69: 1858-1868.

Dungait, J.A.J., Hopkins, D.W., Gregory, A.S. &Whitmore, A.P. 2012. Soil organic matter turnover is governed by accessibility not recalcitrance. *Global Change Biology*, 18: 1781-1796.

Dymond, J. 2014. Ecosystem services in New Zealand. New Zealand, Lincoln, Manaaki Whenua Press. 540 pp.

Elliott, E.T., Paustian, K. & Frey, S.D. 1996. Modeling the measurable or measuring the modelable: A hierarchical approach to isolating meaningful soil organic matter fractionations. In D.S. Powlson, P. Smith & J.U. Smith, eds. *Evaluation of soil organic matter models using existing, long-term datasets*. pp.161-179. NATO ASI Series, Global Environmental Change. Vol. 38. Berlin, Springer Verlag.

European Commission. 2015. *Global Soil Biodiversity Atlas*. Luxembourg, Publications Office of the European Union. 176 pp.

FAO. 2015. Food and Agriculture Organization of the United Nations - Statistic Division (FAOSTAT) (also available at http://faostat 3.fao.org/home/E)

Fierer, N. & Jackson, R.B. 2006. The diversity and biogeography of soil bacterial communities. *Proc. Natl. Acad. Sci. US*, 103: 626-631.

Fierer, N., Ladau, J., Clemente, J.C., Leff, J., Owens, S.M., Pollard, K.S., Knight, R., Gilbert, J.A. & McCulley, R.L. 2013. Reconstructing the microbial diversity and function of pre-agricultural tallgrass prairie soils in the United States. *Science*, 342: 621-624.

Fontaine, S., Mariotti, A. & Abbadie, L. 2003. The priming effect of organic matter: a question of microbial competition. *Soil Biology & Biochemistry*, 35: 837-843.

Furusaka, C. 1993. Global environment and microorganisms. *Bull. Jpn. Soc. Microb. Ecol.*, 8(2): 127-131.

Galloway, J.N., Townsend, A.R., Erisman, J.W., Bekunda, M., Cai, Z., Freney, J.R., Martinelli, L.A., Seitzinger, S.P. & Sutton, M.A. 2008. Transformation of the nitrogen cycle: recent trends, questions, and potential solutions. *Science*, 320: 889-892.

Gans, J., Wolinsky, M. & Dunbar, J. 2005. Computational improvements reveal great bacterial diversity and high metal toxicity in soil. *Science*, 309: 1387-1390.

Gärdenäs, A.I., Ågren, G.I., Bird, J.A., Clarholm, M., Hallin, S. Ineson, P., Kätterer, T., Knicker, H., Nilsson, S.I., Näsholm, T., Ogle, S., Paustian, K., Persson, T. & Stendahl, J. 2011. Knowledge gaps in soil carbon and nitrogen interactions - From molecular to global scale. *Soil Biology and Biogeochemistry*, 43: 702-717.

Global Change Biology. 2013. The microbial efficiency-matrix stabilisation (MEMS) framework integrates plant litter decomposition with soil organic matter stabilization: do labile plant inputs form stable organic matter? *Volume* 19: 988-995.

Griffiths, R.I., Thomson, B.C., James, P., Bell, T., Bailey, M. & Whiteley, A.S. 2011. The bacterial biogeography of British soils. *Environmental Microbiology*, 13(6): 1642-1654.

Janzen, H.H. 2006. The soil carbon dilemma: Shall we hoard it or use it? *Soil Biology and Biochemistry*, 38(3), 419-424.

Jenkinson, D.S., 1971. Studies on the decomposition of 14C-labelled organic matter in soil. *Soil science*, 111: 64-70.

Kalbitz, K. & Kaiser, K. 2012. Cycling downwards – dissolved organic matter in soils. *Soil Biology and Biochemistry*, 52: 29-32.

Kibblewhite, M.G., Ritz, K. & Swift M.J. 2008. Soil health in agricultural systems. *Phil. Trans. Royal Soc.*, 363: 685-701.

Knicker, H., 2007. How does fire affect the nature and stability of soil organic nitrogen and carbon? A review. *Biogeochemistry*, 85: 91-118.

Krull, E.S., Baldock J.A. & Skjemstad J.O. 2003. Importance of mechanisms and processes of the stabilisation of soil organic matter for modelling carbon turnover. *Funct. Plant Biol.*, 30: 207-222.

Kuzyakov, Y. 2002. Review: factors affecting rhizosphere priming effects. *Journal of Plant Nutrition & Soil Science*, 165: 382-396.

Lal, R. 2004. Soil carbon sequestration impacts on global climate change and food security. *Science*, 304: 1623-1627.

Lehmann, J., Kinyangi, J. & Solomon, D. 2007. Organic matter stabilization in soil microaggregates: implications from spatial heterogeneity of organic carbon contents and carbon forms. *Biogeochemistry*, 85: 45-57.

Lehmann, J., Solomon, D., Kinyangi, J., Dathe, L., Wirick, S. & Jacobsen, C. 2008 Spatial complexity of soil organic matter forms at nanometre scales. *Nature Geoscience*, 1: 238-242.

Lehmann, L., Abiven, S., Kleber, M. Pan, G., Singh, B.P., Sohi, S. & Zimmerman, A. 2015. Persistence of biochar in soil. In J. Lehmann & S. Joseph, eds. Biochar for Environmental Management – *Science, technology and implementation.*, 2nd edition. pp. 235-283. Routledge. 944 pp.

Lugato, E., Berti, A. & Giardini, L. 2006. Soil organic carbon (SOC) dynamics with and without residue incorporation in relation to different nitrogen fertilisation rates. *Geoderma*, 135: 315-321.

MacDonald, G. K., Bennett, E. M., Potter, P. A. & Ramankutty, N. 2011. Agronomic phosphorus imbalances across the world's croplands. *Proceedings of the National Academy of Sciences of the United States of America*, 108(7): 3086-3091.

McDowell, R.W., Cox, N., Daughney, C.J., Wheeler, D. & Moreau, M. 2014. A national assessment of the potential linkage between soil, and surface and groundwater concentrations of phosphorus. *Journal of the American Water Resources Association*. In press.

McGill, W.B. 1996. Review and classification of ten soil organic matter (SOM) models In D.S. Powlson, P. Smith and J.U. Smith, eds. *Evaluation of soil organic matter models using existing, long-term datasets*. pp. 111-132. NATO ASI Series, Global Environmental Change, Vol. 38. Berlin Springer Verlag.

Miltner, A., Bombach, P., Schmidt-Brücken, B. & Kästner, M. 2012. Som genesis: Microbial biomass as a significant source. *Biogeochemistry*, 111: 41-55.

Mueller, C.W., Weber, P.K., Kilburn, M.R., Hoeschen, C., Kleber, M. & Pett-Ridge, J. 2013. Advances in the analysis of biogeochemical interfaces: NanoSIMS to investigate soil microenvironments. *Advances in Agronomy*, 121: 2-39.

Nordt, L.C., Wilding, L.P. & Drees, L.R. 2000. Pedogenic carbonate transformations in leaching soil systems: Implications for the global C cycle. In R. Lal, J.M. Kimble, H. Eswaran, & B.A. Stewart, eds. *Global Climate Change and Pedogenic Carbonates*. pp. 43-64. USA, FL, Boca Raton, Lewis Publishers.

Parton, W.J., Schimel, D.S., Cole, C.V. & Ojima, D.S. 1987. Analyses of factors controlling soil organic matter levels in great plain soils. *Soil Science Society of America Journal*, 51: 1173-1179.

Paul, E.A. 2014. *Soil microbiology, ecology and biochemistry*. Academic press. 598 p.

Paul, E.A., Follett, R.F., Leavitt, S.W., Halvorson, A., Peterson, G.A. & Lyon, D.J. 1997. Radiocarbon dating for determination of soil organic matter pool sizes and fluxes. *Soil* Sci.Soc. *Amer.J*. 61: 1058-1067.

Paustian, K. 1994. Modelling soil biology and biogeochemical processes for sustainable agriculture research. In C. Pankhurst, B.M. Doube, V.V.S.R. Gupta & P.R. Grace, eds. *Management of Soil Biota in Sustainable Farming Systems*. pp. 182-196. Melbourne, CSIRO Publ. 262 pp.

Paustian, K., Cole, C.V., Sauerbeck, D. & Sampson, N. 1998. CO_2 mitigation by agriculture: An overview. *Climatic Change*, 40: 135-162.

Radajewski, S., Ineson, P., Parekh N.R. & Murrell J.C. 2000. Stable-isotope probing as a tool in microbial ecology. *Nature*, 403: 646-649.

Rasse, D.P., Rumpel, C. & Dignac, M.F. 2005 : Is soil carbon mostly root carbon? Mechanisms for a specific stabilisation. *Plant and Soil*, 269: 341-356.

Robinson, D.A., Fraser, I., Dominati, E.J., Davíðsdóttir, B., Jónsson, J.O.G., Jones, L. Jones, S.B., Tuller, M., Lebron, I., Bristow, K.L., Souza, D.M., Banwart, S. & Clothier, B.E. 2014. On the value of soil resources in the context of natural capital and ecosystem service delivery. *Soil Sci. Soc. Am. J.* In press.

Robinson, D.A., Hockley, N., Cooper, D.M., Emmett, B.A., Keith, A.M., Lebron, I., Reynolds, B., Tipping, E., Tye, A.M., Watts, C.W., Whalley, W.R., Black, H.I.J., Warren, G.P. & Robinson, J.S. 2013. Natural capital and ecosystem services, developing an appropriate soils framework as a basis for valuation. *Soil Biology and Biochemistry*, 57: 1023-1033.

Robinson, D.A., Lebron, I. & Vereecken, H. 2009. On the Definition of the Natural Capital of Soils: A Framework for Description, Evaluation, and Monitoring. *Soil Sci. Soc. Am. J.* 73: 1904-1911.

Rumpel, C. & Kögel-Knabner, I. 2011. Deep soil organic matter – a key but poorly understood component of terrestrial C cycle. *Plant and Soil*, 338: 143-158.

Sato, T., Qadir, M., Yamamoto, S., Endo, T. & Zahoor, A. 2013. Global, regional, and country level need for data on wastewater generation, treatment, and use. Agric. *Water Manage*. 130: 1-13.

Scharpenseel, H.W., Becker-Heidmann, P., Neue, H.U. & Tsutsuki, K. 1989. Bomb-carbon, ^{14}C dating and ^{13}C measurements as tracers of organic matter dynamics as well as of morphogenetic and turbation processes. *The Science of the Total Environment*, 81/82: 99-110.

Schmidt, M.W.I. & Noack, A.G. 2000. Black carbon in soils and sediments: Analysis, distribution, implications, and current challenges. *Global Biogeochemical Cycles*, 14: 777-793.

Schmidt, M.W.I., Torn, M.S., Abiven, S., Dittmar, T., Guggenberger, G., Janssens, I.A., Kleber, M., Kogel-Knaber, I., Lehmann, J., Manning, D.A.C., Nannipieri, P., Rasse, D.P., Weiner, S. & Trumbore, S.E. 2011. Persistence of soil organic matter as an ecosystem property. *Nature*, 478: 49-56.

Setälä, H., Bardgett, R. D., Birkhofer, K., Brady, M., Byrne, L., de Ruiter, P. C., de Vries, F. T., Gardi, C., Hedlund, K., Hemerik, L., Hotes, S., Liiri, M., Mortimer, S. R., Pavao-Zuckerman, M., Pouyat, R., Tsiafouli, M. & van der Putten, W. H. 2014. Urban and agricultural soils: conflicts and trade-offs in the optimization of ecosystem services. *Urban Ecosyst*, 17: 239–253.

Six, J., Conant, R.T., Paul, E.A., & Paustian, K. 2002. Stabilization mechanisms of soil organic matter: Implications for C-saturation of soils. *Plant and Soil*, 241: 155-176.

Six, J., Elliott, E.T. & Paustian, K. 2000. Soil microaggregate turnover and microaggregate formation: a mechanism for C sequestration under no-tillage agriculture. *Soil Biology and Biochemistry*, 32 (14): 2099-2103.

Smith, J.U., Smith, P., Monaghan, R. & MacDonald, J. 2002. When is a measured soil organic matter fraction equivalent to a model pool? *European Journal of Soil Science*, 53: 405-416.

Smith, P., Ashmore, M., Black, H., Burgess, P.J., Evans, C., Quine, T., Thomson, A.M., Hicks, K. & Orr, H. 2013. The role of ecosystems and their management in regulating climate, and soil, water and air quality. *Journal of Applied Ecology*, 50: 812-829.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H.H., Kumar, P., McCarl ,B., Ogle, S., O'Mara, F., Rice, C., Scholes, R.J., Sirotenko, O., Howden, M., McAllister, T., Pan, G., Romanenkov, V., Schneider, U., Towprayoon, S., Wattenbach, M. & Smith, J.U. 2007. Greenhouse gas mitigation in agriculture. *Philosophical Transactions of the Royal Society,* B., 363: 789-813.

Smith, P., Smith, J.U., Powlson, D.S., McGill, W.B., Arah, J.R.M., Chertov, O.G., Coleman, K., Franko, U., Frolking, S., Jenkinson, D.S., Jensen, L.S., Kelly, R.H., Klein-Gunnewiek, H., Komarov, A.S., Li, C., Molina, J.A.E., Mueller, T., Parton, W.J., Thornley, J.H.M. & Whitmore, A.P., 1997. A comparison of the performance of nine soil organic matter models using datasets from seven long-term experiments. *Geoderma* 81: 153-225.

Sollins, P., Homann, P. & Caldwell, B.A. 1996. Stabilization and destabilization of soil organic matter: mechanisms and controls. *Geoderma*, 74: 65-105.

Sparling, G.S., Cheschire, M.V. & Mundie, C.M. 1982. Effect of barley plants on the decomposition of 14C-labelled soil organic matter. *Journal of Soil Science*, 33: 89-100.

Tarnocai, C., Canadell, J.G., Schuur, E.A.G., Kuhry, P., Mazhitova, G. & Zimov, S. 2009. Soil organic carbon pools in the northern circumpolar permafrost region. *Global Biogeochemical Cycles*, 23, GB2023.

Thévenot, M., Dignac, M.-F. & Rumpel, C. 2010. Fate of lignins in soils: a review. *Soil Biology and Biochemistry*, 42: 1200-1211.

Todd-Brown, K.E.O, Randerson, J.T., Post, W.M., Hoffman, F.M., Tarnocai, C., Schuur, E.A.G. & Allison, S.D. 2013. Causes of variation in soil carbon simulations from CMIP 5 Earth system models and comparison with observations. *Biogeosciences*, 10: 1717-1736.

Trumbore, S. 2009. Radiocarbon and soil carbon dynamics. *Annual Review of Earth Plantary Sciences*, 37: 47-66.

Van Groenigen, J.W., Velthof, G.L., Oenema, O., Van Groenigen, K.J. & Van Kessel, C. 2010. Towards an agronomic assessment of N2O emissions: a case study for arable crops. *Eur J Soil Sci*, 61: 903-913.

Venterea, R.T., Maharjan, B. & Dolan, M.S. 2011. Fertilizer Source and Tillage Effects on Yield-Scaled Nitrous Oxide Emissions in a Corn Cropping System. *J. Environ. Qual.*, 40: 1521-1531.

Vitousek, P. M. & Matson, P. A. 1993. In R. S. Oremland, ed. *The Biogeochemistry of Global Change: Radiative Trace Gases.* pp. 193-208. New York, Chapman and Hall.

Vogel, C., Mueller, C.W., Hoschen, C., Buegger, F., Heister, K., Schulz, S., Schloter, M. & Kogel-Knabner, I. 2014. Submicron structures provide preferential spots for carbon and nitrogen sequestration in soils. *Nature Communications*, 5: 2947. 7 pp.

von Lützow, M., Kögel-Knabner I., Ekschmitt K., Matzner E., Guggenberger G., Marschner B. & Flessa H. 2006. Stabilization of organic matter in temperate soils: Mechanisms and their relevance under different soil conditions – a review. *European Journal of Soil Science*, 57: 426-445.

Wall, D.H., Bardgett, R.D., Behan-Pelletier, V., Herrick, J.E., Jones, T.H., Ritz, K., Six, J., Strong, D.R. & van der Putten, W.H. (eds.). 2012. *Soil Ecology and Ecosystem services*. Oxford University Press, UK. 424 pp.

Walthall, C.L., Hatfield, J., Backlund, P., Lengnick. L., Marshall, E., Walsh, M., Adkins, S., Aillery, M., Ainsworth, E.A., Ammann, C., Anderson, C.J., Bartomeus, I., Baumgard, L.H., Booker. F., Bradley, B,, Blumenthal, D.M., Bunce, J., Burkey, K., Dabney, S.M., Delgado, J.A., Dukes, J., Funk, A., Garrett, K., Glenn, M., Grantz, D.A., Goodrich, D., Hu, S., Izaurralde, R.C., Jones, R.A.C., Kim, S-H., Leaky, A.D.B., Lewers, K., Mader, T.L., McClung, A., Morgan, J., Muth, D.J., Nearing, M., Oosterhuis, D.M., Ort, D., Parmesan, C., Pettigrew, W.T., Polley, W., Rader, R., Rice, C., Rivington, M., Rosskopf, E., Salas, W.A., Sollenberger, L.E., Srygley, R., Stöckle, C., Takle, E.S., Timlin, D., White, J.W., Winfree, R., Wright-Morton, L. & Ziska, L.H. 2012. *Climate Change and Agriculture in the United States: Effects and Adaptation*. Washington, DC, USDA Technical Bulletin 1935. 186 pp.

West, P.C., Gerber, J.S., Engstrom, P.M., Mueller, N.D., Brauman, K.A., Carlson, K.M., Cassidy, E.S., Johnston, M., MacDonald, G.K., Ray, D.K. & Siebert, S. 2014. Leverage points for improving global food security and the environment. *Science* 345: 325-328

Whitmore, A.P., Kirk, G.J.D. & Rawlins, B.G. 2014. Technologies for increasing carbon storage in soil to mitigate climate change. *Soil use and Management*, 31(S 1): 62-71.

Wilkinson, M.T., Richards, P.J. & Humphreys, G.S. 2009. Breaking ground: Pedological, geological and ecological implications of soil biotrubation. *Earth Science Reviews*, 97: 257-272.

Withers, P.J., van Dijk, K.C., Neset, T.S., Sesme, T., Oenema, O., Rubæk, G.H., Schoumans, O.F., Smit, B., Pellerin, S. 2015. Stewardship to tackle global phosphorus inefficiency: the case of Europe. Ambio, 44: 193-206.

3 | Global Soil Resources

3.1 | The evolution of soil definitions

The definition of soil has changed over time. Early definitions (Kraut, 1853; Ramann, 1919) emphasized the geological or substrate aspect of soil as the upper weathering mantle of the earth's crust. At the end of the 19th century, Vasiliy Dokuchaiev formulated the paradigm of soil as a natural body formed by the combined effect of five soil-forming factors (climate, organisms, parent material, time and relief). This formulation effectively made Dokuchaiev the founder of a new science – pedology. His ideas were translated into English and promulgated by Coffey (1912) and Marbut (1921). Jenny (1941) published the equation of soil forming factors as independent variables; S = f(cl, o, r, p, t...). Dudal, Nachtergaele and Purnell (2002) added a human factor of soil formation, implying that soil is not exclusively a natural body.

For digital soil mapping, the soil forming factors were modified by McBratney, Mendonça-Santos and Minasny (2003) as Sc = f(s, c, o, r, p, a, n...) or Sa = f(s, c, o, r, p, a, n...) where Sc is soil classes, Sa is soil attributes, s is the soil or property at a point, and n is the spatial position. Grunwald, Thompson and Boettinger (2011) further expanded the factor model to the STEP-AWBH Model by including space and time to infer soil properties and their evolution in which the factors of human action, atmosphere, and water are added.

Defined in the simplest terms, soil is the upper layer of the Earth's crust transformed by weathering and physical/chemical and biological processes. It is composed of mineral particles, organic matter, water, air and living organisms organized in genetic soil horizons (ISO, 2013).

3.2 | Soil definitions in different soil classification systems

The World Reference Base for Soil Resources (FAO, 2014) classifies as soil any material within 2 m of the Earth's surface that is in contact with the atmosphere, but excluding living organisms, areas with continuous ice not covered by other material, and water bodies deeper than 2 m.

In the United States Soil Taxonomy (Soil Survey Staff, 1999) soil is considered to be a natural body comprised of solids (minerals and organic matter), liquid, and gases that occurs on the land surface, occupies space, and is characterized by one or both of the following: (i) horizons or layers that are distinguishable from the initial material as a result of additions, losses, transfers, and transformations of energy and matter; and (ii) the ability to support rooted plants in a natural environment.

In the Russian Classification System (Shishov *et al.*, 2004), soil is defined as a solid-phase natural-historical body with a system of inter-related horizons composing a genetic profile and which derives from the transformation of the uppermost layer of the lithosphere by the integrity of soil-forming agents.

French pedologists put emphasis on the spatial aspects of soil as an 'objet naturel, continu et tridimensionnel' (a natural, continuous and three dimensional object) (AFES, 2008). A related variant considers that "soil in nature is a three-dimensional continuum, temporally dynamic and spatially anisotropic, both vertically and laterally" (Sposito and Reginato, 1992).

Urban soils including those 'sealed' by concrete or asphalt, strata of composts or other fertile materials applied to construct lawns and gardens, superficial layers, mine spoil or garbage heaps are also considered in some soil classification systems (Rossiter, 2007). The concept of soils as natural bodies also includes very thin films in caves or fine earth patches within desquamation cracks of hard rocks as found in Antarctic endolithic soils (Goryachkin *et al.*, 2012) and in underwater soils (Demas, 1993). Thus, the concept of soil becomes very broad. Soil scientists have even proposed to extrapolate it to other planets (Targulian *et al.*, 2010).

3.3 | Soils, landscapes and pedodiversity

The relationships between soils and landscapes were at the core of the 'zonality' concept developed by Dokuchaev and tested during his excursion to the Caucasus in 1898. He expressed the concept at the global scale in the form of many-coloured soil bands around the Earth. This zonal concept was also used in the United States 1938 classification of zonal, azonal, and intrazonal soils (Baldwin, Kellogg and Thorp, 1938). Along with zonal ideas, concepts of regularities in local soil patterns emerged. The earliest among these was the concept of soil series developed in the United States in 1903 (Simonson, 1952). The work of Neustuev (1931) on soil geography further developed the concept of regularities.

Another set of spatial soil patterns related to topography was recognized by Milne (1935) and Bushnell (1945) who proposed the term 'catena' (chain) and applied it to soil sequences on the slopes of mountains. Different soil catenas in landscapes all over the world were subsequently described and attempts were made to inventory them systematically (Sommer and Schlichting, 1997).

Fridland (1976) gave a new impulse to the theory of the 'soil/landscape' relationship by defining the types of soil systems related to landforms at different scales ('soil associations'). The relationships between soils – their ingredients, taxonomic distances, geometric shape and kinds of boundaries - were described and for some of them mathematic formulas were proposed.

Fridland's was the first attempt to analyse and quantify the pedological diversity of a territory. The concept of soil diversity, or pedodiversity (Ibáñez, Jiménez-Ballesta and García-Álvarez, 1990; Ibáñez *et al.*, 1995; McBratney, 1992), opened a new conceptual window in soil science (Ibañez and Bockheim, 2013; Toomanian and Esfandiarpoor, 2010). Approaches comprised the description and measurement of either the spatial distribution of soils, or their evolutionary stages by indicating rates of soil development. Soil development makes a contribution to the spatial heterogeneity of the soil because, together with other agents, soils with different evolutionary pathways participate in forming the soil cover and so contribute to the creation of specific soilscapes.

The term 'pedodiversity' and many tools for studying pedodiversity were adapted from biology. Pedodiversity, for example, can be measured just as biodiversity is measured - by means of special indices showing the abundance of species and the taxonomic distances between them. A set of mathematical methods, both parametric and non-parametrical, can be applied to quantify soil spatial heterogeneity.

The pedodiversity concept is an updated, quantification-oriented branch of soil geography. Its advantage is its compatibility with GIS and remote sensing technologies and its solid base in mathematics and statistics, which leads to a broad applicability in environmental sciences and biology.

3.4 | Properties of the soil

Because soils have physical, chemical, mineralogical, and biological characteristics, knowledge of the basic sciences of geology, chemistry, physics and biology contributes to understanding basic soil properties. The solid inorganic fraction defines the soil's texture, the amount of sand, silt, and clay. Solid particles are arranged into aggregates to form diverse structures by biological, chemical and physical processes. Structure describes the size, organization, and shape of the soil aggregates. Consistence and strength are how the soil deforms under pressure. Texture and structure influence porosity and bulk density. Gases or solutions occupy the soil pores. Soil reaction (pH), redox status, carbon, nutrients, and cation exchange capacity are key chemical properties. Secondary clay minerals e.g. smectite, vermiculite, illite, influence the soil physical and chemical properties and are the primary source of ionic exchange. The abiotic, inorganic properties create a platform for the biotic soil component.

Properties that are seen or felt are part of the soil morphology. Soil morphology is the object of study both in nature and in laboratories – micro morphology – with the help of microscopy and computer tomography. Soil colour is influenced by the content and type of organic matter and specific minerals including oxides (e.g. Fe oxi-hydroxides), and redox conditions. Horizon and total soil thickness describe internal organization and root and moisture availability.

3.5 | Global soil maps

Local soil investigations started at the end of the 19th century in Russia (see 3.3. above), but only after World War II were efforts geared towards more systematic national soil inventories. The first regional maps were produced in the early 1960s for Europe (FAO/UNESCO, 1962) and for Africa (D'Hoore, 1964).

The development of a global soil map was initiated by the International Soil Science Society in 1960 and implemented by FAO and UNESCO between 1971 and 1980, resulting in the FAO-UNESCO Soil Map of the World.[1]

1 A digital version of this map is downloadable at: http://www.fao.org/geonetwork/srv/en/resources.get?id=14116&fname=DSMW.zip&access=private

This Soil Map of the World was, from 1995 onwards, systematically updated under the Soil and Terrain Database (SOTER) program carried out by FAO, ISRIC and UNEP together with national soil survey services. This resulted in several regional updates, including for Latin America and the Caribbean, large parts of Africa, and Eastern and Central Europe. In parallel, other organizations, notably the Joint Research Centre (JRC) of the European Commision (EC) and the USDA, undertook regional soil updates, while several countries completed national soil inventories and maps (China, Brazil, Botswana and Kenya etc.). This updated information was harmonized with the digitalized Soil Map of the World and published by a consortium of FAO, IIASA, JRC, ISRIC and CAS in 2006 as the Harmonized World Soil Database (HWSD). Although not fully harmonized and consistent, the HWSD contains the most up-to-date and comprehensive soil information that is currently available. The latest version of this database, giving geo-referenced estimates of twenty soil characteristics, is available online.[2]

In 2006, work began on the design and planning for a soil grid of the world at fine resolution (100 m) and this became known as GlobalSoilMap. The intent was to integrate the best available data from local and national sources and deliver the information online. The format and resolution was to be compatible with other fundamental data sets on terrestrial systems (e.g. vegetation, land cover, terrain, remote sensing). The initial focus was Africa (Sanchez *et al.*, 2009) and this led to the establishment of the African Soil Information System (AfSIS).[3] The technical and logistical complexity of the project has been substantial but good progress has been made during the initial research phase of the project and continental coverages are starting to be published.[4] A full summary is provided by Arrouays *et al.* (2014).

Another, more recent initiative that arose from the GlobalSoilMap effort is Soil Grid 1km[5] which is a collection of updatable soil property and class maps of the world at a relatively coarse resolution of 1 km. These maps are being produced using state-of-the-art model-based statistical methods: 3D regression with splines for continuous soil properties and multinomial logistic regression for soil classes. SoilGrids 1km are outputs of a system for automated global soil mapping developed within the Global Soil Information Facilities framework. This system is intended to facilitate global soil data initiatives and to serve as a bridge between global and local soil mapping (Hengl *et al.*, 2014).

Information on the availability of global, regional and national soil maps has been summarized by Omuto, Nachtergaele and Vargas (2012). The plan for developing the global soil information system was endorsed by the Plenary Assembly of the Global Soil Partnership in July 2014 and it is now being implemented.[6]

A simplified global soil map with the major soil groups is given in Figure A 35 (Annex).

3.6 | Soil qualities essential for the provision of ecosystem services

Soil functions depend on a number of physical, chemical and biological soil properties that in combination determine essential soil qualities. These qualities in turn guarantee that the soil can fulfil its ecological and productive services. Soils differ considerably in terms of properties, qualities, limitations and potential. Significant changes may occur over very short distances, making environmental and soil monitoring difficult (Brammer and Nachtergaele, 2015).

2 http://www.fao.org/soils-portal/soil-survey/soil-maps-and-databases/harmonized-world-soil-database-v12/it/

3 http://www.africasoils.net

4 http://www.clw.csiro.au/aclep/soilandlandscapegrid/

5 http://www.isric.org/content/soilgrids

6 http://www.fao.org/fileadmin/user_upload/GSP/docs/plenary_assembly_II/pillar4.pdf

Soil management has a considerable effect on how the soil may fulfil its ecosystem services. Mineral and organic fertilizer may compensate for poor inherent nutrient conditions in a soil; drainage may remedy excessive wetness in soils, or leach salts when these are present; amendments (lime or gypsum) may correct very acid or highly sodic soils. However, these interventions always have a cost in terms of labour and inputs, and they may also have negative side effects, such as groundwater contamination.

In this section a number of soil qualities essential for the provision of ecosystem services are discussed and related to the major soil groups summarized and illustrated in Annex A35.

3.6.1 | Inherent soil fertility

The capability of a soil to provide sufficient nutrients to crops, grasses and trees is a major quality of soils that supports all provisioning services of the ecosystem. Sixteen nutrients are essential for plant growth and living organisms in the soil. These fall into two different categories: macronutrients and micronutrients. Macronutrients are the most important nutrients for plant development and relatively high quantities are required. Macronutrients include: carbon (C), oxygen (O), hydrogen (H), nitrogen (N), phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg), and sulphur (S). Micronutrients, on the other hand, are needed in smaller amounts, but are still crucial for plant development and growth. Micronutrients include iron (Fe), zinc (Zn), manganese (Mn), boron (B), copper (Cu), molybdenum (Mo) and chlorine (Cl). Nearly all plant nutrients are taken up in ionic forms from the soil solution as cations or as anions.

Soil properties directly related to the amount and availability of nutrients in the soil are: (i) soil texture (clayey soils contain more nutrients than sandy ones); (ii) the type of clay minerals present (smectitic clays absorb more ions than kaolinitic ones); (iii) the soil organic carbon content (more SOC corresponds with a larger amount of nutrients); and (iv) the cation exchange capacity that corresponds to the total of Ca, Mg, K, Na (basic ions) and Al and H (acidic ions) exchangeable with the soil solution. A large amount of available nutrients is present in Vertisols, Chernozems (Borolls), Kastanozems (Ustolls) and Phaeozems (Udolls). Also volcanic soils (Andosols) and alluvial soils (Fluvisols/Fluvents) generally have a large nutrient content. On the other hand, sandy soils (Arenosols/Psamments) and highly leached soils (Ferralsols/Oxisols and Acrisols/ Ultisols) generally have a small nutrient content.

The amount of nutrients that a soil can provide to plants within the growing season represents a limit to nutrient mining. Nutrient mining occurs when crops take out a high proportion of the nutrients available in the soil, leaving a nutrient imbalance that threatens the sustained provision of food and ecosystem services. These challenges are discussed in Section 6.8. Figure 3.1 illustrates an estimation of the nutrient availability in soils globally based on information contained in HWSD.

Soil depth to a hard or an impermeable layer is a vital factor that determines the capability of roots to take hold and determines the total volume of nutrients and water available to crops and vegetation. Soils tend to be deeper when strong weathering conditions prevail over a long period and wherever the parent material is readily weathered. Typical soils include Ferralsols and Nitisols). Shallow soils often occur in mountainous areas (Leptosols) and in dry areas characterized by indurated layers of silica, calcium carbonate or gypsum (Durisols/ Durids, Calcisols/Calcids and Gypsisols/Gypsids). Each plant type has its own ideal rooting conditions. Tubers are the most sensitive to soil depth and volume limitations (Fischer *et al.*, 2008; Grossnickle, 2005; Unger and Kaspar, 1994; McSweeney and Jansen, 1984; Myers *et al.*, 2007). Figure 3.2 illustrates global soil rooting conditions

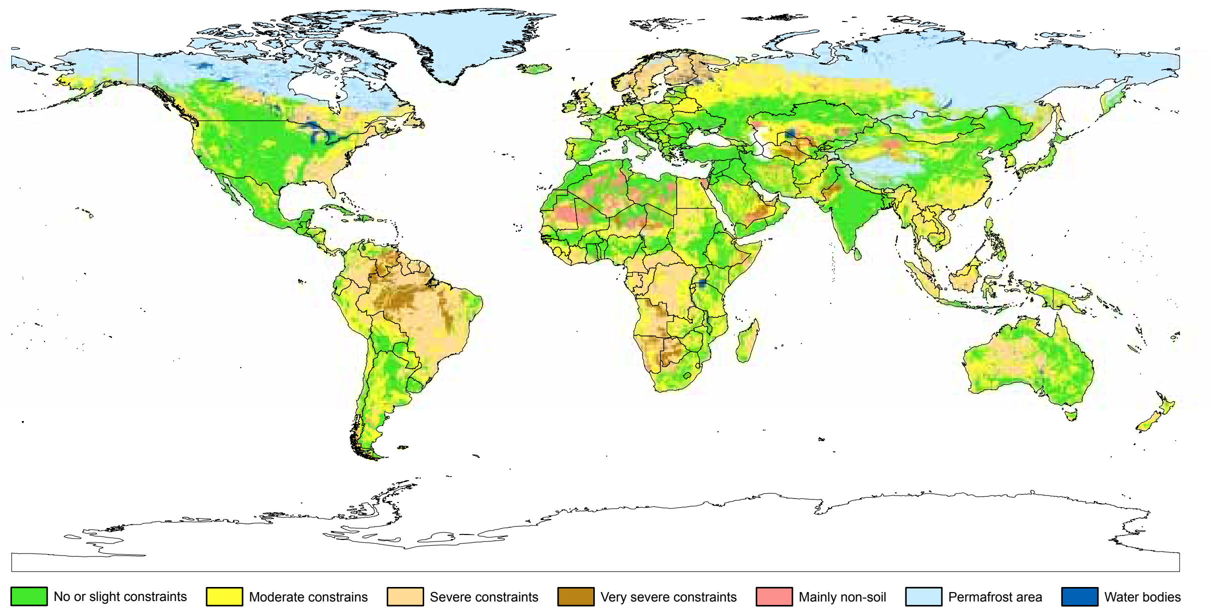


Figure 3.1 | Nutrient availability in soils. Source: Fischer et al., 2008.

The soil pH is a measure of its hydrogen ion concentration and indicates the acidity or alkalinity of the soil. Optimum availability of nutrients occurs around pH=6.5. Toxic concentrations of H and Al occur when the pH drops below 5.5. Values of pH above 7.2 indicate an alkaline reaction and may be symptomatic for the immobilization of nutrients. Very high pH values over 8.5 result in the dispersion of the soil particles and a collapse of structure. High rainfall results in more acid soils (Ferralsols/Oxisols, Alisols, Plinthisols, Acrisols/ Ultisols, Podzols/Spodosols), while drier conditions often lead to the accumulation of Gypsum (Gypsisols/ Gyspsids) or other less soluble salts (Silicon and Calcium Carbonate) in Durisols/Durids and Calcisols/Calcids. The soil pH is also important to the characterization of soil threats to ecosystem services such as acidification (section 6.4) and sodification (Section 6.5). A global map of soil pH is given in Section 6.4.

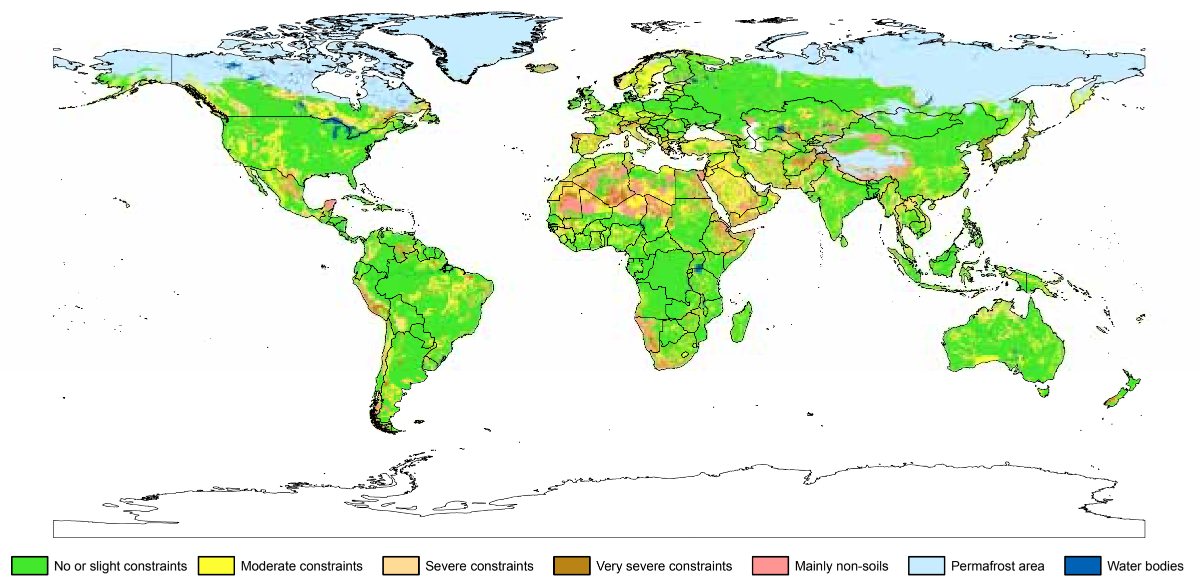


**Figure 3.2 | Global soil rooting conditions.
Source: Fischer et al., 2008.**

Accumulation of water soluble salts: soils in relatively dry areas are often characterized by the accumulation of water-soluble salts (NaCl, Na_2SO_4, etc.) and of less water-soluble salts ($CaCO_3$, $CaSO_4$). These salts may form indurated layers that limit the soil depth available for roots. This accumulation is often a natural process resulting in soils such as Solonchaks/Salids, Calcisols/Calcids, Gypsisols/Gypsids, Durisols/Durids. In irrigation schemes, which are most commonly developed in dry areas, the problem may be human-induced and made worse by the use of saline irrigation water, by insufficient leaching/drainage, or by the conversion to irrigation of soils formed from marine sediments. Most salt-affected soils have moderate to severe limitations for crop production. Section 6.5 deals specifically with salinization and sodification problems.

Toxic elements and other soil fertility problems: some toxic elements such as aluminium occur naturally in acid soils. The parent material may also be a natural source of undesirable elements (for instance cadmium) that may be a problem for human and animal health. Some soils have a high phosphorus adsorption ratio (Andosols and Nitisols/Kandi subgroups) that make P fertilization cumbersome. Atmospheric deposition of toxic elements may also contaminate soils as discussed in section 4.4.

3.6.2 | Soil moisture qualities and limitations

The moisture stored in or flowing through the soil affects **soil formation**, its **structure** and **stability**, and **erosion** run-off. Soil moisture is of primary concern with respect to plant growth. The depth of the groundwater table and the availability of **oxygen** in the soil also affect soil ecosystem functions. The physical properties of soils (texture, structure, porosity, drainage class, permeability) are of prime importance in this respect.

The capacity to store water and moisture in a soil is largely determined by its texture, structure, organic carbon content and depth. Soil moisture provides a buffer for crops during dry periods and is a built-in safeguard against run-off and erosion. Ecological functions of this parameter are discussed in Chapter 7. High soil moisture capacities are typical for deep clayey soils, rich in organic matter and containing modest amounts of $CaCO_3$ (Chernozems, Cambisols). The lowest soil moisture capacities are encountered in sandy soils (Arenosols) or very shallow soils (Leptosols). Very high soil moisture storage occurs in volcanic soils (Andosols) and in many peat soils (Histosols). Figure 3.3 illustrates the distribution of different soil moisture storage classes globally.

Oxygen availability is a critical factor for plant growth. Inadequate oxygen supply to the roots leads to the formation of an underdeveloped root system which is not able to provide sufficient nutrients and water to the plant. Oxygen availability is basically defined by drainage characteristics of soils related to soil type, soil texture, soil phases and terrain slope, all of which play an important role in determining the proportion of gases and water into the soil. Soil phases define specific soil and terrain characteristics. Gleysols/Aquic suborders, Stagnosols and Plinthosols often suffer from temporary saturation with groundwater or rain water, resulting in poor oxygen availability for part of the year. Oxygen availability can be improved by farming practices (e.g. adapted tillage) and by farming inputs such as artificial drainage (Crawford, 1992; Erikson, 1982; Fischer *et al.*, 2008).

3.6.3 | Soils properties and climate change

Soils are both affected by and contribute to climate change. The carbon that is fixed by plants is transferred to the soil via dead plant matter including dead roots and leaves. This dead organic matter creates a substrate which soil micro-organisms respire back to the atmosphere as carbon dioxide or methane depending on the availability of oxygen in the soil. Some of the carbon compounds are easily digested and respired by the microbes, resulting in a relatively short residence time. Others become chemically and/or physically stabilised in soils and have longer residence times (as described in Chapter 2). Soil organic carbon can also be thermally decomposed during fire events and returned to the atmosphere as carbon dioxide. Remaining charred material can persist in soils for long periods (Lehmann *et al.*, 2015).

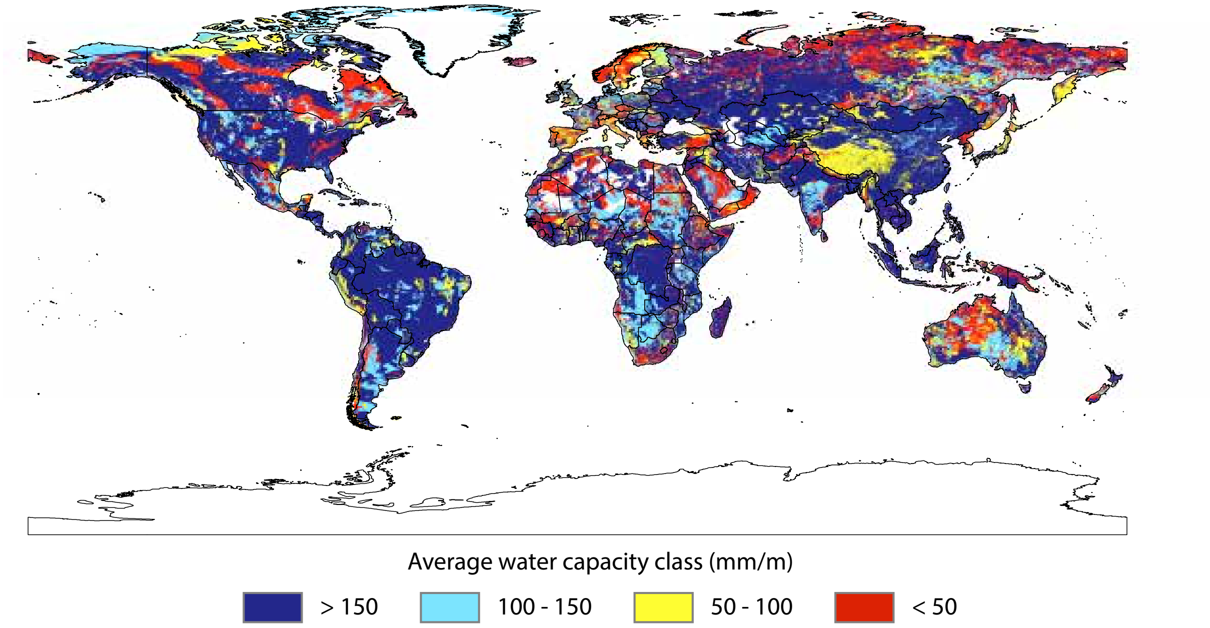


Figure 3.3 | Soil Moisture storage capacity.
Source: Van Engelen, 2012.

Soil organic carbon improves the physical and chemical properties of the soil by increasing the cation exchange capacity and the water-holding capacity. It also contributes to the structural stability of soils by helping to bind particles into aggregates. Soil organic matter (SOM), of which carbon is a major part, holds a great proportion of nutrients, including trace elements, which are of importance to plant growth. SOM mitigates nutrient leaching and contributes to soil pH-buffering capacity. It is widely accepted that the organic matter content of the soil is a major factor contributing to soil functions, including that of organic C storage, which has important feedbacks with the Earth's climate system (Chapter 2).

A large organic carbon content is found in peat soils (Histosols/Histisols), in volcanic soils (Andosols/Andisols) and in steppe soils (Chernozem/Borolls, Kastanozems/Ustolls and Phaeozems/Udolls). Large organic carbon contents are not always indicative of fertile soils because carbon may also accumulate under wet and cold conditions as in Podzols/Spodosols and Cryosols/Gelisols, and in some hydromorphic soils such as Gleysols. Changes in SOC represent one of the major soil threats – see the discussion in section 6.2. The global distribution of soil organic carbon is given in Figure 3.4.

Crysols/Gelisols are soils which are frozen for a large part of the year. In taiga areas they often occur together with Histosols. Global warming in these areas will have a significant effect by allowing agriculture to move more northwards. However, mineralization of organic carbon may be accelerated, with negative consequences for GHG release.

3.6.4 | Soil erodibility and water erosion

The susceptibility of a soil to water erosion is primarily determined by the erosive potential of the rainfall, the slope of the land surface and position of the soil in the catchment, and the vegetative cover on the soil surface. Soil erodibility refers to the susceptibility of soil to erosion by water and is an important secondary control on the intensity of water erosion. Most clay-rich soils (e.g. Vertisols with the exception of erodible self-mulching forms) have a high resilience because they are resistant to detachment. Coarse textured, sandy soils (e.g. Arenosols/Psamments) are also resilient because of low runoff even though these soils are easily detached. Medium textured soils, such as silt loam soils are only moderately resistant to erosion because they are moderately susceptible to detachment and they produce moderate runoff. Soils having a high silt content are the most erodible of all soils. They are easily detached, tend to crust and produce high rates of runoff. Organic matter reduces erodibility because it reduces the susceptibility of the soil to detachment, and increases infiltration, which reduces runoff and thus erosion. Soil structure affects both susceptibility to detachment and infiltration. Permeability of the soil profile affects erodibility because it affects runoff. Past management or misuse of a soil (e.g. by intensive cropping) can increase a soil's erodibility, for example if the subsoil is exposed or if the organic matter has been depleted, or where the soil's structure has been destroyed or soil compaction has reduced permeability. Section 6.1 discusses soil erosion by water in more detail. Soil erodibility worldwide, as characterized by the k factor in the RUSLE equation, is represented in Figure 3.5.

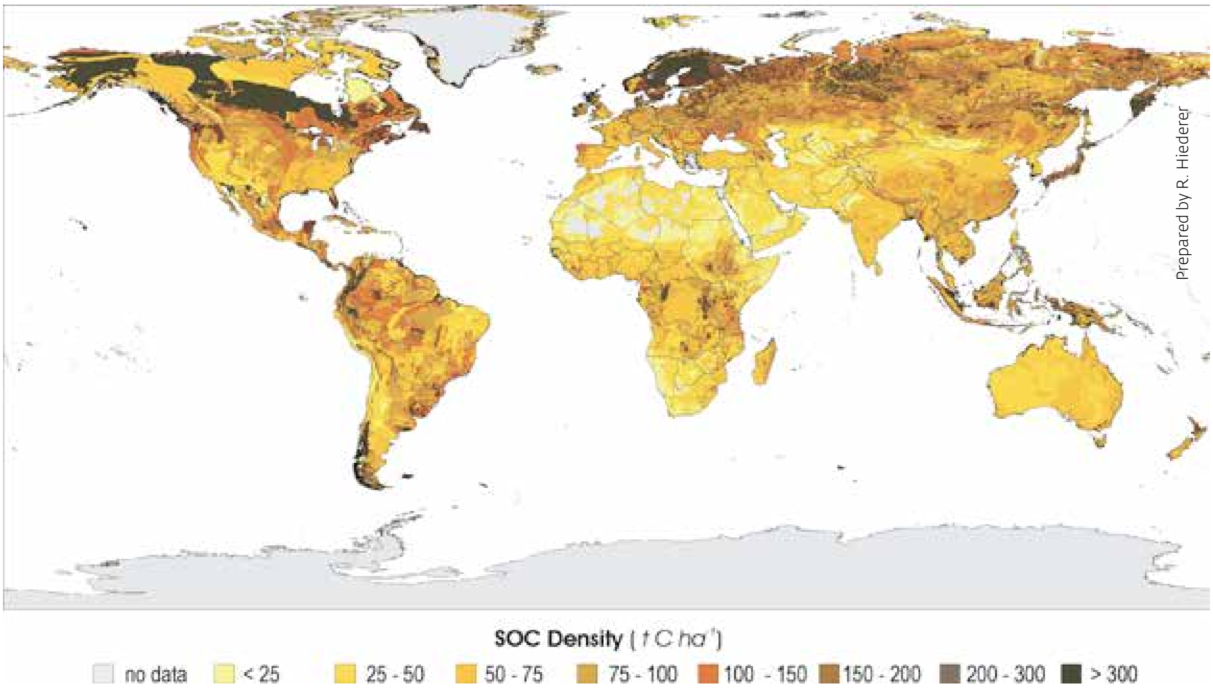


Figure 3.4 | Soil Organic Carbon pool (tonnes C ha^{-1}).

3.6.5 | Soil workability

Soil workability refers to the ease of tillage, which depends on the soil's interrelated characteristics of texture, structure, organic matter content, etc., on the soil's gravel content, and on the presence of continuous hard rock at shallow depth. Depending on the soil characteristics, soil workability also varies with the soil moisture content. Some soils are easy to work regardless of the moisture content, but other soils – such as Vertisols - can be worked only at a specific moisture status. This is true especially for farming systems employing manual cultivation methods or using only light machinery. Soil workability is also related to the type of soil management adopted. While low and intermediate input farming systems mainly face constraints related to soil texture and soil structure, high-level input mechanized farming systems mainly face constraints related to irregular soil depth and stony and rocky soil conditions. Indeed, the use of heavy field equipment is not possible on stony soils or on soils characterized by irregular soil depth. This factor can prevent soil degradation, for example by compaction (Earl, 1997; Fischer *et al.*, 2008; Müller *et al.*, 2011; Rounsevell, 1993). Figure 3.6 shows the distribution of the constraints to soil management and food production due to soil workability worldwide.

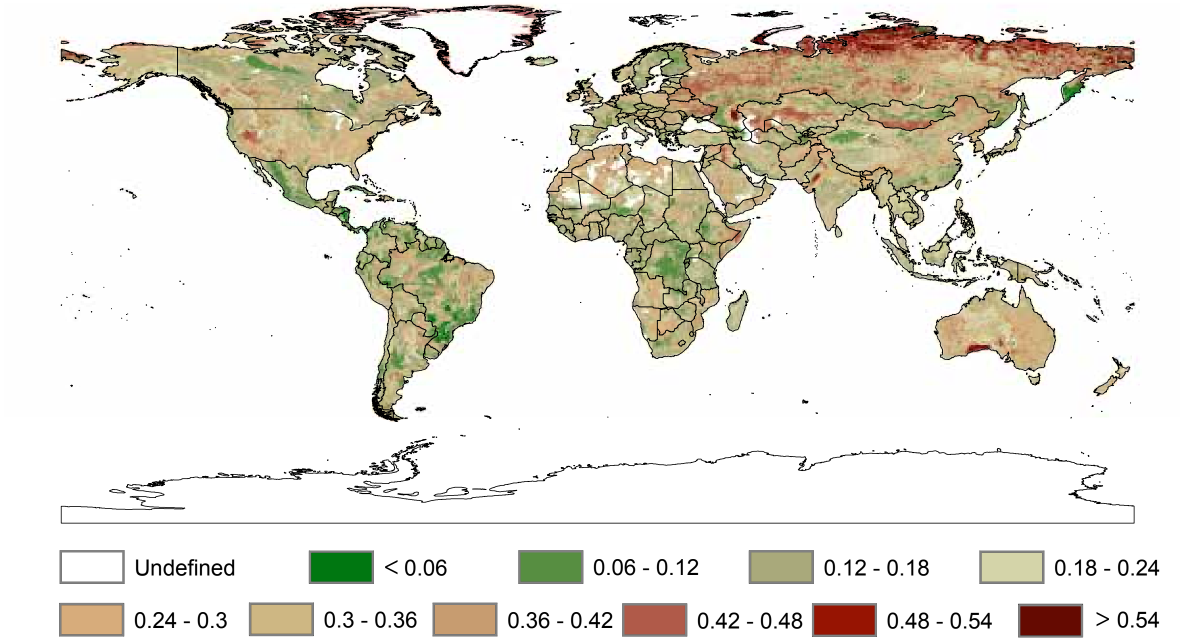


Figure 3.5 | Soil erodibility as characterized by the k factor.
Source: Nachtergaele and Petri, 2011.

3.6.6 | Soils and ecosystem goods and services

Figure 3.7 illustrates the suitability of soils for supporting crops. The evaluation is based on soil health but excludes climatic considerations (except for low temperatures). In Table 3.1, the contribution of the main soil types to major ecosystem services (food security, climate regulation, water regulation and socio-cultural provisions) is estimated at a scale from zero to five. The ratings are based on soil characteristics and quality as measured by: suitability for growing crops; organic carbon content; water holding capacity; and capacity to support infrastructure and store archaeological remains.

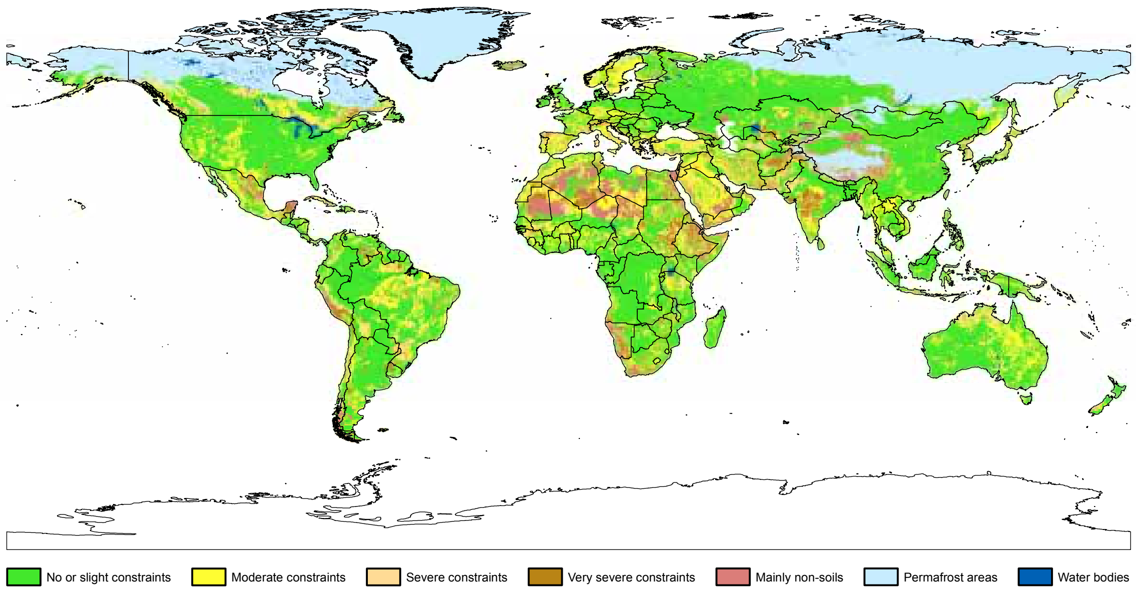


Figure 3.6 | Soil workability derived from HWSD.
Source: Fischer et al., 2008.

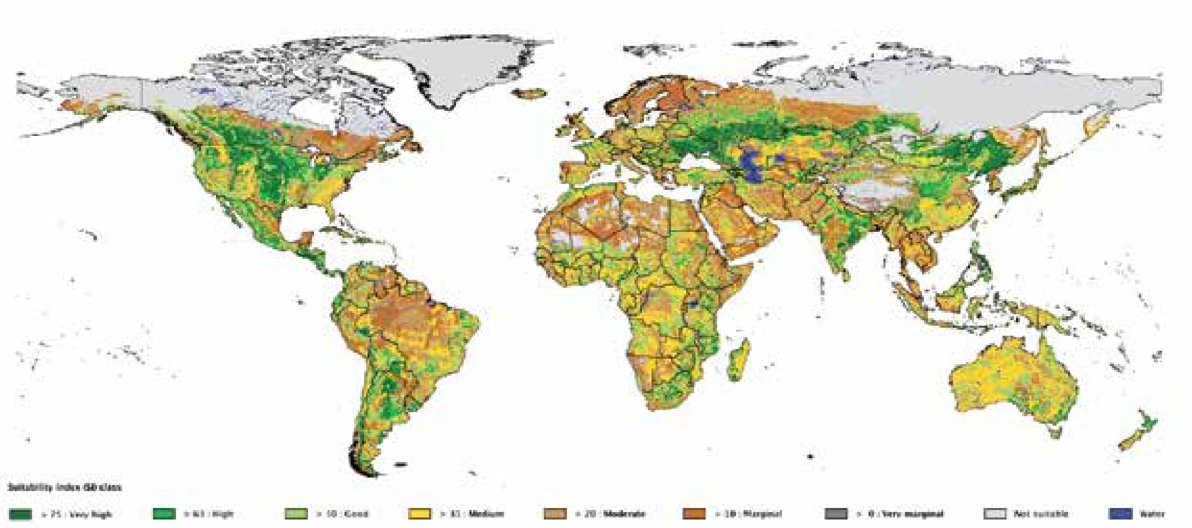


Figure 3.7 | Soil suitability for cropping at low input, based on the global agro-ecological zones study.
Source: Fischer et al., 2008.

Table 3.1 | Generalized ecosystem service rating of specific soil groups (WRB) [7]

	Ecosystem Services					
Reference Soil Groups	Food	Climate	Water	Cultural	SUM	Major Service
Histosols	2	5	5	3	15	Climate Change
Anthrosols	5	5	5	4	19	Food Security
Technosols	1	3	2	4	10	Infrastructure
Cryosols	0	5	2	3	10	Climate Change
Leptosols	1	1	2	1	5	Water runoff
Vertisols	4	2	3	1	10	Food Security
Solonetz	1	1	1	1	4	Very few
Solonchaks	1	1	1	1	4	Very few
Podzols	1	3	1	1	6	Biomass
Ferralsols	2	4	3	1	10	Biomass
Nitisols	4	3	4	1	12	Food Security
Plinthosols	2	1	2	1	6	Biomass
Planosols	1	1	1	1	4	Very few
Gleysols	2	1	3	1	7	Food Security
Stagnosols	2	1	3	1	7	Water storage
Andosols	4	3	5	1	13	Food Security
Chernozems	5	4	4	1	14	Food Security
Kastanozems	3	4	2	1	10	Food Security
Phaeozems	4	4	3	1	12	Food Security
Umbrisols	3	3	3	1	10	Water runoff
Durisols	1	1	1	1	4	Very few
Calcisols	1	1	2	1	5	Very few
Gypsisols	1	1	1	1	4	Very few
Retisols	2	1	2	1	6	Biomass
Acrisols	2	1	2	1	6	Food Security
Lixisols	2	1	2	1	6	Food Security
Alisols	1	1	2	1	5	Biomass
Luvisols	3	2	2	1	8	Food Security
Cambisols	3	2	3	1	9	Food Security
Regosols	2	1	1	1	5	Biomass
Arenosols	1	1	1	1	4	Biomass
Fluvisols	4	2	4	2	12	Food security
Wassents	0	2	2	1	5	Very few

7 Soil Taxonomy equivalents given in the Annex, except for Wassents that are a suborder in Soil Taxonomy

3.7 | Global assessments of soil change - a history

Global assessments of soil and land degradation started more than 40 years ago, but have until now not achieved a clear answer on where soil degradation takes place, what impact it has on the population, and what the cost to governments and land users would be if the decline in soil, water and vegetation resources continued unabated. Although institutional, socio-economic and biophysical causes of soil degradation have been identified locally in many case studies, these have seldom been inventoried systematically at national or regional level. Much of the investment in land reclamation and rehabilitation during recent years has been driven by donor interest to fund action, rather than research to understand the scope of the problem. Even knowledge about what works and what does not work in combating soil degradation is scanty, and there has been little systematic investigation. In recent years, however, the World Overview of Conservation Approaches and Technologies (WOCAT) consortium has begun to make a substantial contribution through its systematic collection of information on sustainable soil and water conservation practices and their impacts.

The first comprehensive assessment of global soil degradation was based on expert opinion only. This was the *GLobal Assessment of human-induced SOil Degradation* - GLASOD, published by UNEP/ISRIC (Oldeman, Hakkeling and Sombroek, 1991). *The Land Degradation Assessment in Drylands* project (LADA) was launched by GEF, implemented by UNEP and executed by FAO between 2006 and 2011 in support of the UNCCD. LADA developed an approach based on remotely-sensed NDVI data (the *Global Land Degradation Assessment* – GLADA). The project also used an ecosystems approach that brought together and interpreted information from pre-existing and newly developed global databases to inform decision makers on all aspects of land degradation at a global scale (GLADIS: the *Global LAnd Degradation Information System*).

During this period other important and broader environmental assessments took place, notably the *Millennium Ecosystem Assessment* (MA, 2005) and the periodical review of the *State of the Environment* by UNEP with the GEO- reports (UNEP, 2012). FAO published a *State of Land and Water* (SOLAW) in 2011. The Economics of Land Degradation (ELD) initiative (ELD, 2015) provided in 2015 a first estimate of the cost of land degradation at global scale based on rather scattered and uncertain information. The annual economic losses due to deforestation and land degradation were estimated at EUR 1.5–3.4 trillion in 2008, equaling 3.3–7.5 percent of the global GDP in 2008. All of these studies used the results of one of the three global inventories: GLASOD, GLADA or GLADIS which are discussed in more detail below.

3.7.1 | GLASOD: expert opinion

An expert consultation on soil degradation convened by FAO and UNEP in Rome in 1974 recommended that a global assessment be made of actual and potential soil degradation. This assessment, which was conducted in collaboration with UNESCO, WMO and ISSS, was based on the compilation of existing data and the interpretation of environmental factors influencing the extent and intensity of soil degradation. The assessment considered such environmental factors as climate, vegetation, soil characteristics, soil management, topography and type of land utilization. The results of this assessment were compiled as a world map of soil degradation. During the next four years FAO, UNESCO and UNEP developed a provisional methodology for soil degradation assessment and prepared a first approximation study identifying areas of potential degradation hazard for soil erosion by wind and water, salinization and sodification. Maps at a scale of 1:5 M covering Africa north of the equator and the Middle East were prepared (FAO/UNEP/UNESCO, 1979). These first efforts were then scaled up into the Global Assessment of Human Induced Soil Degradation Project or GLASOD. The project was initiated by UNEP. It had a duration of 28 months and was executed by ISRIC. In order to cover the whole world, 21 regions and individual countries were defined and experts on these regions were asked to prepare detailed maps of soil degradation. More than 250 soil scientists and environmentalists cooperated in this project (Oldeman, Hakkeling and Sombroek, 1991). The global results of the GLASOD project are available online.[8]

8 http://www.isric.org/UK/About+ISRIC/Projects/Track+Record/GLASOD.htm

A regional follow-up in Southeast Asia resulted in a more detailed database for that region: ASSOD (Van Lynden and Oldeman, 1997).

Since its publication, some expert opinion has faulted GLASOD, questioning the objectivity and reproducibility of an assessment based on expert opinion as an assessment approach (Sonneveld and Dent, 2007). However, at the time GLASOD was developed there were few alternatives available, especially given the overall lack of remotely sensed data at the time. Even today the criticism seems unwarranted as remotely sensed techniques and most modelling approaches have so far failed to come up with more useful assessments. GLASOD results are presented in Figure 3.8.

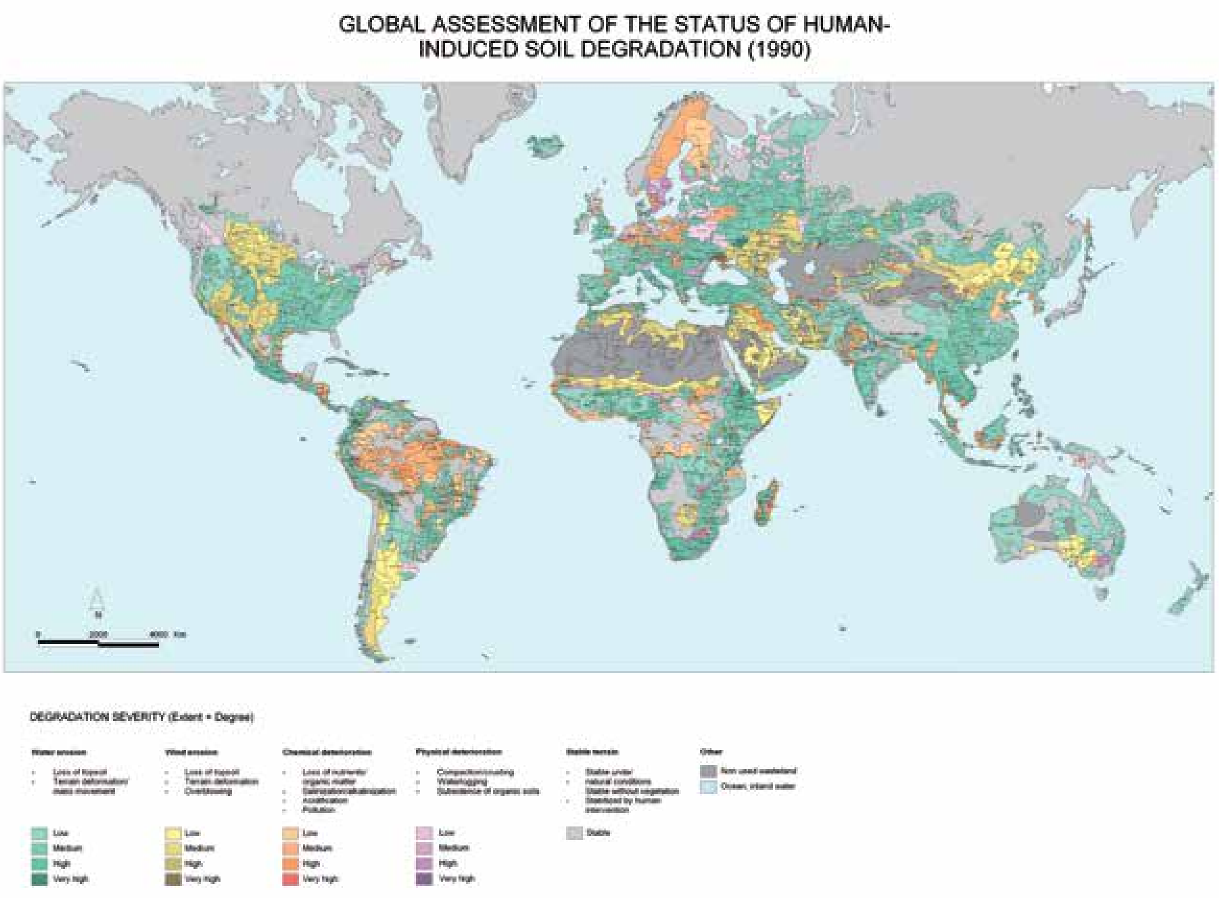


Figure 3.8 | GLASOD results.
Source: Oldeman, Hakkeling and Sombroek, 1991.

3.7.2 | LADA-GLADIS: the ecosystem approach

The first approaches of LADA (see 3.7 above) used remotely-sensed NDVI data to prepare the *Global Land Degradation Assessment* – GLADA (Bai *et al.*, 2008). However, this was soon superseded by a complementary approach that focused on the actual status and trends of land resources in terms of six factors: biomass, water resources, soil health [9], above-ground biodiversity, and economic and social provisions that contribute to ecosystem goods and services (Figure 3.9). The evaluation was based on interpretation of global databases available in the public domain, using documented algorithms to achieve a rating for each of the six factors in terms of status and trends. In order to map the various aspects, a special 'global land use system' was developed (Nachtergaele and Petri, 2011) which allowed cause and effect to be linked. Results were presented in radar diagrams (Figure 3.10) that showed the variability of ecosystem services provided as a function of land use and the need for trade-offs between different factors related to ecosystem goods and services. The GLADIS system is accessible on-line at:

http://www.fao.org/nr/lada/index.php?option=com_content&view=article&id=161&Itemid=113&lang=en

An example of an output for global soil compaction is shown in Figure 3.10.

Criticism of the GLADIS system focused on the unreliability of some of the global databases used and on questions about the downscaling relationships that were developed at local scale (such as the RUSLE). For the specific factor - soil health - the absence of an assessment of wind erosion is certainly a limitation, while the fact that no difference is made between 'natural' and 'human induced' soil erosion is also confusing. These weaknesses have been recognized and should be corrected where possible during the further development of the GLADIS information system which is pending.

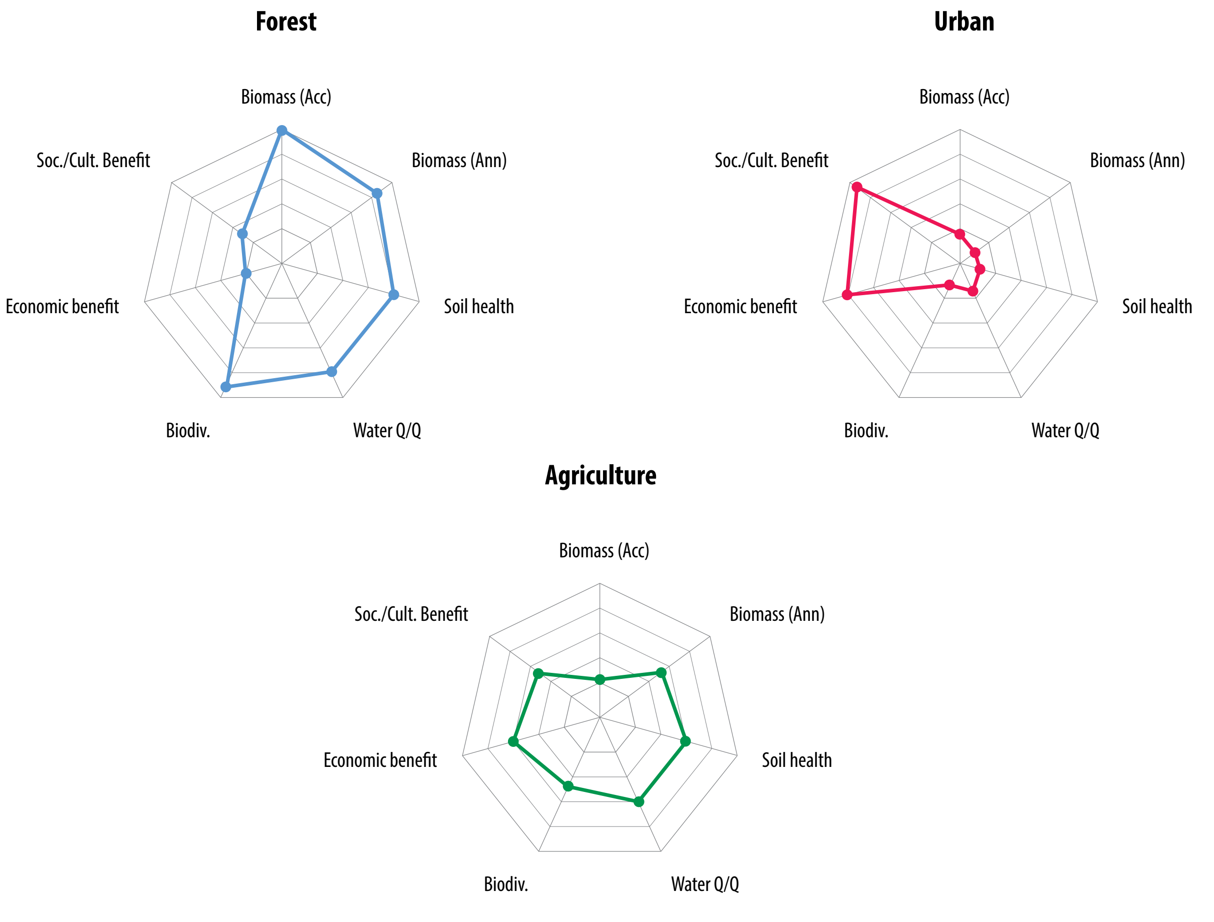


Figure 3.9 | Example of the effect of land use on indicative factors for ecosystem goods and services

9 The soil health status was obtained by comparing the soil suitability for the actual land use. The soil health trend was based on a combination of ratings for the risk of erosion by water, the soil compaction risk, a nutrient balance, and the soil contamination and soil salinization risks.

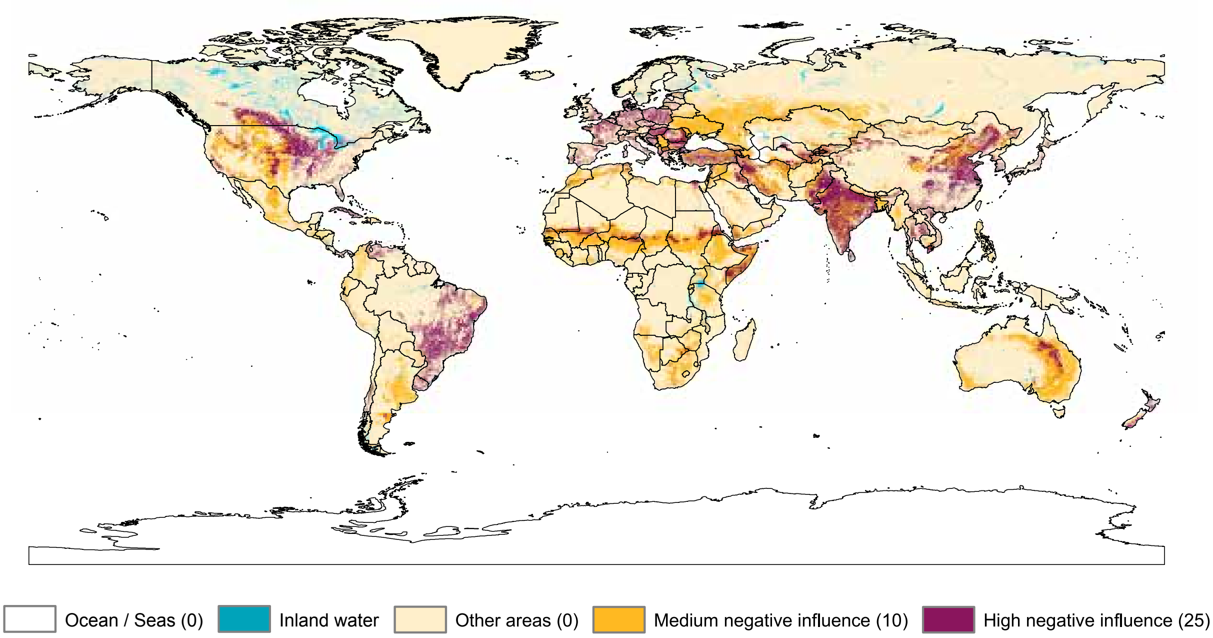


Figure 3.10 | Soil compaction risk derived from intensity of tractor use in crop land and from livestock density in grasslands.
Source: Nachtergaele et al., 2011

3.7.3 | Status of the World's Soil Resources

The present book – The Status of the World's Soil Resources - takes a different approach from the ones described above by focusing on well documented and peer reviewed research data on soil degradation processes, status and trends in scientific literature at all levels. It also draws attention to the uncertainty of estimates made.

The quantity and quality of information on soil degradation is shown to be very variable in different regions. Some regional statements - Africa, Eurasia, Near East, Latin America - still rely on GLASOD or ASSOD. For other regions, such as North America, no regional harmonized approach has been undertaken. Only the EU and the South West Pacific have made progress in establishing new regional updated approaches.

The report also shows the great differences that exist in data and data availability on soil resources and soil change information at national level. Systematic sampling/surveying and monitoring does take place for selected major land uses (forests, arable lands) in most EU countries, the United States and Canada, China, Australia and New Zealand. However, results are not always made available in the public domain. The progress in digital soil mapping may help more countries to produce harmonized data and to make the information public.

The data presented in this book constitute a baseline inventorying the documented knowledge at a point in time: 2015. Future progress can thus be measured against this baseline.

References

AFES. 2008. *Réferentiel pédologique*. France, Éditions Quæ Versailles.

Arrouays, D., Grundy, M.G., Hartemink, A.E., Hempel, J.W., Heuvelink, G.B.M., Hong, S.Y., Lagacherie, P., Lelyk, G., McBratney, A.B., McKenzie, N.J., Mendonca-Santos, M.D.L., Minasny, B., Montanarella, L., Odeh, I.O.A., Sanchez, P.A., Thompson, J.A. & Zhang, G.L. 2014. GlobalSoilMap: Toward a Fine-Resolution Global Grid of Soil Properties. *Advances in Agronomy*, 125: 93-134.

Bai, Z.G., Dent, D.L., Olsson, L. & Schaepman, M.E. 2008. Poxy global assessment of land degradation. *Soil Use and Management*, 24(3): 223–234.

Baldwin, M., Kellogg, C.E. & Thorp, J. 1938. Soil Classification. *In Soils and Men: Yearbook of Agriculture* 1938. pp. 979-1001. Washington, DC, U.S. Government Printing Office.

Brammer, H. & Nachtergaele, F.O. 2015. Implications of soil complexity for environmental monitoring. *International Journal of Environmental Studies*, 72(1): 56-73.

Bushnell, T.M. 1945. The catena cauldron. *Soil* Sc.Soc.Am. *Proc*., 10: 335-340.

Coffey, G.N. 1912. A Study of the Soils of the United States. *U.S. Dept. of Agriculture Bur*., *Soils Bull*. V 85. U.S. Washington, DC, Govt. Printing Office.

Crawford, R.M.M. 1992. Oxygen availability as an ecological limit to plant distribution. *In* M. Begon & A.H Fitter, eds. *Advances in ecological research*. V 23. pp. 93-171. Academic Press. 355 pp.

D'Hoore, J.L. 1964. *Carte des sols d'Áfrique*. *C.C.T.A*. Service pédologique interafricain. Projet conjoint No 11.C.C.T.A Publication No 93.

Demas, G. P. 1993. Submerged soils: a new frontier in soil survey. *Soil Survey Horizons*, 34: 44-46.

Dudal, R., Nachtergaele, F.O. & Purnell, M.F. 2002. The human factor of soil formation. Symposium 18. Vol. II. paper 93. Bangkok, Transactions 17th World Congress of Soil Science.

Earl, R. 1997. Prediction of trafficability and workability from soil moisture deficit. *Soil and Tillage Research*, 40: 155-68.

ELD. 2015. *The value of land: Prosperous lands and positive rewards through sustainable land management* (available at *www.eld-initiative.org*)

Erickson, A.E. 1982. Tillage effects on soil aeration. *Predicting tillage effects on soil physical properties and processes* (special issue), 44: 91-104.

FAO. 2011. *The State of the World's Land and Water Resources for food and agriculture (SOLAW)*. Managing systems at risk. Food and Agriculture Organisation of the United Nations, Rome and London, Earthscan.

FAO. 2014. *World reference base for soil resources 2014*. World Soil Resources Report 106. Rome. 120 pp.

FAO/IIASA/ISRIC/ISS-CAS/JRC, 2009. Harmonized World Soil Database (version 1.1). Rome, FAO & Austria, Laxenburg, IIASA (available at *http://www.fao.org/soils-portal/soil-survey/soil-maps-and-databases/harmonized-*world-soil-database-v 12/it/)

FAO/UNEP/UNESCO. 1979. *A Provisional Methodology for Soil Degradation Assessment*. Rome, Food and Agriculture Organisation of the United Nations.

FAO/UNESCO. 1962. First draft of a unified soil map of Europe. *In* R. Dudal, ed. World Soil Resources Report No 3. pp. 33-38. Rome, Food and Agriculture Organization of the United Nations.

Fischer, G., Nachtergaele, F., Prieler, S., van Velthuizen, H.T., Verelst, L. & Wiberg, D. 2008. *Global Agro-ecological Zones Assessment for Agriculture (GAEZ 2008)*. Laxenburg, Austria, IIASA & Rome, Food and Agriculture Organisation of the United Nations.

Fridland, V.M. 1976. *Pattern of the soil cover*. Jerusalem. 291 pp. (First published in Russian in 1972).

Goryachkin, S. Cherkinsky, A., Mergelov, N. & Shorkunov, I. 2012. Endolithic (bio)weathering and rock varnish in East Antarctica as early Earth analogs of 'protosoils'. *Mineralogical Magazine*. 76(6): 1777.

Grossnickle, S.C., 2005. Importance of root growth in overcoming planting stress. *New forests*, 30: 273-294.

Grunwald, S., Thompson, J.A. & Boettinger, J.L. 2011. Digital soil mapping and modeling at continental scales – finding solutions for global issues. *Soil Sci. Soc. Am. J. (SSSA 75th Anniversary Special Paper)*, 75(4): 1201-213.

Hengl, T., de Jesus, J.M., MacMillan, R.A., Batjes, N.H., Heuvelink, G.B.M., Ribeiro, E., Samuel-Rosa, A., Kempen, B., Leenaars, J.G.B., Walsh, M.G. & Gonzalez, M.R. 2014. SoilGrids1km – Global Soil Information Based on Automated Mapping. *PLoS ONE*, 9(8): e 105992.

Hiederer, R. & Köchy, M. 2011. *Global soil organic carbon estimates and the harmonized world soil database*. European Commission, Joint Research Centre, Institute for Environment and Sustainability. Luxembourg, Publication Office of the European Union. 90 pp.

Ibañez, J.J. & Bockheim, J. 2013. *Pedodiversity*. London-New York, CRC Press. 244 pp.

Ibáñez, J.J., De-Alba, S., Bermúdez, F.F. & García-Álvarez. A. 1995. Pedodiversity: concepts and measures. *Catena*, 24: 215-232.

Ibáñez, J.J., Jiménez-Ballesta, R. & García-Álvarez, A. 1990. Soil Landscapes and drainage basins in mediterranean mountain areas. *Catena*, 17: 573-583.

ISO. 2013. Draft international standard ISO/DIS 11074. 9 pp.

Jenny, H. 1941. *Factors of soil formation: a system of quantitative pedology*. New York, Dover Publications.

Kraut, H. 1853. Boring rocks for blasting. *The Civil Engineer and Architect's Journal*, 20:1857

Lehmann, L., Abiven, S., Kleber, M., Pan, G., Singh, B.P., Sohi, S. & Zimmerman, A. 2015. Persistence of biochar in soil. *In* J. Lehmann & S. Joseph, eds., *Biochar for Environmental Management*. Science and Technology, 2nd edition.

MA. 2005. *Ecosystems and Human Well-Being: Current State and Trends Findings of the Condition and Trends*. Working Group Millennium Ecosystem Assessment. Series Vol. 1. Island Press. 948 pp.

Marbut, C.F. 1921. The contribution of soil surveys to soil science. *Proceedings of the Society for the Promotion of Agricultural Science*, 41: 116-142.

McBratney, A.B. 1992. On variation, uncertainty and informatics in environmental soil management. *Australian Journal of Soil Research*, 30: 913–935.

McBratney, A.B., Mendonça-Santos, M.L. & Minasny, B. 2003. On digital soil mapping. *Geoderma*, 1: 3-52.

McSweenev, K. & Jansen, I.J. 1984. Soil structure and associated rooting behaviour in minesoils. *Soil science society of America journal*, 48: 607-612.

Milne,G. 1935. Composite units for the mapping of complex soil associations. *Transac. 3rd Intern.* Congr.Soil *Sci*. Vol.I pp. 345-347.

Müller, L., Lipiec, J., Kornecki, T.S. & Gebhardt, S. 2011. Trafficability and workability of soils. *Encyclopedia of afrophysics*, pp. 912-924.

Myers, D.B., Kitchen, N.R., Sudduth, K.A., Sharp, R.E. & Miles, R.J. 2007. Soybean root distribution related to claypan soil properties and apparent soil electrical condusctivity. *Crop science*, 47: 1498-1509.

Nachtergaele, F.O. & Petri, M. 2011. *Mapping land use systems at global and regional scales for land degradation assessment analysis v 1.1*. LADA. Rome, FAO.

Nachtergaele, F.O., Petri, M., Biancalani, R., van Lynden, G., van Velthuizen, H. & Bloise, M. 2011. *Global Land Degradation Information System (GLADIS v 1.0)*. LADA Technical Report #17. An information base for land degradation at global level. Rome, FAO.

Neustuev, S.S. 1931. *Elements of Soil Geography*. Leningrad [in Russian].

Oldeman, L.R., Hakkeling, R.T.A. & Sombroek, W.G. 1991. *World Map of the Status of Humaninduced Soil Degradation (GLASOD): An Explanatory Note*. Wageningen, International Soil Reference and Information Centre.

Omuto, C., Nachtergaele, F. & Vargas, R. 2013. *State of the Art Report on Global and Regional Soil Information: Where are we? Where to go?* Global Soil Partnership. Technical Report no 1. Rome, Food and Agriculture Organization of the United Nations.

Ramann, E. 1919. Der boden und sein geographischen Wert. *Mitteilungen der Geographischen Gesellschaft Milnchen*, 13: 1-14.

Rossiter, D.G. 2007. Classification of urban and industrial soils in the world reference base for soil resources. *Journal of soils and sediments*, 7(2): 96-100.

Rounsevell, M.D.A. 1993. A review of soil workability models and their limitations in temperate regions. *Soil use and management*, 9: 15-20.

Sanchez, P.A., Ahamed, S., Carré, F., Hartemink, A.E., Hempel, J., Huising, J., Lagacherie, P., McBratney, A.B., McKenzie, N.J., Mendonça-Santos, M.L., Minasny, B., Montanarella, L., Okoth, P., Palm, C.A., Sachs, J.D., Shepherd, K.D., Vagen, T.G., Vanlauwe, B., Walsh, M.G., Winowiecki, L.A. & Zhang, G.L. 2009. Digital soil map of the world. *Science*, 325(5941): 690-81.

Shishov, L., Tonkonogov, V., Lebedeva, I. & Gerasimova, M. 2004. *Classification and diagnostics of soils of Russia*. Smolensk, Oekumena. 341 pp. [in Russian].

Simonson, R.W. 1952. Lessons from the first half century of soil survey: I. Classification of soils. *Soil Science*, 74: 249–257.

Soil Survey Staff. 1999. Soil taxonomy. A basic system of soil classification for making and interpreting soil surveys. 2nd Edition. Agric. Handbook 436. Washington, DC, Natural Resources Conservation Service, United States Department of Agriculture.

Sommer, M. & Schlichting, E. 1997. Archetypes of catenas in respect to matter – a concept for structuring and grouping of catenas. *Geoderma*, 76(1): 1-33.

Sonneveld, B.G. & Dent, D.L. 2009. How good is GLASOD? *Journal of Environmental Management*, 90(1): 274-283.

Sposito, G. & Reginato R. 1992. *Opportunities in Basic Soil Science Research*. Wisconsin, Madison, SSSA. 109 pp.

Targulian, V., Mergelov N., Gilichinsky, D., Sedov, S., Demidov N., Goryachkin, S. & Ivanov, A. 2010. Dokuchaev's soil paradigm and extraterrestrial 'soils'. In R.J. Gilkes & N. Prakongkep, eds. *Proceedings of the 19th World Congress of Soil Science; Soil Solutions for a Changing World*. pp. 1-4. Division Symposium. Astropedology. Australia, Brisbane, IUSS.

Toomanian, N. & I. Esfandiarpoor. 2010. Challenges of pedodiversity in soil science. *Eurasian Soil Science*, 43: 1486-1502.

UNEP. 2012. *GEO-5 Global Environmental Outlook*. Nairobi, UNEP.

Unger, P.W. & Kaspar, T.C. 1994. Soil compaction and root growth: a review. *Agronomy Journal*, 86: 759-766.

Van Engelen, V. 2012. e-SOTER as a contribution to a Global Soil Observing System. In '*Towards Global Soil Observation: activities within the GEO Task Global Soil Data*'. Workshop Report, GSP, Rome, Food and Agriculture Organization of the United Nations.

Van Lynden, G.W.J. & Oldeman, L.R. 1997. *The assessment of the status of human-induced soil degradation in south and south-east Asia (ASSOD).* Wageningen, United Nations Environment Programme (UNEP), Food and Agricultural Organization of the United Nations (FAO), International Soil Reference and Information Centre (ISRIC).

4 | Soils and Humans

4.1 | Current land cover and land use

The new Global Land Cover Share database (Latham *et al.*, 2014) includes eleven global land cover layers, each representing the major land cover classes defined by the FAO and SEEA legend (Weber, 2010).

Analysis of the database indicates that of the global land mass, artificial surfaces occupy 0.6 percent, croplands 12.6 percent, grasslands 13.0 percent, tree-covered areas 27.7 percent, shrub-covered areas 9.5 percent, herbaceous vegetation 1.3 percent, mangroves 0.1 percent, sparse vegetation 7.7 percent, bare soils 15.2 percent, snow and glaciers 9.7 percent and inland water bodies 2.6 percent.

The intensity of each land-cover type varies substantially across the globe according to numerous factors, including soils, altitude, climatic conditions and anthropogenic influences. For example, while cultivated land is less than 10 percent in most African regions, it accounts for more than 25 percent of the land in the Asia region. A land cover map is given in Figure 4.1. Summary statistics by region, derived from the respective GIS layers are given in Figure 4.2. In the following discussion, attention is focused on three main land cover classes: cropland, grasslands/grazing lands and forests. The management of these three classes has large impacts on soils and ecosystem services. The presence of artificial surfaces is treated in more detail in Section 6.7. More than 25 percent of the land mass carries almost no vegetation because of climatic factors (glaciers, deserts) or topographic or soil conditions.

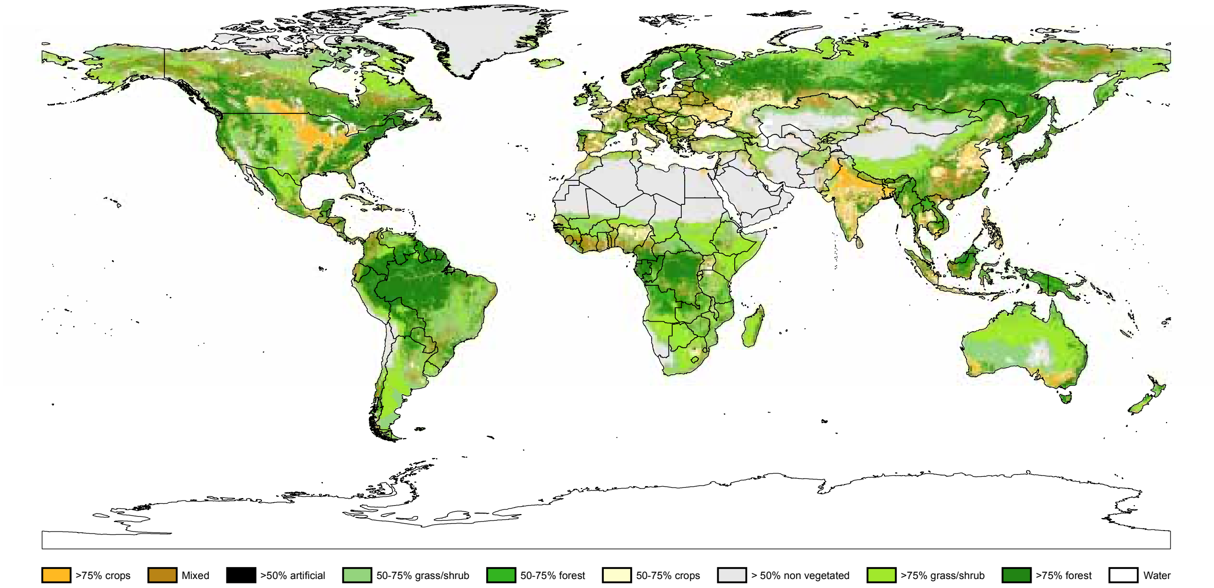


Figure 4.1 | Global Land Cover.
Source: *Latham et al.*, 2014.

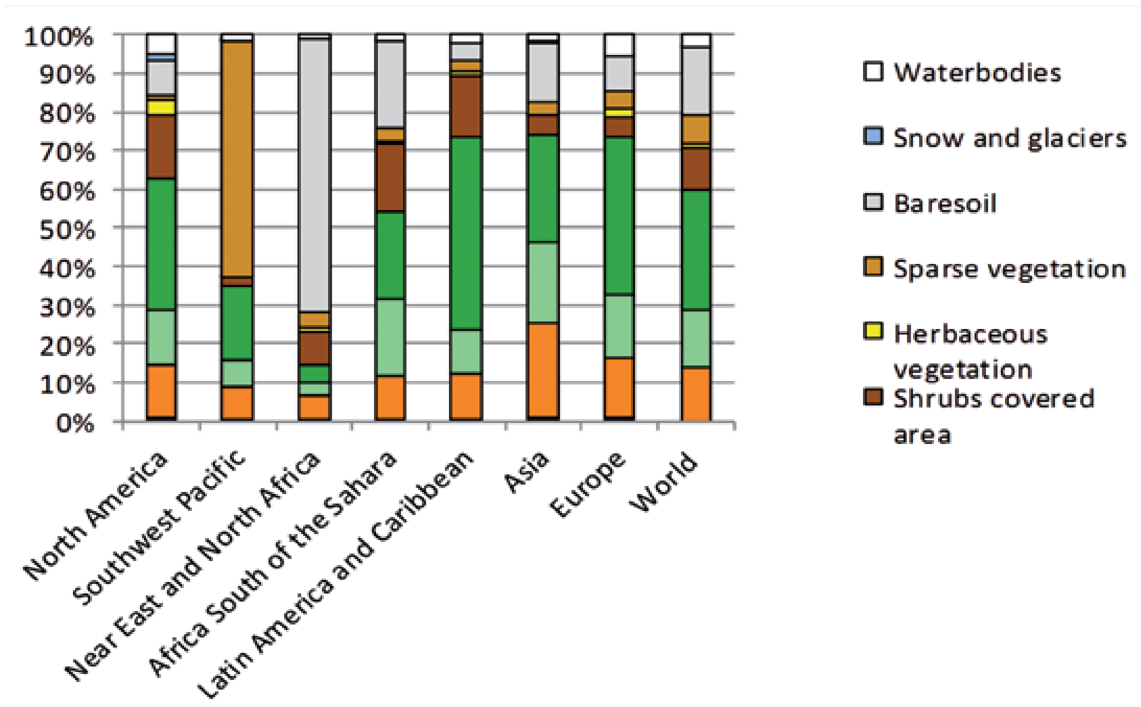


Figure 4.2 | Distribution of land cover in different regions.
Source: *Latham et al.*, 2014.

Cropland

SOLAW (FAO, 2011) established that the cultivated land area in terms of per capita use in 2000 was highest in Australia (more than 2.2 ha per person), followed by North America and Eastern Europe and Russia (about 0.7 ha per person). In contrast, current cultivated land used per capita is only 0.2 ha in Western Europe and in most less developed countries.

By dividing the current cultivated land by the projected populations, the anticipated cultivated land area per capita in 2050 can be estimated. In the more developed countries, the cultivated area per capita would change little. In less developed countries, the cultivated area per capita is expected to halve to 0.1 ha by 2050, unless there is further expansion of the cultivated area.

Further characterization of cropland and land use at a global scale by remote sensing is difficult because:

1. The spatial extent of croplands is highly variable between and within nations. Cropland characteristics such as field size can be highly variable, even for the same crop type. Spatial extent of cropland depends on a host of factors, including the historical, political, social and technological context of agricultural development as well as natural factors such as landscape patterns.
2. Patterns of agricultural intensification – for example, the use of fertilizer – vary greatly, especially between developed and developing nations.
3. Each crop type has a specific growth phenology and structure, with significant seasonal variation between and even within individual crop types.
4. Cropland can be confused with natural vegetation cover types – for example, surveys may confuse cereal grains with tall-grass prairie (Pittman *et al.*, 2010). Better cropland information – in terms of both its extent and the purpose and intensity of its use – is vital to understanding soil change and to formulating adequate responses. Special attention should be paid to irrigated agriculture in developing countries, which covers about one-fifth of all arable land, and accounts for 47 percent of all crop production and almost 60 percent of cereal production (Nachtergaele *et al.*, 2011).

Grazing lands

Grazing lands, including sown pasture and rangeland with various coverage (grasslands, bush/shrublands), are among the largest ecosystems in the world and contribute to the livelihoods of more than 800 million people. They are a source of goods and services such as food and forage, energy and wildlife habitat, and also provide carbon and water storage and watershed protection for many major river systems. Grasslands are also important for in situ conservation of genetic resources. Of a total of 10 000 species, only 100 to 150 forage species have been cultivated, but many more hold potential for sustainable agriculture. Estimates of the proportion of the Earth's land area covered by grasslands vary between 20 and 40 percent, depending on the definition. Those differences are due to a lack of harmonization in the definition of grasslands.

There has been a significant reduction of pasture in Eastern Africa, partially because large grassland areas have been destroyed or converted to agricultural land. In South America, pastures have been lost because of conversion to soybean cultivation. In Europe there has been a gain in grazing lands because European policies such as the 'set-aside' measures oblige farmers to leave a portion of their agricultural land in fallow as a condition for benefiting from direct payments (Suttie, Reynolds and Batello, 2005).

Forests

In 2010, forests covered about 28 percent of the world's total land area. Deforestation affected an estimated 13 million ha per year between 2000 and 2010. Net forest loss was, however, considerably less – 5.2 million ha per year – as losses were compensated by afforestation and some natural expansion (FAO, 2014a). Most deforestation takes place in tropical countries, whereas most developed countries with temperate and boreal forest ecosystems – and more recently, countries in the Near East and Asia – are experiencing stable or increasing forest areas. Between 1990 and 2010, the amount of forest land designated primarily for the conservation of biological diversity increased by 35 percent, indicating a political commitment to conserve forests. These forests now account for 12 percent of the world's forests.

Approximately 13.2 million people worldwide are formally employed in the forestry sector. Many more depend directly on forests and forest products for their living. In developing countries, wood-based fuels are the dominant source of energy for more than two billion mostly poor people. In Africa, over 90 percent of harvested wood is used for energy. Wood accounts for 27 percent of total primary energy supply in Africa, 13 percent in Latin America and the Caribbean and five percent in Asia and Oceania. However, it is also increasingly used in developed countries with the aim of reducing dependence on fossil fuels. For example, about 90 million people in Europe and North America now use wood energy as the main source of domestic heating (FAO, 2014a).

Conclusion

Land cover and land use are essential factors to understand soil change. In particular, better cropland information, in terms of extent, purpose and intensity of use, is vital to understanding soil change and to formulating adequate responses.

4.2 | Historical land cover and land use change

Since the early days of agriculture, human activity has altered vegetation cover and soil properties. 'Land use change' or 'land cover change' typically refers to changing from one type of vegetation cover to another (e.g forest to pasture, natural grassland to cropland). Although the terms land use change and land cover change are often used interchangeably, 'land use' is more typically used to refer to management within a land cover type. Land use is thus "characterised by the arrangements, activities and inputs people undertake in a certain land cover type to produce, change or maintain it" (FAO/UNEP, 1999). Land use change has been accelerated by migration and population increase as food, shelter, and materials are sought and acquired. It is estimated that humans have directly modified at least 70 million km^2, or >50 percent of Earth's ice-free land area (Hooke, Martín-Duque and Pedraza, 2012).

For a long period of human activity, until about a thousand years ago, cropland and pasture occupied less than one to two percent each of the global ice-free land area (based on a range of data sources in Klein Goldewijk *et al.*, 2011 and depicted in Figure 4.3; also see Ramankutty, Foley and Olejniczak, 2002). Subsequently, as the population centres of Europe, South Central Asia and Eastern Asia expanded, more land was converted from natural vegegation to cultivated lands. Cover of croplands and pastures was about two to four percent each by 1700 (Klein Goldewijk *et al.*, 2011). By 1900, agriculture had further expanded in these areas, and spread to North America. Since 1900 rapid expansion has continued, including the arable areas of South America, Africa and Australia. As a result, today nearly all soils and climates suitable for cultivation in industralized countries are in use for crop production. In some of these countries, cropland expansion has been reversed in recent years, as with the EU set-aside programme. South America and Africa continue to convert land use to crop production. By 2000, global cropland cover had reached 11 percent and pasture cover 24 percent, according to Klein Goldewijk *et al.* (2011) based on FAO statistics.

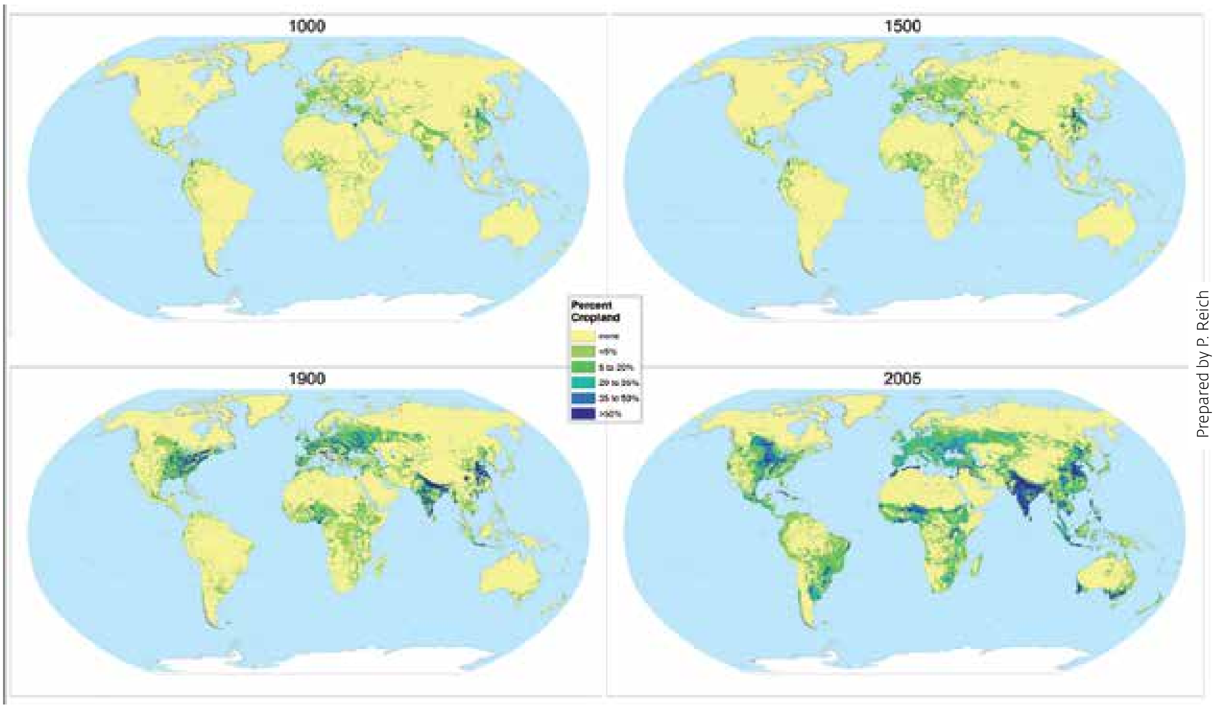


Figure 4.3 | Historical land use change 1000 – 2005.
Source: Klein Goldewijk et al., 2011.

The net loss of natural land has been dominated by loss of tropical forests (3.3 million km²), tropical grasslands (6.8 million km²) and temperate grasslands (5.5 million km²). Quantification from satellite imagery of global forest change over the period 2000-2012 shows that tropical deforestation remains the predominant source of losses (Hansen *et al.*, 2013). However, there has been a reduced rate of deforestation in some regions over the last decade, most notably in Brazil. This is coupled with a rising rate of afforestation in some areas in recent decades, notably in Europe and the United States, and more recently in China, Vietnam and India (FAO, 2013).

4.3 | Interactions between soils, land use and management

Many soils are subject to some degree of direct or indirect human disturbance. However, distinguishing natural from direct and indirect human influence is not always straightforward (Smith, 2005). Nonetheless, some human activities have clear direct impacts. These include land use change, land management, land degradation, soil sealing, and mining. The intensity of land use also has a great impact on soils. Soils are also subject to indirect impacts arising from human activity, such as acid deposition (for example, sulphur and nitrogen) and heavy metal pollution. In this section, we report the state-of-the-art understanding and the knowledge gaps concerning these impacts on soils.

4.3.1 | Land use change and soil degradation

Land cover change (Section 4.2), for example from forest or natural grassland to pasture or cropland, removes biomass and disturbs soils. This in turn leads to loss of soil carbon and other nutrients and to changes in soil properties and in soil biodiversity. Some land cover conversions – for example, afforestation after abandonment of cropland – can result in increases of soil carbon and nutrients. Land use that does not result in a change of cover, such as forest harvest and regrowth, or increasing grazing intensity, can nonetheless result in degradation of soil properties.

Degrading land covers approximately 24 percent of the global land area (35 million km^2). 23 percent of degrading land is broadleaved forest, 19 percent needle-leaved forests, 20–25 percent rangeland (Bai *et al.*, 2008). The scale and nature of the changes are highly variable with type of land cover change, climate, and method of vegetation removal (e.g. land clearing fires, mechanical harvest). This section focuses on meta-analyses of field data and global model results. The effects of land use changes within agricultural lands are dealt with in Section 4.3.2.

Impacts of land cover change

Wei *et al.* (2014) collated observations from 119 publications of 453 paired or chrono-sequential sites in 36 countries where tropical, temperate, and boreal forests were converted to agricultural land. The SOC stocks were corrected for changes in soil bulk density after land-use change and only SOC in the upper 0–30 cm was considered. The SOC stocks decreased at 98 percent of the sites by an average of 52 percent in temperate regions 41 percent in tropical regions and 31 percent in boreal regions. The decrease in SOC stocks and the turnover rate constants both varied significantly according to forest type, cultivation stage, climate and soil factors. A meta-analysis (Guo and Gifford, 2002) of 74 publications across tropical and temperate zones showed a decline in soil C stocks after conversion from pasture to plantation (-10 percent), native forest to plantation (-13 percent), native forest to crop (–42 percent), and pasture to crop (–59 percent). Soil C stocks increased after conversions from native forest to pasture (+8 percent), crop to pasture (+19 percent), crop to plantation (+18 percent), and crop to secondary forest (+53 percent). Broadleaf tree plantations placed onto prior native forest or pastures did not affect soil C stocks whereas pine plantations reduced soil C stocks by – 12 to – 15 percent. In this study, soil depth varied from less than 30 cm to more than 100 cm and was not adjusted to account for changes in bulk density with land use change.

In a meta-analysis of 385 studies on land use changes in the tropics (Don, Schumacher and Freibauer, 2011), SOC decreased when primary forest was converted to cropland (-25 percent), perennial crops (-30 percent) and grassland (-12 percent). SOC increased when cropland was afforested (+29 percent) or under cropland fallow (+32 percent) or converted to grassland (+26 percent). Secondary forests stored 9 percent less SOC than primary forests. Relative changes were equally high in the subsoil as in the surface soil (Don, Schumacher and Freibauer, 2011). In this study, SOC stocks were corrected to an equivalent soil mass and sampling depth was on average 32 cm.

The response of soil organic carbon (SOC) to afforestation in deep soil layers is still poorly understood. Shi *et al.* (2013) compiled information on changes in deep SOC (defined as at least 10 cm deeper than the 0–10 cm layer) after afforestation of croplands and grasslands (total 63 sites from 56 literature). The responses of SOC to afforestation were slightly negative for grassland, and significantly positive for cropland. The SOC in soil depth layers (up to 80 cm) was reduced after afforestation of grassland but not significantly. By contrast, conversion of cropland to forests (trees or shrubs) increased SOC significantly for each soil layer up to 60 cm depth.

Poeplau *et al.* (2011) compiled 95 studies conducted on conversion in temperate climates. One finding was that topsoil (0-30 cm) SOC decreases quickly (~20 years) when cropland is established on grassland (-32 percent) or forest (-36 percent). By contrast, long lasting (> 120 years) sinks are created through conversion of cropland to forest (+16 percent) or grassland (+28 percent). Afforestation of grassland did not result in significant long term SOC stock trends in mineral soils, but did cause a net carbon accumulation in the labile forest floor (e.g. 38 Mg ha^{-1} over 100 years). However, this carbon accumulation cannot be considered as an intermediate or long-term C storage since it may be lost easily after disruptions such as fire, windthrow or clear cut (Poeplau *et al.*, 2011).

Peatlands (organic soils) store a large amount of carbon which is rapidly lost when these peatlands are drained for agriculture and commercial forestry (Hooijer *et al.*, 2010). A rapid increase in decomposition rates leads to increased emissions of CO_2, and N_2O, and vulnerability to further impacts through fire.

The FAO emissions database estimates globally there are 250 000 km^2 of drained organic soils under cropland and grassland, with total GHG emissions of 0.9 Gt CO_2eq yr^{-1}in 2010. The largest contributions are from Asia (0.44 Gt CO_2eq yr^{-1}) and Europe (0.18 Gt CO_2eq yr^{-1}; FAO, 2013). Joosten (2010) estimated that there are >500 000 km^2 of drained peatlands in the world including under forests, with CO_2 emissions having increased from 1.06 Gt CO_2 yr^{-1} in 1990 to 1.30 Gt CO_2 yr^{-1} in 2008. This is despite a decreasing trend in Annex I countries, from 0.65 to 0.49 Gt CO_2 yr^{-1}, primarily due to natural and artificial rewetting of peatlands. In Southeast Asia, CO_2 emissions from drained peatlands in 2006 were 0.61 ± 0.25 Gt CO_2 yr^{-1} (Hooijer *et al.*, 2010).

Soil drainage also affects mineral soils. Meersmans *et al.* (2009) showed that initially poorly drained valley soils in Belgium have lost significant amount of topsoil SOC (e.g. between – 2 and – 4 kg C m^{-2} for the 1960-2006 period). The cause is most probably intensified soil drainage in these environment for cultivation purposes.

A serious consequence of deforestation is extensive loss of carbon from the soil, a process regulated by microbial diversity. Crowther *et al.* (2014) assessed the effects of deforestation on soil microbial communities across multiple biomes, drawing on data from eleven regions ranging from Hawaii to Northern Alaska. The magnitude of the vegetation effect varied between sites. Deforestation dramatically altered the microbial communities in sandy soils, while the effects were minimal in clay-rich soils, even after extensive tree removal. Fine soil particles have a larger surface area to bind nutrients and water. This capacity might buffer soil microbes in clay-rich soils against the disturbance of deforestation. Sandy soils, by contrast, have larger particles with less surface area and so retain fewer nutrients and less organic matter. Microbial community changes were associated with distinct changes in the microbial catabolic profile.

Dynamic Global Vegetation Models (DGVMs) can be used to look at the combined effects of land use change, climate, CO_2, and in some cases N deposition, on vegetation and soil properties over time. In Table 4.1, Figure 4.4 and Figure 4.5 we show results from three vegetation models: *ISAM* (Jain *et al.*, 2013; El-Masri *et al.*, 2013; Barman *et al.*, 2014 a, b), *LPJ-GUESS* (Smith *et al.*, 2001; Pugh *et al.*, 2015) and *LPJmL* (Bondeau *et al.*, 2007; Schaphoff *et al.*, 2013). The ISAM model includes a nitrogen cycle, N deposition and changes in soil N. The ISAM and LPJ-GUESS models were run with the HYDE historical land use change data set (*History Database of the Global Environment*, Klein Goldewijk *et al.*, 2011). The LPJmL group combined three land use change data sets (Klein Goldewijk and Drecht, 2006; Ramankutty *et al.*, 2008; Portmann, Siebert and Döll, 2010) with the global geographic distribution of agricultural lands in the year 2000 (Fader *et al.*, 2010). The models were also run with historical climate and CO_2 (and N deposition in the case of ISAM). Figure 4.4 shows the mineral soil C and N concentration of different land cover types in different geographic ranges while Table 4.1 and Figure 4.5 show the loss of carbon due to historical land use change from 1860 to 2010.

Differences between the models are large for some systems and regions due to different landuse change data, different land cover definitions, different processes included in the models, etc. For example, soil carbon losses are higher in the LPJmL model in part due to greater land cover change in their land cover reconstructions. The highest carbon losses are associated with the conversion of forests to croplands (Figures 4.4 and 4.5). While Table 4.1 shows the global mean soil carbon loss, the effects are not the same everywhere (Figure 4.5). This may be the case, for example, when forests are converted to pastures in regions where pastures strongly favour soil C accumulation.

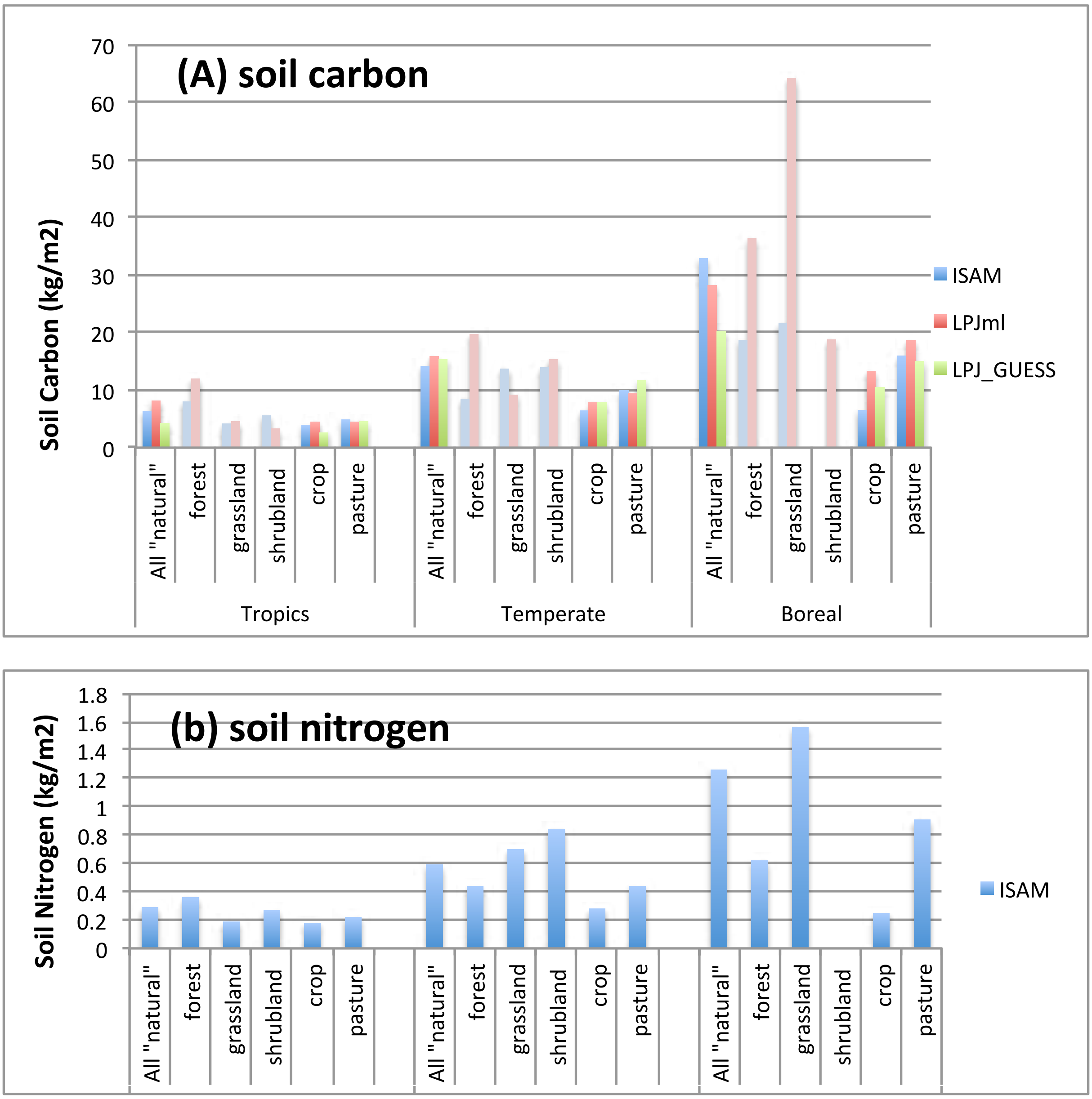
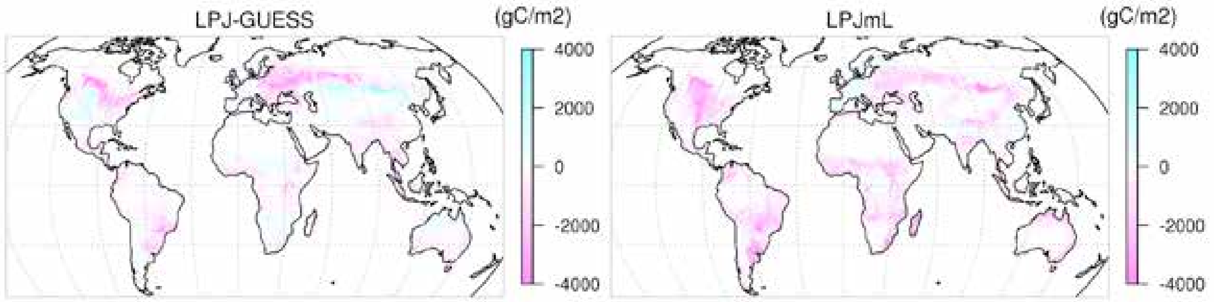


Figure 4.4 | Soil carbon and nitrogen under different land cover types.
Source: Smith et al. (in press).

Panel (a) shows mean soil carbon stocks; Panel (b) shows mean soil nitrogen stocks. Based on three vegetation models ISAM (Jain *et al.*, 2013; El-Masri *et al.*, 2013; Barman, Jain and Liang, 2014 a, b), LPJ-GUESS (Smith *et al.*, 2001; Pugh *et al.*, 2014); and LPJmL (Bondeau *et al.*, 2007; Schaphoff *et al.*, 2013). The soil carbon and soil nitrogen are the average over the period 2001 to 2010 (2003 for LPJmL) in model simulations with historical land-use change, climate, and CO_2 (and N_2 for the ISAM model). All 'natural' land is the mean of all lands without pasture or crop land cover. It includes 'un-managed' forest, grassland and shrubland categories and may include other land cover types depending on the models e.g. bare soil.



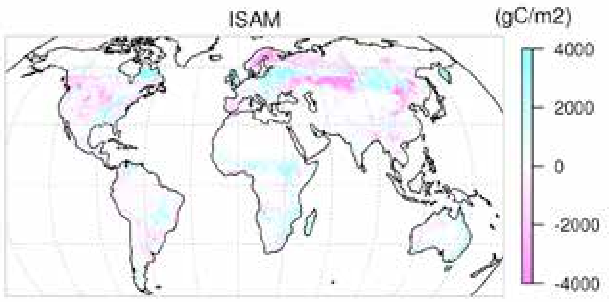


Figure 4.5 | Maps of change in soil carbon due to land use change and land management from 1860 to 2010 from three vegetation models. Pink indicates loss of soil carbon, blue indicates carbon gain. The models were run with historical land use change. This was compared to a model run with only natural vegetation cover to diagnose the difference in soil carbon due to land cover change. Both model runs included historical climate and CO_2 change.
Source: Smith et al. (in press).

Panel (a) of Figure 4.5 shows cropland and pasture coverage in 2003. The model was run with historical land use change. This was compared to a model run with only natural vegetation cover to diagnose the difference in soil carbon due to land cover change up to year 2003 as shown in Panel (b). Both model runs included historical climate and CO_2 change. Pink indicates loss of carbon due to land use, blue indicates areas of carbon gain.

Table 4.1 | Soil carbon lost globally due to land use change over the period 1860 to 2010 (PgC)

Model	Tropical	Temperate	Boreal	Global
LPJ-GUESS	12.63	15.01	0.37	29.85
LPJmL	34.86	25.99	0.05	61.86
ISAM	17.24	37.83	5.28	60.35
Mean	**21.57666667**	**26.27666667**	**1.9**	**50.68666667**

Data are from three vegetation models ISAM (Jain *et al.*, 2013; El-Masri *et al.*, 2013; Barman, Jain and Liang, 2014 a, b); LPJ-GUESS (Smith *et al.*, 2001; Pugh *et al.*, 2015); and LPJmL (Bondeau *et al.*, 2007; Schaphoff *et al.*, 2013). Each model is run with and without historical land use change data and the difference between the 'with land use change' and 'no land use change' runs gives the loss due to land use change. The runs also included historical climate and CO_2 and cover the period from 1900 to 2010.

Impacts of land management and degradation

Logging and fire are the major causes of forest degradation in the tropics (Bryan *et al.*, 2013). Logging removes nutrients. Logging operations also cause soil disturbance affecting soil physical properties and nutrient levels (soil and litter) in tropical (e.g. Olander *et al.*, 2005; Villela *et al.*, 2006; Alexander, 2012) and temperate forests (Perez *et al.*, 2009). Many physical, chemical, mineralogical, and biological soil properties can be affected by forest fires depending on fire regime (Certini, 2005). Increased frequency of fires contributes to degradation and reduces the resilience of the biomes to natural disturbances

A meta-analysis of 57 publications (Nave *et al.*, 2011) showed that fire had significant overall effects on soil C (-26 percent) and soil N (-22 percent). Fires reduced forest floor storage (pool sizes only) by an average of 59 percent (C) and 50 percent (N), but the concentrations of these two elements did not change. Prescribed fires caused smaller reductions in C and N storage (-46 percent and – 35 percent) than wildfires (-67 percent and – 69 percent). Burned forest floors recovered their C and N pools in an average of 128 and 103 years, respectively. Among mineral soil layers, there were no significant changes in C or N storage, but C and N concentrations declined significantly (-11 percent and – 12 percent, respectively). Mineral soil C and N concentrations were significantly reduced in response to wildfires but not after prescribed burning.

A large field study in the Amazon (225 forest plots) examined the effects of anthropogenic forest disturbance (selective logging, fire, and fragmentation) on soil carbon pools. Results showed that the first 30 cm of the soil pool did not differ between disturbed primary forests and undisturbed areas of forest, suggesting a resistance to impacts from selective logging and understory fires (Berenguer *et al.*, 2014). However, impacts of human disturbances on the soil carbon are of particular concern in tropical forests growing on organic soils.

Forest fires produce pyrogenic carbonaceous matter (PCM), which can contain significant amounts of fused aromatic pyrogenic C (often also called black C), some of which can be preserved in soils over centuries and even millennia. This was found to be the reason for similar soil organic C contents modelled for scenarios with and without burning in Australia: the loss in litter C input by fire was compensated by the greater persistence of the pyrogenic C (Lehmann *et al.*, 2008). Dissolved pyrogenic carbon (DPyC) from burning of the Brazilian Atlantic forest continued to be mobilized from the watershed each year in the rainy season, despite the fact that widespread forest burning ceased in 1973 (Dittmar *et al.*, 2012). Fire events are a source of carbonaceous aerosol emissions, and these are considered a major source of global warming (Kaufman, Tanre and Boucher, 2002)

Shifting cultivation practices of clearing land through fire have been used for thousands of years but in recent years increasing demographic pressure has often reduced the duration of the fallow period and so affected system sustainability. A review by Ribeiro Filho, Adams and Sereni Murrieta (2013) reported negative impact on SOC associated with the conversion stage, although impacts depended on the characteristics of the burning. Chop-and-mulch of enriched fallows appears to be a promising alternative to slash-and-burn. A study in the Amazon (Comtea *et al.*, 2012) found that this technique conserves soil bulk density and significantly increases nutrient concentrations and organic matter content compared to burnt cropland and to a control forest.

Climate change and land use dynamics are the major drivers of dryland degradation with important feedbacks through changes in plant community composition – for example shrub encroachment or decrease in vegetation cover (D'Odorico *et al.*, 2013). A review conducted by Ravi *et al.* (2010) indicated soil erosion as the most widespread form of land degradation in drylands, with wind and water erosion of dryland soils accounting for 87 percent of the land degradation. Grazing pressure, loss of vegetation cover, and the lack of adequate soil conservation practices increase the susceptibility of these soils to erosion. An analysis of 224 dryland sites highlighted a negative effect of aridity on the concentration of soil organic C and total N, but a positive effect on the concentration of inorganic P (Delgado-Baquerizo *et al.*, 2013). Because aridity is negatively related to plant cover, the authors argue that these effects might be related to the dominance in arid areas of physical processes such as rock weathering, a major source of P to ecosystems, over biological processes that provide more C and N, such as litter decomposition.

Grasslands, including rangelands, shrublands, pastureland, and cropland sown with pasture and fodder crops, covered approximately 3.5 billion ha in 2000. This represented 26 percent of the global ice-free land area and 70 percent of the agricultural area, and contained about 20 percent of the world's soil organic carbon (C) stocks. Portions of the grasslands on every continent have been degraded due to human activities – about

7.5 percent of grassland worldwide has been degraded because of overgrazing (Conant, 2012). Grassland management and grazing intensity can affect the stock of SOC. A multifactorial meta-analysis of grazer effects on SOC density (17 studies that include grazed and ungrazed plots) found a significant interaction between grazing intensity and grass type. Specifically, higher grazing intensity was associated with increased SOC in grasslands dominated by C_4 grasses (increase of SOC by 6–7 percent), but with lower SOC in grasslands dominated by C_3 grasses (decrease of SOC by an average 18 percent). Impacts of grazing were also influenced by precipitation. An increase in mean annual precipitation of 600 mm resulted in a 24 percent decrease in grazer effect size on finer textured soils, while on sandy soils the same increase in precipitation produced a 22 percent increase in grazer effect on SOC (McSherry and Ritchie, 2013).

4.3.2 | Land use intensity change

Land use intensity has increased in recent decades, largely driven by the need to feed a growing population, by shifts in dietary patterns towards more meat consumption, and by the growing production of biofuels. At the same time, fast urbanization has occupied more of the land, reducing the stock available for agricultural production. Intensification has been widely advocated because of the many negative environmental consequences of clearing natural ecosystems to expand agricultural areas.

However, intensifying management practices, such as fertilization, irrigation, tillage and increased livestock density, can have negative environmental impacts (Tilman *et al.*, 2002). Intensifying land use can potentially reduce soil fertility. Intensification can also reduce soil resilience to extreme weather under climate change, to pests and biological invasion, to environmental pollutants and to other disasters. This section provides an overview of the benefits and consequences of intensifying use of agricultural lands. The section also highlights examples of how negative consequences can be minimized.

Several factors influence the increase in land use intensity during the recent decades. On the demand side, three main factors are at play: (i) the need to meet the food, fibre, and fuel demands of a growing population; (ii) an increase in meat consumption as developing nations become wealthier and tastes change; and (iii) rising demand for crops for biofuels. On the supply side, settlements are occupying more land and so reducing the land available for agriculture.

To meet the increased demand, it is estimated that food production will need to increase by 70-100 percent by 2050 (World Bank, 2008; Royal Society of London, 2009; Keating *et al.*, 2014). Of the two pathways of increasing production—intensification and expansion—intensification is widely promoted as the more sustainable option because of the negative environmental consequences of land expansion through deforestation and conversion of wetlands to cultivation (Foley *et al.*, 2011; MA, 2005). However, the current increase in land use intensity is generally not sustainable. In order to give a clear picture of the effects of increased land use intensity, this section is organized according to the primary management practices that characterize intensification of agricultural lands (see Table 4.2 for summary).

Nutrient management

Nutrient inputs, from both natural and synthetic sources, are needed to sustain soil fertility and to supply the nutrient needs of higher yielding crop production. Intensification in recent years has led to the annual global flows of nitrogen and phosphorus now being more than double the natural levels (Matson *et al.*, 1997; Smil, 2000; Tilman, 2002). The trend is still increasing – in China, for example, N input in agriculture in the 2000s was more than double the levels of the 1980s (State Bureau of Statistics-China, 2005). Nutrient management is particularly intensive in greenhouse production systems. In some parts of Asia, for example, up to six tons of chemical nutrient and hundreds tons of organic fertilizers are applied per hectare each year in order to achieve high yielding multiple cropping of vegetables (Liu *et al.*, 2008). Between 50-60 percent of the nutrient inputs remain in the croplands after harvest (West *et al.*, 2014). When these nutrients are later mobilized, they become a major source of pollution to local, regional and coastal waters (Carpenter *et al.*, 1998). Intensive nutrient input in agriculture has been shown to be a major cause of eutrophication and algae blooming in lakes and inshore waters. In addition, over-use of nitrogen chemical fertilizers has been found in many locations globally to be a cause of acidification and accelerated decomposition of soil organic matter, leading to further soil degradation in over-fertilized soils (Ju *et al.*, 2009; Tian *et al.*, 2012).

Nutrient inputs also affect the earth's climate. Globally, approximately one percent of nitrogen additions are released to the atmosphere as nitrous oxide (N_2O), a gas which has 300 times the warming power of carbon dioxide (Klein Goldewijk and van Drecht, 2006). China, India, and the United States account for ~56 percent of all N_2O emissions from croplands, with 28 percent originating from China alone (West *et al.*, 2014).

One remedy is to increase the efficiency of nutrient use. Nutrient efficiency can be significantly increased – and N_2O emissions can be reduced – through changes in the rate, timing, placement, and type of application of nutrients, and by improving the balance amongst nutrients applied (Venterea *et al.*, 2011). In addition, if best management practices are used, agricultural soils have the potential to be carbon storage areas (Paustian *et al.*, 2004; Smith, 2004). Technological improvements are being made to the production of biochar which converts a fraction of the C present in the original material into a more persistent form through carbonisation. Biochar can then be used as a soil amendment to provide agronomic and environmental benefits (Lehmann and Joseph, 2015). In many cases, the presence of biochar has caused a reduction in N_2O emissions, especially when these originate from denitrification. However, the mechanics of the process are not yet fully understood (Cayuela *et al.*, 2013; 2014).

The effect of pesticides on soil biodiversity

The large-scale use of pesticides may have direct or indirect effects on soil biodiversity. With the intensification of agriculture, the use of pesticides has increased worldwide to approximately two million tonnes per year (herbicides 47.5 percent, insecticides 29.5 percent, fungicides 17.5 percent, other 5.5 percent by De *et al.* (2014)). Studies of the effect that pesticides have on soil biodiversity have shown contradictory results. Effects are dependent on a variety of factors including the chemical composition, the rate applied, the buffering capacity of the soil, the soil organisms in question, and the time-scale. For example, Boldt and Jacobsen (1998) tested the effects of sulfonylurea herbicides on strains of fluorescent pseudomonads cultured from agricultural field soils. They found that the herbicide Metsulfuron methyl was toxic to the majority of fluorescent pseaudomonads (77 strains) in low concentrations, while Chlorsulfuron was only toxic at high concentrations, and Thifensulfuron methyl was toxic only to a few strains, even at high concentrations.

In a review by Bünemann, Schwenke and Van Zwieten (2006) of the effects of pesticide application on soil organisms, there were no data available for 325 of 380 active constituent pesticides registered for use in Australia. The review thus effectively highlighted the huge gap in knowledge. A synthesis of the impact of herbicides on non-target organisms concluded that herbicides did not have a major effect on soil organisms (Bünemann, Schwenke and Van Zwieten, 2006) with the exception of butachlor, which was toxic to earthworms when applied at typical agricultural rates (Panda and Sahu, 2004). In addition, the application of bromoxynil herbicides caused a shift in the communities of four out of five targeted bacterial taxa even after degradation of the herbicide (Baxter and Cummings, 2008). Avoidance behaviour to phendimedipham has also been observed for collembola (Heupel, 2002) and earthworms (Amorim, Rombke and Soares, 2005).

Insecticide application, however, has a much greater effect on soil biota, including changes in microbial community composition (Pandey and Singh, 2004), lower collembolan abundance (Endlweber, Schadler and Scheu, 2005) and earthworm reproduction. Because some species of earthworm such as Eisenia Fetida can be easily bred and because they ingest large quantities of organic matter in the soil, earthworms have often been used as bioindicators of chemical toxicity in soils (Yasmin and D'Souza, 2010). A variety of studies have reported changes in earthworm reproductive rates, growth rates and weight loss when the pesticides Malathion

(Espinoza-Navarro and Bustos-Obregon, 2005), Chlorpyrifos (Zhou *et al.*, 2007; De Silva *et al.*, 2010), Benomyl (Römbke, Garcia and Scheffczyk, 2007), Carbofuran (De Silva *et al.*, 2010) were applied to soil in laboratory experiments. Non-target effects of insecticide applications may be highly dependent on the organism since field application of Chlorpyrifos did not affect the abundance of soil predatory mites (Navarro-Campos *et al.*, 2012). Fungicides have also demonstrated significant negative effects on earthworms (Eijsackers *et al.*, 2005). In particular, copper-based fungicides that are resistant to degradation have caused long-term reductions in earthworm populations (Van Zwieten *et al.*, 2004).

Although an assessment of soil food webs across Europe did not specifically focus on pesticide application, the study demonstrated that land-use intensification was related to decreased diversity of soil fauna and resulted in less diversity among functional groups. Larger soil animals showed the most sensitivity (Tsiafouli *et al.*, 2015). However, there have been no such comprehensive studies to quantify the effects of pesticides on soil organisms at multiple trophic levels across regions. Such studies need to consider also the indirect effect of pesticides, including interactions between pesticides and biotic factors. Since below-ground biodiversity is intimately linked to above-ground vegetation patterns (De Deyn and van der Putten, 2005) and vice versa (Bardgett and van der Putten, 2014), changes in plant diversity resulting from herbicide may cause indirect effects of herbicide application.

Water management

The area of irrigated croplands has doubled in the last 50 years and irrigation now accounts for 70 percent of all water diversions on the planet (Gleick, 2003). Irrigated areas account for 34 percent of crop production, yet only cover 24 percent of all cropland area (Siebert and Doll, 2010). With the increased frequency of drought under climate change, demand for agricultural water is rising in many locations. Not surprisingly, irrigation is most commonly used in more arid areas. Where a high proportion of available water is used for agriculture, this can cause water stress for both people and nature. Water efficiency can be improved through infrastructure and through better management practices. Irrigation can potentially increase soil salinity in dry regions (Ghassemi, Jakeman and Nix, 1995). Where salinization occurs, additional irrigation is needed to 'flush' the salts beyond the root zone of the crops. This additional water requirement can further exacerbate water stress.

Harvest frequency

Land use intensity can also be increased by harvesting a parcel of farmland more frequently (double cropping, triple cropping). Approximately 9 percent of crop production increases from 1961-2007 came from increases in the harvest frequency (Alexandratos and Bruinsma, 2012). As more land was double cropped, the global harvested area increased four times faster than total cropland between 2000 and 2011 (Ray and Foley, 2013). In addition, with global warming, the areas suited for double or even triple cropping are extending into subtropical and warm temperate regions (Liu *et al.*, 2013a). The factors involved in this fast rate of increase include: fewer crop failures; fewer fallow years; and an increase in multi-cropping.

Greenhouse production has allowed multiple cropping around the world. For fruit and vegetable crops, world greenhouse cultivated area reached a total area of 408 890 ha in 2013, which includes as many as five harvests in a single year. This increasing harvest frequency has reduced soil quality through soil compaction and has increased the risk of pathogen diseases. The intensive use of pesticides and herbicides in greenhouses not only affects soil quality but creates risks to human health. In some greenhouse systems, long term multiple cropping has led to soil acidification, salinization and biological deterioration, especially where large amounts of fertilizer and pesticide/herbicide have been used. In these situations, there is a need to improve management practices, using organic matter, balancing nutrient additions and adopting intermittent fallow.

Livestock density

Livestock production is projected to increase to meet the growing demand for livestock products from a rising population and from an increase in per capita consumption. The greatest increases in per capita demand are projected to be in developing and transition countries (Bouwman *et al.*, 2006). Since the 1970s, most increases in livestock production have resulted from intensification, with a shift to a greater fraction of livestock raised in industrial conditions (Bouwmann *et al.*, 2006). For example, 76-79 percent of pork and poultry production is now industrialized (Herrero *et al.*, 2013).

Industrial livestock production systems can be highly polluting. The manure from animals, the inputs for growing animal feed, and the soil loss from intensively managed areas can all be major sources of water pollution to local and downstream freshwater ecosystems. Where natural ecosystems are cleared and converted to pasture, particularly in arid and semi-arid regions, the lands are typically low potential and have a high risk of soil erosion and soil carbon/nutrient depletion (Delgado *et al.*, 1999; Seré and Steinfeld, 1996). The soils capacity for water storage and their biodiversity are also at risk. Moreover, intensified livestock production requires an increased use of veterinary medicines, sulfa-antibiotics and hormones, all of which carry risks of pollution to soil, water and the livestock products themselves, with risks to biological and human health.

Forestry harvest and wetland draining

Forests and wetlands and their soils are massive reservoirs of carbon. In fact, forest soils store approximately the same amount of carbon as the living biomass of the forest itself (FAO, 2010). Wetlands are important not only for the huge carbon pool they contain but also for their role in the hydrological cycle. However, wetlands along big river banks, lakes and estuaries have been increasingly developed for croplands/bioenergy production in recent decades, particularly in Asia. The majority of soil carbon is concentrated in peatlands within the boreal forest as well as tropical forests in Southeast Asia. Around the world, deforestation causes ~25 percent of the total loss of soil carbon (Guo and Gifford, 2002; Murty *et al.*, 2002). This loss largely stems from oxidation of the organic matter and from soil erosion. In China over the last four decades, almost 1.3 million ha of wetlands have been converted to crop production, causing the loss of about 1.5 Pg C of soil carbon (Zhang *et al.*, 2009). Deforestation continues through conversion to agriculture and through extraction of forest products. Between 2000 and 2012, there was a new loss of 1.5 million square kilometres of forests, with the most pronounced trend in the tropics (Hansen *et al.*, 2013). Soil erosion and organic matter oxidation can be reduced through selective tree harvesting rather than clear felling, and by avoiding deforestation on steep slopes. Draining and cultivating wetlands can also affect local and regional water storage.

Research needs

It will be evident from the discussion in this section that much remains to be learned. Amongst the priority research questions are the following:

1. *Sustainable intensification* – How can we get the benefits from intensification while minimizing the associated environmental and social costs?
2. *Trade-offs between soils and efficiency* – How can we manage for resilient soil and related ecosystem services while continuing to maximize efficiency? To what extent can we have both?
3. *Soil degradation and intensification* – What is the extent of degraded soils? There are currently no sound estimates. What portion of degraded soils can be attributed to un-sustainable intensification?
4. *Options and trade-offs for improved soil management* – What can we learn from management practices used in intensification areas to help restore degraded soils? Are there any options that can integrate best management practice for sustainable intensification? What are the short – and long-term trade-offs of resource use and sustainability? What are the environmental and social costs and economic benefits of land use intensification?
5. *Farming practices and soil health* – How do changes in harvest frequency and crop rotation affect soil resilience? How much change is needed to restore degraded soils?

Table 4.2 | Threats to soil resource quality and functioning under agricultural intensification

Land intensification	Sector	Distribution	Major environmental consequence	Knowledge gap
Cropping intensification	Harvest frequency	Globally	Soil quality and resilience	Ecosystem service
	Continuing monoculture	Developing and transition countries	Soil health, pesticide residue	Biological resilience
Nutrient intensification	Over fertilization	Developing countries	Soil acidification, water pollution, N_2O emission and nitrate accumulation	Rate reducing versus balancing?
Irrigation	Submerged Rice	Developing countries, Asia	Water scarcity, methane emission	Trade-offs C and water,
	Dry crops	Arid/semi-arid regions	Secondary salinization, water scarcity	Competition for water
Livestock intensification	Over grazing	Developing countries	Soil degradation, water storage, C loss	Forage versus feed crops?
	Industrial breeding	Industrialized countries	Waste, water pollution, residue of veterinary medicine and antibiotics	Safe waste treatment and recycling
Forest clearance, wetlands drainage	Deforestation. wetland shrink	Developing and transition countries	Biodiversity, natural wealth, C loss	Agro-benefit versus natural value

4.3.3 | Land use change resulting in irreversible soil change

In this section we deal with soil sealing and mining, which have been identified as two important soil degradation processes occurring around the world. The current extent and rate of growth of soil sealing and mining are significant, and create considerable risks to essential ecosystem services. These changes in land use nearly always require a trade-off between various social, economic and environmental needs.

Sealing and land take

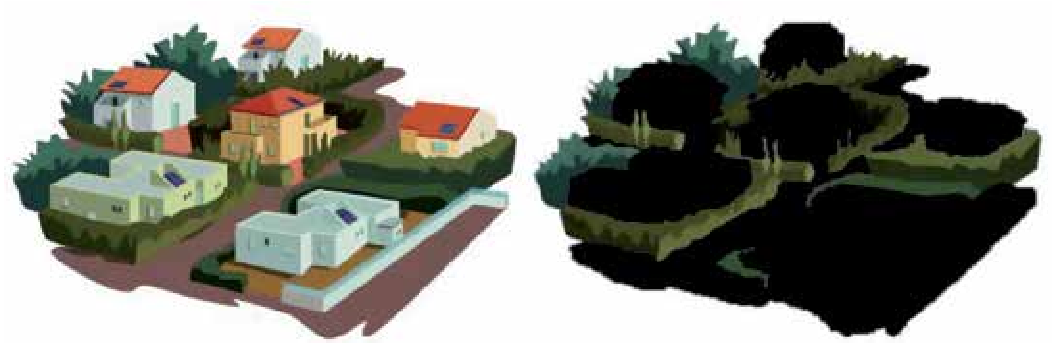
The ongoing urbanization and conversion of the landscape with settlements, infrastructure and services is occurring in many regions. Europe and Asia, in particular, are experiencing high rates of urban expansion and urban sprawl, and there are often insufficient incentives to re-use brownfield sites. These factors are causing an increase in land take and soil sealing. The drivers are essentially economic and demographic growth. In Europe, America and Oceania, at least 70 to 80 percent of the population currently lives in urban areas. The rate of urbanization is expected to continue to increase, particularly in Asia and Africa.

The concept of **land take** covers all forms of conversion for the purpose of settlement, including: the development of scattered settlements in rural areas; the expansion of urban areas around an urban nucleus; the conversion of land within an urban area (densification); and the expansion of transport infrastructure such as roads, highways and railways. Broadly, this discussion considers as land take any conversion of agricultural, natural or semi-natural land cover to an 'artificial' (e.g. human-made) area. Artificial land cover classes are categorized in the Corine Land Cover system – see Table 4.3.

A greater or smaller part of land take will result in soil sealing. **Soil sealing** means the permanent covering of an area of land and its soil by impermeable artificial material such as asphalt or concrete, for example through buildings and roads. As shown in Figure 4.6, the sealed area is only part of a settlement area. Gardens, urban parks, leisure areas and other green spaces within the boundaries of settlements are not covered by an impervious surface or are only partially covered. They thus form part of a land take but do not contribute to soil sealing (Prokop, Jobstmann and Schöbauer, 2011.) The ratio between sealed area and total area for a given land use class is measured by the soil sealing index. An example of this index, calculated for the Italian region of Emilia-Romagna, is shown in Table 4.4.

Table 4.3 | Artificial areas in Corine Land Cover Legend

Corine CODE	LABEL1	LABEL2	LABEL3
111	Artificial surfaces	Urban fabric	Continuous urban fabric
112	Artificial surfaces	Urban fabric	Discontinuous urban fabric
121	Artificial surfaces	Industrial, commercial and transport units	Industrial or commercial units
122	Artificial surfaces	Industrial, commercial and transport units	Road and rail networks and associated land
123	Artificial surfaces	Industrial, commercial and transport units	Port areas
124	Artificial surfaces	Industrial, commercial and transport units	Airports
131	Artificial surfaces	Mine, dump and construction sites	Mineral extraction sites
132	Artificial surfaces	Mine, dump and construction sites	Dump sites
133	Artificial surfaces	Mine, dump and construction sites	Construction sites
141	Artificial surfaces	Artificial, non-agricultural vegetated areas	Green urban areas
142	Artificial surfaces	Artificial, non-agricultural vegetated areas	Sport and leisure facilities



A) Typical structure of settlement B) Sealed areas about 70 percent (black color)

Figure 4.6 | Schematic diagram showing areas sealed (B) as a result of infrastructure development for a settlement (A). Source: European Union, 2012.

Table 4.4 | Artificial areas in Emilia Romagna according to the Corine Land Cover Legend and sealing index.

Classes of artificial areas	Sealing Index
1.1.1.1 Dense resdential areas	0.95
1.1.1.2 Loose residential areas	0.7
1.1.2.0 Discontinous urban areas	0.3
1.2.1.1 Industrial and agro-industrial productive districts	0.75
1.2.1.2 Commercial districts	0.75
1.2.1.3 Service and tertiary districts	0.75
1.2.1.4 Hospitals	0.75
1.2.1.5 Large technological plants	0.75
1.2.2.1 Roads and accessory areas	0.75
1.2.2.2 Railroads and accessory areas	0.25
1.2.2.3 Logistic centers	0.9
1.2.2.4 Telecomunication plants	0.25
1.2.2.5 Areas for production, transformation and transport of energy	0.75
1.2.2.6 Areas for water treatment and distribution	0.25
1.2.3.1 Commercial harbours	0.2
1.2.3.2 Touristic harbours	0.2
1.2.3.3 Fishing harbours	0.15
1.2.4.1 Commercial airports	0.3
1.2.4.2 Leisure airports	0.15
1.2.4.3 Military airports	0.2
1.3.1.1 Active mining areas	0.1
1.3.1.2 Inactive mining areas	0.1
1.3.2.1 Mining and industrial dumping sites	0.1
1.3.2.2 Urban waste disposals	0.1
1.3.2.3 Car cemetries and scra yards	0.3
1.3.3.1 Building and escavation sites	0.05
1.3.3.2 Derelict parcels	0.1
1.4.1.1 Parks and villas	0.1
1.4.1.2 Uncultivated urban areas	0.1
1.4.2.1 Campsites	0.2
1.4.2.2 Sportive areas	0.22
1.4.2.3 Leisure areas	0.3
1.4.2.4 Golf corses	0.05
1.4.2.5 Racecourses	0.05
1.4.2.6 Racetracks	0.2
1.4.2.7 Archeological areas	0.1
1.4.2.8 Bathing areas	0.15
1.4.3.0 Cemeteries	0.4

Impact of land take

Land take, by its definition, is the subtraction of an area from a previous agricultural, natural or semi-natural land use. According to this definition, the most obvious impact on the ecosystem services that can be provided by soil is on the production of biomass, and in particular of food. To clarify the concept, we may imagine that a city expands its urbanized area by a new allotment of 100 ha created at the expense of agricultural land. This area will be covered by buildings, private and public gardens, commercial centres, roads, etc. The entire area will clearly lose most of its capacity to produce food, with the possible minor exception of family horticulture in unsealed areas such as gardens or allotments. Had the entire area been previously cultivated with, say, winter wheat with an average yield of 5 tonnes ha^{-1}, the total loss in terms of food production potential will be equal to 500 tonnes of winter wheat per year.

Other ecosystem services are at risk also. Water infiltration and purification and carbon storage are mainly reduced by the effective sealed area, and not by the entire land taken. Support to biodiversity is clearly affected, although the degree depends on the different groups of organisms and also on the design of the urbanized area. In this context, a positive mitigation role can be played by 'Urban Green Infrastructure' – the incorporation of a network of high-quality green spaces and other environmental features. Green Infrastructure can include natural areas as well as human-made rural and urban elements such as urban green spaces, reforestation zones, green bridges, green roofs, eco-ducts to allow crossing of linear barriers, corridors, parks, restored floodplains, biodiverse farmland.

Regulation of land take and mitigation of its impacts

Where policy aims to minimize land take, measures can be implemented to encourage re-use of existing urban areas such as derelict areas, brownfields and upgrading of degraded neighborhoods. Measures promoting densification of existing urban areas can also contribute to the reduction of land take.

Fiscal measures can prevent speculative urban sprawl. A number of municipalities, and regional governments, especially in Europe, have already adopted policies designed to achieve zero net urban expansion. However, zero expansion becomes more problematic when there is significant demographic pressure and a high rate of rural to urban migration.

Rational and efficient urban planning and intelligent building and infrastructure design can also help reduce land take. In the past, urban planners, architects and civil engineers too often considered soil as a raw material, abundantly available and of limited value. Examples of efficient consideration of the value of soil in urban development include: the construction of parking lots in the basement of buildings; and 'green' covering of areas that are only occasionally used, such as parking lots for exhibitions and fairs etc.

Where expansion of urban and built-up areas is a policy and planning imperative, intelligent urban planning needs to take account of the soil dimension to mitigate the impact of land take. An education process is needed to make urban planners aware of the value of soil quality and land capability and of the options for mitigating negative impacts of land take.

Impacts of soil sealing

Sealing by its nature has a major effect on soil, diminishing many of its benefits. Normal construction practice is to remove the upper layer of topsoil, which delivers most of the soil-related ecosystem services, in order to be able to develop strong foundations in the subsoil or underlying rock to support the building or infrastructure. Where strong foundations are not required, only a thin layer of topsoil is generally excavated and the surfaces are simply covered by a layer of impervious material, such as asphalt or concrete. Both techniques impair or eliminate the soil's capacity to deliver ecosystem services.

The main impacts include the following.

1. Water infiltration and purification are lost, and regulation of the water cycle is completely altered. The concentration time of water flow is shortened, promoting flood events.
2. Soil biodiversity is impaired, as sealing prevents the production, release and recycling of organic material, so affecting the soil biological communities (Marfenina *et al.*, 2008). In addition, the alteration of soil water regimes, soil structure and redox potential have a strong impact on soil biodiversity.
3. Soil carbon storage potential is fundamentally altered (Jones *et al.*, 2005), particularly where topsoil, which normally contains about half of the organic carbon in mineral soils, is stripped off.
4. The urban microclimate is altered. The reduction of evapotranspiration in urban areas due to the loss of vegetation and through alteration of albedo strengthens the 'urban heat island' effect (Früh *et al.*, 2011).

Prevention of soil sealing and mitigation of its impacts

Appropriate mitigation measures can be taken in order to maintain some of the ecosystem functions of soils and to reduce negative effects on the environment and human well-being. Key options available to urban planners and managers include: (i) minimizing conversion of green areas; (ii) re-use of already built-up areas, such as brownfield sites; (iii) using permeable cover materials instead of concrete or asphalt; (iv) supporting Green Infrastructure (see above); and (v) providing incentives to developers to minimize soil sealing.

In practice, planners need to be able to evaluate the tradeoffs and ensure that policy instruments are used to ensure optimal outcomes which consider both human needs for urbanization and the preservation of the integrity of the soil and its services:

1. Existing policies for development of settlements and infrastructure should be reviewed and adapted to take account of the value of soils, particularly where subsidies or other incentives are driving unplanned land take and soil sealing (Prokop, Jobstmann and Schöbauer, 2011).
2. An integrated approach to urban planning should be followed. Existing best practice has demonstrated that soil sealing can be limited, mitigated and compensated. This requires that spatial planning follow an integrated approach and involve the full commitment of all relevant public authorities and governance entities responsible for land management, such as municipalities, counties and regions (Siebielec *et al.*, 2010).
3. Specific regional and local approaches can be developed. These could, for example, take into account unused resources at the local level such as a particularly large number of empty buildings or brownfield sites.

Mining

Ancient mining

Mining is the extraction from the Earth of rocks, valuable minerals, and other geological materials of economic interest. It is one of the most ancient activities in human history (Mighall *et al.*, 2002; Shotyk *et al.*, 1998). Mining for specific materials such as quartz, silex and clays began as far back as the Palaeolithic – the Old Stone Age – when the first stone tools were developed. In the Neolithic era – the New Stone Age – flint mines existed in Belgium, Britain and elsewhere. Landscape records and evidence from bogs show that mining activities became more intense with the development of metal tools in the Bronze Age, and subsequently in the Iron Age (Martínez-Cortizas *et al.*, 2002; Shotyk *et al.*, 1998). Examples of the environmental impact of ancient mining are numerous (Figure 4.7) (López – Merino *et al.*, 2010; Grattan, Huxley and Pyatt, 2003; Fernández Caliani, 2008).

Figure 4.7 | (A) Panoramic view of Las Medulas opencast gold mine (NW Spain). The Roman extractive technique – known as 'ruina montis' – involved the massive use of water that resulted in important geomorphological changes; (B) Weathered gossan of the Rio Tinto Cu mine, considered the birthplace of the Copper and Bronze Ages; (C) typical colour of Rio Tinto ('red river' in Spanish), one of the best known examples of formation of acid mine waters. These are inhabited by extremophile organisms.

Impact of mining

The impact of mining on the environment differs greatly depending on the type of extraction, the ore or material exploited, and the method used to process the material extracted (Moore and Luoma, 1990). Traditional underground mining, which follows profitable veins beneath the earth's surface, has less impact than open cast mining activities – also referred to as strip mining – which grew very rapidly in the last hundred years (Salomons, 1995). In some instances, entire mountains have been literally blasted apart to reach thin ore vein seams within, leaving permanent scars on the landscape. Nonetheless, mining operations themselves affect relatively small areas. By contrast, significant environmental problems are caused by tailing and waste rock deposits and by subsequent smelting operations. Pollutants can be transferred to surrounding areas by acid mine drainage or by atmospheric deposition of wind-blown dust. The incidence of these problems depends on local climatic and hydrologic conditions (Aslibekian and Moles 2003; Batista, Abreu and Serrano, 2007; López, Gónzalez and Romero, 2008). Other environmental effects, in addition to those caused by pollutants, include deforestation, erosion and formation of sinkholes (Meuser, 2010; Hester and Harrison, 2001).

Only a small fraction of the material extracted is valuable ore. The ore needs to be separated by milling and flotation from the large volume of other material discarded as tailings. When the remaining concentrate is refined by processes such as smelting, flue dust and slag are produced (Hutchinson, 1979). Atmospheric contamination has commonly occurred throughout the world during smelting operations, leading to contaminated soils and risks to livestock (Down and Stocks, 1977; Munshower, 1977). Mining for coal, gold, uranium, wolfram, tin, platinoids and, in particular, poly-metallic sulphides has created large environmental impacts on soil, water and biota. Sulphide minerals include iron sulphides such as pyrite and pyrrhothite, and other poly-metallic sulphides, such as those containing Cu, Pb, Zn, Hg, Cd, Tl, Sb, Bi etc. These sulphides can also in some instances combine with arsenides or selenides to form sulfoarsenides or sulfoselenides (Evangelou, 1995; Abreu *et al.*, 2010).

Sulphide minerals oxidise when brought to surface conditions (Nordstrom and Southam, 1997; Nordstrom and Alpers, 1999). The sulphide oxidation can cause extreme changes in Eh and pH (Figure 4.8) – negative pH values (as low as -3.6) have been measured in the acid mine waters of the Richmond mine in California (Nordstrom and Alpers, 1999). Depending on the local geochemical and hydrological conditions, sulphide oxidation can also affect the electrical conductivity of the system and may lead to elevated concentrations of many toxic elements in soils and waters nearby. Waters downstream of these mine systems (Figure 4.7C) are frequently hyperacid, hyperoxidant and hyperconductive. These waters may exhibit high activities of: (i) various metal species such as Al^{+3}, $Al\text{-}SO_4$); (ii) heavy metal species, for example Cu^{+2}, Cd^{+2}, Zn^{+2}, Hg^{+2} y Hg^{o}; and (iii) metalloids, including arseniates, arsenites and seleniates (Sengupta, 1993; Macías, 1996; Monterroso, Alvarez and Macías, 1994; Monterroso *et al.*, 1998, 1999; Azcue, 1999). Smelting operations of sulphide minerals also generate SO_2, which, if not recovered, is released into the atmosphere and thus contributes to acid deposition (described in Section 4.4).

The mining of gold deserves special attention given its contribution to Hg emissions (Drude de Lacerda, 2003). Mercury is used to concentrate the fine gold particles through amalgamation and then the gold is separated from the amalgam by applying heat. When this process is carried out under uncontrolled conditions – as in small-scale gold mining (Drude de Lacerda, 2003) – Hg volatilises to the atmosphere. Tailings from Hg amalgamation are then leached with cyanide, and waste contaminated with metals and cyanide is released into the environment (Veiga *et al.*, 2009). Arsenic exposure has also been recorded in many gold and base metal producing countries (Williams, 2001). However, arsenate and arsenite mobilisation can be controlled with soil colloidal compounds such as reactive Fe and Al (Goldberg, 2002).

As materials from mining are exposed to the environmental conditions of the Earth's surface, these minesoils develop through weathering (Sencindiver and Ammons, 2000). However, their properties differ considerably from the original soil. They contain a high percentage of rock fragments, a low nutrient content, and elevated levels of potentially harmful trace elements. They also usually lack a distinct horizonation. These soils are in fact very young soils characterised by properties that limit their functions and their capability to support vegetation (Macias, 1996; Vega *et al.*, 2004; Abreu and Magalhães, 2009). When the overburden contains sulphidic material such as pyritic mine waste, the major weathering process is the oxidative dissolution of pyrite. Here the rate of soil formation is mainly controlled by the sulphide content and its particle-size distribution, causing strongly acidic conditions, as described above (Neel *et al.*, 2003; Haering, Daniels and Galbraith, 2004). Quite often, restoration of mine soils requires the addition of exogenous material to correct the extreme pH, Eh and/or EC values and the anomalous concentrations of toxic elements common in these systems which are generally bioavailable and susceptible to mobilisation.

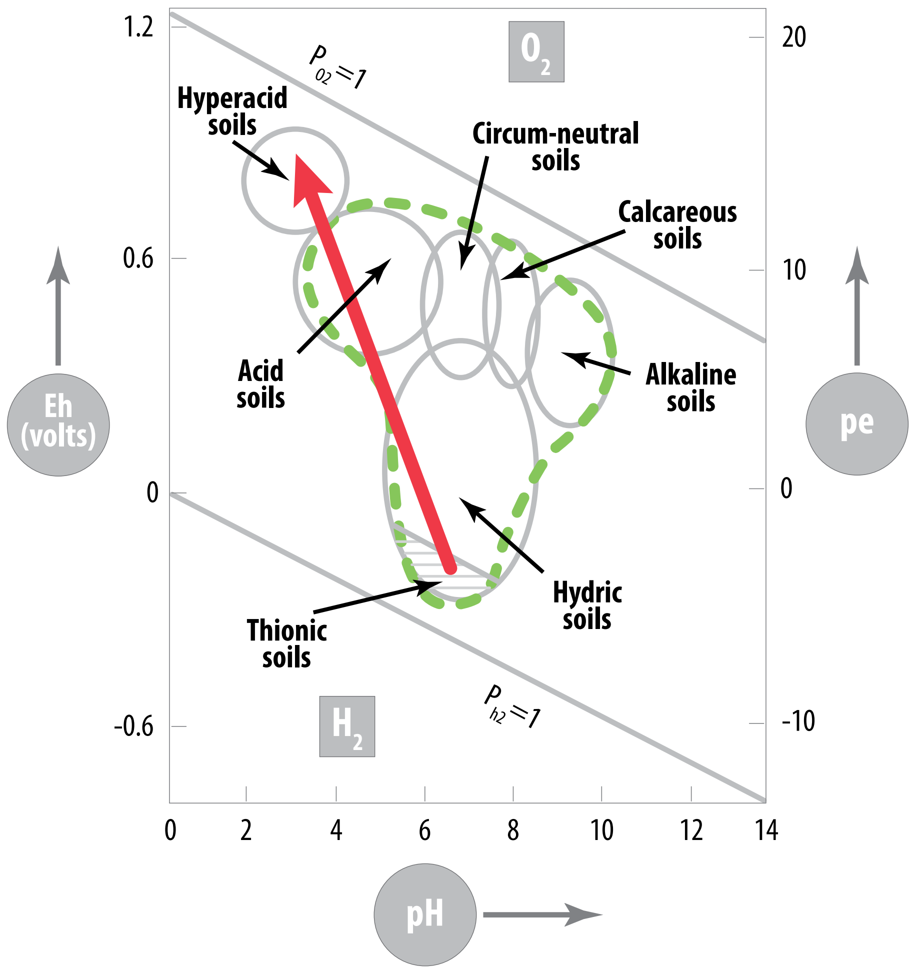


Figure 4.8 | Eh-pH conditions of thionic/sulfidic soils and of hyperacid soils.
Source: Otero et al., 2008.

The formation of sulfidic material requires strongly reducing conditions and slight acidity. Once these are oxidised, and in the absence of minerals with high acid buffering capacity, extremely acid and oxidising conditions are generated. The dashed envelope in Figure 4.8 is the approximate extent of redox-pH conditions of mineral soils (with the exception of hyper-acid soils).

Preventing impacts from mining

The rehabilitation of abandoned mines is a difficult and costly task. In fact, in many instances, the landscape cannot be repaired. Some mining methods may have significant environmental and public health effects. The Aznalcollar pyritic sludge spill (SW Spain) (López-Pamo *et al.*, 1999; Grimalt, Ferrer and McPherson, 1999; Aguilar *et al.*, 2004; Calvo de Anta and Macias, 2009) is such an example. It occurred in 1998 in the surroundings of Doñana Park – the largest reserve of bird species in Europe – as a result of the failure of a tailings dam which contained several million tons of pyrite stockpile, flotation tailings and acid waters. The toxic spill contaminated ca. 26 km^2 of riverbanks and adjacent farmlands, extending 45 km downstream, with an estimated quantity of 16 000 tonnes of Zn and Pb, 10 000 tonnes of As, 4 000 tonnes of Cu, 1 000 tonnes of Sb, 120 tonnes of Co, 100 tonnes of Tl and Bi, 50 tonnes of Cd and Ag, 30 tonnes of Hg, and 20 tonnes of Se.

Mining operations have a responsibility to protect the environment: air, water, soils, ecosystems and landscape. Many countries require reclamation plans for mining sites to follow environmental and rehabilitation codes. Nonetheless, mine restoration is still problematic, mainly because the environmental impacts were only recently understood or appreciated (Azcue, 1999; Sengupta, 1993). In addition, the technology available has not always been adequate to prevent or control environmental damage. Restoration of such systems requires a thorough understanding of material properties and their geochemistry. Only through such an understanding can the current and future behaviour of such systems be predicted and appropriate decisions taken to ensure their restoration (Gil *et al.*, 1990; Macías-García, Camps Arbestain and Macías, 2009; Macías-García *et al.*, 2009).

Development of tailor-made Technosols to restore mine soils

Technosols are defined by the FAO (2014b) as those soils with recent human activities in industrial and urban environments which have resulted in the presence of artificial and human-made objects. Technosols often result from the abandonment of urban, mining or industrial waste. These soils tend to have a large content of artefacts – that is objects that are either human-made, strongly transformed by human activity, or excavated (e.g. mine spoils, rubbles, cinders) (FAO, 2014b).

Throughout history, humans have formed soils – 'anthropogenic soils' - and in certain cases these soils have proved more fertile that natural soils nearby (Sombroek, Nachtergaele and Hebel, 1993). Thus, it is feasible to produce specific Technosols which can fulfil the environmental and productive functions of natural soils – essentially, tailor-made Technosols. This may require the formulation and mixing of artefacts and other waste materials such as manure and biosolids. The production of these Technosols could be a feasible technique through which waste products are reused and the elements they contain are returned to their biogeochemical cycles, while restoring degraded areas and contributing to the sequestration of C in soils and biomass (Macias and Camps Arbestain, 2010).

Environmental problems associated with this use of Technosols may be prevented if: (i) the characteristics of the materials used provide the soil with adequate buffering properties against contaminants, pH and/or redox changes; and (ii) there is a good understanding of how the constituent mixtures will evolve over time under the pedoclimatic conditions of the area to be restored. Figure 4.9 illustrates the benefits of the use of tailor-made Technosols in the restoration of an abandoned Cu mine rich in pyrite (Macías-García, Camps Arbestain and Macías, 2009; Macías-García *et al.*, 2009; Macías and Camps Arbestain, 2010).

Figure 4.9 | Use of different Technosols derived from wastes in the recovery of hyperacid soils and waters in the restored mine of Touro (Galicia, NW Spain).

4.4 | Atmospheric deposition

4.4.1 | Atmospheric deposition

The impacts of the deposition of atmospheric pollutants on soils vary with respect to soil sensitivity to a specific pollutant and to the total pollutant load. Anthropogenic emissions of sulphur, nitrogen and trace elements to the atmosphere mainly derive from fossil fuel and waste combustion in, for example, power generation, incineration, industry and transport. Emissions may also derive from non-combustion processes such as agricultural fertilizers or waste amendments. Mining activities may also contribute, for example Hg mining. Once in the atmosphere, these pollutants can be transported off-site and even cross national borders before being deposited either as dry or wet deposition. Deposition is more accentuated in forests, especially in coniferous forests (because of reduced wind speeds) and in areas of high elevation because of high precipitation rates.

Once in the soil, pollutants can be mobilised by being: (i) released back to the atmosphere; (ii) made available to biota; (iii) leached out to surface waters; or (iv) transported to other areas by soil erosion. Pollutants disrupt natural biogeochemical cycles by altering soil functions. This disruption may come about through direct changes to the nutrient status, acidity, and bioavailability of toxic substances, or through indirect changes to soil biodiversity, plant uptake and litter inputs. Soil sensitivity to atmospheric pollution varies with respect to: (i) key properties influenced by geology and associated pedogenesis such as cation exchange capacity, soil base saturation, aluminium, or rate of base cation supply by mineral weathering); (ii) organic matter content and carbon to nitrogen ratio (C:N); and (iii) position of the water table. When atmospheric pollution is associated with sulphate deposition, the capacity of soils to adsorb sulphate (e.g. soils with a dominance of short-range ordered constituents) plays a key role in buffering the acidification process (Camps Arbestain, Barreal and Macías, 1999; Rodríguez-Lado, Montanarella and Macías, 2007). Harmful effects on soil function and structure occur where deposition exceeds the 'critical load' - the specific amount of one or more pollutants that a particular soil can buffer (Nilsson and Grennfelt, 1988). Estimates and mapping of critical loads of acidity are however strongly dependent on the neutralisation mechanisms considered in the analysis, for example, the inclusion or exclusion of sulphate adsorption (Rodríguez-Lado, Montanarella and Macías, 2007). Spatial differences in soil sensitivity – commonly defined by the 'critical load' – and in pollutant deposition result in an uneven global distribution of impacted soils (Figure 4.10). For instance, global emissions of sulphur and nitrogen have increased 3–10 fold since the pre-industrial period (van Aardenne *et al.*, 2001), yet critical loads for acidification are only exceeded in 7–17 percent of the global natural terrestrial ecosystems area (Bouwman *et al.*, 2002).

4.4.2 | Main atmospheric pollutants: Synopsis of current state of knowledge

Since the 1980s, emissions of pollutants, notably sulphur, across Europe and North America have declined. The decline is due to the establishment of protocols under the 1979 Convention on Long-range Transboundary Air Pollution (LRTAP) and the 1990 United States Clean Air Act Amendments (CAAA) (Greaver *et al.*, 2012; Reis *et al.*, 2012; EEA, 2014). Conversely, emissions in South and East Asia, sub-Saharan Africa and South America are likely to increase in response to industrial and agricultural development (Kuylenstierna *et al.*, 2001; Dentener *et al.*, 2006). Further emission increases are also occurring in remote areas due to mining activity, such as oil sands extraction in Canada (Kelly *et al.*, 2010; Whitfield *et al.*, 2010).

Sulphur deposition

Sulphur emissions primarily result from combustion of coal and oil and are typically associated with power generation and heavy industry. In 2001, deposition exceedances of 20 kg S ha^{-1} yr^{-1} were detected in regions of China and Republic of Korea, Western Europe and eastern North America (Vet *et al.*, 2014; Figure 4.10.(a)). Deposition in unaffected ecosystems is <1 kg S ha^{-1} yr^{-1} (Figure 4.10a). The deployment of sulphur emission protocols led to the reduction of approximately 80 percent in the deposition levels of sulphur across Europe between 1990 and 2010 (Reis *et al.*, 2012). This reduction led to an increase in the use of sulphur fertilizer to combat crop sulphur deficiencies in agricultural soils in Europe (Bender and Weigel, 2011). Sulphur emissions in China also declined between 2005 and 2010 (Fang *et al.*, 2013).

Soil acidification is a natural process that is altered and accelerated by anthropogenic sulphur and nitrogen deposition (Greaver *et al.*, 2012). Sulphur oxide (SO_x) gases react with water vapour in the atmosphere to form sulphuric acid (H_2SO_4). Once in the soil, excess inputs of acidity (H^+) displace base cations (e.g. calcium (Ca_2^+) and magnesium (Mg_2^+)) from soil surfaces to the soil solution, and the base cations are subsequently lost from the soil profile by leaching (Reuss and Johnson, 1986). In mineral soils, these base cation losses can be balanced by rock weathering or atmospheric dust deposition. Thus, the global distribution of acid sensitive soils is mainly associated with conditions that favour development of soils with low cation exchange capacity and base saturation (Bouwman *et al.*, 2002; Figure 4.10b). The exception is where soils are dominated by variable-charge constituents, as in the case of Acrisols, Ferrasols, Nitosols and Andosols. On these soils, sulphate adsorption may become the most important acid-buffering mechanism (Rodríguez-Lado, Montanarella and Macías, 2007). Wetlands can also buffer inputs of acidity through biological sulphate reduction, although acidity can be mobilised again following drought and drainage (Tipping *et al.*, 2003; Laudon *et al.*, 2004; Daniels *et al.*, 2008). Organic acids can also buffer acid deposition in naturally acidic organic soils (Krug and Frink, 1983; Monteith *et al.*, 2007).

Acidification decreases soil fertility due to loss of nutrients and increases the mobilisation of toxic metals, particularly Al and heavy metals. The negative effect of Al species on crop yield is particularly strong in soils with a dominance of 2:1 clay minerals with high CEC and low organic matter content. The atmospheric deposition of acid compounds had a huge impact on Scandinavian ecosystems over the 1960s-80s, including declines in freshwater fish populations and damage to forests (EEA, 2014). Sulphur inputs can also stimulate microbial processes that increase Hg bioavailability, leading to bioaccumulation of Hg in the food chain (Greaver *et al.*, 2012).

The increase in soil pH following the reduction of sulphur emissions shows that the acidification process is reversible, although the recovery time is highly variable and dependent on soil properties. Some areas with organic soils where deposition has declined are showing either slow or no recovery (Greaver *et al.*, 2012; Lawrence *et al.*, 2012; RoTAP, 2012). On agricultural soils, lime can be applied to increase soil pH. However, 50-80 percent of sulphur deposition on land is on natural land (Dentener *et al.*, 2006). Application of lime to naturally acidic forest soils can cause further acidification of deep soil layers by increasing the decomposition in surface litter (Lundström *et al.*, 2003). In acid waters, the addition of liming material may favour the formation of polymeric Al hydroxides (e.g. $Al_{13}OH_{27}^{+12}$), which are highly toxic to aquatic species (Monterroso, Alvarez and Macías, 1994).

Wider effects of acidification are starting to be understood through long-term monitoring. Decreased organic matter decomposition due to acidification has increased soil carbon storage in tropical forests (Lu *et al.*, 2014). In wetland soils, methane (CH_4) emissions have also been suppressed. This is because sulphate-reducing bacteria have a higher affinity for substrate (H_2 and acetate) than methanogenic microbes (Gauci *et al.*, 2004). Conversely, declining sulphur deposition has been associated with increased dissolved organic carbon fluxes from organic soils (Monteith *et al.*, 2007) and decreased soil carbon stocks in temperate forest soils (Oulehle *et al.*, 2011; Lawrence *et al.*, 2012).

Nitrogen deposition

Nitrogen deposition covers a wider geographical area than sulphur deposition. This is because the sources are more varied, including extensive agriculture fertilizer and animal waste application, biomass burning, and fossil fuel combustion (Figure 4.10c). Regions with deposition in excess of 20 kg N $ha^{-1}yr^{-1}$ in 2001 include Western Europe, South Asia (Pakistan, India, Bangladesh) and eastern China (Vet *et al.*, 2014). In addition, extensive areas with deposits of 4 kg N $ha^{-1}yr^{-1}$ or more were found across North, Central and South America and parts of Europe and Sub-Saharan Africa. By contrast, 'natural' deposition in un-impacted areas is as little as 0.5 kg N $ha^{-1}yr^{-1}$ (Dentener *et al.*, 2006). While both nitrogen and sulphur emissions related to fossil fuel combustion have declined across Europe, agricultural sources of nitrogen in the region are likely to stay constant in the near future (EEA, 2014). At the same time, overall global emissions are likely to increase (Galloway *et al.*, 2008). Nitrogen deposition in China in the 2000s was similar to peaks in Europe during the 1980s before Europe embarked on mitigation measures (Liu *et al.*, 2013b).

Deposition of nitrogen induces a 'cascade' of environmental effects, including acidification and eutrophication that can have both positive and negative effects on ecosystem services (Galloway *et al.*, 2003). Soils with low nitrogen content are most sensitive to eutrophication - typically Histosols, Cryosols and Podzols located in cold areas in northern countries such as northern Canada, Scandinavia and northern Russia (Bouwman *et al.*, 2002; Rodríguez-Lado, Montanarella and Macías, 2007; Figure 4.10d). Excluding agricultural areas where nitrogen deposition is beneficial, 11 percent of the world's natural land experiences nitrogen exceedances above 10 kg N $ha^{-1}yr^{-1}$ (Dentener *et al.*, 2006). In Europe, eutrophication has and will continue to impact a larger area than acidification (Rodríguez-Lado and Macias, 2005; EEA, 2014).

Nitrogen fertilisation can increase tree growth (Magnani *et al.*, 2007) and cause changes in plant species and diversity (Bobbink *et al.*, 2010). This can in turn alter the amount and quality of litter inputs to soils, notably the C:N ratio and soil-root interactions (RoTAP, 2012). However, increased global terrestrial carbon sink can be largely offset by increased emissions of the greenhouse gases N_2O and CH_4 (Liu and Greaver, 2009). Long-term changes caused by nitrogen deposition are uncertain as transport times vary between environmental systems. The only way to remove excess nitrogen is to convert it to an unreactive gas (Galloway *et al.*, 2008).

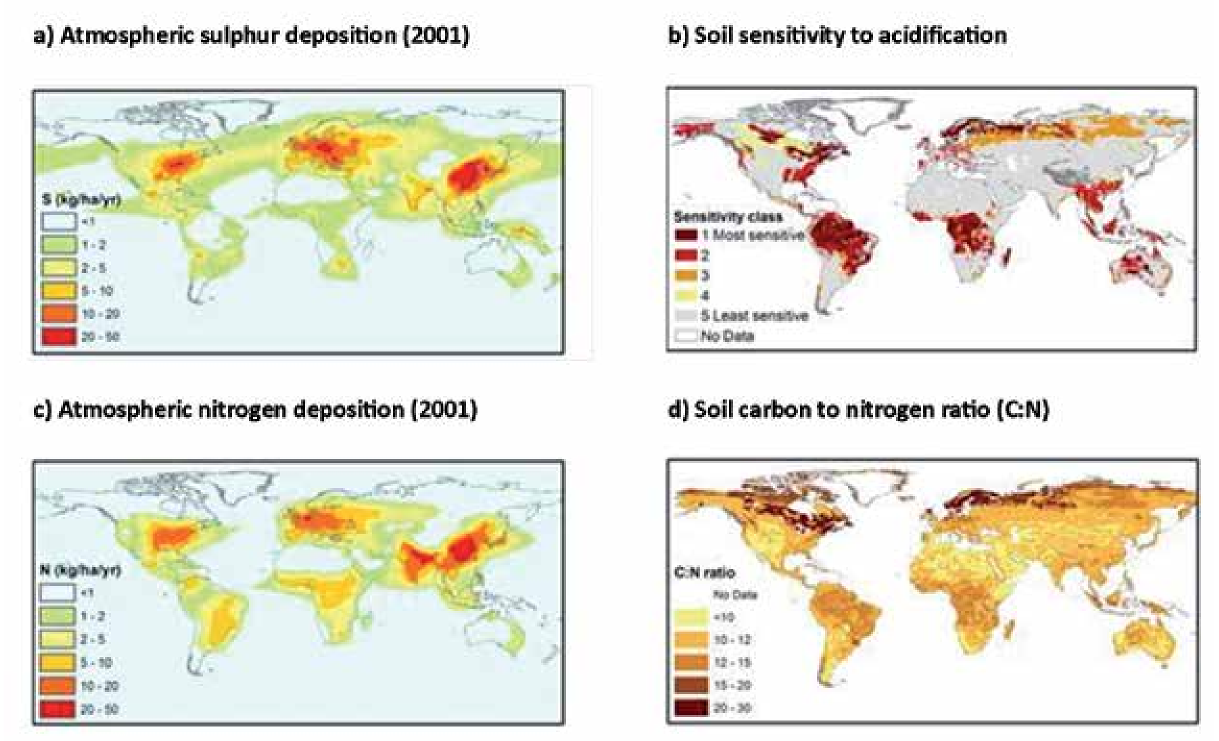


Figure 4.10 | Global distribution of (a) atmospheric S deposition, (b) soil sensitivity to acidification, (c) atmospheric N deposition, and (d) soil carbon to nitrogen ratio (soils most sensitive to eutrophication have a high C:N ratio; eutrophication is caused by N).
Source: Vet et al., 2014; Batjes, 2012; FAO, 2007.

Atmospheric deposition data in (a) and (c) were provided by the World Data Centre for Precipitation Chemistry (*http://wdcpc.org*, 2014) and are also available in Vet *et al.* (2014). Data show the ensemble-mean values from the 21 global chemical transport models used by the Task Force on Hemispheric Transport of Air Pollution (HTAP) (Dentener *et al.*, 2006). Total wet and dry deposition values are presented for sulphur, oxidized and reduced nitrogen. Soil data in (b) and (d) were produced using the ISRIC-WISE derived soil properties (ver 1.2) (Batjes, 2012) and the FAO Digital Soil Map of the World.

Trace element deposition

Global trace element emissions and deposition are poorly understood in comparison to our understanding of emissions of sulphur and nitrogen. Emissions of trace elements are associated with combustion of fossil fuel (V, Ni, Hg, Se, Sn), traffic (Pb), insecticides (As), steel manufacture (Mn, Cr), and mining and smelting (As, Cu, Zn, Hg) (Mohammed, Kapri and Goel, 2011). In the United Kingdom, trace element deposition is responsible for 25-85 percent of total trace element inputs to soils (Nicholson *et al.*, 2003). In Europe, the area at risk from Cd, Hg and Pb deposition in 2000 was 0.34 percent, 77 percent and 42 percent respectively, although emissions are declining (Hettelingh *et al.*, 2006). In China, 43-85 percent of total As, Cr, Hg, Ni and Pb inputs to agricultural soils originate from atmospheric deposition (Luo *et al.*, 2009). In bioavailable form these elements have a toxic effect on soil organisms and plants, influencing the quality and quantity of plant inputs to soils and the rate of decomposition. Significantly, they can also bioaccumulate in the food chain. Activity of trace elements in soils will depend on the specific mobility of the element and this will be influenced by pH, Eh and the concentration of dissolved organic matter with complexing properties (Blaser *et al.*, 2000). Some trace elements will persist for centuries as they are strongly bound to soil particles. However, they can become bioavailable, as observed in peatlands following drought-induced acidification, drainage and soil erosion (Tipping *et al.*, 2003; Rothwell *et al.*, 2005).

4.4.3 | Knowledge gaps and research needs

Atmospheric pollution is a global phenomenon impacting large areas of the land surface. Regional and global scale assessment relies on the use of simple models to: (i) upscale site-specific soil data, in some instances using soil databases collected as long ago as the 1970s; and (ii) estimate where soil sensitivity – the 'critical load' – of a single pollutant is exceeded. There are few locations with long-term soil monitoring data, particularly in comparison to the data available on air, rain and surface water quality. Therefore, the actual global extent and magnitude of polluted soils are unclear. Essentially, we lack data at adequate scales to check the model outputs. A long-term global soil monitoring network is needed.

While the direct impacts of sulphur, nitrogen and trace elements on inorganic soil chemical processes are generally well understood, many uncertainties still exist about pollutant impacts on biogeochemical cycling, particularly interactions between organic matter, plants and organisms in natural and semi-natural systems (Greaver *et al.*, 2012). Process understanding is dominated by research in Europe and North America (e.g. Bobbink *et al.*, 2010). Research is needed in other regions where soil properties and environmental conditions differ from the empirically studied areas in Europe and North America. Models need to be developed to examine the combined effects of air pollutants and their interactions with climate change and feedbacks on greenhouse gas balances and carbon storage (Spranger *et al.*, 2008; RoTAP, 2012). Air quality, biodiversity and climate change polices all impact on soils. A more holistic approach to protecting the environment is needed, particularly as some climate change policies (e.g. biomass burning, carbon capture and storage) have potential to impact air quality and, therefore, soil functions (Reis *et al.*, 2012; RoTAP, 2012; Aherne and Posch, 2013).

References

Abreu, M.M. & Magalhães, M.C.F. 2009. Phytostabilization of Soils in Mining Areas. Case studies from Portugal. *In* L. Aachen, P. Eichmann, eds., *Soil Remediation*. pp 297-344. New York, Nova Science Publishers, Inc.

Abreu, M.M., Batista, M. J., Magalhães, M.C.F. & Matos, J,X. 2010. Acid Mine Drainage in the Portuguese Iberian Pyrite Belt. In B.C. Robinson, ed. *Mine drainage and Related problems.* pp 71-118. New York, Nova Science Publishers, Inc. 275 pp.

Aguilar, J., Dorronsoro, C., Fernández, E., Fernández, J., García, I., Martín, F. & Simón, M. 2004. Soil pollution by a pyrite mine spill in Spain: evolution in time. *Environmental Pollution*, 132: 395-401.

Aherne, J. & Posch, M. 2013. Impacts of nitrogen and sulphur deposition on forest ecosystem services in Canada. *Current Opinion in Environmental Sustainability*, 5: 108–115.

Alexander, A.B. 2012. Soil compaction on skid trails after selective logging in moist evergreen forest of Ghana. *Agriculture and Biology Journal of North America*, 3(6): 262-264

Alexandratos, J. & Bruinsma, J. 2012. *World agriculture towards* 2030/2050: the 2012 revision. Rome, FAO.

Amorim, M.J.B., Rombke, J. & Soares, A.M.V.M. 2005. Avoidance behavior of *Enchytraeus albidus*: Effects of Benomyl, Carbendazim, Phenmedipham and different soil types. *Chemosphere* 59: 501-510.

Aslibekian, O. & Moles, R. 2003. Environmental risk assessment of metals contaminated soils at Silvermines abandoned mine site, Ireland. *Environmental Geochemistry and Health*, 25: 247-266.

Azcue, J.M. 1999. Environmental impacts of Mining Activities. Emphasis on Mitigation and Remedial Measures. Berlin, Springer. 300 pp.

Bai, Z.G., Dent, D.L., Olsson, L. & Schaepman, M.E. 2008. Global assessment of land degradation and improvement. 1. Identification by remote sensing. Report 2008/01, ISRIC – World Soil Information, Wageningen.

Bardgett, R.D. & van der Putten, W.H. 2014. Belowground biodiversity and ecosystem functioning. Nature, 515: 505-511.

Barman, R. Jain, A.K. & Liang, M. 2014b. Climate-driven uncertainties in terrestrial energy and water fluxes: a site-level to global scale analysis. *Global Change Biology*, 20(6): 1885–1900

Barman, R., Jain, A.K. & Liang, M. 2014a. Climate-driven uncertainties in terrestrial gross primary production: a site-level to global scale analysis. *Global Change Biology*, 20(5): 1394–1411

Batista, M.J., Abreu, M.M. & Serrano, M. 2007. Biogeochemistry in Neves-Corvo mining area, Iberian Pyrite belt, Portugal. *Journal of Geochemical Exploration*, 92: 159-176.

Batjes, N.H. 2012. *ISRIC-WISE derived soil properties on a 5 by 5 arc-minutes global grid (ver. 1.2)*. Wageningen, ISRIC - World Soil Information. 52 pp.

Baxter, J. & Cummings, S.P. 2008. The degradation of the herbicide bromoxynil and its impact on bacterial diversity in a top soil. J. *Appl. Microbiol*, 104: 1605-1616.

Bender, J. & Weigel, H.J. 2011. Changes in atmospheric chemistry and crop health: A review. *Agronomy for Sustainable Development*, 31: 81–89.

Berenguer, E., Ferreira, J., Gardner, T.A, Aragão, L.E.O.C., Camargo, P.B., Cerri, C.E., Durigan, M., Oliveira Jr, R.C., Vieira, I.C.G. & Barlow, J. 2014. A large-scale field assessment of carbon stocks in human-modified tropical forests. *Global Change Biology*, 20(12): 3713-3726

Blaser, P., Zimmermann, S., Luster, J. & Shotyk, W. 2000. Critical examination of trace element enrichments and depletions in soils: As, Cr, Cu, Ni, Pb, and Zn in Swiss forest soils. *The Science of the Total Environment*, 249: 257-280.

Bobbink, R., Hicks, K., Galloway, J., Spranger, T., Alkemade, R., Ashmore, M., Bustamante, M., Cinderby, S., Davidson, E., Dentener, F., Emmett, B., Erisman, J.W., Fenn, M., Gilliam, F., Nordin, A., Pardo, L. & De Vries, W. 2010. Global assessment of nitrogen deposition effects on terrestrial plant diversity: a synthesis. *Ecological Applications*, 20: 30–59.

Boldt, T.S. & Jacobsen, C.S. 1998. Different toxic effects of the sulfonylurea herbicides metsulfuron methyl, chlorsulfuron and thifensulfuron methyl on fluorescent pseudomonads isolated from an agricultural soil. *FEMS Microbiology Letter*, 161: 29-35.

Bondeau, A., Smith, P.C., Zaehle, S., Schaphoff, S., Lucht, W., Cramer, W., Gerten, D., Lotze-Campen, H., Müller, C., Reichstein, M. & Smith, B. 2007. Modelling the role of agriculture for the 20th century global terrestrial carbon balance. *Global Change Biology*, 13(3): 679–706.

Bouwman, A.F., Vuuren, D.P., van Derwent, R.G. & Posch, M. 2002. A Global Analysis of Acidification and Eutrophication of Terrestrial Ecosystems. *Water, Air, and Soil Pollution*, 141: 349–382.

Bouwman, L., van der Hoek, K., van Drecht, G. & Eickhout, B. 2006. World Livestock and Crop Production Systems, Land Use and Environment between 1970 and 2030. In F. Brouwer & B. A. McCarl, eds. *Agriculture and Climate Beyond* 2015. pp. 75–89. Netherlands, Springer. 310 pp.

Bryan, J.E., Shearman, P.L., Asner, G.P., Knapp, D.E., Aoro, G. & Lokes, B. 2013. Extreme Differences in Forest Degradation in Borneo: Comparing Practices in Sarawak, Sabah, and Brunei. PLoS ONE, 8(7): e 69679.

Bünemann, E.K., Schwenke, G.D. & Van Zwieten, L. 2006. Impact of agricultural inputs on soil organisms: a review. *Soil Res.*, 44: 379-406.

Calvo de Anta, R. & Macias, F. 2009. Remediation of Soils contaminated with pyritic sludge from a mine spill in Aznalcóllar, Spain. In A. Faz, A.R. Mermut, J. M. Arocena & R. Ortiz, eds. *Land Degradation and Rehabilitation. Dryland Ecosystems*. pp. 295-310. Advances in Geoecology 40. Catena Verlag. 440 pp.

Camps Arbestain, M., Barreal, M.E. & Macías, F. 1999. Relating sulphate sorption in forest soils to lithological classes, as defined to calculate critical loads of acidity. *The Science of the Total Environment*, 241: 181–195.

Carpenter, S.R., Caraco, N.F., Correll, D.L., Howarth, R.W., Sharpley, A.N. & Smith, V.H. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological Applications*, 8(3): 559–568.

Cayuela, M.L., Sánchez-Monedero, M.A., Roig, A., Hanley, K., Enders, A. & Lehmann, J. 2013. Biochar and denitrification in soils: when, how much and why does biochar reduce N_2O emissions? Sci. Rep., 3: 1732.

Cayuela, M.L., van Zwieten, L., Singh, B.P., Jeffery, S., Roig, A. & Sánchez-Monedero, M.A. 2014. Biochar's role in mitigating soil nitrous oxide emissions: a review and meta-analysis. *Agriculture, Ecosystems and Environment*, 191: 5-16.

Certini, G. 2005. Effects of fire on properties of forest soils: a review. *Oecologia*, 143: 1–10.

Comtea, I., Davidson, R., Lucotte, M., Carvalho, C.J.R., Oliveira, F.A., Silva, B.P. & Rousseaug, G. 2012. Physicochemical properties of soils in the Brazilian Amazon following fire-free land preparation and slash-and-burn practices. *Agriculture, Ecosystems and Environment*, 156: 108–115.

Conant, R.T. 2012. Grassland soil organic carbon stocks: status, opportunities, vulnerability. In R. Lal, K. Lorenz, R.F. Hüttl, B.U. Schneider & J. von Braun, eds. *Recarbonization of the Biosphere*. pp. 275-302. Dordrecht, Springer. 465 pp.

Crowther, T.W., Maynard, D.S., Leff, J.W., Oldfield, E.E., McCulley, R.L., Fierer, N. & Bradford, M.A. 2014. Predicting the responsiveness of soil biodiversity to deforestation: a cross-biome study. *Global Change Biology*, 20: 2983–2994.

D'Odorico, P., Bhattachan, A., Davis, K.F., Ravi, S. & Runyan, C.W. 2013. Global desertification: drivers and feedbacks. *Advances in Water Resources*, 51: 326-344.

Daniels, S.M., Evans, M.G., Agnew, C.T. & Allott, T.E.H. 2008. Sulphur leaching from headwater catchments in an eroded peatland, South Pennines, U.K. *The Science of the total environment*, 407: 481–496.

De Deyn, G.B. & van der Putten, W.H. 2005. Linking aboveground and belowground diversity. *Trends in Ecology & Evolution*, 20: 625-633.

De Klein, C., Novoa, R., Ogle, S., Smith, K., Rochette, P., Wirth, T., McConkey, B.G., Walsh, M., Mosier, A., Rypdal, K. & Williams, S. 2006. N_2O emissions from managed soils, and CO_2 emissions from lime and urea application. In H.S. Eggleston, L. Buendia, K. Miwa, T. Ngara & K. Tanabe, eds. IPCC *Guidelines for National Greenhouse Gas Inventories*. Vol. 4: Agriculture, Forestry and Other Land Use. Chapter 11. pp. 1-54. Japan, IGES.

De Silva, P.M., Pathiratne, A., van Straalen, N.M. & van Gestel, C.A. 2010. Chlorpyrifos causes decreased organic matter decomposition by suppressing earthworm and termite communities in tropical soil. *Environ. Pollut*, 158: 3041-3047.

De, A., Bose, R., Kumar, A. & Mozumdar, S. 2014. *Targeted delivery of Pesticides Using Biodegradable Polymeric Nanoparticles*. SpringerBriefs in Molecular Science, pp. 5-6.

Delgado, C., Rosegrant, M., Steinfeld, H., Ehui, S. & Courbois, V. 1999. *Livestock to 2020. The next food revolution*. Food, Agriculture and the Environment Discussion paper 28, International Food Policy Research Institute, Food and Agriculture Organization of the United Nations, International Livestock Research Institute. Washington, DC., IFPRI. 83 pp.

Delgado-Baquerizo, M., Maestre, F.T., Gallardo, A., Bowker, M.A., Wallenstein, M.D., Quero, J.L., Ochoa, V., Gozalo, B., García-Gómez, M., Soliveres, S., García-Palacios, P., Berdugo, M., Valencia, E., Escolar, C., Arredondo, T., Barraza-Zepeda, C., Bran, D., Carreira, J.A., Chaieb, M., Conceição, A.A., Derak, M., Eldridge, D.J., Escudero, A., Espinosa, C.I., Gaitán, J., Gatica, M.G., Gómez-González, S., Guzman, E., Gutiérrez, J.R., Florentino, A., Hepper, E., Hernández, R.M., Huber-Sannwald, E., Jankju, M., Liu, J., Mau, R.L., Miriti, M., Monerris, J., Naseri, K., Noumi, Z., Polo, V., Prina, A., Pucheta, E., Ramírez, E., Ramírez-Collantes, D.A., Romão, R., Tighe, M., Torres, D., Torres-Díaz, C., Ungar, E.D., Val, J., Wamiti, W., Wang, D. & Zaady, E. 2013. Decoupling of soil nutrient cycles as a function of aridity in global drylands. *Nature*, 502: 672-676.

Dentener, F., Drevet, J., Lamarque, J.F., Bey, I., Eickhout, B., Fiore, A.M., Hauglustaine, D., Horowitz, L.W., Krol, M., Kulshrestha, U.C., Lawrence, M., Galy-Lacaux, C., Rast, S., Shindell, D., Stevenson, D., Van Noije, T., Atherton, C., Bell, N., Bergman, D., Butler, T., Cofala, J., Collins, B., Doherty, R., Ellingsen, K., Galloway, J., Gauss, M., Montanaro, V., Müller, J.F., Pitari, G., Rodriguez, J., Sanderson, M., Solmon, F., Strahan, S., Schultz, M., Sudo, K., Szopa, S. & Wild, O. 2006. Nitrogen and sulfur deposition on regional and global scales: A multimodel evaluation. *Global Biogeochemical Cycles*, 20.

Dittmar, T., Rezende, C.E., Manecki, M., Niggemann, J., Ovalle, A.R.C., Stubbins, A. & Bernardes, M.C. 2012. Continuous flux of dissolved black carbon from a vanished tropical forest biome. *Nature Geoscience*, 5: 618-622.

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: 1658–1670.

Down, C.G. & Stocks, J. 1977. Environmental Impacts of Mining. London, Applied Science. 371 pp.

Drude de Lacerda, L. 2003. Updating global Hg emissions from small-scale gold mining and assessing its environmental impacts. *Environmental Geology*, 43: 308-314.

EEA. 2014. *Effects of air pollution on European ecosystems*. Copenhagen, European Environment Agency. 42 pp.

Eijsackers, H., Beneke, P., Maboeta, M., Louw, J.P.E. & Reinecke, A.J. 2005. The implications of copper fungicide usage in vineyards for earthworm activity and resulting sustainable soil quality. *Ecotoxicology and Environmental Safety*, 62: 99-111.

El-Masri, B., Barman, R., Meiyappan, P., Song, Y., Liang, M. & Jain. A. 2013. Carbon dynamics in the Amazonian basin: integration of eddy covariance and ecophysiological data with a land surface model. *Agr. Forest Meteorol.*, 182-183: 156–167.

Endlweber, K., Schadler, M. & Scheu, S. 2005. Effects of foliar and soil insecticide applications on the collembolan community of an early set-aside arable field. *Applied Soil Ecology*, 31: 136–146.

Espinoza-Navarro, O. & Bustos-Obregón, E. 2005. Effect of malathion on the male reproductive organs of earthworms Eisenia foetida. *Asian Journal of Andrology*, 7: 97–101.

European Union. 2012. Guidelines on best practice to limit, mitigate or compensate soil sealing. Publication Office of the European Union, Luxemburg. 65 pp.

Evangelou, V.P. 1995. Pyrite oxidation and its control. *CRC Press*. 293 pp.

Fader, M.M, Rost, S., Müller, C., Bondeau, A. & Gerten, D. 2010 Virtual water content of temperate cereals and maize: Present and potential future patterns. *Journal of Hydrology*, 384(3-4): 218–231.

Fang, Y., Wang, X., Zhu, F., Wu, Z., Li, J., Zhong, L., Chen, D. & Yoh, M. 2013. Three-decade changes in chemical composition of precipitation in Guangzhou city, southern China: has precipitation recovered from acidification following sulphur dioxide emission control? *Tellus B*, 65..

FAO. 2007. Digital Soil Map of the World (DSMW). Rome, FAO. (Also available at *http://www.fao.org/geonetwork/srv/en/metadata.show?id=14116*).

FAO, 2014b. *World Reference Base for Soil Resources. World Soil Resources Reports* # 106. FAO, Rome..

FAO. 2010. *Global Forest Resources Assessment* 2010. FAO Forestry Paper #163. Rome, FAO..

FAO. 2011. *The state of the world's land and water resources for food and agriculture* (SOLAW) – Managing systems at risk. Rome, FAO & London, Earthscan..

FAO. 2013. *FAO statistical yearbook 2013. World Food and Agriculture.* Rome, FAO..

FAO. 2014a. *The State of the World's Forests 2014*. Enhancing the socioeconomic benefits from forests. Rome, FAO..

FAO/UNEP. 1999. *Terminology for Integrated Resources Planning and Management*. United Nations Environmental Programme. Rome, FAO & Kenya, Nairobi.

Fernández Caliani, J.C. 2008. Una aproximación al conocimiento del impacto ambiental de la minería en la faja pirítica ibérica. *MACLA*, 10: 24-28..

Foley, J. A., Ramankutty, N., Brauman, K. A., Cassidy, E. S., Gerber, J. S., Johnston, M. & O'Connell, C. 2011. Solutions for a cultivated planet. *Nature*, 478(7369): 337–342..

Früh, B., Becker, P., Deutschlnder, T., Hessel, J.D., Kossmann, M., Mieskes, I., Namyslo, J., Roos, M., Sievers, U., Steigerwald, T., Turau, H. & Wienert, U. 2011. Estimation of climate change impacts on the urban heat load using an urban climate model and regional climate projections. *J. Appl. Meteorol. Climatol.*, 50: 167-184..

Galloway, J.N., Aber, J.D., Erisman, J.W., Seitzinger, S.P., Howarth, R.W., Cowling, E.B. & Cosby, B.J. 2003. The Nitrogen Cascade. *BioScience*, 53: 341..

Galloway, J.N., Townsend, A.R., Erisman, J.W., Betunda, M., Cai, Z., Freney, J.R., Martinelli, L.A., Seitzinger, S.P. & Sutton, M.A. 2008. Transformation of the Nitrogen Cycle: Recent Trends, Questions, and Potential Solutions. *Science*, 320(5878): 889-892..

Gauci, V., Matthews, E., Dise, N., Walter, B., Koch, D., Granberg, G. & Vile, M. 2004. Sulfur pollution suppression of the wetland methane source in the 20th and 21st centuries. *Proceedings of the National Academy of Sciences of the United States of America*, 101: 12583–12587..

Ghassemi, F., Jakeman A.J. & Nix, H.A. 1995. *Salinisation of land and water resources: Human causes, extent, management and case studies*. Australia, Canberra, Centre for Resource and Environmental Studies. 562 pp..

Gil, A., Macias, F., Monterroso, M.C. & Val, C. 1990. Influence of waste selection in the dump reclamation at Puentes mine. *In* A.K.M. *Rainbow*, ed. *Reclamation, treatment and utilization of Coal Mining Wastes*. pp. 203-208. Rotterdam, Balkema. 527 pp..

Gleick, P.H. 2003. WATER USE. Annual Review of Environment and Resources, 28: 275-314...

Goldberg, S. 2002. Competitive adsorption of arsenate and arsenite on oxides and clay minerals. *Soil Science Society of America Journal*, 66: 413-421..

Grattan, J.P., Huxley, S.I. & Pyatt, F.B. 2003. Modern Bedouin exposures to copper contamination: an imperial legacy? *Ecotoxicology and Environmental Safety*, 55: 108-115.

Greaver, T.L., Sullivan, T.J., Herrick, J.D., Barber, M.C., Baron, J.S., Cosby, B.J., Deerhake, M.E., Dennis, R.L., Dubois, J.-J.B., Goodale, C.L., Herlihy, A.T., Lawrence, G.B., Liu, L., Lynch, J. & Novak, K.J. 2012. Ecological effects of nitrogen and sulfur air pollution in the US: what do we know? *Frontiers in Ecology and the Environment*, 10: 365–372..

Grimalt, J.O., Ferrer, M. & McPherson, E. 1999. The mine tailing accident in Aznalcollar. *The Science of the Total Environment*, 242: 3-11..

Guo, L. B. & Gifford, R. M. 2002. Soil carbon stocks and land use change: a meta analysis. *Global Change Biology*, 8(4): 345–360..

Haering, K.C., Daniels, W.L. & Galbraith, J.M. 2004. Appalachian mine soil morphology and properties: effects of weathering and mining method. *Soil Science Society of America Journal*, 68: 1315-1325...

Hansen, M.C., Potapov, P.V., Moore, R., Hancher, M., Turubanova, S.A., Tyukavina, A., Thau, D., Stehman, S.V., Goetz, S.J., Loveland, T.R., Kommareddy, A., Egorov, A., Chini, L., Justice, C.O. & Townshend, J.R.G. 2013. High-Resolution Global Maps of 21st-Century Forest Cover Change. *Science*, 342(6160): 850–853...

Herrero, M., Havlík, P., Hugo Valin, H., Notenbaert, A., Rufino, M.C., Thornton, P.K., Blümmel, M., Weiss, F., Grace, D. & Obersteiner, M. 2013. Biomass use, production, feed efficiencies, and greenhouse gas emissions from global livestock systems. *Proceedings of the National Academy of Sciences*, 110: 20888–20893..

Hester, R.E. & Harrison, R.M. 2001. *Assessment and Contamination of Contaminated Land*. Royal Society of Chemistry. 164 pp.

Hettelingh, J.P., Sliggers, J., van het Bolcher, M., Denier van der Gon, H., Groenenberg, B.J., Ilyin, I., Reinds, G.J., Slootweg, J., Tranvnikov, O., Visschedijk, A. & de Vries, W. 2006. *Heavy Metal Emissions, Depositions, Critical Loads and Exceedances in Europe.* Den Haag, Netherlands..

Heupel, K. 2002. Avoidance response of different collembolan species to Betanal. Eur. J. *Soil Biol.*, 38: 343-346..

Hooijer A., Page, S., Canadell, J.G., Silvius, M., Kwadijk, J., Wosten, H. & Jauhiainen, J. 2010. Current and future CO_2 emissions from drained peatlands in Southeast Asia. *Biogeosciences*, 7: 1505–1514.

Hooke, R.L., Martín-Duque, J.F. & Pedraza, J. 2012. Land transformation by humans: a review. *GSA Today*, 22(12): 4-10..

Hutchinson, T.C. 1979. Copper contamination of ecosystems caused by smeller activities. In J.O. Nriagu, ed. *Copper in the Environment.* New York, John Wiley and Sons. 451 pp.

Jain, A.K., Meiyappan, P., Song, Y., House, J. 2013. CO_2 emissions from land-use change affected more by nitrogen cycle, than by the choice of land-cover data. *Global Change Biology*, 19: 2893–2906..

Jones, C.D., McConnell, C., Coleman, K., Cox, P., Falloon, P., Jenkinson, D. & Powlson, D. 2005. Global climate change and soil carbon stocks; predictions from two contrasting models for the turnover of organic carbon in soil. *Global Change Biology*, 11: 154-166..

Joosten, H. 2010. The Global Peatland CO_2 Picture. Peatland status and drainage related emissions in all countries of the world. Ede, Wetlands International..

Ju, X., Xing, G., Chen, X., Zhang, S., Zhang, L., Liu, X., Zhen, C., Yin, B., Christie, P., Zhu, Z. & Zhang, F. 2009. Reducing environmental risk by improving N management in intensive Chinese agricultural systems. Proceedings of the National Academy of Sciences, 106(9): 3041–3046.

Kaufman, Y.J., Tanre, D. & Boucher, O. 2002. A satellite view of aerosols in the climate system. Nature, 419: 215-223..

Keating, B.A., Herrero, M., Carberry, P.S., Gardner, J. & Cole, M.B. 2014. Food wedges: Framing the global food demand and supply challenge towards 2050. *Global Food Security*, 3: 125-132..

Kelly, E.N., Schindler, D.W., Hodson, P. V, Short, J.W., Radmanovich, R. & Nielsen, C.C. 2010. Oil sands development contributes elements toxic at low concentrations to the Athabasca River and its tributaries. *Proceedings of the National Academy of Sciences*, 106(52): 22346-22351..

Kelly, E.N., Schindler**, D.W.,** Hodson**, P.V.,** Short**, J.W.,** Radmanovich**, R. &** Nielsen**, C.C.** 2010. Oil sands development contributes elements toxic at low concentrations to the Athabasca River and its tributaries. *Proc. Natl. Acad. Sci.* U.S.A., 107(37): 16178–16183..

Klein Goldewijk, K. & van Drecht, G., 2006. HYDE 3. Current and historical population and land cover. In A.F. Bouwman, T. Kram, K. Klein Goldewijk, eds. *Integrated Modelling of Global Environmental Change. An overview of IMAGE* 2.4. pp. 93-112. The Netherlands, Bilthoven, Netherlands Environmental Assessment Agency (MNP)..

Klein Goldewijk, K., Beusen, A., Van Drecht, G. & De Vos, M. 2011. The HYDE 3.1 spatially explicit database of human-induced global land-use change over the past 12,000 years. *Glob. Ecol. Biogeogr*,20(1): 73–86..

Krug, E.C. & Frink, C.R. 1983. Acid Rain on Acid Soil: A New Perspective. *Science*, 221: 520–525..

Kuylenstierna, J.C., Rodhe, H., Cinderby, S. & Hicks, K. 2001. Acidification in developing countries: ecosystem sensitivity and the critical load approach on a global scale. *Ambio*, 30: 20–8.

Latham, J. Cumani, R., Rosati, I. & Bloise, M. 2014. *Global Land Cover SHARE (GLC-SHARE) database Beta-Release Version 1.0. Rome*, FAO.

Laudon, H., Dillon, P.J., Eimers, M.C., Semkin, R.G. & Jeffries, D.S. 2004. Climate-induced episodic acidification of streams in central ontario. *Environmental science & technology*, 38: 6009–60015..

Lawrence, G.B., Shortle, W.C., David, M.B., Smith, K.T., Warby, R.A.F. & Lapenis, A.G. 2012. Early Indications of Soil Recovery from Acidic Deposition in U.S. Red Spruce Forests. *Soil Science Society of America Journal*, 76: 1407.

Lehmann, J. & Joseph, S. 2015. *Biochar for Environmental Management*. Science and Technology, 2nd edition. New York, Routledge.

Lehmann, J., Skjemstad, J.O., Sohi, S., Carter, J., Barson, M., Falloon, P., Coleman, K., Woodbury, P. & Krull, E. 2008 Australian climate-carbon cycle feedback reduced by soil black carbon. *Nature Geoscience*, 1: 832–835..

Liu, L. & Greaver, T.L. 2009. A review of nitrogen enrichment effects on three biogenic GHGs: the CO_2 sink may be largely offset by stimulated N_2O and CH_4 emission. *Ecology letters*, 12: 1103–1117..

Liu, L., Xu, X., Zhuang, D., Chen, X. & Li, S. 2013a. Changes in the Potential Multiple Cropping System in Response to Climate Change in China from 1960–2010. PLoS ONE, 8(12): e 80990.

Liu, X., Zhang, Y., Han, W., Tang, A., Shen, J., Cui, Z., Vitousek, P., Erisman, J.W., Goulding, K., Christie, P., Fangmeier, A. & Zhang, F. 2013b. Enhanced nitrogen deposition over China. *Nature*, 494: 459–462

Liu, Z.H., Jiang, l.H., Zhang, W.J., Zheng, F.L., Wang, M. & Lin, H.T. 2008. Evolution of fertilization rate and variation of soil nutrient contents in greenhouse vegetable cultivation in Shandong. *Pedologica Sinica*, 45(2): 296-303. [in Chinese with English abstract]

López, M., Gónzalez, I. & Romero, A. 2008. Trace elements contamination of agricultural soils affected by sulphide exploitation (Iberian Pyrite Belt, SW Spain). *Environmental Geology*, 54: 805-818.

López-Merino, L., Pena-Chocarro, L., Ruíz-Alonso, M., López-Saez, J.A. & Sánchez-Palencia, F.J. 2010. Beyond nature: the management of a productive cultural landscape in Las Medulas area (El Bierzo, Leon, Spain) during pre-Roman and Roman times. *Plant Biosystems*, 144: 909-923.

López-Pamo, E., Barettino, D., Anton-Pacheco, C., Ortiz, G., Arranz, J.C., Gumiel, J.C., Martínez-Pledel, B., Aparicio, M. & Montouto, O. 1999. The extent of the Aznalcollar pyritic sludge spill and its effects on soils. *The Science of the Total Environment*, 242: 57-88.

Lu, X., Mao, Q., Gilliam, F.S., Luo, Y. & Mo, J. 2014. Nitrogen deposition contributes to soil acidification in tropical ecosystems. *Global change biology*, 20(12): 3790–3801

Lundström, U.S., Bain, D.C., Taylor, A.F.S. & van Hees, P.A.W. 2003. Effects of Acidification and its Mitigation with Lime and Wood Ash on Forest Soil Processes: A Review. *Water, Air and Soil Pollution*: Focus, 3: 5–28

Luo, L., Ma, Y., Zhang, S., We[i], D. & Zhu, Y.-G. 2009. An inventory of trace element input to agricultural soils in China. *Journal of Environmental Management*, 90: 2524-2530

MA. 2005. *Current State and Trends. Millennium Ecosystem Assessment*. Washington, D.C., Island Press. 839 pp.

Macías, F. & Camps Arbestain, M. 2010. Soil carbon sequestration in a changing global environment. *Mitig. Adapt. Strateg. Glob. Change*, 15(6): 511-529.

Macías, F. 1996. Los suelos de mina: Su recuperación. In J. Aguilar, A. Martinez Raya & A. Roca Roca, eds. *Evaluación y Manejo de Suelos*. pp. 227-243. Granada, CL, Junta de Andalucía.

Macías-García, F., Camps Arbestain, M. & Macías, F. 2009. Utilización de Tecnosoles derivados de residuos en procesos de restauración de suelos de la mina Touro. In Minería Sostenible. pp. 651-661. A Coruña, Cámara Oficial Mineira de Galicia.

Macías-García, F., Fontán, L., Otero, X.L., Pérez Llaguno, C., Camps Arbestain, M. & Macías, F. 2009. Recuperación de aguas ácidas de la mina Touro mediante sistemas integrados de barreras reactivas con diferentes Tecnosoles y humedales. *In Minería Sostenible*. pp. 963-973. A Coruña, Cámara Oficial Mineira de Galicia

Magnani, F., Mencuccini, M., Borghetti, M., Berbigier, P., Berninger, F., Delzon, S., Grelle, A., Hari, P., Jarvis, P.G., Kolari, P., Kowalski, A.S., Lankreijer, H., Law, B.E., Lindroth, A., Loustau, D., Manca, G., Moncrieff, J.B., Rayment, M., Tedeschi, V., Valentini, R. & Grace, J. 2007. The human footprint in the carbon cycle of temperate and boreal forests. *Nature*, 447: 848–850

Marfenina, O.E., Ivanova, A.E., Kislova, E.E. & Sacharov, D.S. 2008. The mycological properties of medieval culture layers as a form of soil "biological memory" about urbanization. *Journal of Soils and Sediments*, 8(5): 340-348.

Martínez-Cortizas, A., García-Rodeja, E., Pontevedra-Pombal, X., Nóvoa-Muñoz, J.C., Weiss, D. & Cheburkin, A. 2002. Atmospheric Pb deposition in Spain during the last 4600 years recorded by two ombrotrophic peat bogs and implications for the use of peat as archive. *Sci. Total Environ.*, 292: 33-44.

Matson, P.A., Parton, W.J., Power, A.G. & Swift, M.J. 1997. Agricultural Intensification and Ecosystem Properties. *Science*, 277(5325): 504–509.

McSherry, M.E. & Ritchie, M.E. 2013. Effects of grazing on grassland soil carbon: a global review. Global Change Biology, 19: 1347–1357

Meersmans, J., van Wesemael, B., De Ridder, F., Fallas Dotti, M., De Baets, S. & van Molle, M. 2009. Changes in organic carbon distribution with depth in agricultural soils in northern Belgium, 1960-1990. *Glob. Change Biol.*, 15: 2739–2750

Meuser, H. 2010. *Contaminated Urban Soils.* Springer Science & Buseness media. 340 pp.

Mighall, T.M., Abrahams, P.W., Grattan, J.P., Hayes, D., Timberlake, S. & Forsyth, S. 2002. Geochemical evidence for atmospheric pollution derived from prehistoric copper mining at Copa Hill, Cwmystwyth, mid-Wales, UK. *Sci. Total Environ.*, 292: 69-80.

Mohammed, A.S., Kapri, A. & Goel, R. 2011. Heavy metal pollution: source, impact and remedies. In S.K. Mohammad, R. Goel & J. Musarrat, eds. *Biomanagement of metal-contaminated soils.* pp. 1-28. New York, Springer. 556 pp

Monteith, D.T., Stoddard, J.L., Evans, C.D., de Wit, H.A., Forsius, M., Høgåsen, T., Wilander, A., Skjelkvåle, B.L., Jeffries, D.S., Vuorenmaa, J., Keller, B., Kopácek, J. & Vesely, J. 2007. Dissolved organic carbon trends resulting from changes in atmospheric deposition chemistry. *Nature*, 450: 537–540.

Monterroso, C., Alvarez, E. & Macías, F. 1994. Speciation and Solubility Control of Al and Fe in Minesoil Solutions. *The Science of the Total Environment*, 158: 31-43.

Monterroso, C., Alvarez, E., Fernández-Marcos, M.L. & Macías, F. 1999. Geochemistry of aluminium and iron in mine soils from As Pontes, Galicia (NW Spain). *Water, Air & Soil Pollution*, 110: 81-102.

Monterroso, C., Macías, F., Gil Bueno, A. & Val Caballero, C. 1998. Evaluation of the land reclamation project at the As Pontes Mine (NW Spain) in relation to the suitability of the soil for plant growth. *Land Degradation & Development*, 9: 441-451.

Moore, J.N. & Luoma, S.N. 1990. Hazardous wastes from large-scale metal extraction. *Environ. Sci. Technol.*, 24: 1278-1285

Munshower, F.F. 1977. Cadmium accumulation in plants and animals on polluted and non-polluted grasslands. *J. Environ. Qual.*, 6: 411-413

Murty, D., Kirschbaum, M.U.F., McMurtrie, R.E. & McGilvray, H. 2002. Does conversion of forest to agricultural land change soil carbon and nitrogen? a review of the literatur. *Global Change Biology*, 8: 105–123.

Nachtergaele, F., Bruinsma, J., Valbo-Jorgensen, J. & Bartley, D. 2011. *Anticipated trend in the use of global land and water resources*. SOLAW Background Thematic Report TR 01. Rome, FAO

Navarro-Campos, C., Pekas, A., Moraza, M.L., Aguilar, A. & Garcia-Mari, F. 2012. Soil-dwelling predatory mites in citrus: Their potential as natural enemies of thrips with special reference to Pezothrips kellyanus (Thysanoptera: Thripidae). *Biological Control*, 63: 201-209

Nave, L.E., Vance, E.D., Swanston, C.W. & Curtis, P.S. 2011. Fire effects on temperate forest soil C and N storage. *Ecological Applications*, 21(4): 1189–1201.

Neel, C., Bril, H., Courtin-Nomade, A. & Dutreuil, J-P. 2003. Factors affecting natural development of soil on 35-year-old sulphide-rich mine tailings. *Geoderma*, 111: 1-20

Nicholson, F.A., Smith, S.R., Alloway, B.J., Carlton-Smith, C. & Chambers, B.J. 2003. An inventory of heavy metals inputs to agricultural soils in England and Wales. *The Science of the total environment*, 311: 205–19

Nilsson, J. & Grennfelt, P. 1988. *Critical Loads for Sulphur and Nitrogen*. Copenhagen, Nordic Council of Ministers.

Nordstrom, D.K. & Alpers, C.N. 1999. Geochemistry of acid mine waters. In G. Plumlee & M. Logsdon eds. *Environmental geochemistry of mineral deposits*. Vol. 6A. pp. 133-156. Rev. Econ. Geol. 580 pp.

Nordstrom, D.K. & Southam, G. 1997. Geomicrobiology of sulphide mineral oxidation. In J.F. Banfield & K.H. Nealson, eds. *Geomicrobiology: interactions between microbes and minerals*. Vol. 35. Reviews in Mineralogy. pp. 361-390. Washington, DC, Mineralogical Society of America. 448 pp.

Olander, L.O., Bustamante, M.C.C., Asner, G.P., Telles, E. & do Prado, Z.A. 2005. Surface soil changes following selective logging in an Eastern Amazon forest. *Earth Interactions*, 9(4): 1-19.

Otero, X.L., Ferreira, T.O., Vidal, P., Macias, F. & Chesworth, W. 2008. Thionic or sulfidic soils. In W. Chesworth, ed. *Encyclopedia of Soil Science*. pp. 777-781. Springer. 902 pp

Oulehle, F., Evans, C.D., Hofmeister, J., Krejci, R., Tahovska, K., Persson, T., Cudlin, P. & Hruska, J. 2011. Major changes in forest carbon and nitrogen cycling caused by declining sulphur deposition. *Global Change Biology*, 17: 3115–3129

Panda, S. & Sahu, S.K. 2004. Recovery of acetylcholine esterase activity of Drawida willsi (Oligochaeta) following application of three pesticides to soil. *Chemosphere*, 5: 283–290.

Pandey, S. & Singh, D.K. 2004. Total bacterial and fungal population after chlorpyrifos and quinalphos treatments in groundnut (Arachis hypogaea L.) soil. *Chemosphere*, 55(2): 197–205

Paustian, K., Babcock, B.A., Hatfield, J., Kling, C.L., Lal, R., McCarl, B.A., McLaughlin, S., Mosier, A.R., Post, W.M., Rice, C.W., Robertson, G.P., Rosenberg, N.J., Rosenzweig, C., Schlesinger, W.H. & Zilberman, D. 2004. *Climate Change and Greenhouse Gas Mitigation: Challenges and Opportunities for Agriculture*. Task Force Report No. 141. Council for Agricultural Science and Technology

Perez, C.A., Carmona, M.R., Fariña, J.M. & Armesto, J.J. 2009. Selective logging of lowland evergreen rainforests in Chiloe´ Island, Chile: Effects of changing tree species composition on soil nitrogen transformations. *Forest Ecology and Management*, 258: 1660–1668

Pittman, K., Hansen, M.C., Becker-Reshef I., Potapov, P.V. & Justice, C.O. 2010. Estimating global cropland extent with Multi-year MODIS data. *Remot. Sens.*, 2(7): 1844–1863.

Poeplau, C., Don, A., Vesterdal, L., Leifeld, J., Wesemael, B., Schumacher, J. & Gensior, A. 2011. Temporal dynamics of soil organic carbon after land-use change in the temperate zone – carbon response functions as a model approach. *Global Change Biology*, 17: 2415–2427

Portmann, F., Siebert, S. & Döll, P. 2010. MIRCA 2000 - Global Monthly Irrigated and Rainfed Crop Areas around the year 2000: A new high-resolution data set for agricultural and hydrological modelling. *Global Biological Cycles*.24(GB 1011): 1-24

Prokop, G., Jobstmann, H. & Schöbauer, A. 2011: Overview on best practices for limiting soil sealing and mitigating its effects in EU-27. Environment Agency Austria. European Communities. 231 pp.

Pugh, T.A.M., Arneth, A., Olin, S., Ahlström, A., Arvanitis, A., Bayer, A.D., Klein Goldewijk, K., Lindeskog, M. & Schurgers, G. 2015. Agricultural management substantially enhances land-use change emissions. [Submitted]

Pugh, T.A.M., Arneth, A., Olin, S., Ahlström, A., Arvanitis, A., Bayer, A., Lindeskog, M., Klein Goldewijk, K. & Schurgers, G. 2014. Accounting for land management notably reduces projections of the terrestrial carbon sink. [submitted

Ramankutty, N., Evan, A.T., Monfreda, C. & Foley, J.A. 2008. Farming the planet: 1.Geographic distribution of global agricultural lands in the year 2000. *Global Biogeochemical Cycles*, 22(GB 1003)

Ramankutty, N., Foley, J.A. & Olejniczak, N.J. 2002. People on the land: Changes in global population and croplands during the 20th century. *AMBIO: A Journal of the Human Environment*, 31(3): 251-257

Ravi, S., Breshears, D.D., Huxman, T.E. & D'Odorico, P. 2010. Land degradation in drylands: Interactions among hydrologic–aeolian erosion and vegetation dynamics. *Geomorphology*, 116: 236–245

Ray, D.K. & Foley, J.A. 2013. Increasing global crop harvest frequency: recent trends and future directions. *Environmental Research Letters*, 8(4), 044041.

Reis, S., Grennfelt, P., Klimont, Z., Amann, M., Apsimon, H., Hettelingh, J.-P., Holland, M., LeGall, A.-C., Maas, R., Posch, M., Spranger, T., Sutton, M.A. & Williams, M. 2012. From acid rain to climate change. *Science*, 338(6111): 1153–1154

Reuss, J.O. & Johnson, D.W. 1986. *Acid Deposition and the Acidification of Soils and Waters*. New York, Springer Verlag. 119 pp

Ribeiro Filho, A.A., Adams, C. & Sereni Murrieta, R.S. 2013. The impacts of shifting cultivation on tropical forest soil: a review. 2013. *Bol. Mus. Para. Emílio Goeldi. Cienc. Hum., Belém*, 8(3): 693-727

Rodríguez-Lado, L. & Macías, F. 2005. Calculation and mapping of critical loads of sulphur and nitrogen for forest soils of Galicia (NW Spain). *Science of the Total environment*, 336: 760-7

Rodríguez-Lado, L., Montanarella L. & Macías F. 2007. Evaluation of the sensitivity of European Soils to the deposition of acid compounds: different approaches provide different results. *Water, Soil and Air Pollution*, 185: 293-303

Römbke, J., Garcia, M.V. & Scheffczyk, A. 2007. Effects of the fungicide benomyl on earthworms in laboratory tests under tropical and temperate conditions. *Archives of Environmental Contamination and Toxicology*. 53: 590–598.

RoTAP. 2012. *Review of Transboundary Air Pollution (RoTAP): Acidification, Eutrophication, Ground Level Ozone and Heavy Metals in the UK*. Edinburgh, Contract Report to the Department for Environment, Food and Rural Affairs. Centre for Ecology & Hydrology

Rothwell, J.J., Robinson, S.G., Evans, M.G., Yang, J. & Allott, T.E.H. 2005. Heavy metal release by peat erosion in the Peak District, southern Pennines, UK. *Hydrological Processes*, 19: 2973–2989

Royal Society of London. 2009. *Reaping the benefits: science and the sustainable intensification of global agriculture*. London, UK

Salomons, W. 1995. Environmental impact of metals derived from mining activities: Process, prediction, prevention. *Journal of Geochemical Exploration*, 52: 5-23.

Schaphoff, S., Heyder, U., Ostberg, S., Gerten, D., Heinke, J. & Lucht, W. 2013. Contribution of permafrost soils to the global carbon budget. *Environmental Research Letters*, 8(1): 014026

Sencindiver, J.C. & Ammons, J.T. 2000. Minesoil genesis and classification. In R.I. Barnhisel, R.G. Darmody & W.L. Daniels, eds. *Reclamation of drastically disturbed lands. Agronomy monograph no.* 41. pp. 595–613. Madison: ASA, CSSA, SSSA

Sengupta, M. 1993. *Environmental impacts of Mining. Monitoring, Restoration and Control*. Lewis Publishers, CRC Pres

Seré, C. & Steinfeld, H. 1996. *World livestock production systems. Current status, issues and trends. Animal Production and Health Paper 127*. Rome, FAO.

Shi, S., Zhang, W., Zhang, P., Yu, Y. & Ding, F.A. 2013. Synthesis of change in deep soil organic carbon stores with afforestation of agricultural soils. *Forest Ecology and Management*, 296: 53–63.

Shotyk, W., Weiss, D., Appleby, P.G., Cheburkin, A.K., Fre[i], R., Gloor, M., Kramers, J.D., Reese, S. & van der Knaap, W.O. 1998. History of atmospheric lead deposition since 12,370 14C yr BP from a peat bog, Jura Mountains, Switzerland. *Science*, 281: 1635-1640.

Siebert, S. & Doll, P. 2010. Quantifying blue and green virtual ¨ water contents in global crop production as well as potential production losses without irrigation. *J. Hydrol.*, 384(3–4): 198-217.

Siebielec, G., Lazar, S., Kaufmann, C. & Jaensch, S. 2010. *Handbook for measures enhancing soil function performance and compensating soil loss during urbanization process.* Urban SMS – Soil Management Strategy project, 37 pp.

Smil, V. 2000. Phosphorus in the environment: natural flows and human interferences. *Annual Review of Energy and the Environment*, 25(1): 53–88.

Smith, B., Prentice, I.C. & Sykes, M.T. 2001. Representation of vegetation dynamics in the modelling of terrestrial ecosystems: comparing two contrasting approaches within European climate space. Global Ecology & Biogeography, 10: 621-637

Smith, P. 2004. Carbon sequestration in croplands: the potential in Europe and the global context. *European Journal of Agronomy*, 20(3): 229–236.

Smith, P. 2005. An overview of the permanence of soil organic carbon stocks: influence of direct human-induced, indirect and natural effects. *European Journal of Soil Science*, 56: 673-680

Smith, P., House, J.I., Bustamante, M., Sobocká, J., Harper, R., Pan, G., West, P., Clark, J., Adhya, T., Rumpel, C., Paustian, K., Kuikman, P., Cotrufo, M.F., Elliott, J.A., McDowell, R., Griffiths, R.I., Asakawa, S., Bondeau, A., Jain, A.K., Meersmens, J. & Pugh, T. (in press) Global Change Pressures on Soils from Land Use and Management. *Global Change Biology*. DOI: 10.1111/gcb.13068

Sombroek, W.G., Nachtergaele, F.O. & Hebel, A. 1993. Amounts, dynamics and sequestering of carbon in tropical and subtropical soils. *Ambio*, 22: 417-426

Spranger, T., Hettelingh, J.-P., Slootweg, J. & Posch, M. 2008. Modelling and mapping long-term risks due to reactive nitrogen effects: an overview of LRTAP convention activities. *Environmental Pollution*, 154: 482–487

State Bureau of Statistics- China. 2005. 50 *Years Rural Statistics of New China*. China, Beijing, China Statistics Press

Suttie, J.M., Reynolds, S.G. & Batello, C. 2005. Grasslands of the World. FAO *Plant production and protection series* No. 34. Rome, FAO

Tian, H.Q., Lu, C.Q., Melillo, J., Ren, R., Huang,Y., Xu, X.F., Liu, M.L., Zhang, Z., Chen, G. S. & Pan, S.F. 2012. Food benefit and climate warming potential of nitrogen fertilizer uses in China. *Environm Res. Letters*, 7: 044020

Tilman, D., Cassman, K. G., Matson, P. A., Naylor, R. & Polasky, S. 2002. Agricultural sustainability and intensive production practices. *Nature*, 418(6898): 671–677

Tipping, E., Smith, E., Lawlor, A., Hughes, S. & Stevens, P. 2003. Predicting the release of metals from ombrotrophic peat due to drought-induced acidification. *Environmental Pollution*, 123: 239–253

Tsiafouli, M.A., Thebault, E., Sgardelis, S.P., de Ruiter, P.C., van der Putten, W.H., Birkhofer, K., Hemerik, L., de Vries, F.T., Bardgett, R.D., Brady, M.V., Bjornlund, L., Jørgensen, H.B., Christensen S., D' Hertefeldt, T., Hotes S., Hol, W.H.G., Frouz, J., Liiri, M., Mortimer, S.R., Setälä H., Tzanopoulos, J., Uteseny K., Pižl, V., Stary, J., Wolters, V., & Hedlund, K. 2015. Intensive agriculture reduces soil biodiversity across Europe. *Global Change Biology*, 21(2): 973-985

Van Aardenne, J.A., Dentener, F.J., Olivier, J.G.J., Klein Goldewijk, C.G.M. & Lelieveld, J. 2001. A 1°×1° resolution data set of historical anthropogenic trace gas emissions for the period 1890-1990. *Global Biogeochemical Cycles*, 15: 909–928

Van Zwieten, L., Rust, J. Kingston, T., Merrington, G. & Morris, S. 2004. Influence of copper fungicide residues on occurrence of earthworms in avocado orchard soils. *The Science of the Total Environment*, 329: 29-41.

Vega, F.A., Covelo, E.F., Andrade, M.L. & Marcet, P. 2004. Relationships between heavy metals content and soil properties in mine soils. *Analytica Chimica Acta*., 524: 141-150.

Veiga, M.M., Nunes, D., Klein, B., Shandro, J.A., Colon Velasquez, P. & Sousa R.N. 2009. Mill leaching: a viable substitute for mercury amalgamation in the artisanal gold mining sector? *Journal of Cleaner Production*, 17: 1373-1381.

Venterea, R.T., Maharjan, B. & Dolan, M.S. 2011. Fertilizer Source and Tillage Effects on Yield-Scaled Nitrous Oxide Emissions in a Corn Cropping System. *J. Environ. Qual.*, 40:1521–1531.

Vet, R., Artz, R.S., Carou, S., Shaw, M., Ro, C.-U., Aas, W., Baker, A., Bowersox, V.C., Dentener, F., Galy-Lacaux, C., Hou, A., Pienaar, J.J., Gillett, R., Forti, M.C., Gromov, S., Hara, H., Khodzher, T., Mahowald, N.M., Nickovic, S., Rao, P.S.P. & Reid, N.W. 2014. A global assessment of precipitation chemistry and deposition of sulfur, nitrogen, sea salt, base cations, organic acids, acidity and pH, and phosphorus. *Atmospheric Environment*, 93: 3–10

Villela, D.M., Nascimento, M.T., Aragão, L.E.O.C. & Gama, D.M. 2006. Effect of selective logging on forest structure and nutrient cycling in a seasonally dry Brazilian Atlantic forest. *Journal of Biogeography*, 33: 506–516.

Weber, J.L. 2010. *Land cover classification in the revised SEEA*. Land Cover Classification for Land Cover Accounting, LG/15/9, position paper drafted by Jean-Louis Weber (EEA), 15th Meeting of the London Group on Environmental Accounting, Copenhagen, 4 October 2010

Wei, X., Shao, M., Gale, W. & Li, L. 2014. Global pattern of soil carbon losses due to the conversion of forests to agricultural land. *Scientific Reports* 4: 4062

West, P. C., Gerber, J. S., Engstrom, P. M., Mueller, N. D., Brauman, K. A., Carlson, K. M., Cassidy, E.C.

Johnston, M., MacDonald, G.K., Ray, D.K. & Siebert, S. 2014. Leverage points for improving global food security and the environment. *Science*, 345(6194): 325–328.

Whitfield, C.J., Aherne, J., Watmough, S.A. & Mcdonald, M. 2010. Estimating the sensitivity of forest soils to acid deposition in the Athabasca Oil Sands Region, Alberta. *Journal of Liminology*, 69: 201–208.

Williams, M. 2001. Arsenic in mine waters: an international study. *Environmental Geology*, 40: 267-278

World Bank. 2008. Agriculture for Development. *World Development Report* 2008.Washington, DC, World Bank.

Yasmin, S. & D'Souza, D. 2010. Effects of pesticides on the growth and reproduction of earthworm: A review. *Appl. Environ. Soil Sci.*, 2010(678360): 1-9.

Zhang ,Y., Xu, M., Chen, H. & Adams, J. 2009. Global pattern of NPP to GPP ratio derived from MODIS data: effects of ecosystem type, geographical location and climate. *Global Ecology and Biogeography*, 18: 280–290

Zhou, S.-P., Duan, C.-Q., Fu, H., Chen, Y.-H., Wang, X.-H. & Yu, Z.-F. 2007. Toxicity assessment for chlorpyrifos-contaminated soil with three different earthworm test methods. *Journal of Environmental Sciences*, 19: 854–858.

Global Soil Change

Drivers, Status and Trends

Coordinating Lead Authors:
André Bationo (Burkina Faso), Srimathie Indraratne (Sri Lanka)

Contributing Authors:
Sayed Alavi (ITPS/Iran), Abdullah Alshankiti (ITPS/Saudi Arabia), Dominique Arrouays (ITPS/France), Charles Bielders (United States), Keith Bristow (Australia), Marta Camps Arbestain (ITPS/New Zealand), Lucrezia Caon (Italy), Brent Clothier (New Zealand), Tandra Fraser (United States), Ciro Gardi (Italy), Gerard Govers (Belgium), Roland Hiederer (Germany), Jeroen Husing (TSBF-CIAT), Joyce Jefwa (TSBF-CIAT), Shawntine Lai (United States/Taiwan, Province of China), Rattan Lal (United States), John P. Lamers (Germany), Dar-Yuan Lee (Taiwan, Province of China), Fredah Maina (Kenya), Luca Montanarella (ITPS/EC), Joseph Mung'atu (TSBF-CIAT), Freddy Nachtergaele (Belgium), Peter F. Okoth (TSBF-CIAT), Asad Qureshi (ICBA), Shabbir Shahid (ICBA), Manuela Ravina da Silva (Sweden), Justin Sheffield (United Kingdom), Tran Tien (Vietnam), Kristof Van Oost (Belgium), Boris Vrscaj (Slovenia), Diana Wall (United States), Boaz Waswa (Kenya), Jeewika Weerahewa (Sri Lanka), Kazuyuki Yagi (ITPS/Japan), Ted Zobeck (United States).

Reviewing Authors:
Dominique Arrouays (ITPS/France), Richard Bardgett (United Kingdom), Marta Camps Arbestain (ITPS/New Zealand), Tandra Fraser (Canada), Ciro Gardi (Italy), Neil McKenzie (ITPS/Australia), Luca Montanarella (ITPS/EC), Dan Pennock (ITPS/Canada) and Diana Wall (United States).

5 | Drivers of global soil change

Drivers in general comprise the factors that bring about socio-economic and environmental changes. They operate at various spatial and temporal levels in society. They differ from one region to another, and within and between nations. Drivers are diverse in nature and they include: demographics; economic factors; scientific and technological innovation; markets and trade; wealth distribution; institutional and socio-political frameworks; value systems; and climate and climate change (UNEP, 2007; IAASTD, 2009). Drivers have an impact on natural resources including soil services and functions, with impacts on biodiversity, environmental health and ultimately human well-being. Globalization has particularly affected these drivers, leading to an increase in human mobility with social, economic and environmental implications. Patterns of settlement and consumption result in pressures on ecosystem services, including those provided by soils. Rural-urban migration and associated livelihood changes contribute to changing patterns of energy use and shifts in diet – for example, towards meat – which can intensify pressures on land and soils in producing areas (UNEP, 2012). In addition, climate change may have significant impacts on soil resources through changes in water availability and soil moisture, as well as through sea level rise (IPCC, 2014b).

5.1 | Population growth and urbanization

5.1.1 | Population dynamics

Changing global population trends

The world population of 7.2 billion in mid-2013 is projected to increase by almost one billion by 2025. By 2050 it is expected to reach 9.6 billion, and to rise to 10.9 billion by 2100 (UN, 2014). The principal factor in this continual rise is the rapid increase in the population of developing countries, in particular in Africa, where the population is projected to increase from the current 1.1 billion to reach 2.4 billion by 2050 (Table 5.1). Many countries of Sub-Saharan Africa are still experiencing fast population growth with high fertility rates. Other countries with similar trends include India, Indonesia, Pakistan, the Philippines and the United States. By 2030

India's population is expected to surpass China's, to become the most populous country in the world. Nigeria's population is expected to surpass the United States population in 2045 to become the world's third most populous country. Nigeria's population is likely to rival that of China by the end of the century (Table 5.2). Over the period 2013–2100, eight countries are expected to account for over half of the world's projected population increase: Nigeria, India, the United Republic of Tanzania, the Democratic Republic of Congo, Niger, Uganda, Ethiopia and the United States of America. On the other hand, Europe's population is projected to decline, since fertility rates are far below the level for replacement of population in the long run. As fertility decreases and life expectancy rises, population ageing is a challenge for Europe (UN, 2014). Other developing countries with young populations but lower fertility (e.g. China, Brazil and India) are also likely to face challenges of an ageing society by the end of this century (Gerland *et al.*, 2014).

Table 5.1 | World population by region

Region	Area (millions of km²)	2013 Population (millions)	Percent of world population	Density (p/km²)	2050 population. (projected)	% of world pop.	% Change 2013-2050
Asia	31.9	4 298	60.0	135	5 164	54.1	20
Africa	31.0	1 110	15.5	36	2 393	25.1	115
Europe	23.0	742	10.4	32	709	7.4	-4
LAC	20.5	617	8.6	30	782	8.2	27
North America	21.8	355	5.0	16	446	4.7	26
Oceania	8.6	38	0.5	4	57	0.6	48
World	**136.8**	**7 162**	**100.0**	**52**	**9 551**	**100**	**33**

Table 5.2 | The ten most populous countries 1950, 2013, 2050 and 2100 (population in millions). Source: United Nations, 2014.

Country	Population in 1950	Country	Population in 2013	Country	Projected Population in 2050	Country	Proj. Popul. 2100
China	544	China	1 386	India	1 620	India	1 547
India	376	India	1 252	China	1 385	China	1 086
United States	158	United States	320	Nigeria	440	Nigeria	914
Russian Federation	103	Indonesia	250	United States	401	United States	462
Japan	82	Brazil	200	Indonesia	321	Indonesia	315
Indonesia	73	Pakistan	182	Pakistan	271	Tanzania	276
Germany	70	Nigeria	174	Brazil	231	Pakistan	263
Brazil	54	Bangladesh	157	Bangladesh	202	Dem. Rep of Congo	262
United Kingdom	51	Russian Federation	143	Ethiopia	188	Ethiopia	243
Italy	46	Japan	127	Philippines	157	Uganda	205

5.1.2 | Urbanization

In tandem with the rate of population increase is the rising rate of urbanization. According to the United Nations, by 2014 more people were living in urban areas (54 percent) than in rural areas. The urbanization trend is expected to continue in all regions, and by 2050, 66 percent of the world's population is projected to be urban. Today some of the most urbanized regions are Northern America (82 percent), Latin America and the Caribbean (80 percent) and Europe (73 percent). However, Africa and Asia are now the fastest urbanizing regions, with the share of the population urbanized expected to rise from today's 40 and 48 percent respectively to 56 and 64 percent by 2050. Three countries together are expected to account for 37 percent of the growth of the world's urban population between 2014 and 2050: India (adding 404 million), China (adding 292 million) and Nigeria (adding 212 million). Whereas in the past mega-cities were located in more developed regions, today's large cities are principally found in lower income countries. Since 1990 the number of these mega-cities has nearly tripled globally; and by 2030, the world is projected to have 41 global agglomerations, housing more than 10 million inhabitants each. In developing countries, the competition between demand for agricultural land and the needs of growing cities is a mounting challenge (Jones *et al.*, 2013).

The rural population globally is now close to 3.4 billion but is expected to decline to 3.2 billion by 2050. Africa and Asia are home to nearly 90 percent of the world's rural population. India has the world's largest rural population (857 million), followed by China (635 million). Rural/urban migration continues to feed urban growth, causing environmental changes including effects on land use and soils. Policy and poverty also drive the threats to land and soils. Many rural poor live under regimes of weak land policy and insecure tenure systems. They often farm marginal lands of low agricultural productivity, typically employing traditional farming methods. This may aggravate soil degradation and biodiversity decline, with resulting yield losses and food insecurity (Jones *et al.*, 2013; Barbier, 2013). In addition, land grabs may lead to eviction of farming families, for whom rural to urban migration may be the only option, so accelerating the pace of urbanization (Holdinghausen, 2015).

5.2 | Education, cultural values and social equity

Education influences decisions regarding land use and land management. Farmers' decisions result from many factors, including incentives, access to capital and risk management, but also from knowledge and level of education, all of which may affect land use and management practices (MA, 2005). Land use and management is dependent on the sum total of all decisions taken by individual farmers of different education and gender groups in a community (IAASTD, 2009).

Women play a key role in agriculture. They represent 43 percent of the agricultural labour force world-wide, ranging from around 20 percent in Latin America to 50 percent in parts of Africa and Asia (FAO, 2011b). Women are responsible for half of the world's food production, providing between 60 and 80 percent of the food in most developing countries (World Bank/FAO/IFAD, 2009). However, evidence shows that women still own less land and have smaller landholdings with generally poor soil quality. Improving women's access to land and secure tenure can have direct impacts on farm productivity and in the long run improve household welfare (FAO, 2013). FAO's Gender and Land Rights Database (2010) suggests that less than one quarter of agricultural land holdings in developing countries are operated by women. Latin America and the Caribbean have the largest mean share of female agricultural land holders, exceeding 25 percent in Chile, Ecuador and Panama. In North Africa and West Asia, female landholders represent fewer than five percent of the owners. In sub-Saharan Africa the average rate is 15 percent, although there are wide variations within the region, from less than five percent in Mali to over 30 percent in Botswana, Cape Verde and Malawi (Figure 5.1).

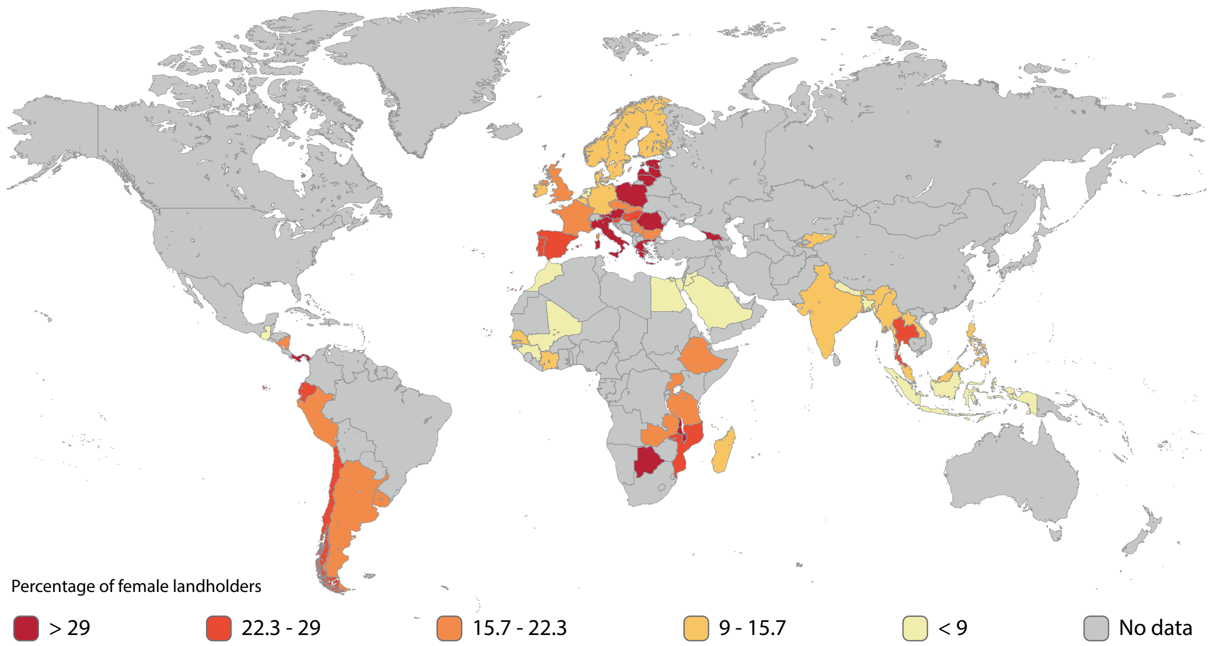


Figure 5.1 | Percentage of female landholders around the world.
Source: FAO, 2010.

5.3 | Marketing land

Today land is used more intensively than ever. The expansion of markets, rising population, and economic development and higher incomes have pushed up demand for land for both agriculture and for settlements and so driven unprecedented land use change (Section 4.1, this volume). The most dramatic changes have been in reduction in forest cover, in expansion and intensification of cropland, and in urbanization (UNEP, 2007). In agriculture, production of crops and livestock products for markets is fundamental to the economies of many countries. One new segment of market-driven production is biofuels, where incentives have strengthened as a result of higher and volatile oil prices and because a number of countries have introduced renewable energy promotion policies (Rulli *et al.*, 2013). North America is leading global biofuel production, with 48 percent of the global market. The second largest producer of biofuels is Brazil, producing 24 percent of the world's biofuels (OECD/FAO, 2011). The growth of biofuels production is driving an increase in deforestation and other land use changes.

It is widely accepted by economists that when land markets function in an efficient manner, the resulting land use patterns provide the highest possible benefits to the society. However, empirical research findings reveal that the functioning of land markets in many developing countries is inefficient and the resulting land use patterns are sub-optimal (Pinstrup-Anderson and Watson II, 2011). Amongst the causes of inefficiencies the following have been cited: lack of well-defined property rights (Allen, 1991; Alston, Libecap and Schneider, 1995; Besley, 1995); higher bargaining power exercised by different groups of buyers (Sengupta, 1997; Ghebru and Holden, 2012); non-existence or under-functioning of insurance markets to absorb risk and uncertainties in the natural environment (such as climate change) (Dayton-Johnson, 2006; Auffret, 2003); and environmental externalities like soil erosion.

Land grabbing - large scale land acquisitions - started initially in response to the 2007-2008 increase in food prices. Since then the phenomenon has intensified (IMF, 2008). Foreign states and companies and national investors, often with the support of the national government, see land as an attractive asset in order to meet the demands of food supply and energy. Experience in Africa, Eastern Europe, South America and South and Southeast Asia has shown that in an unregulated environment this 'land grab' can lead to the displacement of local farmers (Rulli *et al.*, 2013). Since fertile land is a limited resource, competition for it may lead to a rise in poverty, violence and social unrest in countries with weak regulatory systems or power imbalances (Nolte and Ostermeier, 2015).

Large areas of arable land have been bought or leased in recent years, mainly in developing countries (Figure 5.2). According to the Land Matrix Global Observatory database, since the year 2000 over 1 000 land deals involving foreign investors have been struck, covering 39 million ha, while another 200 deals cover 16 million ha. The main driver of large land-scale acquisitions continues to be agricultural production, with 40 percent of deals for food crop production and livestock farming, followed by agrofuels as the second most important driver with 190 deals, and forestry projects which have increased by 50 percent (Land Matrix Newsletter, 2014). Other acquisitions have been for urban expansion, mining, infrastructure projects and tourism (Nolte and Ostermeier, 2015).

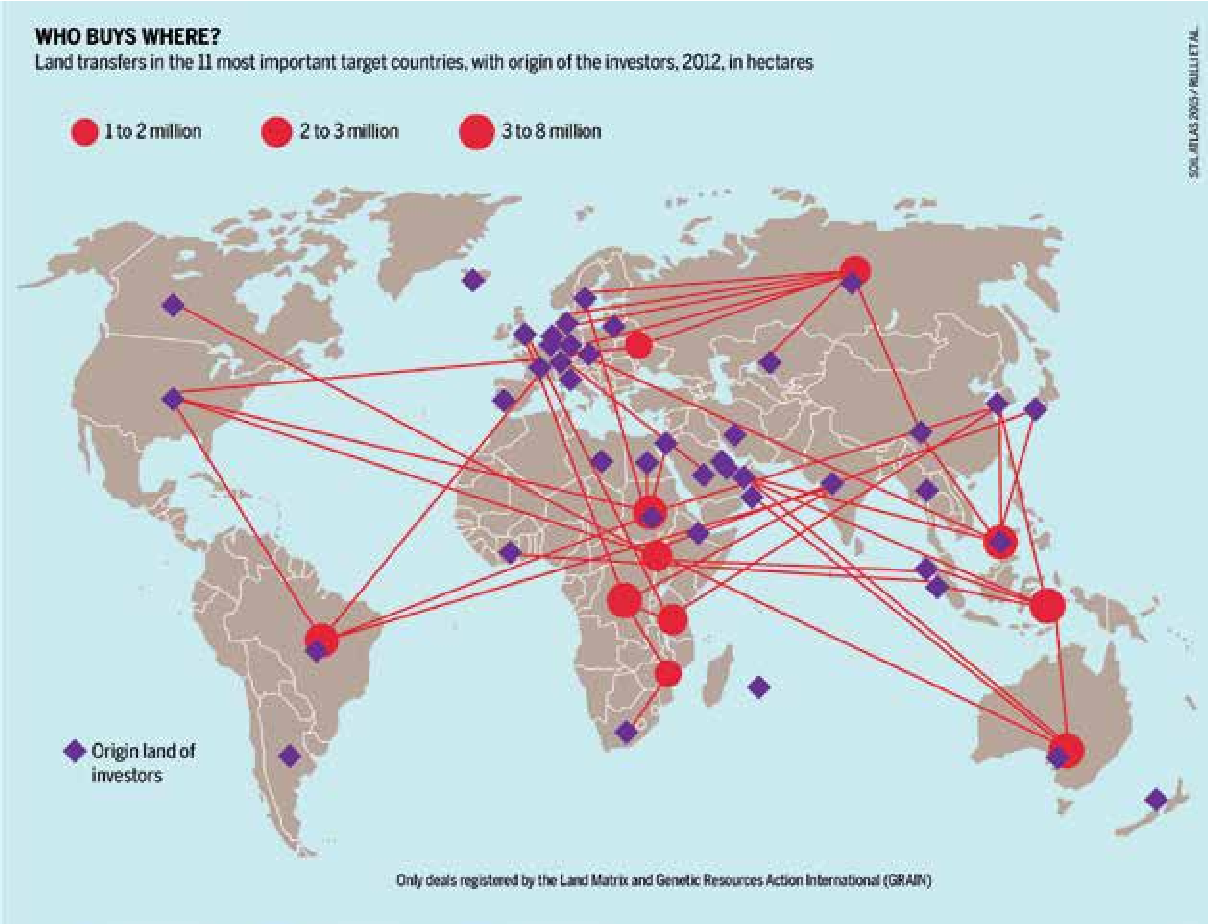


Figure 5.2 | Major land deals occurring between countries in 2012.
Source: Soil Atlas, 2015/Rulli et al., 2013.

In addition to these commercial land transactions, policy responses for climate change adaptation and mitigation have led to market-based approaches which attach a value to ecosystem services. In this context, there is the allocation of land for environmental ends, for example, offsetting emissions in the industrialized North by protecting forests in the South. These approaches in practice have sometimes required curtailment of customary or community access rights to forest and water. In other cases, these approaches have encouraged the shift of smallholder labour from subsistence farming and cash crop production to carbon sequestration (UNEP, 2012). A number of projects focusing on soil health and carbon sequestration in Africa have aimed to benefit individual farmers and at the same time to mitigate climate change. These pioneering projects have faced implementation challenges, including high unit costs, small land sizes, and weak land tenure rights. In addition, the incentive framework has been weakened by the small size of the cash payments the projects can offer for carbon sequestration, due to the low value of carbon credits and periodic market volatility in international voluntary carbon markets. Nevertheless, the non-carbon benefits gained from the projects, such as improved agricultural productivity through sustainable land management and soil health and strengthening of community solidarity, are important results to be prioritized in future projects (Shames, 2013).

5.4 | Economic growth

Economic growth and urbanization generate immense benefits to humankind but also contribute to unsustainable consumption patterns. They may lead to increased levels of emissions from mining, manufacturing, sewage, energy and transport and to the consequent release of persistent pollutants to land, air and water (UNEP, 2012). By 2050 the population around the globe is expected to be generally wealthier and more urbanized, resulting in an increase and shift in consumption and food demand and consequently in a rise in pollution risk. In developing economies, livestock production is already increasing at a rapid rate as a result of structural change in diets and consumption. FAO predicts that the total demand for animal products will increase at more than double the rate of increase in demand for food of vegetable origin, such as cereals (FAO, 2011a). This will lead to the expansion of land dedicated to livestock, both pasture and feed production. The largest expansion is predicted in the tropics, particularly in South America and Africa where vast areas of tropical forests, semi-arid lands, savannah, grassland and wetland ecosystems could be exploited for livestock - with potentially devastating environmental results (Laurance, Sayer and Cassman, 2014). In developing countries, difficult decisions on the trade-offs between preserving natural ecosystems and economic development will be required. In any case, it is likely that agricultural expansion and biofuel production will continue to trigger deforestation, and consequently soil degradation, pollution of land and water and increase in greenhouse gas emissions (FAO, 2003; Alexandratos and Bruinsma, 2012).

Despite improvements in income growth in many countries, poverty and access to food remain problematic. According to FAO (2014), an estimated 842 million people around the world are currently undernourished. The 2007-2008, 2010 and 2012 price hikes in commodity markets evidenced how price shocks can trigger prolonged crises leading to food insecurity amongst the most vulnerable (FAO, 2011b). Global agendas such as those stated in the Sustainable Development Goals and the Post-2015 Agenda argue that environmental stewardship and sustainable management of natural resources provide opportunities to decrease inequality while increasing production of goods and services. However, this is a complicated agenda, as links between human well-being and natural resources, including soils, are influenced by a host of factors, including economic wealth, trade, technology, gender, education etc. Turning these lofty themes into real policies and programs to reduce poverty equitably and sustainably remains the key development challenge for the coming decades (UNEP, 2007).

5.5 | War and civil strife

In the course of history, many conflicts over fertile land have occurred. Until the twentieth century, most of these conflicts were local and had relatively little impact on the soils themselves. However, modern warfare makes use of non-degradable weapons of destruction and of chemicals that may remain in the affected soils for centuries after the conflict. The impacts of war and civil strife on the environment in general, and on soils in particular, are both direct physical impacts and indirect socio-economic impacts.

Direct physical impacts of war on the soil resource include weapons and bombs remaining in the soil, the destruction of structures with consequent terrain deformation, heavy military transport that results in compaction, and chemical spraying that leads to contamination of both soils and groundwater. Socioeconomic impacts of war include local desertification and displacement of large populations of refugees towards safe regions, resulting in pressure on the environment and soils in the receiving sites (Owona, 2008).

Extensive areas in the world are still affected by remnants of past and present war events. Especially affected are zones where land mines have been buried (Box 5.1), making these soils unsuitable for any exploitation and provisioning of services. There are approximately 110 million mines and other unexploded ordnance (UXO) scattered in sixty-four countries on all continents, remnants of wars from the early twentieth century to the present (Kobayashi, 2013). Africa alone has 37 million landmines in at least 19 countries. Angola is by far the most affected zone with 15 million landmines and an amputee population of 70 000, the highest rate in the world. The problem persists despite campaigns to raise awareness, including the International Campaign to Ban Landmines (ICBL), which was awarded the 1997 Nobel Peace Prize. Removal of mines is proceeding, but at a glacial pace due to the danger, the cost (US$300 to US$1 000 per mine removed), and lack of international agreement on priorities.

Box 5.1 | Minefields

Minefields are one of the main constraints to the development of rural areas in Bosnia and Herzegovina (BIH), where large tracts (ca. 4 000 km²) of agricultural land and forest areas cannot be used because they remain mined after the war that ended two decades ago (ICBL, 2002). BIH is the most mine-affected country in Europe with an estimated one million mines (mostly antipersonnel) remaining in the soil, and only 60 percent of which are located (Bolton 2003; Mitchell, 2004). This affects about 1.3 million people, roughly one third of the population. At current rates of de-mining, it will take several generations before rural areas are again safe.

The displacement of people as a result of wars and conflicts has also created severe environmental and soil problems (Box 5.2).

Box 5.2 | Migration/Refugee Camps

Acholiland in northern Uganda has suffered from persistent insecurity since the mid-1980s. The massive disruption, dislocation and displacement and suffering of the people in the region are well-known. As a way of protecting the local people, the government placed most inhabitants of those districts in camps popularly referred to as Internally Displaced Peoples (IDP) camps. As a result, land has been abandoned and farming and other socio-economic activities are only possible near the protected camps in a restricted radius not exceeding seven kilometres. War creates refugees, leaves government and environmental agencies impaired or destroyed, and substitutes short-term survival for longer-term environmental considerations. This means that ecosystems continue to suffer even after the fighting has stopped (Owona, 2008). The Uganda analysis shows that the creation of 157 IDP camps has significantly affected the environment in terms of deforestation (140 km²), soil erosion, habitat destruction and pollution (Owona, 2008).

Often war results in a combination of negative effects on soils. These may include soil compaction, soil contamination, soil sealing and enhanced wind erosion and dust fall out (Box 5.3).

Box 5.3 | Combined effects of war and strife on soils.

During the 1991 Gulf War in Iraq and Kuwait, there were massive impacts on the environment, resulting from heavy vehicle movements, hundreds of oil well fires, numerous oil lakes and spill-outs (Stephens and Matson, 1993; El-Baz and Makharita, 1994; El-Gamily, 2007). The desert ecosystem was severely damaged by the war: the rate of sand dune movement increased while in addition, new sand sheets and sand dunes were formed in several areas where there had been no sheets or dunes previously (Misak *et al.*, 2002; Misak, Al-Ajmi and Al-Enezi, 2009). The building of many fortifications exposed huge amounts of fine particles to wind erosion Off-site impacts included an increase in the rates of sand transport and dust fallout (Misak, Al-Ajmi and Al-Enezi, 2009). The damage remains years afterwards below the surface.

5.6 | Climate change

The IPCC Fifth Assessment Report reveals that the globally-averaged combined land and ocean surface temperature data show a linear trend of global warming due to increases in anthropogenic emissions of greenhouse gases of 0.85°C (90 percent uncertainty intervals of 0.65 to 1.06°C). Human influence has been detected in warming of the atmosphere and the ocean, in changes in the global water cycle, in reductions in snow and ice, in global mean sea level rise, and in changes in some climate extremes. Continued emission of greenhouse gases will cause further warming and long-lasting changes in all components of the climate system, increasing the likelihood of severe, pervasive and irreversible impacts for people and ecosystems (IPCC, 2014a).

Climate change will have significant impacts on soil resources and food production in both irrigated and rainfed agriculture across the globe. Changes in water availability due to changes in the quantity and pattern of precipitation will be a critical factor (Turral, Burke and Faurès, 2011). Also, higher temperatures, particularly in arid conditions, entail a higher evaporative demand. Where there is sufficient soil moisture, for example in irrigated areas, this could lead to soil salinization if land or farm water management, or irrigation scheduling or drainage are inadequate. The amount of water stored in the soil is fundamentally important to agriculture and is an influence on the rate of actual evaporation, groundwater recharge, and generation of runoff. Soil moisture contents directly simulated by global climate models give an indication of possible directions of change (IPCC, 2014b). For example, using the $HadCM_2$ climate model, Gregory, Mitchell and Brady (1997) show that a rise in greenhouse gas concentrations is associated with reduced soil moisture in Northern Hemisphere mid-latitude summers. This results from higher winter and spring evaporation caused by higher temperatures and reduced snow cover, and from lower rainfall inputs during summer.

The local effects of climate change on soil moisture, however, will vary not only with the degree of climate change but also with soil characteristics (IPCC, 2014b). The water-holding capacity of soil will affect possible changes in soil moisture deficits; the lower the capacity, the greater the sensitivity to climate change. Climate change also may affect soil characteristics, perhaps through changes in waterlogging or cracking, which in turn may affect soil moisture storage properties. Infiltration capacity and water-holding capacity of many soils are influenced by the frequency and intensity of freezing. Boix-Fayos *et al*. (1998), for example, show that infiltration and water-holding capacity of soils on limestone are greater with increased frost activity. From this, they infer that increased temperatures could lead to increased surface or shallow runoff. Komescu, Erkan and Oz (1998) assess the implications of climate change for soil moisture availability in southeast Turkey, finding substantial reductions in availability during summer.

The probable effects on soil characteristics of a gradual eustatic rise in sea-level will vary from place to place depending on a number of local and external factors, and interactions between them (Brammer and Brinkman, 1990). In principle, a rising sea level would tend to erode and move back existing coastlines. In coastal lowlands which are insufficiently defended by sediment supply or embankments, tidal flooding by saline water will tend to penetrate further inland than at present, extending the area of perennially or seasonally saline soils.

Climate change such as uncharacteristic droughts or rainfall and flooding have detrimental influences on soil microorganisms, changing the natural growing conditions for a region (Gschwendtner, 2014).

Soil formation is strongly dependent on environmental conditions of both the atmosphere and the lithosphere. Soil temperature is an important factor in this physical, chemical and biological process. Soil temperature is also an important parameter for plant growth. For example, excessive high temperature is harmful to roots and causes lesions of stems, while extreme low temperatures impede intake of nutrients. Extreme low and high soil temperatures also influence the soil microbial population and the rate of organic matter decomposition. Recent studies have shown that soil temperature is one of the main climate factors that influence CO_2 emission. High soil temperatures accelerate soil respiration and thus increase CO_2 emission

(Brito *et al.*, 2005). This has implications for the landscape and for land use. On convex slopes and hilltops, emission is greater than in foothills, where temperatures are normally lower. Soil surface wetness and canopy cover strongly influence the soil's energy balance and soil temperature, whereas variation in soil porosity and soil thermal conductivity have little effect on soil temperature (Luo, Loomis and Hsiao, 1992). The various ways to calculate soil temperature and its trends are discussed in Alavipanah (2006) and Alavipanah *et al.* (2007).

References

Alavipanah, S.K. 2006. *Thermal Remote Sensing and Its Application in the Earth Sciences.* Iran, Tehran, Tehran University Press. 522 pp.

Alavipanah, S.K., Saradjian, M. Savaghebi, G.R. Komaki, C.B. Moghimi, E. & Reyhan, M.K. 2007. Land surface temperature in the Yardang region of Lut Desert (Iran) based on field measurements and Landsat thermal data. *J. Agric*. Sci., 9: 287–303.

Alexandratos, N. & Bruinsma, J. 2012. *World agriculture towards 2030/2050: the 2012 revision.* ESA Working paper No. 12-03. Rome, FAO

Allen, D.W. 1991. Homesteading and Property Rights; or, How the West Was Really Won. *Journal of Law and Economics*, 34: 1-23

Alston, L.J., Libecap, G.D. & Schneider, R. 1995. Property Rights and the Preconditions for Markets: The Case of the Amazon Frontier. *Journal of Institutional and Theoretical Economics*, 151(1): 89-111

Auffret, P. 2003. *Catastrophe Insurance Market in the Caribbean Region: Market Failures and Recommendations for Public Sector Interventions.* Policy Research Working Paper No. 2963. Washington, DC, World Bank

Barbier, E.B. 2013. Structural Change, Dualism and Economic Development: T*he Role of the Vulnerable Poor on Marginal Lands*. Washington, DC, World Bank

Besley, T. 1995. Property rights and investment incentives: Theory and evidence from Ghana. *Journal of Political Economy*, 103(5): 903-937.

Boix-Fayos, C., Calvo-Cases, A., Imeson, A.C., Soriano-Soto, M.D. & Tiemessen, I.R. 1998. Spatial and short-term temporal variations in runoff, soil aggregation and other soil properties along a Mediterranean climatological gradient. *Catena*, 33: 123–138

Bolton, M. 2003. Mine actions in Bosnia's special district: a case study. *Journal of Mine Action*, 7 (2)

Brammer, H. & Brinkman, R. 1990. Changes in soil resources in response to a gradually rising sea-level. In H.W. Scharpenseel, M. Schomaker & A. Ayoub, eds. *Soils on a Warmer Earth.* pp. 145-156. Elsevier Ltd

Brito, L.F., La Scala Jr, N., Merques Jr, J. & Pereira, G.T. 2005. Variabilidade temporal da emissão de CO_2 do solo e sua relação com a temperatura do solo em diferentes posições na paisgem em área cultivada com cana-de açúcar. *In Simpósio sobre Plantio direto e Meio ambiente. Seqüestro de carbono e qualidade da agua.* pp. 210-212. Anais. Foz do Iguaçu

Dayton-Johnson, J. 2006. *Natural disaster and vulnerability.* Policy brief No. 29. OECD

El Baz, F. & Mekharta, R.M. 1994. *The Gulf War and the Environment.* Amsterdam, Gordon and Breach

El Gamily, H. 2007. Utilization of multi-dates LAndsat-TM data to detect and quantify the environmental damages in the southeastern region of Kuwait from 1990 to 1991. *International Journal of Remote Sensing*, 28(8): 1773-1788

FAO. 2003. *World Agriculture: Towards 2015/2030: An FAO Perspective.* In J. Bruinsma, ed. UK, Earthscan.

FAO. 2010. *Gender and Land rights Database*. (Also available at *http://www.fao.org/gender/landrights/home/en/*)

FAO. 2011a. *World Livestock 2011* – Livestock in food security. Rome, FAO

FAO. 2011b. *The Role of women in Agriculture*. ESA Working Paper No. 11- 02, Agricultural Economics Division, Food and Agricultural Organization of the United Nations. Rome, FAO. 48 pp

FAO. 2013. FAO *Policy on Gender Equality: Attaining Food Security Goals in Agriculture and Rural Development*. Rome, FAO

Gerland, P., Raftery, A.E. (co-first authors); Ševčíková, H., Li, N., Gu, D., Spoorenberg, T., Alkema, L., Fosdick, B.K., Chunn, J.L., Lalic, N., Bay, G., Buettner, T., Heilig, G.K. & Wilmoth, J. 2014. World Population Stabilization Unlikely This Century. *Science*, 346: 234-237

Ghebru, H.H. & Holden, S.T. 2012. *Reverse share-tenancy and marshallian inefficiency: bargaining power of landowners and the sharecropper's productivity*. International Association of Agricultural Economists (IAAE) Triennial Conference, Foz do Iguaçu, Brazil

Gregory, J.M., Mitchell, J.F.B. & Brady A.J. 1997. Summer drought in northern midlatitudes in a time-dependent CO_2 climate experiment. *Journal of Climate*, 10: 662–686

Gschwendtner, S., Tejedor, J., Bimueller, C., Dannenmann, M., Knabner, I.K. & Schloter M. 2014. Climate change induces shifts in abundance and activity pattern of bacteria and archaea catalysing major transformation steps in nitrogen turnover in a soil from a mid-European beech forest. PLoS ONE, 9(12): e 114278.

Holdinghausen, H. 2015. Big Business: Fighting back against foreign acquisitions. In Heinrich Böll Foundation, ed. Soil Atlas 2015- *Facts and Figures about earth, land and fields*. pp. 42-43. Institute for Advanced Sustainability Studies (IASS).

IAASTD. 2009. *Agriculture at as crossroad. International assessment of agricultural knowledge, science and technology for development*. Global Report. Washington, DC, Island Press.

ICBL. 2002. *Bosnia Herzegovina Landmine Monitor Report 2002.* International Campaign to Ban Landmines.

IMF. 2008. *Food and Fuel Prices, Recent Developments, Macroeconomic Impact and Policy responses*. Washington, DC, International Monetary Fund.

IPCC. 2014a. *Climate Change 2014: Synthesis Report*. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Switzerland, Geneva, IPCC. 151 pp

IPCC. 2014b. *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects*. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. UK, Cambridge, Cambridge University Press & USA, New York. 1132 pp.

Jones, A., Breuning-Madsen, H., Brossard, M., Dampha, A., Deckers, J., Dewitte, O., Gallali, T., Hallett, S., Jones, R., Kilasara, M., Le Roux, P., Micheli, E., Montanarella, L., Spaargaren, O., Thiombiano, L., Van Ranst, E., Yemefack, M. & Zougmore, R. 2013. *Soil Atlas of Africa*. Luxembourg, European Commission, Publications Office of the European Union. 176 pp

Kobayashi, A. (ed.) 2013. *Geographies of Peace and Armed Conflict*. Routledge. 248 pp

Komescu, A.U., Erkan, A. & Oz, S. 1998. Possible impacts of climate change on soil moisture availability in the Southeast Anatolia Development Project Region (GAP): an analysis from an agricultural drought perspective. *Climatic Change*, 40: 519–545

Land Matrix Newsletter. Land Matrix Newsletter. October 2014. (Also available at *https://www.landmatrix.org*)

Laurance, W.F., Sayer, J. & Cassman, K. 2013. Agricultural expansion and its impacts on tropical nature. *Trends in Ecology & Evolution*, S 0169-5347(13): 00292-9.

Laurance, W.F., Sayer, J. & Cassman, K.G. 2014. Agricultural expansion and its impact on tropical nature. *Trends in Ecol. Evol.*, 29: 107-116

Luo, Y., Loomis, R.S. & Hsiao, T.C. 1992. Simulation of soil temperature in crops. *Agricultural and Forest Meteorology*, 61(1): 23-38

MA. 2005. *Ecosystems and Human Well-being: Synthesis*. A report of the Millennium Ecosystem Assessment. Washington, DC, Island Press

Misak, R., Al-Ajmi, D. & Al-Enezi, A. 2009. War-Induced Soil Degradation, Depletion, and Destruction (The Case of Ground Fortifications in the Terrestrial Environment of Kuwait). In T.A., Kassim, & D., Barceló, eds. *Environmental Consequences of War and Aftermath* pp. 125-139. Springer Berlin Heidelberg

Misak, R.F., Al-Awadhi, J.M., Omar, S.A. & Shahid, S.A. 2002. Soil degradation in Kabd area, southwestern Kuwait City. *Land degradation & development*, 13(5): 403-415

Mitchell, S.K. 2004. Death, disability, displaced persons and development. The case of landmines in Bosnia and Herzegovina. *World Development*, 32(12): 2105-2120

Nolte, K. & Ostermeier, M. 2015. Land Investments: A new type of territorial expansion. In Heinrich Böll Foundation, ed. Soil Atlas 2015- *Facts and Figures about earth, land and fields.* pp. 38-39. Institute for Advanced Sustainability Studies (IASS).

OECD/FAO. 2011. OECD-FAO *Agricultural Outlook* 2011-2020. OECD Publishing & FAO.

Owona, J.C. 2008. *Land degradation and internally displaced person's camps in Pader District—Northern Uganda*. Pader Pader District Local Government. (Also available at *http://www.unulrt.is/static/fellows/document/joel.pdf*

Pinstrup-Andersen, P. & Watson II, D.D. 2011. *Food Policy for Developing Countries: The Role of Government in Global, National, and Local Food Systems*. Cornell University Press.

Rulli, M.C., Saviori, A. & D'Odorico, P. 2013. Global land and water grabbing, Proc. Natl. Acad. Sci., 110(3): 892-897

Sengupta, K. 1997. Limited liability, moral hazard and share tenancy. Journal of Development Economics, 52: 393-407

Shames, S. 2013. *How can small-scale farmers benefit from carbon markets?* CCAFS Policy Brief no. 8. Copenhagen, Denmark, CGIAR Research Program on Climate Change, Agriculture and Food Security (CCAFS).

Soil Atlas. 2015. *Soil Atlas: Facts and Figures about earth, land and fields*. Berlin, Institute for Advanced Sustainability Studies,Heinrich-Böll-Stiftung

Stephens, G. & Matson, M. 1993. Monitoring the Persian Gulf War with NOAA AVHRR data. *International Journal of Remote Sensing*, 14 (7): 1423-2

Turral, H., Burke, J. & Faurès, J.-M. 2011. *Climate change, water and food security.* FAO Water Report 36, Rome, FAO

UNEP. 2007. *Global Environmental Outlook*, GEO 4. Nairobi & New York, UNEP.

UNEP. 2012. *Global Environmental Outlook*: Fifth Edition. Nairobi & New York, UNEP.

United Nations. 2014. *World Urbanization Prospects: The 2014 Revision, Highlights.* Department of Economic and Social Affairs, Population Division, ST/ESA/SER.A.352

World Bank/FAO/IFAD. 2009. *Gender in Agriculture Sourcebook*. Washington, DC, World Bank.

6 | Global soil status, processes and trends

6.1 | Global status, processes and trends in soil erosion

6.1.1 | Processes

Soil erosion is broadly defined as the accelerated removal of topsoil from the land surface through water, wind or tillage. Water erosion on agricultural land occurs mainly when overland flow entrains soil particles detached by drop impact or runoff, often leading to clearly defined channels such as rills or gullies. Wind erosion occurs when dry, loose, bare soil is subjected to strong winds. Wind erosion is common in semi-arid areas where strong winds can easily mobilize soil particles, especially during dry spells. This dynamic physical aeolian process includes the detachment of particles from the soil, transport for varying distances depending on site, particle and wind characteristics, and subsequent deposition in a new location, causing onsite and offsite effects. During wind erosion events, larger particles creep along the ground or saltate (bounce) across the surface until they are deposited relatively close to field boundaries (Hagen *et al.*, 2007). Finer particles (< 80 µm) can travel great distances, with the finest particles entering global circulation (Shao, 2000). Tillage erosion is the direct down-slope movement of soil by tillage implements where particles only redistribute within a field.

6.1.2 | Status of Soil Erosion

Over the last decade, the figures published for water erosion range over an order of magnitude of ca. 20 Gt (gigaton) yr^{-1} to over 200 Gt yr^{-1}. While this huge variation may at first seem to suggest that our estimates of global soil erosion are very uncertain, a more detailed analysis shows that estimates exceeding ca. 50 Gt yr^{-1} are not realistic. In most cases, excessively high estimates can be traced back to conceptually flawed approaches and/or inappropriate model applications. Considering only those estimates that are not manifestly affected by such problems, the most likely range of global soil erosion by water is 20–30 Gt yr^{-1}, while tillage erosion may amount to ca. 5 Gt yr^{-1}.

Total erosion rates for wind erosion are highly uncertain. Estimates of the total amount of dust that is yearly mobilized on land place an upper limit on dust mobilization by wind erosion on arable land at ca. 2 Gt yr^{-1}. However, wind not only mobilizes dust but also coarser soil particles (sand), implying much higher total wind erosion rates. A large number of studies have made global estimates of wind erosion and dust transport. Approximately 430 million ha of drylands, which comprise 40 percent of the Earth's surface (Ravi *et al.*, 2011), are susceptible to wind erosion (Middleton and Thomas, 1997). In a survey of global estimates of present-climate dust emissions, Shao *et al.* (2011) described 13 studies that estimated global dust emissions in a range from 500 to ~ 3320 Tg yr^{-1}. Ginoux *et al.* (2012). The studies used global-scale high-resolution satellite imagery to study dust sources. They found that natural dust sources do account for about 75 percent of dust emissions and the remaining 25 percent of emissions were attributed to anthropogenic sources. The fraction of dust sources was highly variable. For example, although North Africa accounted for about 55 percent of the global dust emissions, only 8 percent originated from anthropogenic sources. In contrast, anthropogenic dust sources contributed 75 percent of the dust emissions in Australia (Ginoux *et al.*, 2012).

Translating these global estimates into accurate local soil erosion rates is not straightforward as soil erosion is highly variable, both in time and in space. However, typical soil erosion rates by water can be defined for representative agro-ecological conditions. Hilly croplands under conventional agriculture and orchards without additional soil cover in temperate climate zones are subject to erosion rates up to 10-20 tonnes ha^{-1} yr^{-1} , while average rates are often < 10 tonnes ha^{-1} yr^{-1}. Values during high-intensity rainfall events may reach 100 tonnes ha^{-1} and lead to muddy flooding in downstream areas. Erosion rates on hilly croplands in tropical and subtropical areas may reach values up to 50-100 tonnes ha^{-1} yr^{-1}. Average rates, however, are lower and often 10-20 tonnes $ha^{-1}yr^{-1}$. These high rates are due to the combination of an erosive climate (high intensity rainfall) and slope gradients which are generally steeper than those on cultivated land in the temperate zones. The incidence of erosion on steep slopes is due not only to specific topographic conditions, but also to the combination of a high population pressure with low-intensity agriculture, leading to the cultivation of marginal steeplands.

Rangelands and pasturelands in hilly tropical and sub-tropical areas may suffer erosion rates similar to those of tropical croplands. Due to the lack of field boundaries, which often act as barriers for sediment and runoff and promote infiltration, these rangelands may also be particularly vulnerable to gully formation. This may not affect topsoil so much but may make land inaccessible and hence unusable. Rangelands and pasturelands in temperate areas are characterized by erosion rates which are generally much lower and are most often below 1 tonnes ha^{-1} yr^{-1}. These rangelands are less intensively used and better managed than (sub-) tropical rangelands.

It is possible to identify the areas in the world where soil erosion by water is problematic based on a relatively simple modelling approach combining information on soil type, land use, topography and climate (Doetterl, Van Oost and Six, 2012; Van Oost *et al.*, 2007). Soil erosion by water is problematic in much of the hilly areas that are used as croplands on all continents, even where there have been significant conservation efforts as in the Mid-West of United States. Cropland in Europe is characterized by somewhat lower, yet still very significant soil erosion rates (Figure 6.1).

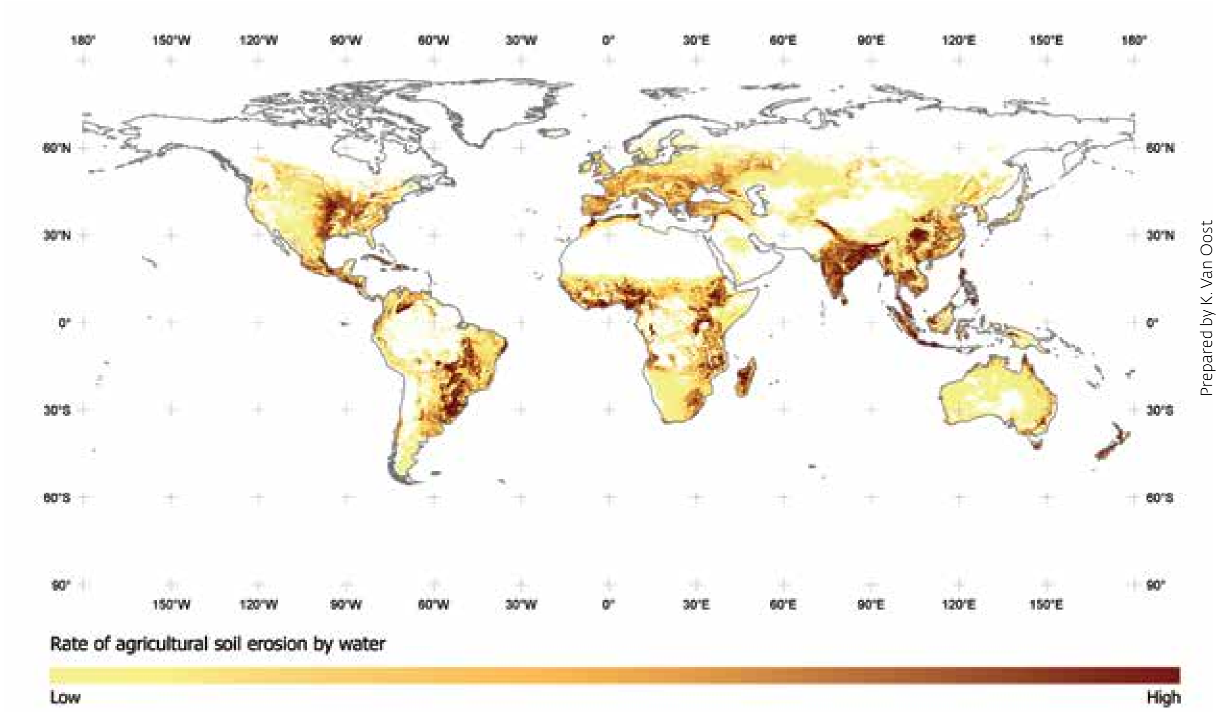


Figure 6.1 | Spatial variation of soil erosion by water. High rates (>ca. 20 t $ha^{-1}y^{-1}$) mainly occur on cropland in tropical areas.

The map gives an indication of current erosion rates and does not assess the degradation status of the soils. The map is derived from Van Oost et al., 2007 using a quantile classification.

The redistribution of soil within fields due to tillage erosion rates may lead to (very) high erosion rates on convexities (knolls) exceeding 30 tonnes ha^{-1} yr^{-1}; and to deposition rates in hollows and at down slope field borders exceeding 100 tonnes ha^{-1} yr^{-1}. These rates are not directly comparable to wind or water erosion rates, as soil eroded by tillage will not leave the field. However, tillage erosion may significantly reduce crop productivity on convexities and near upslope field or terrace borders.

Evidence of past erosion is extensive. This is demonstrated by wind-blown sands of sandstone bedrock, extensive loess accumulations of silt-sized aeolian sediments, and other formerly aeolian-affected landscapes. Large areas of sand seas, dune fields and other aeolian features and observations of activity provide further evidence of past wind erosion (Figure 6.2).

USDA estimates place wind erosion rates at ca. 2.5 tonnes ha^{-1} yr^{-1} on average over all cropland of the United States while the average erosion rate for pastureland is ca. 0.1 tonnes ha^{-1} yr^{-1}. There are very few quantitative assessments of wind erosion rates on arable land outside of the United States.

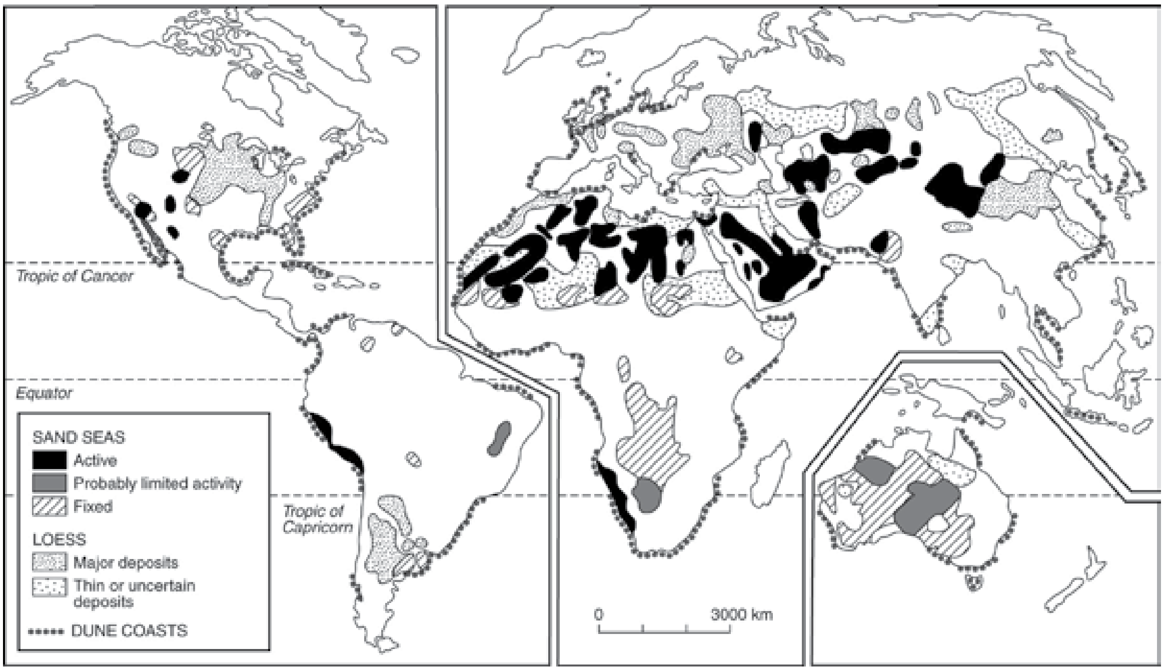


Figure 6.2 | Location of active and fixed aeolian deposits.
Source: Thomas and Wiggs, 2008.

6.1.3 | Soil erosion versus soil formation

The accelerated loss of topsoil through erosion from agricultural land was recognized as an important threat to the world's soil resource many decades ago. Furthermore, it was feared that soil was, in many areas, eroding much faster than that it could be replaced through soil formation processes. More recent studies have confirmed that these early observations were not just perceptions. Estimated rates of soil erosion of arable or intensively grazed lands have been found to be 100-1000 times higher than natural background erosion rates. These erosion rates are also much higher than known soil formation rates which are typically well below 1 tonnes ha^{-1} yr^{-1} with median values of ca. 0.15 tonnes ha^{-1} yr^{-1}. The large difference between erosion rates under conventional agriculture and soil formation rates implies that we are essentially mining the soil and that we should consider the resource as non-renewable.

The imbalance between erosion rates under conventional agriculture and the rate of soil formation implies that conventional agriculture on hilly land is not sustainable because the soil resource is mined and will ultimately become depleted. This has most likely already happened in many areas around the Mediterranean Sea and in tropical mountain regions. So-called soil loss tolerance levels may help to set objectives for short-term action. However, long-term sustainability requires that soil erosion rates on agricultural land are reduced to near-zero levels.

Figure 6.3 | Soil relict in the Jadan basin, Ecuador.
Photo by G. Govers

In this area overgrazing led to excessive erosion and the soil has been completely stripped from most of the landscape in less than 200 years, exposing the highly weathered bedrock below. The person is standing on a small patch of the B-horizon of the original soil that has been preserved. Picture credit: Gerard Govers.

6.1.4 | Soil erodibility

Soil erodibility refers to the degree or intensity of a soil's state or susceptibility to being eroded (SSSA, 2008). A critical review of research into the factors controlling susceptibility of soils to wind erosion ('soil wind erodibility') has been provided by Webb and Strong (2011). Factors controlling soil wind erodibility include physical, chemical and biological characteristics of the soil, including texture, aggregation, stability, crusting, the amount of loose erodible sediment available, soil water content, roughness due to surface features (including tillage marks and vegetation) etc. The factors controlling soil wind erodibility differ somewhat among land uses and management approaches. For example, the factors controlling erodibility on rangeland differ from the factors controlling erodibility on farmland. In cropped soils at the field scale, disturbance due to tillage modifies the soil surface roughness, the amount and distribution of surface cover, soil water content aggregation and other properties, all of which affect soil erodibility for short periods of time (Zobeck, 1991; Zobeck and Van Pelt, 2011). In arid and semiarid rangeland ecosystems, wind erosion depends on vegetative cover and patchiness (Okin *et al.*, 2009) and on surface soil texture and crusting, characteristics that change more slowly unless disturbed. Not only do the factors controlling erosion by wind differ among land uses, but differences occur in their spatial and temporal variability. Natural and anthropogenic disturbances such as grazing, fire and other activities alter the surface and vegetation on rangeland while crop management practices often control erodibility of farmland. Webb and Strong (2011) described the dynamics of soil erodibility as a continuum that responds to changes in climate variability and disturbance. Factors such as rain and crusting on some soils may initially produce low erodibility that will subsequently increase with disturbance and drying. The exact timing and variability of changes in erodibility will vary with inherent soil physical properties such as soil texture.

6.1.5 | Soil erosion and agriculture

Soil erosion has direct, negative effects for global agriculture. Soil erosion by water induces annual fluxes of 23-42 Mt (megaton) N and 14.6-26.4 Mt P off agricultural land. These fluxes may be compared to annual fertilizer application rates, which are ca. 112 Tg for N and ca. 18 Tg of P. These nutrient losses need to be replaced through fertilization at a significant economic cost. Using a United States farm price of ca. US$ 1.45 per kg of N and ca. US$ 5.26 per kg of P implies an annual economic cost of US$ 33-60 billion for N and US$ 77-140 billion for P[1]. It is therefore clear that compensation for erosion-induced nutrient losses requires a massive investment in fertilizer use. In poor regions such as sub-Saharan Africa, the economic resources to achieve compensations for nutrient losses do not exist. As a consequence, the removal of nutrients by erosion from agricultural fields is much higher than the amount of fertilizer applied.

The detrimental removal of soil and nutrients from upland fields may be partly offset through the deposition of the eroded soil and nutrients in depositional areas. While this is true, such gains should not be exaggerated: the deposition of sediments and nutrients in large floodplains is not directly coupled to actual agricultural soil erosion, as in most cases sediments are provided by other sources (natural erosion, landslides) and the residence time of such sediments in large river systems is several thousands of years. In other words: the sediments that are currently being deposited in the Nile valley are not coming from the soils that are currently being eroded in Ethiopia. On a smaller scale, the deposition of eroded sediment may indeed locally increase local crop productivity, but such effects may be overshadowed by other factors, such as water availability.

Soil erosion does not induce an important carbon loss from the soil to the atmosphere: erosion mostly induces a transfer of carbon from eroding locations to depositional locations. Net losses are limited as the carbon lost at eroding locations is partially replaced through dynamic replacement whereas the soil carbon that is deposited in colluvial and alluvial settings may be stored there for several centuries.

1 www.ers.usda.gov/data-products/fertilizer-use-and-price.aspx#26727

High soil erosion rates will also have significant negative effects over longer time spans: the loss of topsoil will result in a reduction in the soil's capacity to provide rooting space and, more importantly, in the capacity to store water that can be released to plants. This may reduce soil productivity. However, these changes occur relatively slowly: the reduction in soil water holding capacity and and/or root space accommodation results in yield declines of ca. 4 percent per 0.1m of soil lost. Except for areas where erosion rates are very high (e.g. exceeding 50 tonnes ha^{-1} yr^{-1} or ca. 4 mm yr^{-1}) this means that effects of erosion on crop productivity will be relatively small on the decennial or centennial time scale, provided that nutrient losses due to erosion are compensated. Over longer time spans, however, the effect of these losses becomes very significant.

On the positive side, transported dust affects distant ecosystems, increasing plant productivity by providing nutrients not provided by the parent soil, as seen in Hawaii (Chadwick *et al.*, 1999) and the Amazon (Mahowald *et al.*, 2008). Transported dust can also provide chemical constituents that affect soil development, as seen in the development of terra rossa soils in Bermuda and Spain (Muhs *et al.*, 2010, 2012).

6.1.6 | Soil erosion and the environment

The direct negative effects of soil erosion are not limited to agriculture. The sediment produced by erosion also pollutes water streams with sediment and nutrients, thereby reducing water quality. In addition, sediment contributes to siltation in reservoirs and lakes. However, not all sediments trapped in reservoirs originate from agricultural land. Other processes such as bank erosion, landslides and natural surface erosion which contribute to reservoir sedimentation are also very important and are often dominant at large scales.

Wind erosion and dust transport have been studied for many years. For example, in 1646, Wendelin first described purple rain in Brussels that we now recognize as coloured dust transported to Europe from Africa (Wendelin, 1646 as cited in Stout, Warren and Gill, 2009). Charles Darwin studied dust that fell on the HMS Beagle in the 1830s and 1840s (Darwin, 1845, 1846) and the dust collected was found to contain viable microbes even today (Gorbushina *et al.*, 2007).

Wind erosion can originate from natural landscapes and from landscapes affected by anthropogenic (human-related) activities (Figure 6.4). Aeolian processes impact soil development, mineralogy, soil physical and biogeochemical properties, and redistribution of soil nutrients, organic materials, and sequestered contaminants. Wind erosion also affects landscape evolution, plant productivity, human and animal health (Ravi *et al.*, 2011), atmospheric properties including effects on solar radiation and cloud attributes (Shao *et al.*, 2011), air quality, and other factors (Field *et al.*, 2010; Ravi *et al.*, 2011). The effects of wind erosion occur at the field, landscape, regional, and global scales.

Figure 6.4 | Dust storm near Meadow, Texas, USA

At the field and landscape scales, wind erosion winnows the finer and more chemically active portion of the soil which carries biogeochemicals, including plant nutrients, soil carbon and microbial products. In some cases, wind erosion processes modify the surface properties by causing increases in sand content while reducing the soil water holding capacity and plant productivity (Zobeck and Van Pelt, 2011). Although some of this eroded sediment is deposited relatively close to field boundaries, often much of it enters into suspended mode and may be transported high in the atmosphere to travel great distances. This long-range transport of dust produces effects at the global and regional scales Atmospheric dust produced by wind erosion profoundly influences the energy balance of the Earth system by carrying organic material, iron, phosphorus and other nutrients to the oceans, affecting ocean productivity and subsequent ocean-atmosphere CO_2 exchange (Shao *et al.*, 2011).

6.1.7 | Effects of hydrology and water

Wind erodibility and subsequent erosion and dust emissions are affected by hydrology and water in several ways. Remote sensing studies of dust sources by Prospero *et al.* (2002) showed that many major dust sources originate from deep alluvial deposits formed by intermittent flooding during the Quaternary and Holocene. These sources, now in drylands, originated when water was more plentiful and produced an ample supply of wind-erodible sediment (Ginoux *et al.*, 2012). In many areas, particularly in areas with more limited erodible sediment supply, dust emissions increase after new inundations of ephemeral water supplies provide additional erodible sediment. However, many fluvial-related dust sources have also developed from the exposure, due to anthropogenic factors, of the bottoms of former lakes such as at Owens Lake in the United States (Reheis, 1997) and the Aral Sea Basin in Uzbekistan (Singer *et al.*, 2003). In these cases, usually water has been extracted from the lake for irrigation or human consumptive needs. This issue will be accentuated as increasing demand for water in dryland regions is met from reservoirs.

Near-surface soil water content has long been recognized to have a significant effect on the threshold wind velocity needed for wind erosion (Akiba, 1933; Chepil, 1956). Soil water acts to bind particles together to resist the shearing force of wind on the particles. In addition, soil water affects vegetative growth, which also affects wind erosion. Research has shown a time-dependent change in the controlling factors for sediment emission and transport from soil water to wind speed (Wiggs, Baird and Atherton, 2004). The change of controlling factors was found to be very sensitive to the prevailing water conditions and, for the sandy soil tested, took place in a very short period of time. They found the soil water content where wind erosion commenced was between 4 and 6 percent (Wiggs, Baird and Atherton, 2004). However, the effect of soil water on wind erosion of dry soils is also sensitive to changes in air relative humidity (Ravi *et al.*, 2006). Recent work on atmospheric dust concentrations have confirmed this sensitivity, finding that dust concentration increased with relative humidity, reaching a maximum around 25 percent and thereafter decreased with relative humidity (Csavina *et al.*, 2014). Climate-induced changes in hydrology and water may produce profound changes in wind erosion and dust emissions as the soil erodibility is altered.

6.1.8 | Vegetation effects

The effect of vegetation on wind erosion is complex. In native conditions, the wind influences patterns of vegetation and soils and these patterns, in turn, affect wind erosion at patch to landscape scales (Okin, Gillette and Herrick, 2006; Okin *et al.*, 2009; Munson, Belnap and Okin, 2011). In agricultural systems, the vegetation is manipulated by managers and its effects vary spatially and temporally from non-managed systems. The protective effects of vegetation are well known. A wide variety of methods and models has been devised to describe the protective effects of vegetation. In general, as vegetation height and cover increase, wind erosion of erodible land decreases. Vegetation affects wind erodibility by: (1) acting to extract momentum from the wind and thereby reducing the wind energy applied to the soil surface; (2) directly sheltering the soil surface from the wind by covering part of the surface and reducing the leeside wind velocity; and (3) trapping windborne particles, so reducing the horizontal and vertical flux of sediment (Okin, Gillette and Herrick, 2006). Trapping of sediment leads to redistribution of nutrients and modifies surface soil properties such as water infiltration rate and soil bulk density.

Vegetation cover affects nutrient removal, which in turn affects plant productivity. A study of the effects of grass cover on wind erosion in a desert ecosystem found increased wind erosion removed 25 percent of the total soil organic carbon and nitrogen from the top 5 cm of soil after only three windy seasons (Li *et al.*, 2007). Studies of agricultural crops on severely eroded cropland found 40 percent reductions in cotton and kenaf yields and 58 percent reduction in grain yield in sorghum (Zobeck and Bilbro, 2001). The eroded areas in this study had statistically significantly less phosphorus than the adjacent non-eroded areas. Climatic changes that reduce the cover of vegetation in drylands will increase wind erosion and dust emissions, and likely result in increased soil degradation and reduced plant productivity.

6.1.9 | Alteration of nutrient and dust cycling

Recognition of a dust cycle, along with other important cycles such as the energy, carbon and water cycles, has become an emerging core theme in Earth system science (Shao *et al.*, 2011). Dust cycles are dependent upon the soil and climate systems within which they operate. The dust cycle is a product, in part, of the soil system. As dust is transported globally, it interacts with other cycles by participating in a range of physical and biogeochemical processes. The dust carries important nutrients to otherwise sterile soils and so may improve productivity (Chadwick *et al.*, 1999; Mahowald *et al.*, 2008). Dust may also transport soil parent material (Reynolds *et al.*, 2006), trace metals (Van Pelt and Zobeck, 2007), soil biota (Gardner *et al.*, 2012) and toxic anthropogenics (Larney *et al.*, 1999) among ecosystems. Although the fact is not widely recognized, the global dust cycle is intimately tied to the global carbon cycle (Chappell *et al.*, 2013). Wind and water erosion both redistribute soil organic carbon within terrestrial, atmospheric and aquatic ecosystems. This carbon is selectively removed from the soil. This was recently demonstrated in an Australian study where the soil organic carbon in dust was from 1.7 to over seven times that of the source soil (Webb *et al.*, 2012). Changing climate will alter these cycles, producing complex and uncertain environmental effects.

6.1.10 | Trends in soil erosion

While rates of soil erosion are still very high on extensive areas of cropland and rangeland, erosion rates have been significantly reduced in several areas of the world in recent decades. The best documented example is the reduction of erosion rates on cropland in the United States. Average water erosion rates on cropland were reduced from 10.8 to 7.4 tonnes ha^{-1} yr^{-1} between 1982 and 2007, while wind erosion rates reduced from 8.9 to 6.2 tonnes ha^{-1} yr^{-1} over the same time span. Another example is the reduction of soil erosion in Brazil through the application of no-tillage from ca. 1980 onwards. This is estimated to have led to a reduction of erosion rates by 70-90 percent over large parts of Brazilian cropland. Studies have shown that erosion rates can be greatly reduced in nearly every situation through the application of appropriate management techniques and structural measures such as terrace and waterway construction (see, for example, Pansak *et al.*, 2008).

However, in many areas of the world, adoption of soil conservation measures is slow. While the reasons for this are diverse, a key point is that the adoption of soil conservation measures is generally not directly beneficial to farmers. This is as true in intensive mechanized systems in the West as it is for smallholder farming in the developing world. This is not surprising: the implementation of conservation measures does not, as such, directly increase yields or efficiencies while the detrimental effects of erosion on the soil capital only become visible over time scales that range from decades to centuries. Hence, farmers do not have a direct incentive to adopt soil conservation measures.

In some cases, this problem may be overcome through financial incentives or by regulation. It is clear, however, that this is not always possible. We need, therefore, to rethink our vision on soil conservation. Essentially, further adoption of soil conservation measures will not in the first place depend on refinement or optimization of technologies. Technology already exists to reduce erosion to acceptable levels under most circumstances. What is critically important is to work out how to incorporate soil conservation in an agricultural system that, as a whole, increases the net returns of farmers. In developing approaches that build in incentives to soil conservation, it is vital to account for local conditions, including the extent to which local markets can provide incentives to sustainable agriculture.

The potential for agricultural intensification is key here. In many areas around the world, crop yields are low and more land is cultivated than is strictly necessary. As a result, large tracts of steep, marginal land are at present used for agriculture without the implementation of proper soil conservation technology, with the result that these areas are subject to high erosion rates. Intensification of production on higher potential land is an option. This not only reduces extension into marginal, highly erodible areas but may also benefit biodiversity and overall carbon storage at the landscape scale.

Erosion can also be checked by forestation. In many areas there is now a net gain of forest area. This reforestation, which is largely of marginal land, is related to four main factors: agricultural intensification; diminishing need for firewood; an increase in exchange and trade making it possible to grow products in the most suitable areas; and an increased public awareness of the problems caused by deforestation. Development of conservation policies should consider these tendencies and stimulate them wherever possible.

6.1.11 | Conclusions

Soil erosion has been recognized as a main problem threatening the sustainability of agriculture for a long time and the magnitude of the problem can now be correctly quantified. The technology to reduce erosion now exists and, over the last decades, significant efforts have been to reduce erosion rates. These efforts have been partially successful. However, erosion rates are still high on much of the agricultural land of the globe, and this is related to the lack of economic incentives for today's farmers to conserve the soil resource for future generations. Tackling this problem requires the soil erosion problem to be reframed. Solutions need to be embedded in policies and programs that support the development of more sustainable agricultural systems.

6.2.1 | Introduction

An evaluation of the various functions of carbon (C) stored in the soil and its role in the global C-cycle require knowledge of the amount and geographic distribution of C stored in the soil. The functions of soil C are determined by the chemical and physical properties of the components that contain C. Chemical properties of soil C determine properties such as soil pH, nutrient storage and availability and regulating functions affecting the water cycle. Soil physical properties related to functions of C are: soil structure, particle agglomeration, and stability. These properties in turn influence water infiltration rates and resistance to water and wind erosion.

Soil C is separated into: (i) inorganic chemical substances (soil inorganic carbon: SIC), mainly carbonates; and (ii) C as part of organic compounds (soil organic carbon: SOC). The amount of SIC in the first one meter of soil was estimated at 695 - 748 Pg carbonate C (Batjes, 1996). The C stored as SOC is about twice the C stored as SIC (1 502 Pg C; Jobbágy and Jackson, 2000). Carbonates are less susceptible to react to anthropogenic changes to the environment than are organic compounds. In addition, the amount and type of organic C compounds are interdependent with environmental conditions, such as land use and management practices. These two characteristics have led to the definition of the persistence of SOC as an ecosystem property (Schmidt *et al.*, 2011). Thus, assessments of soil C stocks and their spatial distribution often concentrate on SOC alone.

Although SOC mainly originates from plant material there is no simple correlation between the amount of C stored in the above-ground plant material and the SOC stocks (Amundson, 2001; Smith, 2012). In fact, the processes involved in decomposing organic material and their mineralization are complex and details are not yet fully understood (Schmidt *et al.*, 2011). However, the effects on SOC of anthropogenic activities of land use change and management practices are known. Given the large amount of C stored in soils and the possibility of influencing this amount through land management to act as a source or sink for atmospheric C, strategies for maintaining SOC have been devised. These strategies follow two main approaches: (i) seeking to enhance existing SOC by increasing the amount of biomass of the terrestrial biosphere; (ii) seeking to decrease the loss of SOC by reducing the respiration rate (Smith *et al.*, 2008). To provide a quantitative appraisal of the possible gains or losses in SOC from measures taken either to increase the input of organic material or decrease losses of soil organic matter (SOM), an assessment of the current situation is needed.

Studies on historic developments in SOC stocks concentrate on the effect of changes in land use. These changes mainly concern the transformation of land from a natural state to an agricultural ecosystem, which in fact now covers more than one third of the global terrestrial area. For the conversion of forest to cropland, losses in SOC stocks of 25-30 percent were observed for temperate regions, with higher losses recorded for the tropics. Estimates of future trends in SOC stocks concentrate on the effect of changing climatic conditions on rates of organic matter accumulation and decomposition. As options for changes in land use are relatively limited, approaches to mitigation of climate change effects have focussed largely on management practices.

6.2.2 | Estimates of global soil organic carbon stocks

It is important to know past, current and likely future SOC stocks because of their importance to climate change and food security. When assessing the amount of C stored in the soil, studies often concentrate on C contained in dead and decomposed organic material or in organic matter located within the soil profile to a given depth and for a specific area. The mass of C stored in the SOM is also termed SOC stocks. SOC stocks are computed as a function of organic C-content, bulk density, depth and the amount of soil remaining after removing the volume taken up by coarse fragments in a unit of volume. Any of these factors can introduce uncertainties to the estimates.

Global estimates of SOC stocks have been published for many decades. One of the earliest estimates was published in 1951 (Rubey, 1951), indicating a global SOC stock of 710 Pg C. This estimate remained current for 25 years until the FAO-UNESCO soil map became available. Analysis of the map data led to a much higher estimate of 3 000 Pg C in the soil (Bohn, 1976). Several studies of global SOC stock followed with varying estimates (Bohn, 1982: 2 200 Pg organic C; Post *et al.*, 1982: 1 395 Pg organic C) An estimate of 1 576 Pg of SOC to 1 m depth was put forward by Eswaran, Van Den Berg and Reich, 1993. Global SOC stocks to 1 m depth of 1 462 – 1 548 Pg of SOC were estimated by Batjes, 1996. An estimate of 1 502 Pg organic C for the first 1 m of soil is often used (Jobbágy and Jackson, 2000; Batjes, 2002). The estimate of 1 500 Pg of SOC for the top 1 m of soil was adopted by the IPCC (IPCC, 2000). Current global estimates, derived from the Harmonized World Soil Database (HWSD; FAO/IIASA/ISRIC/ISS-CAS/JRC, 2009), suggest that approximately 1 417 Pg of SOC are stored in the first meter of soil and about 716 Pg organic C in the top 30 cm (Hiederer and Köchy, 2011).

Fewer estimates of global SOC stock estimates are available for a depth below 1 m. Global SOC stocks to a depth of 3 m are estimated at 2 344 Pg of SOC (Jobbágy and Jackson, 2000) or 3 000 Pg of SOC (Jansson *et al.*, 2010). Recent estimates of SOC in Cryosols may further increase the estimates of global SOC stocks (Tarnocai *et al.*, 2009).

In a comparison of 27 studies on global SOC stock published between 1951 and 2011, the estimates published were found to range from 504 to 3 000 Pg of SOC (Scharlemann *et al.*, 2014). The median of all published estimates is 1 460 Pg of SOC to 1 m depth. Spatially explicit estimates were found to span over 1 965 Pg of SOC. Large uncertainties over SOC stocks concern Histosols since soil data are often limited to a depth of 1 m (Eswaran, Van Den Berg and Reich, 1993). Particularly affected are the soils of the Arctic (Tarnocai *et al.*, 2009) and peatlands in South Asia (Couwenberg, Dommain and Joosten, 2010). The range in the estimates of global SOC stocks correspond to or exceed the amount of C held in the atmosphere, which was estimated at 720 Pg C (Falkowski *et al.*, 2000) and at 820 Pg C for present conditions (Mackey *et al.*, 2013).

With respect to the uncertainty in the estimates of global SOC stocks, various approximations are observed. For an estimated SOC stock of 1 395 Pg of SOC Post *et al.* (1982) assume a standard deviation of ± 200 Pg organic C, provided that the SOC density data are the only source of uncertainty. For the estimate of 1 502 Pg organic C to 1 m depth, Jobbágy and Jackson (2000) suggested an error of the mean of ± 320 Pg C at 1 standard deviation, provided that the SOC content data are the only source of uncertainty. The different assumptions on the causes of uncertainty between the studies (SOC density or content) are quite significant. Based on the HWSD, Todd-Brown *et al.* (2013) provide an interval of estimated global SOC stock of 890 to 1 660 Pg of SOC to a depth of 1 m with a 95 percent confidence level. This range corresponds to approximately 385 Pg SOC at 2 standard deviations from the mean.

With a small number of large-scale data sets available, the variations in SOC stock estimates may be attributed to the analysis method applied as much as to the data used. It also implies that various global SOC stock estimates are not independent and that the variability in the estimates could not necessarily be reduced by an increase in the number in such estimates.

One problem common to all large databases is that the properties were assessed decades ago and stretched over long periods. For example, the DSMW or the ESDB, of which components are included in the HWSD, originated from data collected during the 1950s and 1960s. With the dependence of SOC on climatic conditions and anthropogenic activities, SOC stocks established decades apart are likely to represent significantly different levels, notably in areas where changes in land use or management occurred, such as conversion of natural grassland and forest to agricultural land or urban areas. In extreme cases draining peatlands can lead to a loss of organic material to the degree that the soil no longer qualifies as peat because the organic C content decreases below 12 percent content and the thickness of the remaining organic layer is less than 40 cm (FAO/ISRIC/ISSS, 1998). An example of this change is given by the agricultural areas in north-eastern Netherlands, where the drainage in the 1960's of areas previously classified as peatland caused the SOC content to fall to 7.5 percent (Panagos *et al.*, 2013). Without further adjustments of SOC stock estimates to take account of local changes in the factors that influence SOC, no clear timestamp can be attached to the global estimates. This lack of a clear timestamp of SOC stocks is of consequence when estimating temporal changes in SOC stocks. Estimates of changes in SOC stock therefore concentrate on modelling variations in SOC from changes in land use and cover.

6.2.3 | Spatial distribution of SOC

Different methods of combining point data from soil profiles with soil spatial layers and ancillary ecological data can be applied to derive spatial estimates of SOC stocks (Kern, 1994). SOC density and stock estimates from soil profile data were combined with spatial data of major ecosystems by Post *et al*. (1982). The total SOC stocks for all life zones to a depth of 1 m was 1 395 Pg of SOC. A combination of soil profile data with ancillary information on climate, vegetation and land use was used by Jobbágy and Jackson (2000) to estimate SOC stocks in 11 biomes. The estimates for the biomes were further divided into increments of 1 m soil depth and of 20 cm for the first meter. The distribution of SOC stocks by ecological regions has also been presented, for example by Amundson (2001), who used life zones as the study unit. Eglin *et al*. (2010) used the SOC stock estimates to a depth of 3 m from Jobbágy and Jackson (2000) and modified SOC stocks in permafrost areas (Tarnocai *et al.*, 2009). These SOC stock estimates were combined with estimates provided by the IPCC (IPCC, 2000) of C in vegetation to derive estimates of C in soil and biomass for 10 biomes, with an explicit class for peatlands. A step towards adding a temporal dimension to spatial SOC stock estimates, assessing historical and future trends, was made possible by the availability of SOC models. Combining the models with historic land use and climate data has allowed estimation of SOC stocks with a timestamp and with regional variations (Eglin *et al.*, 2010; Schmidt *et al.*, 2011).

Carré *et al*. (2010) produced estimates of SOC stocks and density using climate data, IPCC methodology and the Harmonized World Soil Database. The results by IPCC Climate Region are presented in Table 6.1.

Table 6.1 | Distribution of Soil Organic Carbon Stocks and Density by IPCC Climate Region

* *Differences in topsoil and subsoil sum are due to data rounding*
** *Total includes 1.4 Pg C in undefined climate regions*

IPCC Climate Region	IPCC	HWSDa			
	Tier 1	Topsoil	Subsoil	Soil	Density
	0-30 cm	0-≤30 cm	30-≤100 cm	0-≤100 cm	0-≤30 cm
	Pg C	Pg C	Pg C	Pg C*	Mg C ha^{-1}
Tropical Wet	52.4	62.6	65.4	128.0	66.5
Tropical Moist	94.5	78.6	72.3	150.9	45.0
Tropical Dry	99.9	67.3	69.0	136.2	22.0
Tropical Montane	49.8	29.6	26.5	56.1	40.3
Warm Temperate Moist	41.7	33.3	29.7	63.0	60.2
Warm Temperate Dry	42.9	38.9	39.6	78.5	30.8
Cool Temperate Moist	110.6	104.1	106.2	210.3	88.2
Cool Temperate Dry	56.9	52.2	50.0	102.2	42.7
Boreal Moist	137.3	162.0	194.7	356.7	117.6
Boreal Dry	30.3	32.0	37.0	68.1	84.0
Polar Moist	26.8	30.6	21.7	52.4	40.4
Polar Dry	7.2	8.0	4.3	12.3	40.5
Total	**750.3****	**699.3**	**716.4**	**1415.7**	**52.1**

The table shows that according to the processed data from HWSD, most SOC (356.7 Pg C) is stored in the 'Boreal Moist' climatic region. The second largest stock is found in the 'Cool Temperate Moist' region (210.3 Pg C). With 117.6 Mg C ha^{-1} and 88.2 Mg C ha^{-1}, these climate regions also have the highest SOC densities. These figures compare poorly with those presented by Post *et al*. (1982). A major source for the deviation is the difference in the definition of the life zones as compared to the climatic regions, which lead to the delineation of different areas.

Using the IPCC Tier 1 default values for organic C in mineral soils and retaining the stocks for organic soil gives global organic C stock in the upper 30 cm of soil of 750.3 Pg C. This estimate is 51 Pg C (7.3 percent) higher than the estimates derived from the HWSDa topsoil layer.

When comparing the two spatial SOC stock estimates by IPCC climate region, the stocks within each region are broadly similar. A notable difference is for soils in the 'Tropical Dry' climate region. The IPCC Tier 1 SOC map gives 99.9 Pg C for this zone, compared to 67.3 Pg C found in the HWSDa.

In the interpretation of the figure for SOC stocks of the IPCC Tier 1 Soil Organic Carbon layer, it should be considered that organic soils were only added to the mineral soil layer in places where this soil type is not found in association with mineral soils. Using all organic soil data is likely to increase the global SOC stocks. However, the Tier 1 default values are calculated over a constant depth of 30cm, although some soils are shallower, which in turn would reduce the stocks.

6.2.4 | Spatial distribution of carbon in biomass

A global map of C stored in biomass following the IPCC Tier 1 approach was produced by the European Commission Joint Research Centre (Carré at al., 2010; EU, 2004; Hiederer *et al.*, 2010). The C stocks are determined for above- and below-ground biomass and include dead organic matter for the relevant vegetation types. The default factors largely follow the IPCC specification, with specific attention given to agricultural areas. The underlying vegetation data are based on the GlobCover V2.2 (ESA, 2011). Because the GlobCover data limits cropland to areas below 57º N in Europe the data were merged with the M 3-Cropland (Ramankutty *et al.*, 2008) and Crops (Monfreda, Ramankutty and Foley, 2008). In a comparison of the geographic distribution of IPCC vegetation classes between the GlobCover and the M 3 Cropland and Pasture data, some notable differences were identified (Hiederer *et al.*, 2010). Some of the differences were attributed to the dissimilar definition of the vegetation classes in the data sets, although others, such as the separation of shrub land from open forest or confusion between cropland and pastures, seem to be the result of the classification algorithm used or of sensor characteristics.

The global biomass map thus generated by the Joint Research Centre (JRC) estimates the storage of C in the above-ground and below-ground vegetation and dead organic matter to be 456 Pg C. The JRC estimates are thus 44 Pg C (8.8 percent) lower than those of the 'New IPCC Tier-1 Global Biomass Carbon Map for the Year 2000' (Ruesch and Gibbs, 2008). The difference is not evenly distributed between geographic regions. A comparison of carbon in C by climatic region is given in Figure 6.5.

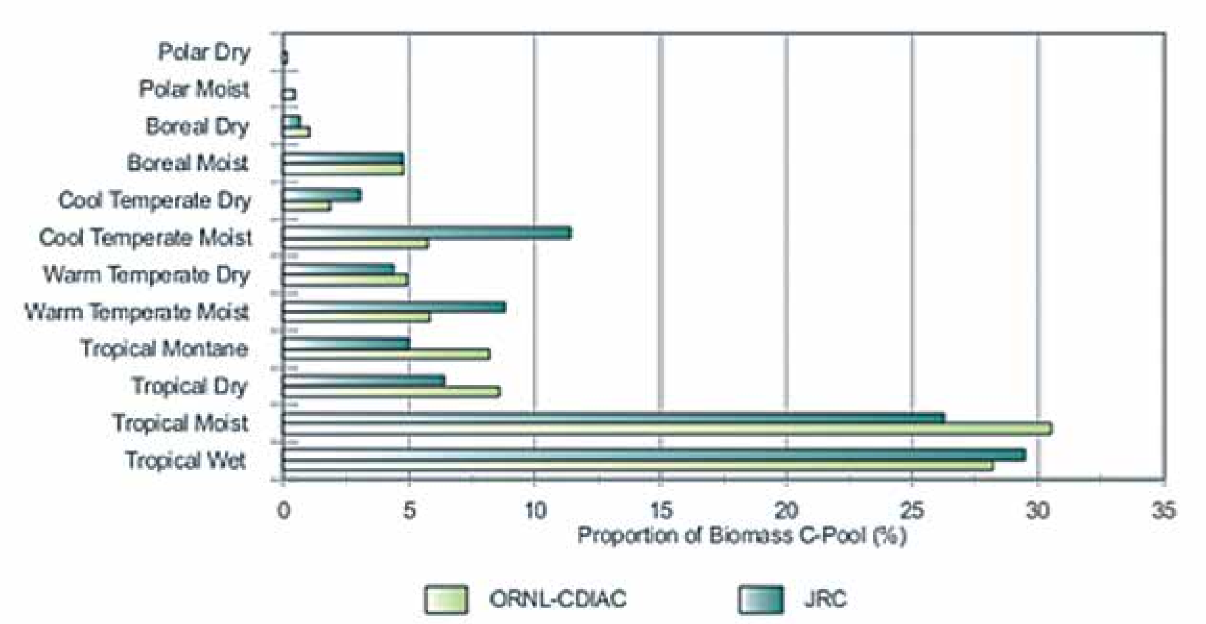


Figure 6.5 | Distribution of carbon in biomass between ORNL-CDIAC Biomass and JRC Carbon Biomass Map

The graph shows that the ORNL-CDIAC Biomass and the JRC Carbon Biomass map are mostly comparable, but the JRC map places relatively more C in the biomass in 'Cool Temperate Moist' (11.4 percent of the total C stock in biomass; 51.8 Pg C) and 'Warm Temperate Moist' (8.7 percent of the total C stock in biomass; 39.9 Pg C) climate regions at the expense of other regions. By contrast, the ORNL-CDIAC Biomass map locates only 5.7 percent of the total C stock in biomass (28.4 Pg C) in the 'Cool Temperate Moist' and 5.7 percent of the total C stock in biomass (28.7 Pg C) in the 'Warm Temperate Moist' climate region.

For the total terrestrial pool of organic C, biomass is the more important pool only in the climate regions 'Tropical Wet' and 'Tropical Moist'. For all other climatic regions, the soil stores more organic C than the biomass (Scharleman *et al.*, 2014).

6.2.5 | Distribution of terrestrial carbon pool by vegetation class

Areas where SOC or biomass C dominate could be identified by computing the difference between the two layers. The resulting layer is presented in Figure 6.6.

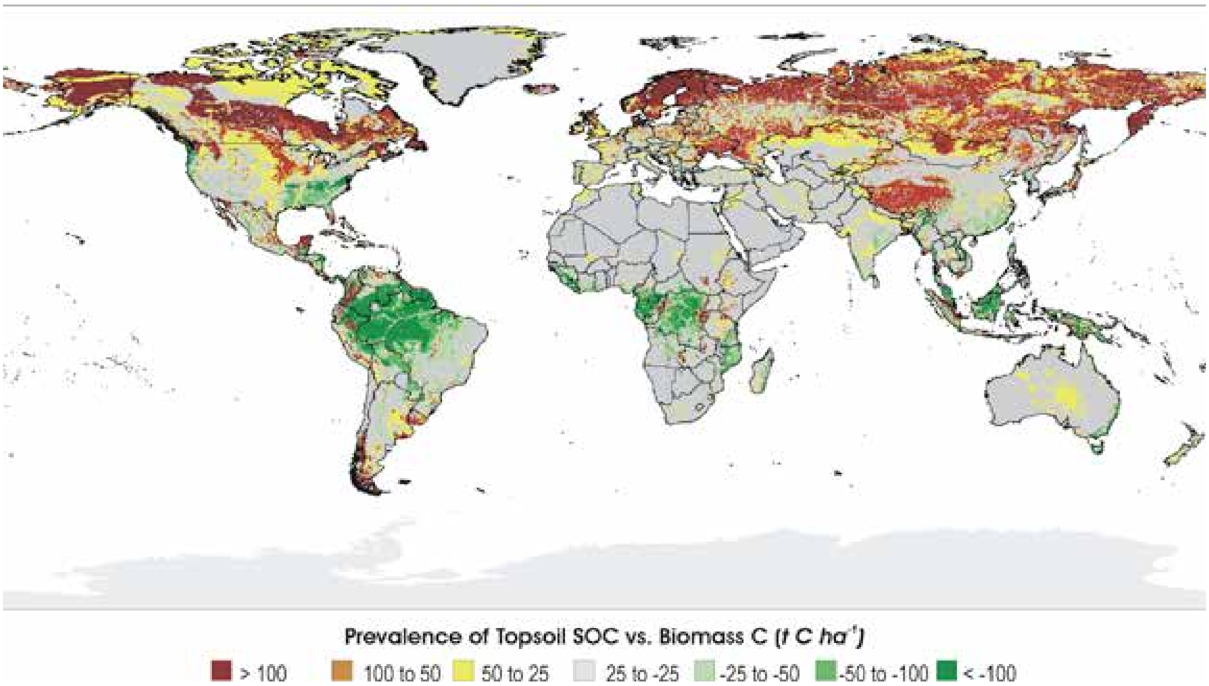


Figure 6.6 | Prevalence of carbon in the topsoil or biomass

The figure shows that, as a general propensity, soil dominates the terrestrial C pool in cooler climates while vegetation forms the dominant pool of terrestrial C in tropical regions.

In an attempt to provide C stock estimates for broad land use activities, global GLC 2000 data layers were used. The GLC 2000 categories were re-classified according to the assignments for these classes given by Ruesch and Gibbs (2008). A difference in the assignment was applied to GLC 2000 classes 16 (Cultivated and Managed Areas) and 23 (Irrigated Agriculture). In the broad classification these areas were grouped with other areas mainly associated with an absence of soil or biomass (bare areas, glaciers, etc.). For the analysis

Table 6.2 | Distribution of terrestrial organic carbon by stock and broad vegetation class

Vegetation Classes	Topsoil	Subsoil	Soil		Biomass		Terrestrial C Stock	
	Pg C	Pg C	Pg C	percent	Pg C	percent	Pg C	percent
Broadleaf Forest	124.7	112.4	237.1	16.8	272.2	54.4	509.4	26.6
Evergreen Forest	126.8	139.7	266.4	18.8	46.4	9.3	312.9	16.3
Mixed Forest	40.5	47.8	88.3	6.2	21.8	4.4	110.1	5.7
Burnt Forest and Natural Forest Mosaic	27.4	36.2	63.6	4.5	10.9	2.2	74.5	3.9
Forest/Cropland Mosaic	23.2	23.4	46.6	3.3	28.0	5.6	74.6	3.9
Forest	342.6	359.5	702.0	49.6	379.3	75.9	1081.5	56.4
Shrub Cover	89.2	102.4	191.6	13.5	51.8	10.4	243.4	12.7
Grasslands	60.5	52.1	112.6	8.0	18.0	3.6	130.5	6.8
Sparse Grassland and Grassland Mosaic	69.0	65.5	134.5	9.5	12.7	2.5	147.2	7.7
Grassland	218.7	220.0	438.7	31.0	82.5	16.5	521.1	27.2
Agriculture and managed areas	80.8	79.4	160.2	11.3	26.7	5.3	186.9	9.8
Other Classes	57.3	57.4	114.7	8.1	11.4	2.3	126.2	6.6

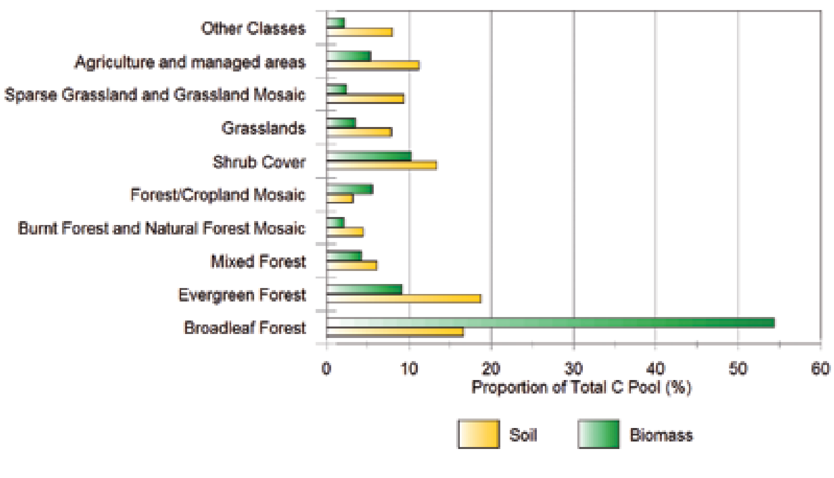


Figure 6.7 | Proportion of carbon in broad vegetation classes for soil and biomass carbon pool

of the distribution of organic C, a separate class of 'Agriculture and Managed Areas' was created by merging the GLC2000 classes 16 and 23. For each of the broad vegetation classes the organic C stock was extracted by pool. The results are presented in Table 6.2.

The single largest stock for terrestrial C is attributed to areas with broadleaf forest (509.4 Pg C). This forest type contains approximately one quarter of all terrestrial organic C in either the soil or the biomass.

The proportions of the C stored in the soil and biomass stocks by broad vegetation class is graphically presented in Figure 6.7.

For the biomass C stock alone, broad forests account for over 50 percent of the C in that pool, but only 16.8 percent of the organic C is stored in the soils under this vegetation type. With the exception of the 'Forest/Cropland Mosaic', in all other vegetation classes the soil stores more C than the biomass.

6.2.6 | Historic trends in soil carbon stocks

The SOC stocks are more susceptible to anthropogenic activities and natural factors than are SIC stocks. Conversion of natural to agro-ecosystems in the past has led to decline in the SOC stock of the surface layers and also in SOC in the total profile for most soils. The magnitude of the historic loss, however, differs among soils and climates. The magnitude and rate of loss are higher for soils within the tropics than for those of temperate climates. Losses are also higher for coarse-textured than for heavy-textured soils, higher for soils containing higher SOC stocks, and higher for soils under subsistence or 'extractive' farming than for those farmed with more science-based agricultural practices. Depletion is also exacerbated by drainage of wetlands, by ploughing, and by biomass burning or removal. Some soils in the tropics can lose 50 percent of their previous pool within five years following deforestation and conversion to agricultural land use. The rate and magnitude of SOC loss are exacerbated in soils vulnerable to accelerated erosion, salinization, nutrient depletion or imbalance, structural decline and compaction, acidification, elemental toxicity, pollution and contamination.

Estimates of the magnitude of historic SOC loss vary widely. The historic loss has been estimated at 40 Pg by Houghton (1995), 55 Pg by IPCC (1995) and Schimel (1995), 150 Pg by Bohr (1978), 500 Pg by Wallace (1994) and 537 Pg by Buringh (1984). The average of these estimates is 223 Pg C year^{-1}. Lal (1999) estimated the magnitude of SOC loss since 1850 at 47 to 104 Pg for different biomes (Table 6.3); 66 to 90 Pg for major soils (Table 6.4);

Table 6.3 | Estimate of the historic SOC depletion from principal biomes. Source: Lal, 1999.

Biome	Change in Area 10^6 ha	Historic SOC Loss Pg C
Forests	1300	23 - 53
Woodlands	180	3 - 7
Shrublands	140	1 - 4
Grasslands	660	20 - 40
Total		**47 - 104**

Table 6.4 | Estimates of historic SOC depletion from major soil orders. Source: Lal, 1999; Hillel and Rosenzweig, 2009.

Soil Order	Historic Area 10^6 ha	Present SOC Pool Pg C	Historic SOC Loss Pg C
Alfisols	1330	91	15 - 18
Andisols	110	30	5 - 7
Aridsols	1560	54	0.2 - 0.3
Entisols	2170	232	0.8 - 1.3
Histosols	160	312	?
Inceptisols	950	324	8 - 13
Mollisols	920	120	7 - 11
Oxisols	1010	99	22 - 27
Spososols	350	67	1 - 3
Ultisols	1170	98	6 - 7
Vertisols	320	18	1 - 2
Gelisols	1120	238	0
Others	1870	17	0.2 - 3
Total	**13050**	**1700**	**66 - 90**

Table 6.5 | Estimates of historic SOC loss from accelerated erosion by water and wind. Source: Lal, 1999.

Erosion	Area		Historic SOC Loss Pg
	Water 10^6 ha	Wind 10^6 ha	
Light	343	269	2 - 3
Moderate	527	254	10 - 16
Strong	224	26	7 - 12
Total			**19 - 31**

and 19 to 31 Pg by erosional processes (Table 6.5). While the historic loss from Gelisols or permafrost soils is zero, these soils, which contain a vast amount of SOC reserves, are vulnerable to projected warming and the attendant positive feedback.

Estimates of the historic C loss are useful as a reference point for assessing the technical potential of C re-sequestration in soil. While the loss of SOC can be rapid, especially in soils of the tropical ecosystems, the rate of re-carbonization is extremely slow. The slow rate of re-sequestration is a major challenge to identifying appropriate land use and to promoting adoption of soil/water/animal/plant management systems that could create a positive soil/ecosystem C budget.

6.2.7 | Future loss of SOC under climate change

Projected changes in climate (temperature and precipitation) are likely to affect the SOC stock both directly and indirectly. Directly, the rate of decomposition by microbial processes is affected by both soil temperature and moisture regimes. Indirectly, changes in climate affect plant growth, net primary productivity, above and below-ground biomass, and the type and amount of residues with differential amounts of materials recalcitrance. Further, the rate and susceptibility to accelerated erosion, salinization and other degradation processes may be exacerbated by an increase in frequency of extreme events. Indeed, climate change can impact several soil forming factors, including rainfall, temperature, micro-organisms/biota and vegetation, thus affecting the rate of SOC accumulation (Jenny, 1930). Climate change may also alter species composition, and the rate of litter fall. However, disagreement exists regarding the effect of warming on SOC stock.

The annual rate of litter return, on which the rate of SOC accretion depends, varies among biomes (White, 1987; Grunwald, 1999). The rate of litterfall (Mg ha^{-1} yr^{-1}) is estimated at 0.1 to 0.4 for alpine and arctic regions, 2 - 4 for temperate grassland, 1.5 - 3 for coniferous forest, 1.5 - 4 for deciduous forest, 5 - 10 for tropical rainforest, and 1 to 2 for arable land (White, 1987). Increase in soil temperature may exponentially increase the rate of soil respiration (Tóth *et al.*, 2007; Lenton and Huntingford, 2003). However, because of increase in the number and activity of soil fungi in the warmer soil, there may also be increase in the relative amount of lignin and other recalcitrant compounds (Simpson *et al.*, 2007). The SOM decomposition is also more temperature-sensitive at low than at high temperature (Kirschbaum, 1995, 2000, 2006).

Thus, knowledge about the temperature–sensitivity of diverse SOC fractions, and their change in the soil under climate change, is important. Change in temperature by 1º Celsius may decrease the turnover times of 4-11 percent and 8-16 percent for the intermediate and stabilized fractions, respectively (Hakkenberg *et al.*, 2008). The decomposition rate is also influenced by the presence of physicochemical protection mechanisms (Conant *et al.*, 2011), especially occlusion within aggregates and by association with mineral surfaces (Freedman, 2014). It is argued that CO_2 emissions from soil response to climate warming are over-estimated, because the decomposition of old SOM is tolerant to temperature (Liski *et al.*, 1999). Thus, the effects of warming on SOM decomposition are governed by complex and interactive factors, and are difficult to predict. Despite much research, no consensus has yet emerged on the temperature sensitivity of SOM decomposition (Davidson and Janssens, 2006).

6.2.8 | Conclusions

Global SOC stocks have been estimated at about 1 500 Pg C for the topmost 1 m. However, a large uncertainty attaches to this estimate, which cannot easily be assigned to a specific period in time. Local variations may also be high, for example for SOC stocks in arctic regions and peatlands. Estimates of SOC stocks below 1 m depth are still evolving, with a tendency for more recent estimates to be higher than previous values. Estimates of the historic loss of SOC pools are also highly variable, ranging from 40 to 537 Pg. The global loss of SOC pool since 1850 is estimated at about 66 ±12 Pg. The projected response of SOC stock to climate change is a debatable issue. While an increase in temperature may increase the rate of respiration at low soil temperature, it may also shift microbial populations to fungi, increase relative proportions of lignin and other recalcitrant fractions, and increase protective mechanisms such as aggregation and reaction with mineral surfaces.

6.3.1 | Introduction

Soil contamination as a result of anthropogenic activities has been a widespread problem globally (Bundschuh *et al.*, 2012; DEA, 2001; EEA, 2014; Luo *et al.*, 2009; SSR, 2010). Soil contamination can be local or diffuse. Local soil contamination occurs where intensive industrial activities, inadequate waste disposal, mining, military activities or accidents introduce excessive amounts of contaminants. Diffuse soil contamination is the presence of a substance or agent in the soil as a result of human activity and emitted from dispersed sources. Diffuse contamination occurs where emission, transformation and dilution of contaminants in other media have occurred prior to their transfer to soil. The three major pathways responsible for the introduction of diffuse contaminants into soil are atmospheric deposition, agriculture, and flood events. These pathways can also cause local contamination in some instances. Causes of diffuse contamination tend to be dominated by excessive nutrient and pesticide applications, heavy metals, persistent organic pollutants and other inorganic contaminants. As a result, the relationship between the contaminant source and the level and spatial extent of soil contamination is indistinct.

While some soil degradation processes are directly observable in the field (erosion, landslides, sealing or even decline of organic matter), soil contamination as well as soil compaction or decline in soil biodiversity cannot be directly assessed, which makes them an insidious hazard. Moreover, diffuse contamination is linked to many uncertainties. The diversity of contaminants (particularly of the persistent organic pollutants, which are in constant evolution due to agrochemical developments) and the transformation of organic compounds in soils by biological activity into diverse metabolites make soil surveys to identify contaminants difficult and expensive. The effects of soil contamination also depend on soil properties, as these have an impact on the mobility, bioavailability, residence time and levels of contaminants. Direct effects of pollutants may not be immediately revealed because of the capacity of soils to store, immobilize and degrade them. Effects can, however, suddenly emerge after changes such as changes in land use that may alter environmental conditions (Stigliani *et al.*, 1991 - see also Chapter 7 on processes impacting service provision). Contaminants include inorganic compounds such as metallic trace-elements and radionuclides, and organic compounds like xenobiotic molecules. The application of some organic wastes to soils – for example, untreated biosolids – also increases the risk of spread of infectious diseases. A new challenge is that the so-called 'chemicals of emerging concern' (CECs) – for example, veterinary and human therapeutic agents such as antibiotics and hormones – are present in amendments added to soils, such as manures. These CECs can have an adverse effect on ecosystems and on human health (Jjemba, 2002; Osman, Rice and Codling, 2008; Jones and Graves, 2010).

6.3.2 | Global status of soil contamination

In most developed countries, waste disposal and treatment, industrial and commercial activities, storage, transport spills on land, military operations, and nuclear operations are the key sources of local soil contamination. Management of local soil contamination requires surveys to seek out sites that are likely to be contaminated, site investigations where the actual extent of contamination and its environmental impacts are defined, and implementation of remedial and after-care measures. By contrast, diffuse soil contamination is much harder to manage: in many instances it is not directly apparent but it may cover very large areas and represent a substantial threat. Despite the fact that most developed countries have implemented long-term soil surveys, even these countries still lack a harmonized soil monitoring system, and the real extent of diffuse soil contamination is not known.

According to the most recent data provided by the European Environmental Agency (EEA, 2014), total potentially contaminated sites in Europe are estimated to be more than 2.5 million, of which 340 000 are thought to be actually contaminated. Approximately one third of the high risk sites have been positively identified as contaminated, and of these only 15 percent have so far been successfully remediated (EEA, 2014). While trends vary across Europe, it is clear that the remediation of contaminated sites is still a significant undertaking. Waste disposal and industrial activities are the most important sources of soil contamination overall in Europe. The most frequent contaminants are heavy metals and mineral oils (EEA, 2014).

In the United States, sites contaminated with complex hazardous substances that impact soil, groundwater or surface water are placed on the Superfund National Priorities List (NPL). As of September 29, 2014, there were 1 322 final sites on the NPL. On 1 163 of these sites, measures to address the contamination threat have been completed. An additional 49 sites have been proposed. In addition, the Office of Solid Waste and Emergency Response (OSWER) has cleaned up over 540 000 sites and 9.3 million ha of contaminated land, all of which can be put back into use. In Canada, a total of 12 723 soil contaminated sites has been identified, with 1 699 sites related to surface soil contamination (Treasury Board of Canada Secretariat, 2014). The key soil contaminants include metals, petroleum hydrocarbons (PHCs), and polycyclic aromatic hydrocarbons (PAHs).

The pattern of contamination in Australia is similar to that of other developed countries. Industry, including the petroleum industry, mineral mining, chemical manufacture and processing facilities, and agricultural activities with their use of P fertilizer and pesticides, have caused soil contamination with heavy metals, hydrocarbons, mineral salts, particulates, etc. The total number of contaminated sites is estimated at 80 000 across Australia (DECA, 2010), with approximately 1 000 actual or potentially contaminated sites in South Australia (SKM, 2013).

Developing countries are undergoing significant industrialization. If appropriate legal and regulatory frameworks and enforcement capability are not developed, this may lead to soil contamination and pose risks to the environment and human health. In large conurbations, there is also a need for adequate provision of sanitation and drainage so that household wastes are collected and managed safely.

Asian countries experience considerable contamination of agricultural soil and crops by trace elements, and this contamination is becoming a threat to human health and the long-term sustainability of food production in the contaminated areas. In China, it is estimated that nearly 20 million ha of farmland (approximately one fifth of China's total farmland) is contaminated by heavy metals (Wei and Chen, 2001). This may result in a reduction of more than 10 million tons of food supplies each year in China (Wei and Chen, 2001). Atmospheric deposition (mainly from mining, smelting and fly ash) and livestock manures are the main sources of trace elements contaminating arable soil (Luo *et al.*, 2009). Among the different trace elements contaminating Chinese agricultural soils, Cd is the biggest concern. Due to its high mobility in the soil (except in poorly drained soil where sulphides are present), it can be easily transferred to the food chain and so poses risks to human health. Arsenic is also naturally present in groundwater in many regions of Southeast Asia. This represents a threat to agriculture, particularly in rice paddy fields where anaerobic conditions prevail (Smedley, 2003; Hugh and Ravenscroft, 2009). Asia is also the largest contributor to the atmosphere of anthropogenic Hg, which originates from the chemical industry, from Hg mining and from gold mining (Li *et al.*, 2009). All across Asia, areas under rapid economic development are experiencing moderate to severe contamination by heavy metals (Ng, 2010).

In many parts of Latin America, the results of anthropogenic activities, such as tailings and smelting operations in mining areas, have resulted in arsenic contamination in the soil. These operations enhance the mobilization of arsenic and cause adverse environmental impacts (see Section 4.3). Also in Latin America, the problem of arsenic contamination in water has been identified in 14 of the continent's 20 countries: Argentina, Bolivia, Brazil, Chile, Colombia, Cuba, Ecuador, El Salvador, Guatemala, Honduras, Mexico, Nicaragua, Peru and Uruguay. The number of exposed people in these countries is estimated to be about 14 million (Bundschuh *et al.*, 2012; Castro de Esparza, 2006). It is also estimated that during the late 1980s and early 1990s, 3 000 to 4 000 tonnes of Hg were deposited in the Amazon basin as a result of artisanal gold-mining activities, mainly in Brazil, Bolivia, Venezuela and Ecuador (de Lacerda, 2003). In addition, intensive use of fertilizers and pesticides in many parts of Latin America contributes to soil contamination and causes a range of environmental pollution and human health problems (UNEP, 2010).

In Africa, soil contamination has resulted from mining, spills, and improper handling of waste (Gzik *et al.*, 2003; SSR, 2010; EA, 2010). The Nigerian federal government reported more than 7 000 spills between 1970 and 2000. In Botswana and Mali, over 10 000 tonnes of pesticides, including DDT, aldrin, dieldrin, chlordane and heptachlor, have leaked from damaged containers and contaminated the soil (SSR, 2010). Soil contamination in the Near East and North Africa is linked to oil production and heavy mining. In arable land, a common source of soil pollution is the use of contaminated groundwater or wastewater for irrigation.

6.3.3 | Trends and legislation

In developed countries, legislation on contaminated land and the related regulatory mechanisms are well established. As a result, the extent of contaminated land is thoroughly reported. The European countries have created a common framework in the Thematic Strategy on Soil Protection (COM (2006) 231), which aims at sustainable use of soil, preservation of soil as a resource, and remediation of contaminated soil. The EC has also created networks such as CLARINET, NICOLE and SNOWMAN (Vicent, 2013). Investigations of suspected contaminated sites continue in Europe and as a result the total of contaminated sites listed is expected to increase by 50 percent by 2025 (EEA, 2007, 2012; EC, 2013). The number of remediated sites is expected to grow as well. In addition, regulation now requires industrial plants to control their wastes and prevent accidents, limiting the introduction of contaminants into the environment. As noted above, the United States has introduced a regulatory regime and has made significant progress on site clean-up.

In Asia, early legislation on contaminated land management (CLM) focused on contamination of agricultural land caused by industrialization and urbanization. Thus Japan, Taiwan, Province of China and South Korea have developed comprehensive CLM frameworks of laws, regulations and guidelines. Other Asian countries, however, are still at early stages of developing a CLM framework (Ng, 2010).

Atmospheric deposition (Section 4.4.1) is an important input of pollutants (Lofts *et al.*, 2007) and air quality regulation to decrease the load of contaminants on soils is therefore important. In most developed countries, relevant legislation is well established. In the case of long-range atmospheric pollution, international agreements are needed. In this regard, the Convention on Long-Range Transboundary Air Pollution (LRTAP) was signed in 1979. Conceived in response to the detrimental impact of acid rain in Europe, the Convention entered into force in 1983. Over the past 30 years, the Convention has been extended by eight further protocols that target pollutants such as S, NO_x, persistent organic pollutants, volatile organic compounds, ammonia and heavy metals. More recently, a global treaty to protect human health and the environment from the adverse effects of mercury - the 2013 Minamata Convention on Mercury - has been established.

CECs require due attention and they can include, but are not limited to, nanoparticles, pharmaceuticals, personal care products, estrogen-like compounds, flame retardants, detergents, and some industrial chemicals (including those in products and packaging) with potential significant impact on human health and aquatic life (Jones and Graves, 2010). Electronic waste (also referred to as 'e-waste') is of great concern given the increasing volumes generated each year, the hazardous nature of some of the components, and the exportation of this waste from industrialized countries to recycling centres in China, India and Pakistan (UNEP DEWA/GRID-Europe, 2005). This chain risks violating the Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and their Disposal, which was adopted in 1989 and came into force in 1992.

Recently some countries have implemented policies and programmes to encourage waste minimization. These programmes of 'Extended Producer Responsibility' make producers responsible for the costs of managing their products at the end of their life. This approach is expected to encourage the manufacture of more environmentally-friendly electronic products (UNEP DEWA/GRID-Europe, 2005).

6.4 | Soil acidification status and trends

6.4.1 | Processes and causes of acidification

Soil acidity increases with the build-up of hydrogen (H^+) and aluminium (Al_3^+) cations in the soil or when base cations such as potassium (K^+), calcium (Ca_2^+), magnesium (Mg_2^+) and sodium (Na^+) are leached and replaced by hydrogen or aluminium (Bolan, Hedley and White, 1991; Helyar and Porter, 1989; von Uexküll and Mutert, 1995). The main causes of soil acidification are: (1) long term rainfall that results in on-site leaching of base cations; (2) draining of potentially acid sulphate soils; (3) acid deposition when urbanization, industrialization, mining, construction or dredging release acid substances into the air or water, causing off-site acidification; (4) excessive application of ammonium-based fertilizers (e.g. ammonium sulphate) as part of intensive agriculture cropping practices; and (5) deforestation and other land use practices that remove all harvested materials, often resulting in a drop of the pH in the topsoil. Only the first of these five causes is a natural phenomenon; all others are human-induced.

In natural ecosystems, soils become more acid with time. Consequently old soils, particularly in humid climates or those developed from acidic rocks, are more weathered and acidic than younger soils or soils of dry climates or those developed from more basic rocks (Helyar and Porter, 1989; von Uexküll and Mutert, 1995). 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, such as acid soils in highly weathered tropical areas (Harter, 2007; Johnson, Turner and Kelly, 1982). Soil texture and soil organic matter content play an important role in the buffering capacity of a soil and hence in determining how prone a soil is to acidification (Helyar and Porter, 1989; Steiner *et al.*, 2007). Light sandy soils poor in organic matter are the least buffered against acidification.

Acid sulphate soils contain metal sulphides which, when exposed to oxidation, produce sulphuric acid. Inland, acid sulphate soils form naturally in aquatic ecosystems and also as a consequence of human-induced changes to land use and hydrology. Structures regulating water flow such as dams, weirs and locks prevent flushing of metals, salts and organic matter, and promote the build-up of acid sulphate soils. Acid sulphate soils also form in coastal areas and are common in mangrove forests, saltmarsh, floodplains, and salt- and freshwater wetlands (Lin and Melville, 1994; Pons, van Breemen and Driessen, 1982; Pannier, 1979). Due to the abundance of metal sulphides in rocks, mining activities also foster the formation of acid sulphate soils (Dent, 1986).

The atmospheric deposition of sulphur dioxide (SO_2), nitrogen oxides (NO_x) and ammonia (NH_3) leads to acid deposition. This can affect not only areas near to the urban, industrial and mining sites where the oxides are produced and released into the environment, but also sites located far away (Fanning *et al.*, 2004; Menz and Seip, 2004; Mylona, 1996; Orndorff and Daniels, 2004). The term 'acid deposition' includes both wet and dry (gaseous) precipitation, usually in the form of acid rain or fog. Besides affecting the chemistry of soil and water resources, acid deposition directly harms plants and fish. Acid deposition is currently a major concern in fast-developing countries such as China (Chen, 2007).

Land use and soil management play a crucial role in determining the chemical characteristics of the soil. Intensive farming practices that employ large inputs of nitrogen fertilizers and remove large quantities of products increase soil acidity (Barak *et al.*, 1997; Bolan, Hedley and White, 1991). Indeed, the conversion of ammonium to nitrate releases hydrogen ions (H^+) into the soil solution that can potentially lower the soil pH. This is a problem in soils with low ability to buffer the increase in H^+ such as those poor in lime and negatively charged organic matter and clay. Harvesting has the potential to increase soil acidity by removing base cations from the soil. This is an issue in both agricultural and forested areas wherever large amounts of biomass are removed by crop harvesting and deforestation (Cavelier *et al.*, 1999; von Uexküll and Mutert, 1995).

6.4.2 | Impact of soil acidification

On acid soils (pH < 5.5), crops and pastures suffer from the resulting increased phytotoxicity (Al, Fe, Mn, etc.), from the reduced availability of nutrients, and from decreased microbiological activity (Cronan and Grigal, 1995; Robson and Abbott, 1989; Slattery and Hollier, 2002; Sverdrup and Warfvinge, 1993; Whitfield *et al.*, 2010). Onsite soil acidification reduces net primary productivity and carbon sequestration by accelerating leaching of nutrients such as manganese, calcium, magnesium and potassium, resulting in nutrient deficiencies for plants (Haynes and Swift, 1986). On-site soil acidification is also responsible for the development of subsoil acidity (Tang, 2004), for the breakdown and subsequent loss of clay materials from the soil (Chen, 2007), and for the erosion which results from decreased groundcover (Slattery and Hollier, 2002). Soil acidification also leads to off-site effects such as surface water acidification through sediment losses, and groundwater enrichment of soluble metals. In turn, these processes mobilize heavy metals into water resources and the food chain (Driscoll *et al.*, 2003; Reuss and Johnson, 1986; Schindler *et al.*, 1980; Slattery and Hollier, 2002; Voegelin, Barmettler and Kretzschmar, 2003).

6.4.3 | Responses to soil acidification

Soil acidification is an insidious process. It develops slowly and, if not corrected by lime applications for example, can continue until the soil is irreparably damaged (Edmeades and Ridley, 2003; Liu and Hue, 2001; Slattery and Hollier, 2002). Biological recovery can potentially be improved by an increase in pH and acid-neutralising capacity (ANC) (Marschner and Noble, 2000). Of main concern is subsoil acidity, which is particularly difficult to correct with conventional methods (Farina, Channon and Thibaud, 2000; Liu and Hue, 2001; Hue and Licudine, 1999). Actions to mitigate global warming can reduce the emission of pollutants such as sulphur dioxide (SO_2) which contribute to soil acidification (NADP, 2014; Smith, Pitcher and Wigley, 2001; Vestreng *et al.*, 2007). However, soil response to decreases in acid deposition is slow and acid-affected sites may require many decades to recover (Zhao *et al.*, 2009).

6.4.4 | Global status and trends of soil acidification

Soil acidity is a serious constraint to food production worldwide. Traditionally it has been counteracted by applying lime to the topsoil but little could be done to increase the pH of the subsoil. Programmes to improve soil pH have been undertaken largely in developed countries, which are able to implement soil management plans to preserve soil properties and to bear the cost of lime to buffer soil acidity. However, even in developed counties, for example Australia, there have been cases where subsoil acidity increased due to failures in correcting topsoil acidity. In developing countries the situation is more stark as the use of lime is constrained by poverty. As a result, the farmed area affected by acidification is on the rise (Sumner and Noble, 2003). Soil acidification affects not only agricultural areas but also forests and grasslands.

According to Sumner and Noble (2003), topsoil acidity (pH <5.5) affects around 30 percent of the total ice-free land area of the world, and subsoil acidity affects as much as 75 percent. Figure 6.8 illustrates the pH status worldwide. The most acidic topsoils (pH <3.5) in the world are located in South America in areas where deforestation and intensive agriculture are practiced, and also in river deltas populated by mangroves, for example the Amazon and Orinoco Deltas (Marchand *et al.*, 2006; Moormann, 1963). Elsewhere, the regions with the highest presence of acid soils are: northern and eastern regions of North America; South-East Asia; Central and South Africa; and northern regions of Europe and Eurasia.

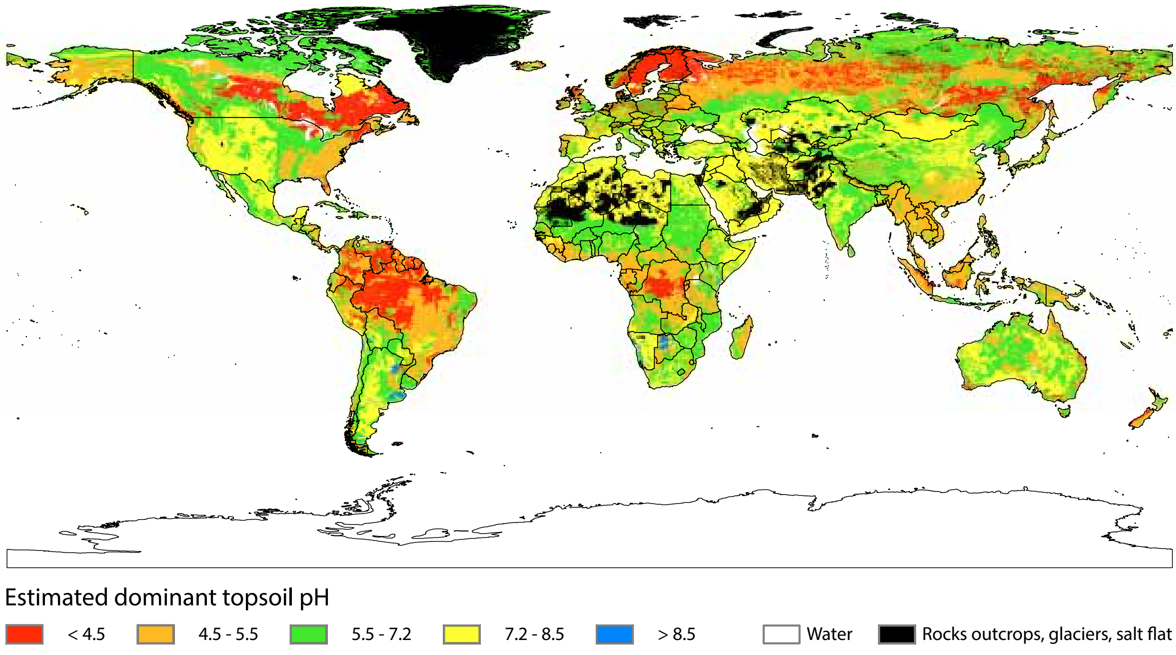


Figure 6.8 | Estimated dominant topsoil pH.
Source: FAO/IIASA/ISRIC/ISS-CAS/JRC, 2009.

The main causes of soil acidification vary by region:

- Regions where soil acidification occurs because of soil texture – parts of North America, Southeast, East and South Asia (Aherne and Posch, 2013; Eswaran *et al.*, 1996; Hicks *et al.* 2008; Shamshuddin *et al.*, 2014; Ouimet *et al.*, 2006)
- Regions where proximity to deltas and coastal plains is a primary cause – parts of West Africa (Bullock *et al.*, 1996),
- Regions where weather conditions are a main cause – parts of Africa and Asia (Breuning-Madsen and Awadzi, 2005; Drees, Manu and Wilding, 1993; Eswaran *et al.*, 1996; Kottek *et al.*, 2006; Wilke, Duke and Jimoh, 1984),
- Regions where acid deposition is an important factor – parts of East Asia and North America (Aherne and Posch, 2013; Quinn, 1989; Wolt, 1981)
- Regions where the massive application of ammonium-based fertilizers plays an important role – parts of East and South Asia (Guo *et al.*, 2010; Wang, Zhang and Zhang, 2010).

In Europe, soil acidification is an issue only in some highly urbanised and industrialized hotspots (EEA, 2010; Kopáček *et al.*, 2004; Menz and Seip, 2004; Moser and Hohensinn, 1983). In the Southwest Pacific, soil acidification is of concern only in intensively farmed areas (Brennan, Bolland and Bowden, 2004; Hartemink, 1998; Xu *et al.*, 2002; Lockwood *et al.*, 2003; NLWRA, 2001). Thus soil acidification affects all regions to some extent, but it is of main concern in poor and developing countries which are growing rapidly but are unable to buffer the decrease in soil pH through conventional means.

6.5 | Global status of soil salinization and sodification

6.5.1 | Status and extent

Salt-affected soils occur in more than 100 countries and their worldwide extent is estimated at about 1 billion ha. Salt-affected soils include those affected by salinity, where the electrical conductivity of the soil exceeds 4dSm–1; and those affected by sodicity, where the exchangeable sodium percentage (ESP) exceeds 6 (Ghassemi, Jakeman and Nix, 1995). Saline soils contain excessive soluble salts, mainly sodium chloride (NaCl) and sodium sulphate (Na_2SO_4) or other neutral salts. These salts increase osmotic pressure, diminish water availability and inhibit plant growth. Sodic soils generally have a low salt content but a high ESP, which causes dispersion of clay particles and results in deterioration of the soil structure. These soils generally have low air and water permeability and a pH above 8.2.

Salinity problems are encountered in all climates and are a consequence of both natural (primary) and human-induced (secondary) processes. Soil salinity and sodicity problems are more common where rainfall is insufficient to leach salts and excess sodium ions out of the rhizosphere. Salt-affected soils often occur on irrigated lands, especially in arid and semiarid regions, where annual rainfall is insufficient to meet the evapotranspiration needs of plants and to provide for leaching of salt. In humid areas, soluble salts are carried down through the soil profile by percolating rainwater and ultimately are transported to sea.

Although salt-affected soils are widespread and an increasingly severe problem, no accurate recent statistics are available on their global extent. The best available estimates suggest that about 412 million ha are affected by salinity and 618 million ha by sodicity (UNEP, 1992), but this figure does not distinguish areas where salinity and sodicity occur together. The Soil Map of the World (FAO/UNESCO, 1980) depicted a similar extent of 953 Mha affected by salinity (352 million ha) and sodicity (580 million ha). Table 6.6 shows the distribution of dryland salinity in different continents.

Human-induced salinity, mainly caused by irrigation without adequate drainage, affects a much smaller area than natural salinity. According to GLASOD, the extent of human-induced salinity is about 76 million ha (Oldeman, Hakkeling and Sombroek, 1991) of which 52.7 million ha occurs in Asia. In Europe, significant parts of Spain and areas in Italy, Hungary, Greece, Portugal, France and Slovakia are also affected by human-induced salinization.

Table 6.6 | Distribution of salt-affected soils in drylands different continents of the world. Source: UNEP, 1992.

Continent	Saline soils (million ha)	Sodic soils (million ha)	Total (million ha)
Africa	122.9	86.7	209.6
South Asia	82.3	1.8	84.1
North and Central Asia	91.5	120.2	211.7
Southeast Asia	20.0	-	20.0
South America	69.5	59.8	129.3
North America	6.2	9.6	15.8
Mexico/Central America	2.0	-	2.0
Australasia	17.6	340.0	357.6
World total	**412.0**	**618.0**	**1030**

In 2006 the global area equipped for irrigation stood at 301 million ha. At present in developing countries, irrigated agriculture covers about one fifth of all arable land, but accounts for nearly half of all crop production and 60 percent of cereal production. About 70 percent of the world area equipped for irrigation is in Asia where it accounts for 39 percent of the cultivated area. India and China each have 62 million ha equipped for irrigation (FAO, 2011). An estimated 60 million ha (or 20 percent of the total irrigated area) are affected by soil salinity, of which 35 million ha are located in four countries e.g. Pakistan (3.2 million ha), India (20 million ha), China (7 million ha) and the United States (5.2 million ha). Other countries with large amounts of salt-affected lands in irrigation districts include Afghanistan, Egypt, Iraq, Kazakhstan, Turkmenistan, Mexico, Syria and Turkey (Squires and Glenn, 2011).

Australiasia has the largest extent of naturally sodic soils of any continent (Table 6.6).

6.5.2 | Causes of soil salinity

The distribution of salt-affected soils varies geographically with climate, landscape type, agricultural activities, irrigation methods and policies related to land management.

Natural causes of salinity and sources of salt

1. *Rock weathering*: Significant quantities of sodium, and to a lesser extent chloride, occur widely in the parent rocks from which soils form. Over time, rock weathering can lead to appreciable salt accumulation in soils if leaching is restricted. Rock weathering is the primary source of salt in seawater.
2. *Sea water and accession of salt in marine sediments*: Saline soils can form from sediments and parent materials that were once under the sea. Likewise, the salts can be due to tidal inundation. Typical examples include the pseudo-delta of Senegal and the Gambia and in the Philippines where coastal tideland reclamation has created about 0.4 million ha of agricultural salt-affected soils. In the United Arab Emirates, areas along the coastal sabkha (salt marshes or lagoonal deposits) are highly salinized (28.8 dS m^{-1}). In the coastal region of the Abu Dhabi Emirate, salinity is more than 200 dS m^{-1} (Abdelfattah and Shahid, 2007)
3. *Atmospheric deposition*: Salt derived from the sea, either deposited via rain or dry fallout, is the primary source of salt across large areas: for example, many millions of hectares in southern Australia. In arid areas, salt can also be derived from dry lake beds and then blown considerable distances by wind (e.g. Eurasia and parts of Australia).

Human-induced causes

1. The management of land and water resources is responsible for the development of human-induced saline and sodic soils. The main causes are:
2. Poor drainage facilities which induce a rise of the groundwater table. This is a major cause of soil salinization in India, Pakistan, China, Kenya and the Central Asian countries.
3. The use of brackish groundwater for irrigation. This is a major cause of secondary salinization in parts of Asia, Europe and Africa.
4. The intrusion of seawater in coastal areas, for example in Bangladesh.
5. Poor on-farm water management and cultural practices in irrigated agriculture.
6. Continuous irrigation over very long periods, particularly in the Middle East.
7. Replacement of deep rooted perennial vegetation with shallower rooted annual crops and pastures that use less water leading to the rise of saline groundwater, for example southern Australia.

6.5.4 | Trends and impacts

Soil salinity is becoming a significant problem worldwide. From the very scattered information on the extent and characteristics of salt-affected soils, salinity and sodicity are rapidly increasing in many regions, both in irrigated and non-irrigated areas. Increasing soil salinity problems are taking an estimated 0.3 to 1.5 million ha of farmland out of production each year and decreasing the production potential of another 20 to 46 million ha. The annual cost of salt-induced land degradation was estimated in 1990 at US$ 264 ha^{-1}. By 2013, the inflation-adjusted cost of salt-induced land degradation was reported as US$ 441 ha^{-1} (Qadir *et al.*, 2014).

6.5.5 | Responses

There are many available responses to contain the salinity threat. These include: (1) direct leaching of salts; (2) planting salt tolerant varieties; (3) domestication of native wild halophytes for use in agro-pastoral systems; (4) phytoremediation (bioremediation); (5) chemical amelioration; and (6) the use of organic amendments.

In several Asian countries, a blend of engineering, reclamation and biological approaches has been adopted to address salinity and waterlogging problems. In Pakistan, engineering solutions included large-scale Salinity Control and Reclamation Projects (SCARPs), which covered 8 million ha at an estimated cost of US$2 billion (Qureshi *et al.*, 2008). Two big drainage water disposal projects were also undertaken. Measures to address the saline soil problem included leaching of salts by excess irrigation, use of chemicals (such as gypsum and acids), the addition of organic matter, and biological measures such as salt-tolerant plants, grasses, and shrubs.

Improvements in on-farm water and crop management have also been practiced. In North America, changes in land use and management practices have reduced the risk of salinization and helped to improve soil health and agri-environmental sustainability.

In Iraq and Egypt, surface and subsurface drainage systems have been installed to control rising water tables and arrest soil salinity. In Iran, Syria and other Gulf countries, crop-based management, and fertilizers are used to combat salinization (Qadir, Qureshi and Cheraghi, 2007). In Iran, Haloxylon aphyllum, Haloxylon persicum, Petropyrum euphratica and Tamarix aphylla are potential species for saline environments (Djavanshir, Dasmalchi and Emararty, 1996). Also in Iran, Atriplex has been shown to be a potential fodder shrub in the arid lands which could bring annual income as high as US$ 200 ha^{-1} (Koocheki, 2000; Nejad and Koocheki, 2000). Breeding of salt tolerant crop varieties (e.g. wheat, barley, alfalfa, sorghum etc.) is also a recognized management response for saline environments. However, most results have been obtained in controlled environments, with few real field results so far.

The use of organic amendments in Egypt showed that the mixed application of farmyard manure and gypsum (1:1) significantly reduces soil salinity and sodicity (Abd Elrahman *et al.*, 2012). Recently, phytoremediation or plant based reclamation has been introduced in the Near East region. In Sudan good responses for control of sodicity have been obtained through phytoremediation. The production of H^+ proton in the rhizosphere during N-fixation from legumes such as the hyacinth bean (Dolichos lablab L.) removed as much Na^+ as gypsum application. This indicates the importance of this technology in calcite dissolution of calcareous salt affected soils (Mubarak and Nortcliff, 2010).

6.6 | Soil biodiversity status and trends

6.6.1 | Introduction

Over the last few decades the importance of soil biota for terrestrial functioning and ecosystem services has emerged as an important focus for soil science research. Current evidence shows that soil biota constitute an important living community in the soil system, providing a wide range of essential services for the sustainable functioning of global terrestrial ecosystems and thereby impacting human wellbeing, directly and indirectly (van der Putten *et al.*, 2004). Soil organisms (e.g. bacteria, fungi, protozoa, insects, worms, other invertebrates and mammals) shape the metabolic capacity of terrestrial ecosystems and many soil functions. Below-ground biodiversity represents one of the largest reservoirs of biodiversity on earth (Bardgett and van der Putten, 2014). Essential services provided by soil biota include: regulating nutrient cycles; controlling the dynamics of soil organic matter; supporting soil carbon sequestration; regulating greenhouse gas emissions; modifying soil physical structure and soil water regimes; enhancing the amount and efficiency of nutrient acquisition by vegetation through symbiotic associations and nitrogen fixation by bacteria; and influencing plant and animal health through the interaction of pathogens and pests with their natural predators and parasites.

Fungi and bacteria are important decomposers in the soil. They are remarkably efficient. The smaller the pieces to be decomposed, the faster these microorganisms are able to do their job. Organic waste such as leaf matter and the droppings of herbivores first feed a host of small animals including insects, earthworms and other small invertebrates which live in the plant litter. The combined fauna break up the organic matter, digesting part of it, and thus facilitating the task of the microorganisms and invertebrates that complete the process of decomposition. In turn, soil macro-fauna affect soil organic matter dynamics through organic matter incorporation, decomposition and the formation of stable aggregates that protect organic matter against rapid decomposition. Successive decomposition of dead material and modified organic matter results in the formation of a more complex organic matter called humus, which affects soil properties by increasing soil aggregation and aggregate stability, increasing the cation-exchange capacity (the ability to attract and retain nutrients), and increasing the availability of N, P and other nutrients.

Many scientists have reported the role of macro-fauna in the accumulation of soil organic matter. For example the work by Snyder, Baas and Hendrix (2009), showed that millipedes and earthworms, both by themselves and taken together, reduce particulate organic matter. In addition, earthworms create significant shifts in soil aggregates from the 2000–250 and 250–53 µm fractions to the > 2000 µm size class. Earthworm-induced soil aggregation was lessened in the 0-2 cm layer in the presence of millipedes. Further, Hoeksema, Lussenhop and Teeri (2000) found that in high-N soil with twice-ambient CO_2 there was a higher density of predator/omnivores, lower diversity, and a larger value of Bonger's Maturity Index compared to ambient CO_2. In this experiment, fine root biomass and turnover were significantly greater under elevated CO_2. This indicates higher vigour in plant root development and growth and hence increased carbon sequestration conditioned by enhanced soil biota activity.

Studies also show the role of soil biota (including fungi, bacteria and plant parasitic nematodes) as pathogens and parasites or herbivores in decreasing root and plant productivity or reducing fruit quality. Recent research has focussed on the use of nematode and fungal resistant plant species or of other soil organisms as suppressive agents to modify the pathogens.

6.6.2 | Soil biota and land use

Losses in soil biodiversity have been demonstrated to affect multiple ecosystem functions including plant diversity, decomposition, nutrient retention and nutrient cycling (Wagg *et al.*, 2014). Links between above-ground and below-ground communities (Wardle *et al.*, 2004; De Deyn and van der Putten, 2005; Bardgett and van der Putten, 2014) suggest that factors affecting above-ground extinction may also be affecting soil organisms.

Agricultural intensification, in particular, may reduce soil biodiversity, leading to decreased food-web complexity and fewer functional groups (Tsiafouli *et al.*, 2015). Other driving forces that influence biodiversity in agricultural soils include the influence of crops/plants, fertilizers and pH, tillage practices, crop residue retention, pesticides, herbicides and pollution (Breure *et al.*, 2004; Bardgett and van der Putten, 2014). Soil biological and physical properties (e.g. temperature, pH, and water-holding characteristics) and microhabitat are altered when natural habitat is converted to agricultural production (Crossley *et al.*, 1992; Bardgett and van der Putten, 2014). Changes in these soil properties may be reflected in the distribution and diversity of soil meso fauna. Organisms adapted to high levels of physical disturbance become dominant within agricultural communities, thereby reducing richness and diversity of soil fauna (Paoletti *et al.*, 1993).

The management practices used in many agro-ecosystems (e.g. monocultures, extensive use of tillage, chemical inputs) degrade the fragile web of community interactions between pests and their natural enemies. The intensification of agricultural management may result in increased incidence of pests and diseases, with numerous studies reporting declines in the biodiversity of soil fauna (Decaens and Jimenez, 2002; Eggleton *et al.*, 2002). In addition, the contribution of soil fauna globally to organic matter decomposition rates may be highly dependent on the temperature and moisture of an ecosystem (Wall *et al.*, 2008). This underlines the need for global-scale assessments. In a global study of soil fungi using 365 soil samples from natural ecosystems, Tedersoo *et al.* (2014) found that distance from the equator and annual precipitation had considerable effect on fungal species richness. They also identified various other controls on soil fungi and this is starting to provide a benchmark for assessing the impacts of human activities on an important component of soil biodiversity.

Soil management strongly influences soil biodiversity, resulting in changes in abundance of individual species. Using a soil biodiversity pressure index calculation from the European Soil Data Centre, Gardi, Jeffery and Saltelli (2013) estimated that 56 percent of soils within the European Union have some degree of threat to soil biodiversity. Based on a questionnaire completed by 20 experts, the study found that the main anthropogenic pressures on soil biodiversity are (in order of importance): (1) intensive human exploitation; (2) reduced soil organic matter; (3) habitat disturbance; (4) soil sealing; (5) soil pollution; (6) land-use change; (7) soil compaction; (8) soil erosion; (9) habitat fragmentation; (10) climate change; (11) invasive species; and (12) GMO pollution (Gardi, Jeffery and Saltelli, 2013).

There is some experimental evidence that there may be threshold levels of soil biodiversity below which functions decline (e.g. Van der Heijden *et al.*, 1998; Liiri *et al.*, 2002; Setälä and McLean, 2004). However, in many instances this is at experimentally prescribed levels of diversity that rarely prevail in nature. Although some studies demonstrate some functional redundancy in soil communities (e.g. Setälä, Berg and Jones, 2005), high biodiversity within trophic groups may be advantageous since the group is likely to function more efficiently under a variety of environmental circumstances, due to an inherently wider potential. In a synthesis of diversity-function relationships of soil biodiversity focusing on carbon cycling, Nielsen *et al.*, (2011) concluded that although there is considerable functional redundancy in soil communities for general processes, change may readily have an impact on specialized processes. However, data to support this conclusion are still limited. More diverse systems may be more resilient to perturbation since, if a proportion of components are removed or compromised in some way, others that prevail will be able to compensate (Kibblewhite, Ritz and Swift, 2008).

6.6.3 | Conclusions

A comprehensive global-assessment on below-ground biodiversity has yet to be carried out. Although there is a Global Soil Biodiversity Atlas (EU/JRC, in press), no benchmark values exist on a global scale. This makes it difficult to quantify changes or future losses that may result from natural or anthropogenic-induced changes. Although progress is being made, few monitoring programs exist that quantify soil biodiversity across regions and at multiple trophic levels, especially outside of Europe. Regarding the threats to soil biodiversity and the effects on ecosystem functioning, more comparative and coordinated studies (from local to global scales) are needed across all ecosystems. These studies should quantify threats and determine the consequences of soil biodiversity loss to ecosystem functions, as well as the effects of interactions between threats. In addition, there is a need for standardization of methods in soil biodiversity studies so that multiple datasets can be synthesized and benchmark values for global soil biodiversity may be established. The use of DNA-based approaches is accelerating the speed at which data is being collected for all organisms. However, although sequencing data must be deposited into a public database (e.g. Genbank) before publication, the majority of morphological data still remains inaccessible and hence largely unavailable for meta-analysis. International initiatives such as the Global Soil Biodiversity Initiative[2], ECOFINDERS, and the EU-sponsored Global Soil Biodiversity Atlas are steps in the right direction but a common database of soil biodiversity data for both morphological and molecular data is still needed (Orgiazzi *et al.*, 2015).

6.7 | Soil sealing: status and trends

For millennia, the vast majority of people lived a rural life, largely dependent on agriculture and other rural occupations. Only over the last two centuries has the ratio between the urban and non-urban population

2 www.globalsoilbiodiversity.org

started to change rapidly. In 1800, only 3 percent of the world's population lived in cities; in 1900 14 percent, 47 percent in 2000, 50 percent in 2007, and 54 percent in 2014. The proportion of the urban population is expected to rise to 66 percent by 2050 (Figure 6.9).

The world's urban population is growing and cities are expanding in order to accommodate the increasing population and economic activity. It is not known with any certainty what share of the Earth's land surface (ca. 144 million km^2) is now occupied by cities or how much land will be required to accommodate the expected urban expansion (Potere *et al.*, 2009). One of the most accurate estimates of the extent of urban areas at global scale, based on the use of MODIS satellite data at a resolution of 500 m, indicates for the year 2000 an area of 657 000 km^2 (Potere *et al.*, 2009), equivalent to 0.45 percent of the Earth's land surface. Urbanization is an important contributor to regional and global environmental change (Foley *et al.*, 2005, Ellis and Ramankutty, 2008). The growth of cities has a vast impact on the landscape and significant impact on soil resources (Chen, 2007; Gardi *et al.*, 2014).

Between 1990 and 2000, the total extent of urban areas worldwide increased by 58 000 km^2. During this period, 2.8 percent of Europe's total land was affected by land use change, including a significant increase in urban land. Of the total land take in the EU between 1990 and 2000, 71 percent was for agriculture. Between 2000 and 2006, the equivalent figure was only 53 percent. Had the land taken for urban expansion been devoted to agriculture instead,

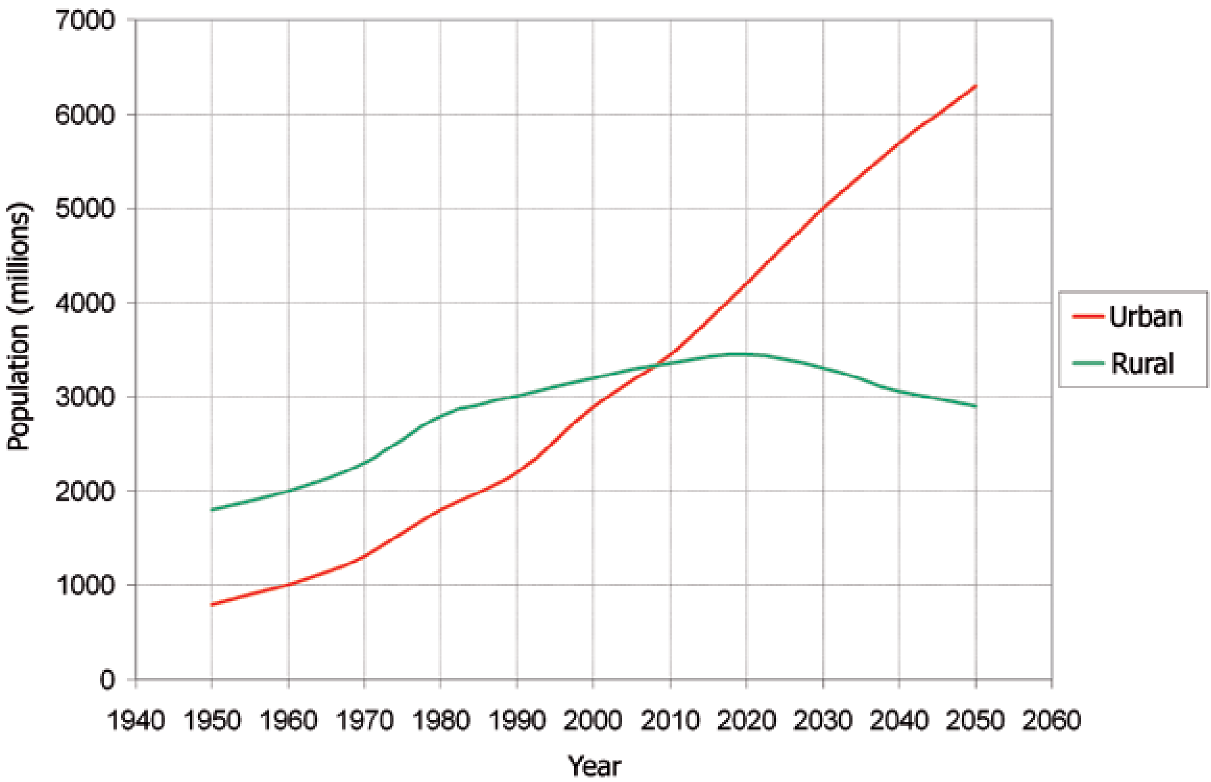


Figure 6.9 | Historical and predicted shift of the urban/rural population ratio.
Source: UN, 2008.

Figure 6.10 | Urbanisation of the best agricultural soils.

the land would have produced more than 6 Mt of wheat. More generally, the best quality soil in alluvial plains is often sealed by expanding cities and the rate of conversion is expected to increase rapidly, especially in developing countries.

The term 'soil sealing' is defined as the permanent covering of the soil surface with an impermeable material. Urbanisation affects the inner urban ecosystem as well as the neighbouring ecosystems. Besides the economic and social effects, negative environmental effects are predominantly linked to land consumption, the loss of high quality agricultural soil (Figure 6.10), the destruction of habitat, fragmentation of existing ecosystems, increased fuel consumption, air, water and soil pollution, and the alteration of microclimate.

Soil sealing is in practice equivalent to total soil loss – virtually all services and functions are lost except the carrying capacity as a platform for supporting infrastructure. The main negative impacts on ecosystem services include: virtually total loss of food and fibre production; a significant decrease or total loss of the soil's water retention, neutralization and purification capacities; the loss of the carbon sequestration capacity; and a significant decrease in the ability to provide (micro) climate regulation. The results include the loss of habitat for soil organisms, loss of soil biodiversity and nutrient cycling, and often a diminished landscape and natural heritage.

Urban expansion is, of course, both beneficial and essential. Historically, the beginning of the most important civilizations was associated with both the development of agriculture and the creation of urban settlements. As early as 3000 BC, cities had arisen in the Fertile Crescent, on the banks of Nile, in the Indus River valley and along major rivers in China. However, the very rapid urban expansion of recent times is creating the need for trade-offs, including decisions regarding soil health and the rate of soil sealing.

6.8 | Soil nutrient balance changes: status and trends

6.8.1 | Introduction

Though changes in soil nutrient balances may possibly affect all types of terrestrial ecosystems, rapid changes are more likely to occur in managed ecosystems as a result of the export of biomass or the addition of nutrients to sustain productivity. These managed ecosystems include cropland, intensively or extensively grazed rangelands or meadows, and forests. Monitoring changes in soil nutrient content is of particular relevance in managed ecosystems because it provides a means to evaluate future changes in the ability of soils to maintain their ecosystemic functions. On the one hand, negative balances ('nutrient mining') ultimately translate into crop nutrient deficiencies (, food production deficits and human nutritional imbalances. On the other hand, positive balances may lead to negative environmental and health externalities. Eutrophication, increased frequency and severity of algal blooms, hypoxia and fish kills and loss of habitat and biodiversity have been related to excessive inputs of N and P into fresh and coastal waters. Excess application of N has also lead to widespread contamination of groundwater by NO_3. Gaseous emissions of ammonia and nitrous oxide may also degrade air quality and contribute to acidification, eutrophication, ground-level ozone and climate change (Oenema, 2004; Chadwick *et al.*, 2011). In addition, strongly positive balances may reflect poor economic management of managed ecosystems. Nutrient balances can thus be viewed as indicators of sustainability of human-induced land use changes and land use practices.

Soil nutrients include the macronutrients nitrogen (N), phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg) and sulphur (S). In addition, the soil supplies micronutrients (boron, copper, iron, manganese, chloride, molybdenum, zinc), whose concentrations in plants are typically one or two orders of magnitude less than those of macronutrients. In most cases, N, P and K taken individually or in combination are the most limiting nutrients for plant growth. This section will therefore focus on these three elements. In soils, these nutrients may be present in different pools. Because the amount of nutrients in certain pools may vary strongly and erratically over short time intervals, stocks and mass balances are generally calculated on the basis of total nutrient content, without distinction among different forms (Roy *et al.*, 2003).

6.8.2 | Principles and components of soil nutrient balance calculations

Because the magnitude of the nutrient fluxes is often small compared to the total stock of nutrients in the soil profile, changes in soil nutrient stocks can be rather slow and difficult to detect over short time scales

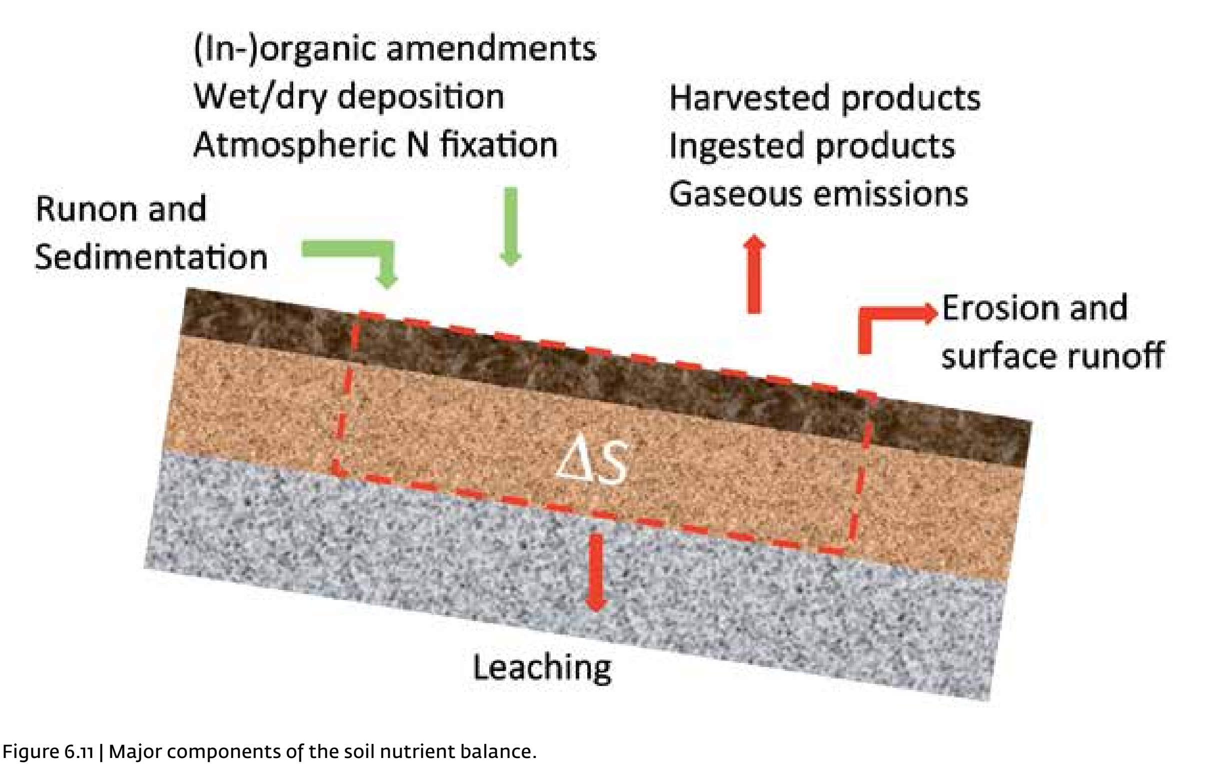


Figure 6.11 | Major components of the soil nutrient balance.

The red discontinuous line marks the soil volume over which the mass balance is calculated. Green arrows correspond to inputs and red arrows to losses. ΔS represents the change in nutrient stock.

(< decades). Hence calculating nutrient balances from nutrient flows rather than from changes in nutrient stocks has been preferred in many studies (Figure 6.11).

Table 6.6 lists the main inputs and outputs used for calculating the mass balances of N, P and K. Inorganic amendments are mostly composed of mineral fertilizers, but also comprise urine or minerals contained in irrigation water. Organic amendments include liquid, semi-solid or solid manures, compost, mulching material not produced on-site, and household refuse. It also includes faeces dropped by animals. In systems such as urban gardening, the re-use of waste water may also input organic compounds. Biological fixation by bacteria is restricted to N. Wet deposition refers to nutrients supplied with rainwater, whereas dry deposition refers to nutrients deposited as dust and aerosols. Dry deposition is a particularly important phenomenon in the case of K in areas downwind of major dust producing areas (e.g. West Africa;). Sedimentation refers to the deposition of sediment eroded upstream or to sediment deposited during river flooding. Additional fluxes may exist in specific situations (e.g. nutrients in subsurface lateral flows ; supply of NO_3 from groundwater ().

The main losses are related to nutrients contained in exported harvested products (crops or fodder), and nutrients contained in food ingested by primary grazers (Table 6.7). Nutrients may also be lost by gaseous emissions (NH_3, N_2, N_2O), through erosion and in surface runoff, or by leaching. The latter applies mostly to

Table 6.7 | Major components of soil nutrient mass balances for N, P and K

	N	P	K
Nutrient inputs			
Inorganic amendments	Yes	Yes	Yes
Organic amendments	Yes	Yes	Yes
Biological fixation	Yes	No	No
Dry or wet deposition	Yes	Yes	Yes
Sedimentation and run-on	Yes	Yes	Yes
Nutrient outputs			
Harvested products	Yes	Yes	Yes
Grazed products	Yes	Yes	Yes
Leaching	Yes	Generally negligible	Low
Gaseous emissions	Yes	No	No
Erosion and runoff	Yes	Yes	Yes

NO_3-N, to a lesser extent to NH_4-N and K, and to a very limited extent to PO_4-P except in coarse textured soils saturated with P.

6.8.3 | Nutrient budgets: a matter of spatial scale

The larger the spatial scale, the more certain nutrient flows are internalized. For instance, in a self-sufficient, well-managed farm, the net balance may be nil or close to nil. However, different parts of the farm may well have very different balances. Likewise, in extensively-managed agropastoral systems, nutrient flows mediated through livestock occur between rangelands and croplands. At a regional scale, the balances may thus be nil or only slightly negative, whereas large imbalances exist within the region (see Box 6.1).

At the global scale, fertilizer use and the growing of leguminous crops have resulted in a doubling of the rate at which N enters the terrestrial ecosystems as compared to pre-industrial levels. Likewise, the use of P fertilizers, animal feed supplements and detergents has led to a doubling of P inputs in the environment as compared to background P release from weathering. This is indicative of a net positive balance but hides large regional disparities. Bouwman, Beusen and Billen (2009) calculated global soil N and P balances for the year 2000. Outputs were restricted to harvested and grazed crops and grasses, whereas inputs included manure, fertilizers, N deposition and N fixation. These authors estimated the inputs to soils at 249 Tg N and 31 Tg P yr^{-1} and losses through harvest and grazing at 93 Tg N and 16 Tg P yr^{-1}. Assuming no build-up of N in the soil, their model predicted that 16 percent (41 Tg yr^{-1}) of the inputs may be lost by erosion and leaching, thereby contributing to a loss in environmental quality. In the case of P, their calculations predicted a net accumulation of P at a rate of 12 Tg yr^{-1} and losses of P through leaching and erosion of 2 Tg yr^{-1}. On a continental scale, considering both natural and agro-ecosystems, balances were always positive and comprised between 8.5 (North Asia) and 35 (Europe) kg N ha^{-1} yr^{-1}, and between 0.22 (Africa) and 5.5 (Europe) kg P ha^{-1} yr^{-1}. Focusing specifically on P and cropland, but restricting the balance calculations to fertilizer and manure inputs and harvest outputs, highlighted large P deficits in South America, northern United States and eastern Europe. Large P surpluses were found in the coastal United States, western and southern Europe, East Asia and southern Brazil.

Within the same continent, large variations in nutrient balances may occur. For 13 African countries, estimated balanced or negative nutrient budgets for N, P and K. At the national level, estimated soil nutrient balances for the year 2000 ranged from -2 to -60 kg N ha^{-1} yr^{-1}, from 0 to -11 kg P ha^{-1} yr^{-1}, and from -2 to -61 kg K ha^{-1} yr^{-1}. A later study at 1 km^2 resolution confirmed the overall negative balances but highlighted larger variability over short distances. The rate of nutrient mining by crops was generally low or moderate, because of low land productivity (low yields), but accumulated over many decennia nutrient depletion may become severe and may be strongly aggravated by soil erosion.

Based on a review of 57 nutrient budget studies related to the African continent, confirmed that N budgets at field and farm scale were largely negative whereas for phosphorus negative balances were reported in only 56 percent of the studies. Going from the continental scale to the plot scale, there was a tendency for the variability in nutrient budgets to increase. This is to be expected, as land uses and management practices in smallholder agriculture in Africa are highly diversified between farms, within farms and even within plots. The study did not find a clear trend in the magnitude of the nutrient budgets from plot to continental scales. This is in contrast to other studies which did report increasingly negative balances as the scale increased.

At even smaller scales, differences in soil fertility may arise from differential nutrient budgets. Strong gradients in soil fertility have been reported around villages, compounds, trees and shrubs as a result of higher levels of inputs (litter, household refuse, human excreta, manure and urine from resting animals, sedimentation, etc.) near these features. These are referred to as 'fertility rings' or 'fertility islands'.

Box 6.1 | Livestock-related budgets within village territories in Western Niger
(Schlecht et al., 2004)

In the Sahelian zone of West Africa, between 1.5 and 9 kg N ha^{-1} yr^{-1} and between 0.06 and 0.7 kg P ha^{-1} yr^{-1} are taken in by grazing livestock. The quantity varies by location and land use type (rangeland, cropland, fallow). However, up to 95 percent of the nutrients consumed by livestock are recycled through faeces. About 40-50 percent of these faeces end up being spatially concentrated at corralling spots or in farmyards, which represent only a few percent of the total village lands. Though nutrient in- and out-flows related to livestock account for only a small fraction of the nutrient flows in Sahelian crop-livestock systems, livestock thus plays a major role in the spatial redistribution of nutrients. Negative balances occur on rangelands and variable (positive or negative) balances are found in croplands depending on the intensity of application of organic amendments.

6.8.4 | Nutrient budgets: a matter of land use system, land use type, managementand household equity

Nutrient balances vary greatly across land use (LU) systems. Intensive growing of industrial crops in Europe is generally characterized by excess inputs of N, despite a recent tendency towards reduced fertilization driven by EU regulations and the economics of fertilizer use. As a result of the decoupling of livestock and land and because livestock are increasingly fed with imported feed, pastures are commonly exposed to excessive applications of manure (e.g. in Normandy in France, and in Denmark and Holland). Regarding P, after decades of excess application of P, there is nowadays a tendency for farmers to reduce their P application rates, or even to stop applying P altogether and to rely only on accumulated soil reserves and P released from soil mineral weathering.

At the other extreme, subsistence farming in developing countries is commonly characterized by negative balances, reflecting nutrient mining (Roy *et al.*, 2003). examined nutrient balances for different land uses in a Kenyan district. N deficits in excess of -100 kg ha^{-1} yr^{-1} were found for maize, sugar cane, and pyrethrum. P deficits in excess of -10 kg ha^{-1} yr^{-1} were found for sugar cane, pyrethrum, and beans, but P excesses occurred in tea and maize-bean plots. Except for coffee, tea and seasonal fallow, K deficits in excess of -50 kg ha^{-1} yr^{-1}

occurred in all systems. These observed differences reflect differences in the use of (in-) organic amendments, but also nutrient transfers across LU types. In the case of coffee for instance, mulching is recommended, which is done by using residues from other crops (e.g. bananas) or grasses from fallow land.

In Asia, both strongly positive and strongly negative balances have been reported. K deficits have been reported for rice-based systems across several Asian countries ranging from -25 to -70 kg ha^{-1} yr^{-1}. also reported K deficits in 71 paddy farms in south China, but found N and P surpluses. Based on negative nutrient balances for Bangladesh, Vietnam, Indonesia, Myanmar, the Philippines, and Thailand, and positive balances for Japan, Malaysia and Korea, it has been argued that lower-income countries with large and growing population were more likely to present negative balances whereas higher income countries with stable populations tended to have positive balances. In sub-Saharan Africa, the larger the population density, the more negative the N and P balances.

For similar systems, differences in nutrient balances may also arise from variable access of farmers to external inputs. In the Sudanian zone of west Africa, cultivated plots near hamlets tended to have less negative or more positive balances than plots near larger villages because farmers in hamlets cared better for their crops, earned more income from sales and therefore could invest more in fertilizers. Generally, cultivated plots near hamlets and villages benefit from greater additions of household refuse and human and animal faeces. However, social inequality in access to resources has been found to have an equally large or even larger effect on nutrient balances than distance from the village. For instance, positive N, P and K balances were observed for Fulani cropland because their large herds supply them with abundant manure. Likewise, nutrient budgets ranging from strongly negative to strongly positive were reported for banana-based systems in Tanzania depending on access to cattle and cattle management (Roy *et al.*, 2003). Especially in small-holder agriculture, site-specific management may also induce large fertility gradients over short distances.

(Peri-)urban agriculture is characterized by large excesses in nutrients, especially N. This is commonly driven by the market-oriented nature of this production system, which allows farmers to invest in external inputs. In addition, these systems often rely heavily on the re-use of urban solid waste and waste water. Hence, (peri-) urban production systems exemplify another form of large scale fertility transfer, from rural areas to urban areas. Food produced by nutrient mining in rural areas is consumed in cities, leading to strong soil enrichment of urban soils, especially at urban vegetable production sites (see Box 6.2).

6.8.5 | What does the future hold?

Soil nutrient budgets depend on the local socio-economic conditions but also on market prices of inputs and on policies. In Western Europe for instance, rising prices of fertilizers and the strengthening of environmental policies has led to reductions in N and P inputs into farmland, and this trend is expected to continue. Dwindling P resources and climate change may further affect soil nutrient balances, in managed but also in natural ecosystems.

Bouwman, Beusen and Billen (2009) evaluated the impact of four future development scenarios on nutrient balances for the year 2050. The scenarios, describing contrasting future development in agriculture nutrient use under changing climate, are based on the Millennium Ecosystem Assessment. In the most

Box 6.2 | Nutrient balances in urban vegetable production in West African cities

Based on a two year study of urban gardening sites in Niamey (Niger), it was found that N, P and K balances were all positive, with values for high and low input gardens respectively of 1133 and 290 kg ha^{-1} for N; 223 and 125 kg ha^{-1} for P; and 312 and 351 kg ha^{-1} for K. Similar N and P balances were reported for urban vegetable gardens in Kano (Nigeria), Bobo Dioulasso (Burkina Faso) and Sikasso (Mali). However, at these latter sites, K balances tended to be negative. Overall, urban vegetable production sites appear to be major nutrient sinks from which large environmental externalities can be expected.

pessimistic case, the global N balance may increase by 50 percent in the coming decades. In case of proactive policies aiming at closing the nutrient balance, the N balance is expected to remain constant at 150 Tg yr^{-1}. Regarding P, all scenarios predict a future increase in global soil P balance. These global balances hide large variations across regions and even across land uses. Unfertilized rangelands are likely to maintain negative P balances. Scenarios with a reactive approach to environmental problems portray significant increases in N and P balances in Asia, Central and South America and Africa, which can be strongly reduced by a proactive approach. For North America, Europe and Oceania, a shift from reactive to proactive environmental policies could allow limiting the increase in N and P balances, or even a decrease in the overall nutrient balance.

Whereas large positive nutrient balances sustained for extended periods of time in industrialized countries have resulted in negative environmental externalities, positive nutrient balances should not be viewed as necessarily environmentally harmful. Indeed, in many developing regions (e.g. sub-Saharan Africa), positive P balances are needed to restore soil fertility potential depleted by long lasting nutrient mining and to boost the often very low crop yields. Inputs of N in organic form may also be beneficial as part of a strategy to restore the soils' organic carbon stocks. Possible negative environmental externalities should be weighed against the benefits of food security, economic welfare and social well-being. To minimize the negative externalities, the best nutrient management approaches should be promoted through judicious policies.

6.9 | Soil compaction status and trends

Soil compaction is an important problem affecting productivity of soils across the globe. A hidden problem of soils occurring on or below the surface, compaction impairs the function of the subsoil by impeding root penetration and water and gaseous exchanges (McGarry and Sharp, 2003). Soil compaction reduces soil macroporosity e.g. from an optimum of 6 to 17 percent, and hence reduces pasture and crop yield (Drewry, Cameron and Buchan, 2008).

Soil compaction in most circumstances is a function of soil type (texture, mineralogy, organic matter), soil-water content and land management (e.g. tillage practices, traffic, grazing intensity). The problem is not limited to crop land but is also prevalent in rangelands and grazing fields, and even in natural non-disturbed systems. Soil compaction occurs when compressible soils are subjected to traction e.g. in forest harvesting, amenity land use, pipeline installation, land restoration, wildlife trampling (Batey, 2009) or winter grazing (Tracy and Zhang, 2008).

Trampling mechanically disrupts soil aggregates and reduces aggregate stability (Warren *et al.*, 1986) and its effect increases with stocking intensity (Willatt and Pullar, 1983). The degree of damage associated with trampling at a particular site depends on soil type (Van Haveren, 1983), soil water content, seasonal climatic conditions (Warren *et al.*, 1986), and vegetation type (Wood and Blackburn, 1984). Climate is therefore an important determinant of the effects of compaction. Where soil moisture deficits are large, a restriction in root depth may have severe effects but the same level of compaction may have a neglible effect where soil moisture deficits are small (Batey, 2009).

Soil compaction effects are long lasting or even permanent (Håkansson and Lipiec, 2000). Especially in cultivated land, soil compaction is exacerbated by low soil organic matter content. Intensive use of farm machinery including tillage implements such as the mould board, disc ploughs and disc harrows contributes to soil compaction, depending on the pattern of load and stress applied and the number of passes. The initial condition of the soil also plays a role, including soil moisture, organic matter content, bulk density, particle size distribution (including high silt content), and aggregate stability (Materechera, 2008; Horn *et al.*, 2005; Imhoff, Da Saliva and Fallow, 2004). Alfisols, a major soil used for crop production in the tropics and covering approximately 4 percent of the African land mass, are particularly vulnerable. They are strongly weathered and inherently of low organic matter and nutrient status, have a weak structure, and are highly susceptible to crusting, compaction and accelerated erosion (Lal, 1987).

Soil compaction decreases soil physical fertility by impairing storage and supply of water and nutrients, and by increasing erosion hazards and the transport of phosphorus and other nutrients out of the farming system. Soil compaction can reduce crop yields by as much as 60 percent (Sidhu and Duiker, 2006). The range of yield effects is variable, and depends partly on the crop. Cotton was found to be more sensitive to soil compaction than were soybeans, corn or Brachiaria brizantha (Busscher, Frederick and Bauer, 2000). Yields of sugarcane (Saccharum officinarum L.) were reduced by 40 percent with sub-surface compaction of a clay soil (Jouve and Oussible, 1979), while in a clay loam soil wheat yields were reduced by 12 to 23 percent (Oussible, Crookstone and Larson, 1992). The compaction effects on yield are greatest when the crop is under stress, such as from drought or an excessively wet growing season (Sidhu and Duiker, 2006). Krmenec (2000) observed stand count reductions of 20 to 30 percent, plant height decreases of up to 50 percent and yield reductions of about 19 percent in compacted compared to non-compacted plots. The study of Voorhees, Nelson and Randall (1986) illustrates that a one-time compaction event can lead to reduced crop yields up to 12 years later. In another study, soil compaction reduced grass yield by up to 20 percent due to N-related stresses (Smith, McTaggart and Tsuruta, 1997; Douglas, Campbell and Crawford, 1998). In addition, the creation of waterlogged zones or of dry zones caused by shallow rooting can deny plants access to deeper reserves of water (Batey and McKenzie, 2006).

Additional consequences include chemical changes, such as the amount of greenhouse gases (nitrous oxide and methane) emitted from or taken up in a soil (Hansen, Maehlum and Bakken, 1993; Ruser *et al.*, 1998), and reduced root growth and consequently lower crop yields. A study by Gray and Pope (1986) showed also that the incidence of Phytophthora root rot in soybeans (Glycine max. L.) was greater with soil compaction. Soil compaction increases the abundance of anaerobic microsites and decreases the proportion of coarse pores, which may favour emissions of both CH_4 and N_2O (Ball, Scott and Parker, 1999a). Only rarely has soil compaction been associated with positive impacts, such as increasing the plant-available water capacity of sandy soils (Rasmussen, 1985) or reducing nitrate leaching (Badalıkova and Hruby, 1998) or benefiting soybean grown in areas prone to iron deficiency chlorosis in wet years (DeJong-Hughes *et al.*, 2001).

6.9.1 | Effect of tillage systems on compaction

While all tillage methods tend to reduce soil bulk density and penetration resistance to the depth of tillage (Erbach *et al.*, 1992), equipment used in modern agriculture causes soil compaction of topsoil and subsoil. Working the soil to avoid compaction requires timing of tillage in relation to soil water moisture content and soil texture (Håkansson and Lipiec, 2000). No-tillage (NT) agriculture is gaining wide acceptance and is among the top options in the portfolio of technologies to reduce tillage costs, conserve soil and water, increase soil organic carbon (SOC) pools, and reduce net CO_2 emissions, which contribute to global warming (Lal *et al.*, 2004). Despite the numerous benefits of NT, there is no consensus yet on its role in alleviating soil compaction: some researchers report increased compaction associated with the practice (Bueno *et al.*, 2006) and others a decrease in compaction (Gregory, Shea and Bakko, 2005). Increasing soil organic matter, as practiced in conservation agriculture, reduces soil compactibility (Thomas, Haszler and Blevins, 1996), but residue availability remains a key challenge, especially in Africa.

6.9.2 | What is the extent of deep soil compaction?

Soil compaction affects mainly topsoils (Balbuena *et al.*, 2000; Flowers and Lal, 1998) but can also affect subsoils at depths >30 cm. Most subsoil compaction occurs when the soil is wet and field equipment weights exceed 10 tons per axle. The average weight and power of vehicles used on farms has approximately tripled since 1966 and maximum wheel loads have risen by a factor of six (Chamen, 2006). While remediation of shallow compaction is possible, for example by ripping and subsoiling, correcting soil compaction at depths below 45 cm is challenging (Batey, 2009; Berli *et al.*, 2004). Both topsoil and subsoil compaction have been acknowledged by the European Union as a serious form of soil degradation, estimated to be responsible for degradation of up to 33 million ha in Europe (Akker and Canarache, 2001). Similar compaction problems have been reported elsewhere, including in Australia, Azerbaijan, Japan, Russia, China, Ethiopia and New Zealand (Hamza and Anderson, 2005). The total amount of compacted soil worldwide has been estimated at approximately 68 million ha or around 4 percent of the total land area (Oldeman, 1992; Soane and Van Ouwerkerk, 1994). Nearly 33 million ha is located in Europe, where the use of heavy machinery is the main cause. Cattle trampling and insufficient cover of the top soil by natural vegetation or crops account for compaction of 18 million ha in Africa, and 10 million ha in Asia (Flowers and Lal, 1998; Hamza and Anderson, 2003). Agricultural mismanagement (80 percent) and overgrazing (16 percent) are the two major causative factors of human induced soil compaction (Oldeman, 1992).

6.9.3 | Solutions to soil compaction problems

Soil compaction, like soil chemical characteristics, should be monitored routinely and corrected as part of soil management (Batey, 2009). Although soil compaction effects on soil biodiversity and related functions and processes depend on several site and soil properties, a threshold of effective bulk density of 1.7 g cm–3 is the maximum above which only negative effects are observed (Beylich *et al.*, 2010). Managing soil compaction can be achieved through appropriate application of some or all of the following techniques: (a) addition and maintenance of adequate amount of soil organic matter to improve and stabilize soil structure (Heuscher, Brandt and Jardine, 2005); (b) guiding, confining and minimizing vehicular traffic to the absolutely essential by reducing the number and frequency of operations, and performing farm operations only when the soil moisture content is below the optimal range for the maximum proctor density (Kroulik *et al.*, 2009); (c) mechanical loosening such as deep ripping (Hamza and Anderson, 2005); and (d) selecting a rotation which includes crops and pasture plants with strong tap roots able to penetrate and break down compacted soils (Hamza and Anderson, 2005). Promoting macrofauna activity can accelerate creation of channels for water infiltration and root growth. Arbuscular mycorrhiza can to some extent alleviate the stress of soil compaction. This effect has been observed on wheat growth following increased root/shoot ratio of wheat under compaction (Miransari *et al.*, 2008). In the long-term, soil compaction can be reduced by natural processes that cause the soil to shrink and swell such as wetting and drying (Shiel, Adey and Lodder, 1988), and freezing and thawing (Miller, 1980).

Soil moisture lower than the plastic limit is desirable for cultivation. Traffic should be avoided or restricted when condition are otherwise. For farmers, a simple test to avoid soil compaction involves squeezing a small lump of soil into a ball and rolling it into a rod about 3 mm in diameter. If a rod can be made easily, the soil is too wet and will compact if it is worked or has animals or machinery on it. If the rod is crumbly the water content should allow traffic and cultivation without compaction. If a rod will not form at all, the soil could be too dry for tillage in a sandy or loamy soil. This test should be run at several points over the full depth of any proposed cultivation.

6.10 | Global soil-water quantity and quality: status, processes and trends

The world relies on its freshwater for ecosystem health and human well-being and prosperity. Yet only 2.5 percent of the world's water is fresh, and of that, 68.7 percent is in the form of ice. Groundwater comprises 30.1 percent of the freshwater, and just 0.4 percent of the world's freshwater is in lakes, rivers and the soil.

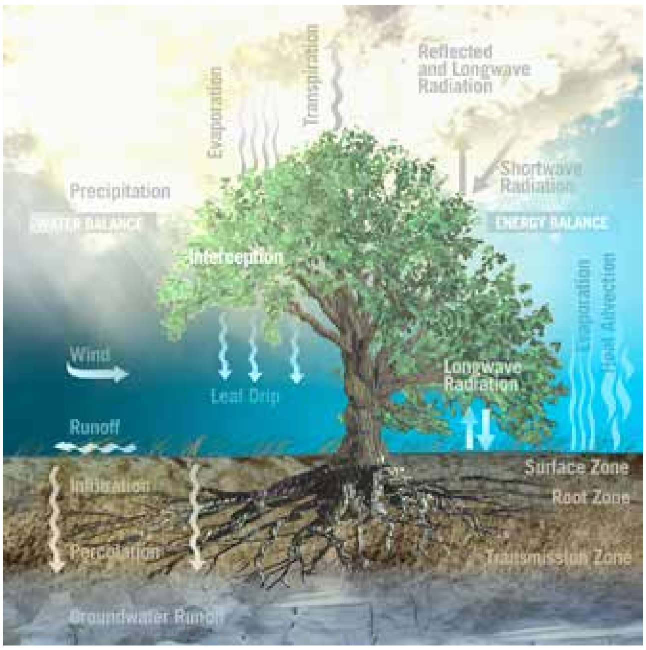
6.10.1 | Processes

Soil water comprises only 0.05 percent of the world's store of freshwater. However, the upward and downward fluxes of water and energy through the soil are massive, and they are strongly linked. The flows are upward in the form of water vapour, long-wave radiation and reflected short-wave radiation, and downward in the form of liquid water and short-wave radiation (Figure 6.12). The soil-vegetation system is the first receiver of the rain and energy that fall on our lands. The soil-vegetation system, which encompasses the upper reaches of the groundwater or basement rock to just above the soil-vegetative layer, is the critical zone for controlling terrestrial water quantity and quality.

Rodell *et al.* (2015) estimate the total annual precipitation onto continents to be 116 500±5 100 km^3 yr^{-1} – equivalent to approximately five-times the water stored in the Great Lakes of North America. Sixty percent of this (70 600±5 000 km^3 yr^{-1}) returns to the atmosphere through evapotranspiration. The remaining 40 percent (45 900±4 400 km^3 yr^{-1}) leaves the continents as runoff, with the greatest proportion either running off the surface of the soil or returning to streams via the groundwater flow system after passing through the soil. Thus small changes due to human intervention and climate change that alter these fluxes can have very large impacts on the store of soil water.

The quantity, quality and flow of water over and through soil affect the spatial and temporal availability and usage of water. The quantity of soil water in a particular layer of soil can be determined by the soil-water retention curve, the so-called 'soil-water characteristic' (Figure 6.13). This curve describes the relationship between the amount of water a particular soil can hold and the energy, or matric potential, required to overcome adhesive and cohesive forces to extract water from the soil. Soils of different textures have very differing characteristic curves (Figure 6.13) and this affects the movement and storage of water in the landscape.

The quality of the soil's water is determined by the impurities and pollutants present in the soil water, which may, or may not be adsorbed to and/or exchanged in some part with the soil's reactive matrix materials.



© European Space Agency

Figure 6.12 | The flows of water and energy through the soil-vegetation horizon

The flow of soil water is determined by the gradient in the matric potential, and the soil's hydraulic conductivity, K (cm day^{-1}) (Figure 6.14), which describes the ease with which water flows through the soil pore space. The hydraulic conductivity curve is highly non-linear and strongly dependent on the soil's water content, θ, and hence matric potential (Figure 6.14). Soil water flow can vary from very slow in soil with small pores, to very fast in soil with large interconnected pores.

The soil-water characteristic (Figure 6.14) is an important factor affecting soil microbiology and rhizosphere ecology. It controls the stability of the spatial and temporal geometry of the soil pore space, which in turn

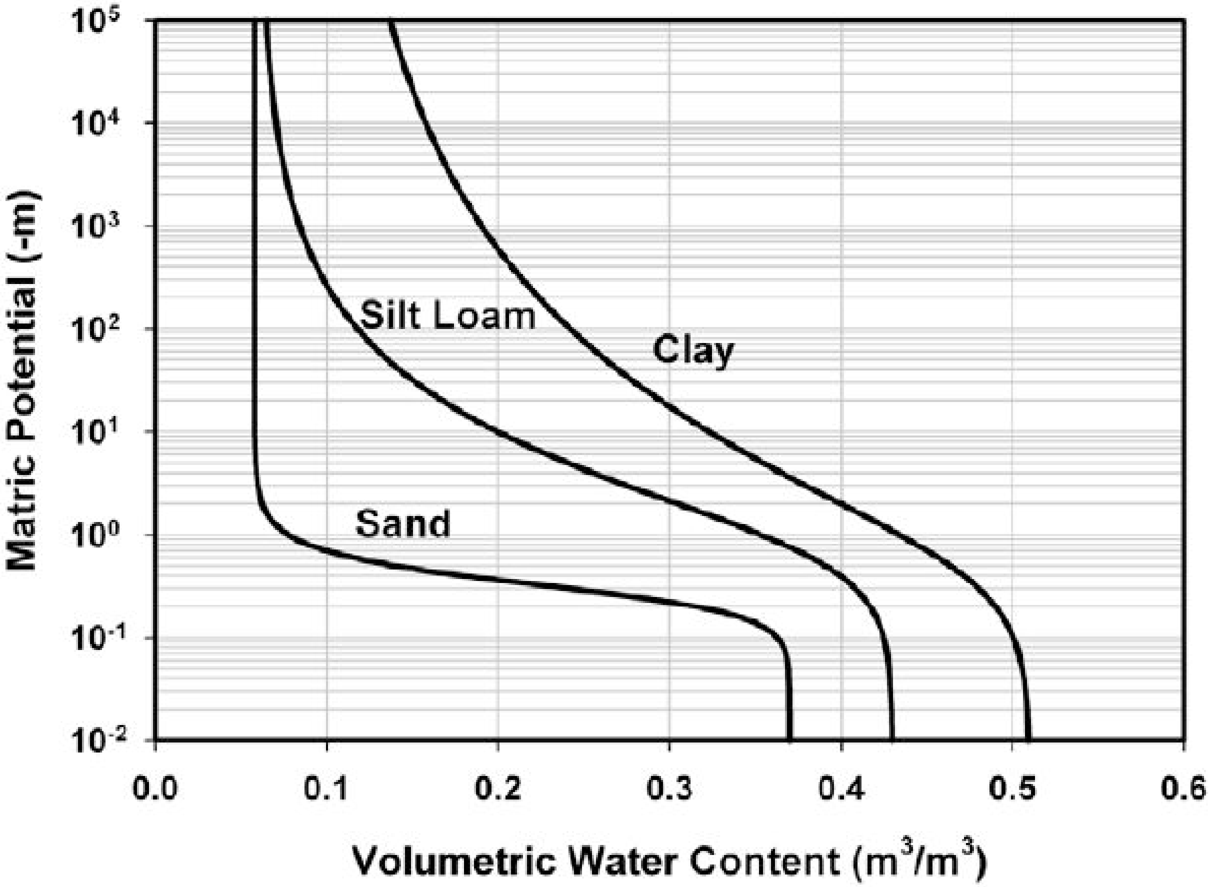


Figure 6.13 | The soil-water characteristic curve linking matric potential, to the soil's volumetric water content.
Source: Tuller and Or, 2003.

defines the allocation of resources to soil biota, the transport of liquids, gases and solutes to and from roots, and the diversity of microbial habitats (Hinsinger *et al.*, 2009). The soil micro-organisms are largely aquatic in nature and do not inhabit the air-filled pores. They live instead in the liquid phase of the pores, the thickness of which is controlled by the matric potential, which also controls the size and distribution of water-filled pores that provide the hydraulic connectivity through soils.

The interactions between the structure and physical, chemical and biological components of the soil control

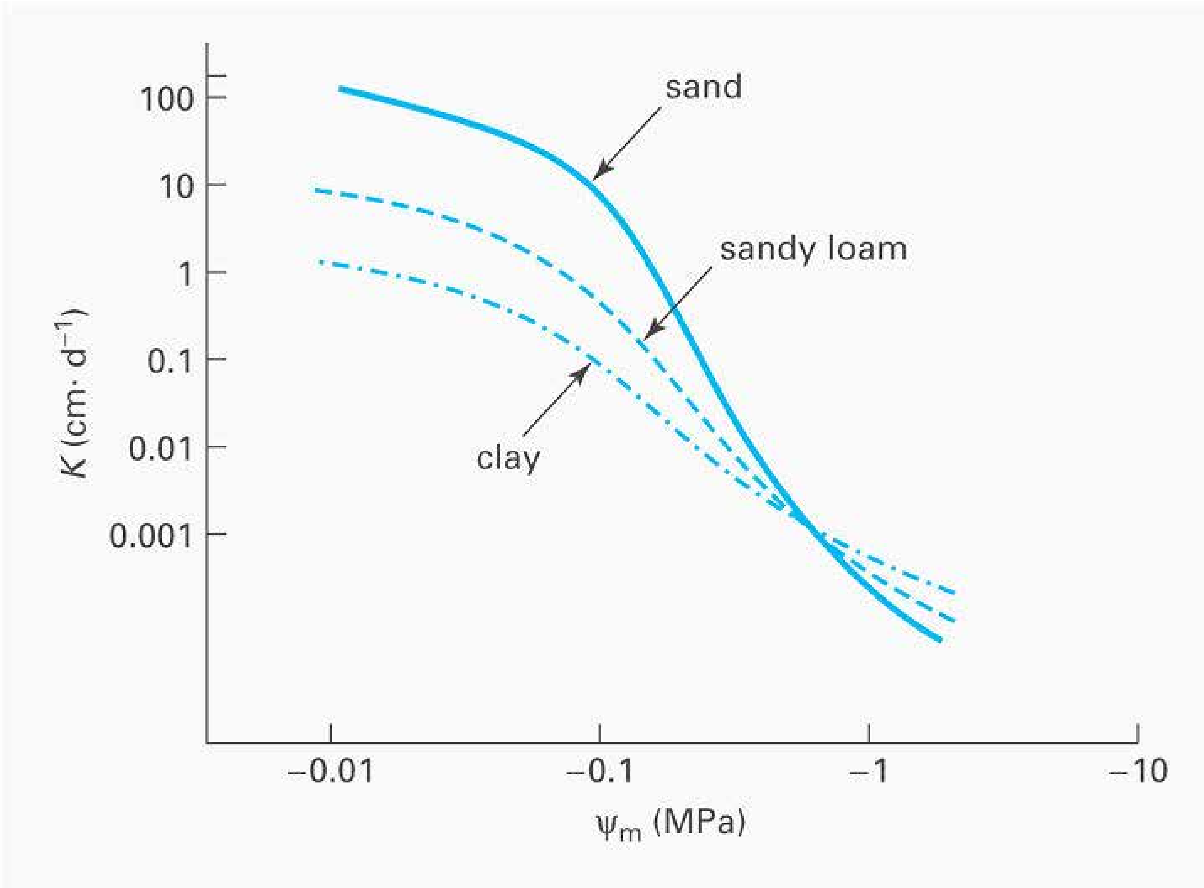


Figure 6.14 | The soil's hydraulic conductivity, K (cm day[-1]) in relation to the matric potential, ψ (MPa). As the matric potential becomes more negative the soil's water content drops (see Figure 6.16) which increases the tortuosity and slows the flow of water.
Source: Hunter College.[3]

the myriad soil functions and processes that are essential for healthy soils, ecosystems and human well-being.

The soil acts as buffer and filter. Indeed, our soil is the world's largest water filter. And through this buffering and filtering, soil controls the quantity and quality of the world's liquid freshwater.

6.10.2 | Quantifying soil moisture

Soil water varies on multiple time and space scales, driven by climate, weather variability, land cover, topography and soil type and structure (Figure 6.15). Measuring variations in soil water is challenging especially at large scales where the cost of direct measurement would be very high. Long-term measurement networks have historically been limited to a few locations globally (Robock *et al.*, 2000). However, with the recognition of soil water as an essential climate variable and the realization that in-situ measurements are necessary for the calibration and validation of remote sensing, the number of operational monitoring networks is increasing (Dorigo *et al.*, 2011). There are also short-term experimental campaigns with multi-scale soil water sampling (Crow *et al.*, 2012). For example, the Soil Climate Analysis Network (SCAN) in the United States provides soil water measurements for 174 sites across the United States, with some measurements dating back to 1992. New technologies such as the COsmic-ray Soil Moisture Observing System (COSMOS) cosmic-ray neutron probes (Zreda *et al.*, 2012) have enabled more efficient and larger measurement footprints of the order of several hundreds of square meters.

At continental scales, the only practical means of estimating soil water is from satellite sensors or simulation models. Satellite-based measurements of soil water are generally based on measuring microwave

3 http://www.geo.hunter.cuny.edu/tbw/soils.veg/lecture.outlines/soils.chap.5/soils_chapter.5.htm

emissions that vary because of the sensitivity of the soil dielectric constant to its wetness. These approaches use radiative transfer models to simulate the transfer of radiation emitted from the soil through the vegetation canopy and atmosphere to the satellite sensor. However, measurements have generally been restricted to the top centimetre of the soil column because of the penetration depth of microwave signals for current sensors (> 6 GHz). They are also restricted to sparsely vegetated regions. The recently launched Soil Moisture Ocean Salinity (SMOS) (Kerr *et al.*, 2001) and Soil Moisture Active Passive (SMAP) (Entekhabi *et al.*, 2010) satellite missions improve on this by using L-band (1-2 Ghz) sensors that have penetration depths of the order 5 cm and are less restricted by dense vegetation. Estimates from land surface models have also contributed to understanding the variation of soil water at large scales (Sheffield and Wood, 2008). These simulation models are driven by observations of precipitation, temperature and other meteorology and simulate the surface hydrological cycle with soil water as a prognostic state variable. Recent efforts have developed long-term simulations of soil water at regional to global scales (Sheffield and Wood, 2007, 2008; Haddeland *et al.*, 2011), although uncertainties exist because of missing process representation in the models and because of errors in model structure, parameters and the meteorological forcings.

6.10.3 | Status and trends

Understanding variations in soil water is critical for a range of applications including drought risk management, agricultural decision making, and understanding and attributing climate change impacts. Currently, long-term (multi-decadal) time series of soil water which have been developed from models and satellite retrievals are being used to understand variability and long-term changes in soil water (Sheffield and Wood, 2008; Dorigo *et al.*, 2012). Figure 6.16(a) shows the spatial variability of soil water globally from model simulations, ranging from high values in the wet tropics and northern boreal forests, to the desert

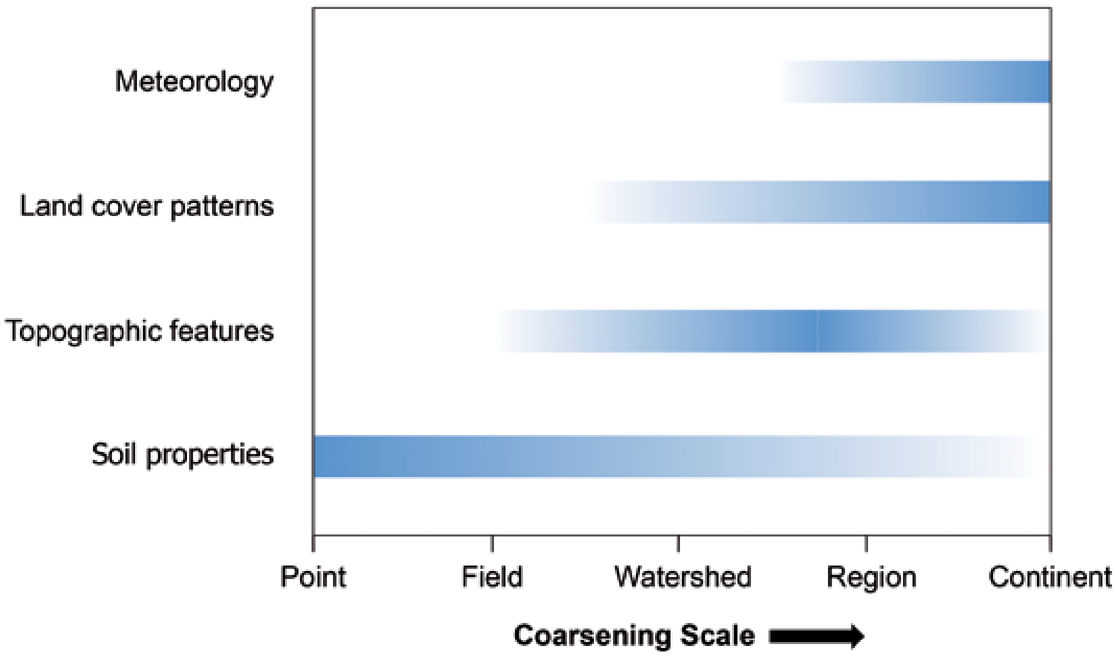


Figure 6.15 | Factors controlling soil water spatial variability and the scales at which they are important.
Source: Crow et al., 2010.

regions, such as North Africa, the Middle East, central Asia and Australia. Seasonally, soil water varies with changes in precipitation (Figure 6.16 b) with the largest variations in the monsoonal regions of south and southeast Asia, west and central Africa and the Amazon. From year-to-year, the El Niño Southern Oscillation (ENSO) is the main driver of soil water variability globally (Sheffield and Wood, 2011), often leading to drought conditions in the Amazon, south Asia, eastern Australia and southern Africa during El Niño years, and to drought in the United States southwest and the Horn of Africa in La Niña years.

Longer-term changes in soil water are mostly driven by changes in precipitation (Figure 6.16 c and d). Global warming may be playing a role in drying soil water in some regions, although this is a subject of debate. Over the past 60 years, soil water has been generally wetting over the western hemisphere and drying over the eastern hemisphere, mostly in Africa, East Asia and Europe. Trends over the past 20 years (Figure 6.16 e and f) indicate intensification of drying in northern China and southeast Australia, and switches from wetting to drying across much of North America, and southern South America, in part because of several large-scale and lengthy drought events.

6.10.4 | Hotspots of pressures on soil moisture

Hotspots of pressures on soil water quantity and quality have emerged around the globe. These result from changes in soil water driven by climate change and variability, coupled with human pressures on soil water through, for example, agricultural intensification and extensification. We describe three hotspots: the North China Plain, the Horn of Africa, and the southwestern United States.

The North China Plain has seen rapid expansion of agriculture driven by population growth and increasing demand for food. This area is relatively dry with around 500 mm yr^{-1} of precipitation and so irrigation from groundwater has become an important feature of agricultural intensification. However, groundwater has been used at unsustainable rates, with the result that groundwater levels are dropping by over 1 m per year in some parts (Kendy *et al.*, 2003). Furthermore, precipitation has decreased over the past few decades (Figure 6.16 f). Coupled with intensive irrigation and fertilizer application, this has led to declines in soil water quality through salinization and nitrogen leaching (Kendy *et al.*, 2003).

Drought has plagued many parts of Africa because of high climate variability from year to year. Severe droughts in the 1970s and 1980s led to the deaths of hundreds of thousands of people across the Sahel

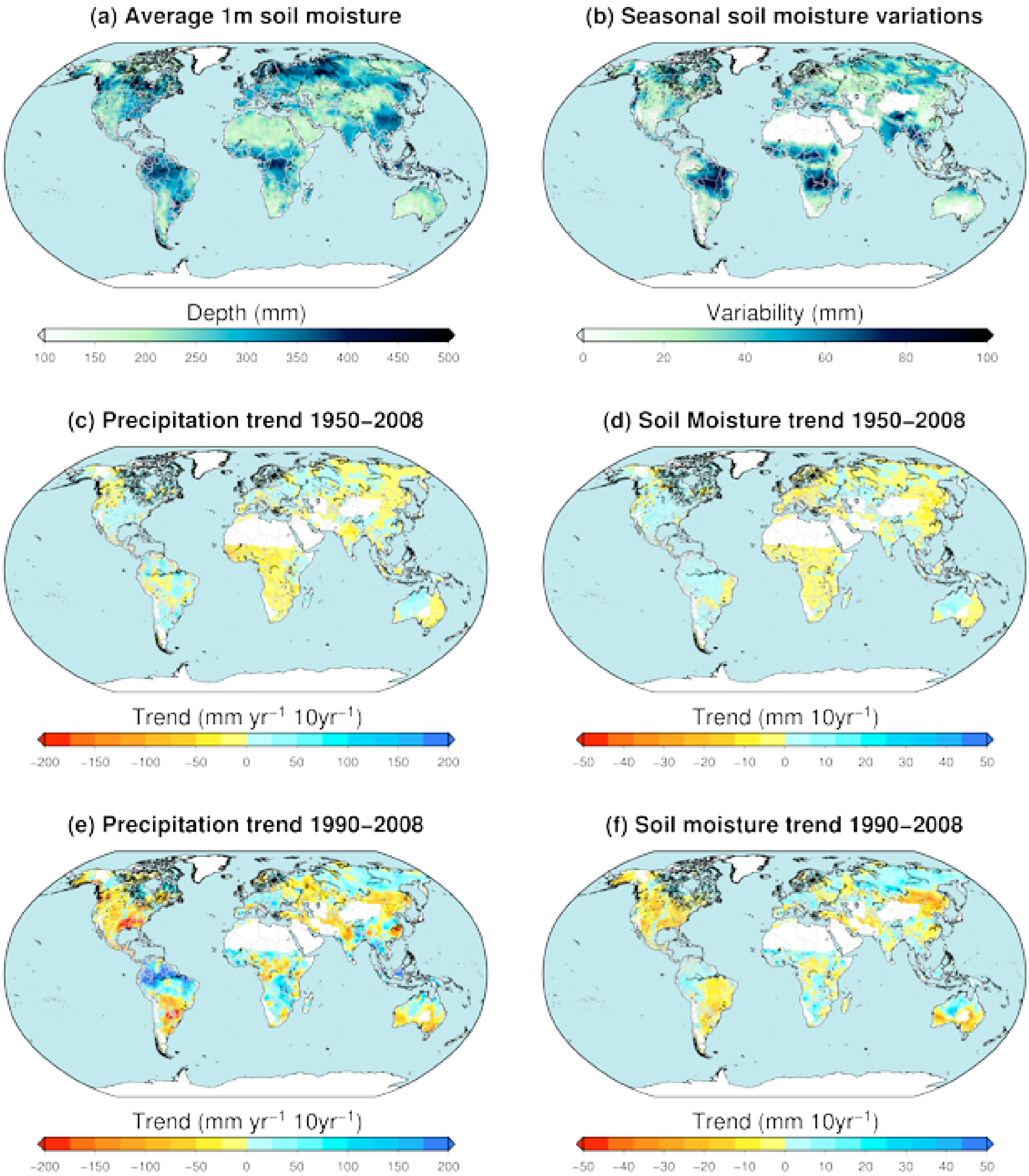


Figure 6.16 | (a) Global distribution of average soil moisture depth in the top 1 m of the soil. (b) Seasonal variability in soil moisture calculated as the standard deviation of monthly mean soil moisture over the year. (c-d) Global trends (1950-2008) in precipitation and 1 m soil moisture. (e-f) As for (c-d) but for 1990-2008. Results for arid regions and permanent ice sheets are not shown. Source: Sheffield and Wood, 2007.

(Sheffield and Wood, 2011). Recent droughts in the Horn of Africa have continued to affect millions of people (Ledwith, 2011; UN, 2011), driven by an overall decline in rainfall that is expected to continue and may be linked to anthropogenic warming of the Indian Ocean (Funk *et al.*, 2008; Williams *et al.*, 2011). Monitoring soil water and its impacts on food security in the Horn of Africa is particularly difficult because of the lack of ground measurements. Nonetheless, the use of satellite and modelling technologies has the potential to provide drought and famine early warning (Anderson *et al.*, 2012; McNally *et al.*, 2013; Sheffield *et al.*, 2014).

Soil water in the southwestern United States has been affected over the past two decades by frequent severe drought events (2000-2002, 2007, 2009), culminating in a three year drought in California (2011-2014) with state-wide impacts on agriculture (Howitt *et al.*, 2014). A shortfall in irrigation water owing to a depleted mountain snowpack was partly offset by increasing groundwater pumping. Recent analysis using Gravity Recover and Climate Experiment (GRACE) satellites has confirmed the resulting massive losses of groundwater since the 1980s from the aquifers underlying California's agriculturally important Central Valley (Famiglietti and Rodell, 2013). McNutt (2014) concludes that "… it is this underground drought we can't see that is enduring, worrisome, and in need of attention".

6.10.5 | Conclusions

Soil water is vital for the health of terrestrial ecosystems and human well-being. Although only a small fraction of the world's water is stored in the soil, the fluxes of water through the soil are massive.

On the time-scale of years, the El Niño Southern Oscillation is the prime control on the global variability in soil water. At longer time-scales, the global pattern of precipitation is the dominant driver in controlling changes in soil water. This pattern may be influenced by climate change.

Global analysis of the changing patterns of soil water has revealed the emergence of three global hotspots in terms of quantity and quality. These are the North China Plan, the Horn of Africa and the southwestern United States. There will be great challenges to address in these hotspot regions and in other pockets where declining soil water quantity and quality is threatening ecosystem health and human well-being.

References

Abd Elrahman, S.H., Mostafa, M.A.M., Taha, T.A., Elsharawy, M.A.O. & Eid, M.A. 2012. Effect of different amendments on soil chemical characteristics, grain yield and elemental content of wheat plants grown on salt-affected soil irrigated with low quality water. *Annals Agric. Sci*., 57: 175-182.

Abdelfattah, M.A. & Shahid, S.A. 2007. A comparative characterization and classification of soils in Abu Dhabi coastal area in relation to Arid and Semi-Arid conditions using USDA and FAO soil classification System. *Arid Land Research & Management*, 21: 245-271.

Abdulkadir, A., Sangaré, S.K., Amadou, H. & Agbenin, J.O. 2014. Nutrient balances and economic performance in urban and peri-urban vegetable production systems of three west African cities. *Experimental Agriculture*, 51: 126–150

Aherne, J. & Posch, M. 2013. Impacts of nitrogen and sulphur deposition on forest ecosystem services in Canada. *Current Opinion in Environmental Sustainability*, 5: 108–115

Akiba, M. 1933. The threshold wind speed of sand grains on a wetted sand surface. *Journal of Agricultural Engineering in Japan*, 5: 157–174.

Akker, J.J.H. & Canarache, A. 2001. Two European concerted actions on subsoil compaction. *Landnutzung* *und Landentwicklung*, 42: 15–22.

Alcamo, J., Van Vuuren, D., Cramer, W., Alder, J. & Bennett, E. 2006. Changes in ecosystem services and their drivers across scenarios. In S.R. Carpenter, ed. *Ecosystem and human well-being: scenarios*, pp. 279–354. Washington, DC, Island Press.

Amundson, R. 2001. The carbon budget in soils. *Annu. Rev. Earth Planet Sci*, 29: 535-562.

Anderson, W.B., Zaitchik, B.F., Hain, C.R., Anderson, M.C., Yilmaz, M.T., Mecikalski, J. & Schultz, L. 2012. Towards an integrated soil moisture drought monitor for East Africa. *Hydrol. Earth Syst. Sci.*, 16: 2893-2913.

Badalıkova, B. & Hruby, J. 1998. Physical soil properties at different systems of sugar beet cultivation. *In* Proc.: Soil condition and crop production. Gödöllö, Hungary, pp 83–85.

Bakker, M.M., Govers, G. & Rounsevell, M.D.A. 2004. The crop productivity-erosion relationship: an analysis based on experimental work. *Catena*, 57(1): 55-76.

Bakker, M.M., Govers, G., Jones, R.A. & Rounsevell, M.D.A. 2007. The effect of soil erosion on Europe's crop yields. *Ecosystems*, 10(7): 1209-1219.

Balbuena, R.H., Terminiello, A.M., Claverie, J.A., Casado, J.P. & Marlats, R. 2000. Soil compaction by forestry harvester operation. Evolution of physical properties. *Revista Brasileira de Engenharia Agricola e Ambiental*, 4: 453-459.

Ball, B.C., Scott, A. & Parker, J.P. 1999a. Field N_2O, CO_2 and CH_4 fluxes in relation to tillage, compaction and soil quality in Scotland. *Soil & Tillage Research*, 53: 29-39.

Barak, P., Jobe, B.O., Krueger, A.R., Peterson, L.A. & Laird, D.A. 1997. Effects of long-term soil acidification due to nitrogen fertilizer inputs in Wisconsis. *Plant and Soil*, 197: 61-69.

Bardgett, R.D. & van der Putten, W.H. 2014. Belowground biodiversity and ecosystem functioning. *Nature*, 515: 505-511.

Batey, T. & McKenzie, D.C. 2006. Soil compaction: identification directly in the field. *Soil Use and Management*, 22(2): 123–131

Batey, T. 2009. Soil compaction and soil management – a review. *Soil Use and Management*, 25: 335–345

Batjes, N.H. 1996. Total carbon and nitrogen in the soils of the world. *European Journal of Soil Science*, 47: 151–163.

Batjes, N.H. 2002. *Soil parameter estimates for the soil types of the world for use in global and regional modelling* (Version 2.1). ISRIC Report 2002/2c. Wageningen, International Food Policy Research Institute (IFPRI) and International Soil Reference and Information Centre (ISRIC)

Berli, M., Kulli, B., Attinger, W., Keller, M., Leuenberger, J., Fluhler, H., Springman, S.M. & Schulin, R. 2004. Compaction of agricultural and forest subsoils by tracked heavy construction machinery. *Soil & Tillage Research*, 75: 37–52

Bernoux, M., Cerri, C.C., Cerri, C.E.P., Neto, M.S., Metay, A., Perrin, A.-S., Scopel, E., Razafimbelo, T., Blavet, D. & Piccolo, M.D.C. 2006. Cropping systems, carbon sequestration and erosion in Brazil, a review. *Agronomy for Sustainable Development*, 26(1): 1-8

Beylich, A., Oberholzer, H., Schrader, S., Hoper, H. & Wilke, B. 2010. Evaluation of soil compaction effects on soil biota and soil biological processes on soils. *Soil and Tillage Research*, 109: 133-143.

Bielders, C.L., Rajot, J.-L. & Michels, K. 2004. L'érosion éolienne dans le Sahel nigérien: influence des pratiques culturales actuelles et méthodes de lutte. *Science et changements planétaires/Sécheresse*, 15: 1–14

Bohn, H.L. 1976. Estimate of Organic Carbon in World Soils. *Soil Sci. Soc. Am. J.*, 40: 468–470.

Bohn, H.L. 1982. Estimate of Organic Carbon in World Soils: II. *Soil Sci. Soc. Am. J.*, 46: 1118–1119.

Bohr, H. 1978. On organic soil C and CO_2. *Tellus*, 30: 472-475

Bolan, N.S., Hedley, M.J. & White, R.E. 1991. Processes of soil acidification during nitrogen cycling with emphasis on legume based pastures. *Plant and soil*, 134: 53-63

Bouwman, A.F., Beusen, A.H.W. & Billen, G. 2009. Human alteration of the global nitrogen and phosphorus soil balances for the period 1970-2050. *Global Biogeochemical Cycles*, 23(4): 1-16

Brennan, R.F., Bolland, M.D.A. & Bowden, J.W. 2004. Potassium deficiency, and molybdenum deficiency and aluminium toxicity due to soil acidification, have become problems for cropping sandy soils in south-western Australia. *Australian Journal of Experimental Agriculture*, 44: 1031-1039

Breuning-Madsen, H. & Awadzi, T.W. 2005. Harmattan dust deposition and particle size in Ghana. *Catena*, 63: 23-38

Breure, A.M., Mulder, C.H., Rutgers, M., Schouten, A.J. & Van Wijnen, H.J. 2004 Belowground biodiversity as an indicator of sustainability for soil use. Land use systems in grassland dominated regions. *Grassland Science in Europe*, 9: 195-197

Bueno, J., Amiama, C., Hernanz, J.L. & Pereira, J.M. 2006. Penetration resistance, soil water content, and workability of grasslands soils under two tillage systems. *Trans. ASAE*, 49: 875–882.

Bullock, P., Houerou, L., Hoffman, M.T., Rounsevell, M., Sehgal, J. & Varallay, G. 1996. Land degradation and desertification. In R.T. Watson, M.C. Zinyowera, R.H. Moss & D.J. Dokken, eds. *Climate change 1995: impacts, adaptations and mitigation of climate change: scientific-technical analyses*, pp. 171-190. Contrinution of working group II to the Second Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press.

Bundschuh, J., Litter, M.I., Parvez, F., Román-Ross, G., Nicolli, H.B., Jean, J., Liu, C., López, D., Armienta, M.A., Guilherme, L.R.G., Cuevas, A.G., Cornejo, L., Cumbal, L. & Toujaguez, R. 2012. One Century of Arsenic Exposure in Latin America: A Review of History and Occurrence from 14 Countries. *Science of the Total Environment*, 429: 2–35.

Buringh, P. 1984. Organic carbon in soils of the World. In G.M. Woodwell, ed. *The role of terrestrial vegetation in the global carbon cycle*, pp. 41-109. SCOPE 23. Chichester, J.Wiley & Sons.

Busscher, W.J., Frederick, J.R. & Bauer, P.J. 2000. Timing effects of deep tillage on penetration resistance and wheat and soybean yield. *Soil Science Society America Journal*, *Madison*, 64(3): 999-1003

Carpenter, S.R. 2008. Phosphorus control is critical to mitigating eutrophication. *Proceedings of the National Academy of Sciences of the United States of America*, 105: 11039–11040.

Carré, F., Hiederer, R., Blujdea, V. & Koeble, R. 2010. *Background Guide for the Calculation of Land Carbon Stocks in the Biofuels Sustainability Scheme Drawing on the 2006*. IPCC Guidelines for National Greenhouse Gas Inventories. EUR 24573 EN. Luxembourg, Office for Official Publications of the European Communities. 109 pp.

Castro de Esparza, M.L. 2006. *The Presence of Arsenic in Drinking Water in Latin America and Its Effect on Public Health*. International Congress.

Cavelier, J., Aide, T.M., Dupuy, J.M., Eusse, A.M. & Santos, C. 1999. Long-term effects of deforestation on soil properties and vegetation in a tropical lowland forest in Colombia. *Ecotropicos*, 12: 57-68.

Cerdan, O., Govers, G., Le Bissonnais, Y., Van Oost, K., Poesen, J., Saby, N., Gobin, A., Vacca, A., Quinton, J., Auerswald, K., Klik, A., Kwaad, F.J.P.M., Raclot, D., Ionita, I., Rejman, J., Rousseva, S., Muxart, T., Roxo, M.J. & Dostal, T. 2010. Rates and spatial variations of soil erosion in Europe: A study based on erosion plot data. *Geomorphology*, 122(1-2): 167-177.

Chadwick, D., Sommer, S., Thorman, R., Fangueiro, D., Cardenas, L., Amon, B. & Misselbrook, T. 2011. Manure management: implications for greenhouse gas emissions. *Anim. Feed. Sci. Technol.*, 166-67: 514-531.

Chadwick, O.A., Derry, L.A., Vitousek, P.M., Huebert, B.J. & Hedindon, L.O. 1999. Changing sources of nutrients during four million years of ecosystem development. *Nature*, 397: 491–497.

Chamen, W.C.T. 2006. Controlled traffic farming on a field scale in the UK. In R. Horn, H. Fleige, S. Peth & X.H. Peng, eds. *Soil Management for Sustainability*. Advances in Geoecology 38, pp. 251–260. Germany, Reiskirchen, GeoScience Publisher. 502 pp.

Chappell, A., Webb, N.P., Butler, H.J., Strong, C.L., McTainsh, G.H., Leys, J.F. & Rossel, R.A.V. 2013. Soil organic carbon dust emission: an omitted global source of atmospheric CO_2. *Global Change Biology*, 19: 3238-3244.

Chen, J. 2007. Rapid urbanization in China: a real challenge to soil protection and food security. *Catena*, 69: 1-15.

Chepil, W.S. 1956. Influence of moisture on erodibility of soil by wind. *Proceedings of the Soil Science Society of America*, 20: 288–292.

Cobo, J.G., Dercon, G. & Cadisch, G. 2010. Nutrient balances in African land use systems across different spatial scales: A review of approaches, challenges and progress. *Agriculture, Ecosystems & Environment*, 136: 1–15.

Cochet, H. 1996. Gestion paysanne de la biomasse et développement durable au Burundi. *Cahiers des Sciences Humaines*, 32: 133–151.

Conant, R.T., Ryan, M.G., Ågren, G.I., Birge, H.E., Davidson, A.S., Eliasson, P.E., Eveans, S.E., Frey, S.D., Giardina, C.P., Hopkins, F.M., Hyvönen, R., Kirchbaum, M.U.F., Lavallee, J.M., Leifeld, J., Parton, W.J., Steinweg, J.M., Wallenstain, M.D., Wetterstedt, J.A.M. & Bradford, M.A. 2011. Temperature and soil organic matter decomposition rates—synthesis of current knowledge and a way forward. *Global Change Biology*, 17: 3392-3404.

Cordell, D., Drangert, J.-O. & White, S. 2009. The story of phosphorus: global food security and food for thought. *Global Environmental Change*, 19: 292–305.

Couwenberg, J., Dommain, R. & Joosten, H. 2010. Greenhouse gas fluxes from tropical peatlands in south-east Asia. *Global Change Biology*, 16(6): 1715–1732.

Cronan, C.S. & Grigal, D.F. 1995. Use of calcium/aluminum ratios as indicators of stress in forest ecosystems. *Journal of Environmental Quality*, 24: 209–226.

Crossley, D.A., Meuller, B.R. & Perdue, J.C. 1992. Biodiversity of microarthropods ina agricultural soils: relation to processes. *Agriculture, Ecosystems and Environment*, 40: 37-46.

Crow, W., Miralles, D.& Cosh, M. 2010. A quasi-global evaluation system for satellite-based surface soil moisture retrievals. IEEE T. *Geosci. Remote Sens.*, 48: 2516-2527.

Crow, W.T., Berg, A.A., Cosh, M.H., Loew, A., Mohanty, B.P., Panciera, R., de Rosnay, P., Ryu, D. & Walker, J.P. 2012. Up-scaling sparse ground-based soil moisture observations for the validation of coarse-resolution satellite soil moisture products. *Review of Geophysics*, 50(2).

Crowder, B.M. 1987. Economic costs of reservoir sedimentation: A regional approach to estimating cropland erosion damage. *Journal of soil and water conservation*, 42(3): 194-197.

Csavina, J., Field, J., Félix, O., Corral-Avitia, A.Y., Sáez, A.E. & Betterton, E.A. 2014. Effect of wind speed and relative humidity on atmospheric dust concentrations in semi-arid climates. *Science of the Total Environment*, 487: 82–90.

Darwin, C.R. 1845. *Journal of Researches into the Natural History and Geology of the Countries Visited during the Voyage of H.M.S. Beagle Round the World, under the Command of Capt. Fitz Roy*, R.N. 2nd edition. London.

Darwin, C.R. 1846. An account of the fine dust which often falls on vessels in the Atlantic Ocean. *Quarterly Journal of the Geological Society*, 2(1–2): 26–30.

Davidson, E.A. & Janssens, I.A. 2006. Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. Nature, 440: 165-173.

De Deyn, G.B. & van der Putten, W.H. 2005. Linking aboveground and belowground diversity. *Trends in Ecology & Evolution*, 20: 625-633.

de Lacerda, L.D. 2003. Updating global Hg emissions from small-scale gold mining and assessing its environmental impact. *Environmental Geology*, 43: 308-314.

De Noni, G., Prat, C., Quantin, P., Viennot, M. & Zebrowski, C. 2000. Erosion et conservation, après récupération, des sols volcaniques indurés de l'Equateur et du Mexique. *Etude et Gestion des Sols*, 7: 25-36.

Decaens, T. & Jimenez, J.J. 2002. Earthworm communities under an agricultural intensification gradient in Columbia. *Plant Soil*, 240: 133-143.

DeJong-Hughes, J.M., Swan J.B., Moncrief J.F. & Voorhees, W.B. 2001. *Soil Compaction: Causes, Effects and Control* (Revision). University of Minnesota Extension Service BU-3115-E.

Dent, D. 1986. *Acid sulfate soils: a baseline for research and development*. ILRI Publication No 39. The Netherlands, Wageningen, International Institute of Land Reclamation and Improvement.

Department of Environment and Conservation, Australia (DECA). 2010. *Assessment Levels for Soil, Sediment and Water*. Australia.

Department of the Environment, Australia (DEA). 2001. *Land Theme Report*. *Australia State of the Environment Report* 2001. Australia, CSIRO.

Dercon, G., Deckers, J., Poesen, J., Govers, G., Sanchez, H., Ramirez, M., Vanegas, R., Tacuri, E. & Loaiza, G. 2006. Spatial variability in crop response under contour hedgerow systems in the Andes region of Ecuador. *Soil & Tillage Research*, 86(1): 15-26.

Diaz, R.J. & Rosenberg, R. 2008. Spreading dead zones and consequences for marine ecosystems. *Science* (New York, NY), 321: 926–929.

Diogo, R.V.C., Buerkert, A. & Schlecht, E. 2009. Horizontal nutrient fluxes and food safety in urban and peri-urban vegetable and millet cultivation of Niamey, Niger. *Nutrient Cycling in Agroecosystems*, 87: 81–102.

Djavanshir, K., Dasmalchi, H. & Emararty, A. 1996. Ecological and ecophysiological survey on sexual Euphrate poplar and Athel trees in Iranian deserts. *Journal of Biaban* (Iran), 1: 67–81.

Dobermann, A., Cruz, P.C.S. & Cassman, K.G. 1996. Fertilizer inputs, nutrient balance, and soil nutrient-supplying power in intensive, irrigated rice systems. I. Potassium uptake and K balance. *Nutrient Cycling in Agroecosystems*, 46: 1–10.

Doetterl, S., Van Oost, K. & Six, J. 2012. Towards constraining the magnitude of global agricultural sediment and soil organic carbon fluxes. *Earth Surface Processes and Landforms*, 37(6): 642-655.

Dorigo, W.A., de Jeu, R.A.M., Chung, D., Parinussa, R. M., Liu, Y. Y., Wagner, W. & Fernandez-Prieto, D. 2012. Evaluating global trends (1988-2010) in harmonized multi-satellite surface soil moisture. *Geophys. Res. Lett*., 39(L18405): 1-7.

Dorigo, W.A., Wagner, W., Hohensinn, R., Hahn, S., Paulik, C., Drusch, M., Mecklenburg, S., van Oevelen, P., Robock, A. & Jackson, T.J. 2011. The International Soil Moisture Network: A data hosting facility for global in situ soil moisture measurements. Hydrol. *Earth Syst. Sci. Discuss*., 8: 1609–1663.

Douglas, J.T., Campbell, D.J. & Crawford, C.E. 1998. Soil and crop responses to conventional, reduced ground pressure and zero traffic systems for grass silage production. *Soil & Tillage Research*, 24: 421-439.

Drechsel, P., Gyiele, L., Kunze, D. & Cofie, O. 2001. Population density, soil nutrient depletion, and economic growth in sub-Saharan Africa. *Ecological Economics*, 38: 251–258.

Drechsel, Pay, Giordano, Mark, Gyiele, & Lucy. 2004. *Valuing nutrients in soil and water: Concepts and techniques with examples from IWMI studies in the developing world*. Research report 82. Sri Lanka, Colombo, IWMI.

Drees, L.R., Manu, A. & Wilding, L.P. 1993. Characteristics of Aeolian dusts in Niger, West Africa. *Geoderma*, 59: 213-233.

Drewry, J.J., Cameron, K.C. & Buchan, G.D. 2008. Pasture yield and soil physical property responses to soil compaction from treading and grazing — a review. *Soil Research*, 46(3): 237–256

Driscoll, C.T., Driscoll, K.M., Mitchell, M.J. & Raynal, D.J. 2003. Effects of acidic deposition on forest and aquatic ecosystems in New York State. *Environmental Pollution*, 123: 327-336.

Edmeades, D.C. & Ridley, A.M. 2003. Using lime to ameliorate topsoil and subsoil acidity. *In Z. Rengel, ed., Handbook of soil acidity.* Switzerland, Basel, Eastern Hemisphere Distribution. ISBN 0-8247-0890-3.

Eggleton, P., Bignell, D.E., Hauser, S., Dibog, L., Norgrove, L. & Madong, B. 2002. Termite diversity across an anthropogenic disturbance gradient in the humid forest zone of West Africa. *Agriculture, Ecosystems & Environment*, 90: 189-202.

Eglin, T., Ciais, P., Piao, S.l., Barre, P., Bellassen, V., Claude, P., Chen, C., Grasser, T., Koven, C., Reichstein, M. & Smith, P. 2010. Historical and future perspectives of global soil carbon response to climate and land-use changes. *Tellus B*, 62: 700–718.

Ellis, E.C. & Ramankutty, N. 2008. Putting people in the map: anthropogenic biomes of the world. *Frontiers in Ecology and the Environment*, 6: 439–447.

Entekhabi, D., Njoku, E.G., O'Neill, P.E., Kellogg, K.H., Crow, W.T., Edelstein, W.N., Entin, J.K., Goodman, S.D., Jackson, T.J., Johnson, J., Kimball, J., Piepmeier, J.R., Koster, R.D., Martin N., McDonald, K.C., Moghaddam, M., Moran, S., Reichle, R., Shi, J.C., Spencer, M.W., Thurman, S.W., Tsang, L. & Van Zyl, J. 2010. The Soil Moisture Active Passive (SMAP) Mission. *Proceedings of the IEEE*, 98(5): 704-716.

Environmental Affairs (EA). 2010. *Framework for the Management of Contaminated Land*. Republic of South Africa.

Erbach, D.C., Benjamin, J.G., Cruse, R.M., Elamin, M.A., Mukhtur, S. & Choi, C.H. 1992. *Soil and corn response to tillage with paraplow. Transactions of the ASAE*, 35: 1347–1354.

ESA. 2011. GLOBCOVER 2009. *Products Description and Validation Report. Belgium*, Université catholique de Louvain. 53 pp.

Eswaran, H., Almaraz, R., van den Berg, E. & Reich, P. 1996. *An Assessment of the Soil Resources of Africa in Relation to Productivity.* World Soil Resources, Soil Survey Division. Washington, DC, USDA Natural Resources Conservation Service.

Eswaran, H., Van Den Berg, E. & Reich, P. 1993. Organic carbon in soils of the world. *Soil Science Society of America Journal*, 57: 192-194.

EU/JRC. (In press). Global Soil Biodiversity Atlas.

European Commission (EC). 2013. *Science for Environmental Policy In-Depth Report. Soil Contamination. Impacts on Human Health*. European Commission.

European Commission. 2004. *European Soil Database – Distribution Version* 2.0. Italy, Ispra, Institute for Environment and Sustainability, European Commission Joint Research Centre.

European Environment Agency (EEA) 1. 2007. *Progress in management of contaminated sites* (CSI 015). European Environment Agency.

European Environment Agency (EEA). 2010. The European Environment State and Outlook 2010: *Freshwater Quality.* Denmark, Copenhagen.

European Environment Agency (EEA). 2012. *The State of Soil in Europe*. European Environment Agency.

European Environment Agency (EEA). 2014. *Progress in Management of Contaminated Sites. European Environment Agency.*

European Fertilizer Manufacturers Association. 2009. *Forecasts of food, farming and fertilizer use in the European Union* 2008-2018. Brussels.

Falkowski, P., Scholes, R.J., Boyle, E., Canadell, J., Canfield, D., Elser, J., Gruber, N., Hibbard, K., Högberg, P., Linder, S., MacKenzie, F.T., Moore, B., Pedersen, T., Rosenthal, Y., Seitzinger, S., Smetacek, V. & Steffen, W. 2000. The Global Carbon Cycle: A Test of Our Knowledge of Earth as a System. *Science*, 290(5490): 291–296.

Famiglietti, J.S & Rodell, M. 2013. Water in the Balance. *Science*, 340(6138): 1300-1301

Fanning, D., Rabenhorst, M., Coppock, C., Daniels, W. & Orndorff, Z. 2004. Upland active acid sulphate soils from construction of new Stafford County, Virginia, US, airport. *Aust. J. Soil Res.*, 42: 527–536.

FAO. 2011. The state of the world's land and water resources for food and agriculture (SOLAW) - Managing systems at risk. Rome, Food and Agriculture Organization of the United Nations & London, Earthscan.

FAO/IIASA/ISRIC/ISS-CAS/JRC. 2009. Harmonized World Soil Database (version 1.1). Rome, FAO & Austria, Laxenburg, IIASA.

FAO/ISRIC/ISSS. 1998. *World Reference Base for Soil Resources*. World Soil Resources Report, No.84. Rome, FAO.

FAO-UNESCO. 1980. *Soil Map of the World. Paris*, UNESCO.

Farina, M.P.W., Channon, P. & Thibaud, G.R. 2000. A comparison of strategies for ameliorating subsoil acidity II. Long-term soil effects. *Soil science society of America journal*, 64: 652-658.

Field, J.P., Belnap, J., Breshears, D.D., Neff, J.C., Okin, G.S., Whicker, J.J., Painter, T.H., Ravi, S., Reheis, M.C. & Reynolds, R.R. 2010. The ecology of dust. *Frontiers in Ecology and the Environment*, 8(8): 423–430.

Flowers, M. & Lal, R. 1998. Axle load and tillage effects on soil physical properties and soybean grain yield on a mollic ochraqualf in northwest Ohio. *Soil Till. Res.*, 48: 21–35.

Foley, J.A., De Fries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E.A., Kucharik, C.J.,Monfreda, C., Patz, J.A., Prentice, I.C., Ramankutty, N. & Syder, P.K. 2005. Global consequences of land use. *Science*, 309: 570–574.

Freebairn, D. & Wockner, G. 1986. A study of soil erosion on vertisols of the eastern Darling Downs, Queensland. I. Effects of surface conditions on soil movement within contour bay catchments. *Australian Journal of Soil Research*, 24(2): 135-158.

Freedman, B. 2014. Soil organic matter dynamics, climate change effects. Handbook of Global *Environmental Pollution*, 1: 317-323.

Funk, C., Dettinger, M. D., Michaelsen, J., Verdin, J. P., Brown, M. E., Barlow, M. & Hoell, A. 2008. Warming of the Indian Ocean threatens eastern and southern African food security but could be mitigated by agricultural development. *PNAS*, 105(32): 11081–11086.

Galloway, J.N., Dentener, F.J., Capone, D.G., Boyer, E.W., Howarth, R.W., Seitzinger, S.P., Asner, G.P., Cleveland, C.C., Green, P.A., Holland, E.A., Karl, D.M., Michaels, A.F., Porter, J.H., Townsend, A.R. & Vosmarty, C.J. 2004. Nitrogen Cycles: Past, Present, and Future. *Biogeochemistry*, 70: 153–226

Gardi, C., Jeffery, S. & Saltelli, A. 2013. An estimate of potential threats levels to soil biodiversity in EU. *Global Change Biology*, 19:1538-1548.

Gardi, C., Panagos, P., van Liedekerke, M., Bosco, C. & de Brogniez, D. 2014. Land take and food security: Assessment of land take on the agricultural production in Europe. J. *Environ. Plan. Manag.*, 8(5): 898-912.

Gardner, T., Acosta-Martinez, V., Calderón, F.J., Zobeck, T.M., Baddock, M., Van Pelt, R.S., Senwo, Z., Dowd, S. & Cox, S. 2012. Pyrosequencing reveals bacteria carried in different wind-eroded sediments. *Journal of Environmental Quality*, 41(3):744–753.

Gebremichael, D., Nyssen, J., Poesen, J., Deckers, J., Haile, M., Govers, G. & Moeyersons, J. 2005. Effectiveness of stone bunds in controlling soil erosion on cropland in the Tigray Highlands, northern Ethiopia. Soil Use Manage., 21(3): 287-297.

Ghassemi, F., Jakeman, A.J. & Nix, H.A. 1995. *Salinization of Land and Water Resources: Human Causes, Extent, Management and Case Studies*. Wallingford, CABI Publishing. 517 pp.

Giller, K.E., Rowe, E.C., de Ridder, N. & van Keulen, H. 2006. Resource use dynamics and interactions in the tropics: Scaling up in space and time. *Agricultural Systems*, 88: 8–27.

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: RG3005.

Gorbushina, A.A., Kort, R., Schulte, A., Lazarus, D., Schneteger, B., Brumsack, H.-J. Broughton, W.J. & Favet. J. 2007. Life in Darwin's dust: Intercontinental transport and survival of microbes in the nineteenth century. *Environmental Microbiology*, 9: 2911–2922.

Govers, G., Merckx, R., Van Oost, K. & Van Wesemael, B. 2013. *Managing Soil Organic Carbon for Global Benefits*. STAP Technical Report. Washington, DC, STAP. 70 pp.

Gray, L.E. & Pope, R.A. 1986. Influence of soil compaction on soybean stand, yield, and Phytophthora root rot incidence. *Agron*. J., 78: 189-191.

Gregory, M.M., Shea, K.L. & Bakko, E.B. 2005. Comparing agroecosystems: Effects of cropping and tillage patterns on soil, water, energy use and productivity. *Renewable Agric. Food Syst*., 20: 81–90.

Grunwald, S. 1999. *Soil Organic Matter* (SOM). University of Florida. (Also available at *http://soils.ifas.ufl.edu/faculty/grunwald/teaching/eSoilScience/organic.shtml*)

Guo, J.H., Liu, X.J., Zhang, Y., Shen, J.L., Han, W.X., Zhang, W.F., Christie, P., Goulding, K.W.T., Vitousek, P.M. & Zhang, F.S. 2010. Significant acidification in major Chinese croplands. *Science*, 327: 1008-1010.

Gzik, A., Kuehling, M., Schneider, I. & Tschochner, B. 2003. Heavy Metal Contamination of Soils in a Mining Area in South Africa and its Impact on Some Biotic Systems. J. *Soils & Sediments*, 3(1): 29-34.

Haddeland, I., Clark, D.B., Franssen, W., Ludwig, F., Voß, F., Arnell, N.W., Bertrand, N., Best, M., Folwell, S., Gerten, D., Gomes, S., Gosling, S.N., Hagemann, S., Hanasaki, N., Harding, R., Heinke, J., Kabat, P., Koirala, S., Oki, T., Polcher, J., Stacke, T., Viterbo, P., Weedon, G.P. & Yeh, P. 2011. Multimodel Estimate of the Global Terrestrial Water Balance: *Setup and First Results. Journal of Hydrometeorology*, 12(5): 869-884.

Hagen, L.J., Van Pelt, R.S., Zobeck, T.M. & Retta, A. 2007. Dust deposition near an eroding source field. *Earth Surface Processes & Landforms*, 32: 281–289.

Haileslassie, A., Priess, J.A., Veldkamp, E. & Lesschen, J.P. 2007. Nutrient flows and balances at the field and farm scale: Exploring effects of land-use strategies and access to resources. *Agricultural Systems*, 94: 459–470.

Håkansson, I. & Lipiec, J. 2000. A review of the usefulness of relative bulk density values in studies of soil structure and compaction. *Soil Till. Res*., 53: 71–85.

Hakkenberg, R., Churkina, G., Rodeghiero, M., Börner, A., Steinhof, A. & Cescatti, A. 2008. Temperature sensitivity of the turnover times of soil organic matter in forests. *Ecological Applications*, 18: 119-131.

Hamza, M. & Anderson, W. 2003. Responses of soil properties and grain yields to deep ripping and gypsum application in a compacted loamy sand soil contrasted with a sandy clay loam soil in Western Australia. *Aust. J. Agr. Res*., 54: 273–282.

Hamza, M.A. & Anderson, W.K. 2005. Soil compaction in cropping systems. A review of the nature, causes and possible solutions. *Soil* Till.Res., 82(2): 121–145.

Hansen, S., Maehlum, J.E. & Bakken, L.R. 1993. N_2O and CH_4 fluxes in soil influenced by fertilization and tractor traffic. *Soil Biology and Biogeochemistry*, 25(5): 621-630

Hartemink, A.E. 1998. Acidification and pH buffering capacity of alluvial soils under sugarcane. *Experimental Agriculture*, 34: 231-243

Harter, R.D. 2007. *Acid soils of the tropics*. ECHO Technical Note. 11 pp.

Haynes, R.J. & Swift, R.S. 1986. Effects of soil acidification and subsequent leaching on levels of extractable nutrients in a soil. *Plant and soil*, 95: 327-336.

Heimsath, A.M., DiBiase, R.A. & Whipple, K.X. 2012. Soil production limits and the transition to bedrock-dominated landscapes. *Nature Geoscience*, 5(3): 210-214.

Helyar, K.R. & Porter, W.M. 1989. Soil acidification, its measurement and the processes involved. In A. Robson, ed. *Soil acidity and plant growth*. Academic Press Australia.

Heuscher, S.A., Brandt, C.C. & Jardine, P.M. 2005. Using soil physical and chemical properties to estimate bulk density. *Soil Sci. Soc. Am. J.*, 69: 51–56.

Hicks, W.K., Kuylenstierna, J.C.I., Owen, A., Dentener, F., Seip, H.-M. & Rodhe, H. 2008. Soil sensitivity to acidification in Asia: status and prospects. *Ambio*, 37: 295-303.

Hiederer, R. & Köchy, M. 2011. *Global Soil Organic Carbon Estimates and the Harmonized World Soil Database*. EUR 25225 EN. Publications Office of the European Union.79 pp.

Hiederer, R., Ramos, F., Capitani, C., Koeble, R., Blujdea, V., Gomez, O., Mulligan, D. & Marelli, L. 2010. *Biofuels: a New Methodology to Estimate GHG Emissions from Global Land Use Change - A methodology involving spatial allocation of agricultural land demand and estimation of CO_2 and N_2O emissions*. EUR 24483 EN. Joint Research Centre. Luxembourg, Publications Office of the European Union. 150 pp.

Hiernaux, P., Fernandez-Rivera, S., Schlecht, E., Turner, M.D. & Williams, T.O. 1998. Livestock-mediated nutrient transfers in Sahelian agro-ecosystems. In G. Renard, A. Neef, K. Becker, M. & von Oppen, eds., Soil fertility management in West African land use systems, pp. 339–347. Germany, Weikersheim, Margraf Verlag.

Hillel, D. & Rosenzweig, C. 2009. Soil carbon and climate change. CSA *News*, 54(6): 1-8.

Hinsinger, P., Bengough, A.G., Vetterlein, D. & Young, I.M. 2009. Rhizosphere: biophysics, biogeochemistry and ecological relevance. *Plant Soil*, 321: 117-152.

Hoeksema, J.D., Lussenhop, J. & Teeri, J.A. 2000. Soil nematodes indicate food web responses to elevated atmospheric CO_2. *Pedobiologia* (Jena), 44(6): 725-735.

Horn, R., Fleige, H., Richter, F.-H., Czyz, E.A., Dexter, A., Diaz-Pereira, Dumitru, E., Enarche, R., Mayol, F., Rajkai, K., De la Rosa, D. & Simota, C. 2005. SIDASS project Part 5: Prediction of mechanical strength of arable soils and its effects on physical properties at various map scales. *Soil* Till.Res., 82: 47-56.

Houghton, R.A. 1995. Changes in the storage of industrial carbon since 1850. In R. Lal, J.M. Kimble, E. Levine & B.A. Stewart, eds. *Soils and global change*, pp.45-65. Boca Raton, FL, CRC/Lewis.

Howarth, R., Billen, G., Swaney, D., Townsend, A., Jaworski, N., Lajtha, K., Downing, J.A., Elmgren, R., Caraco, N., Jordan, T., Berendse, F., Freney, J., Kudeyarov, V., Murdoch, P. & Zhao-Liang, Z. 1996. Regional nitrogen budgets and riverine N & P fluxes for the drainages to the North Atlantic Ocean: Natural and human influences. *Biogeochemistry*, 35: 75–139.

Howitt, R.E., Medellin-Azuara, J., MacEwan, D., Lund, J. R. & Sumner, D.A. 2014. *Economic Analysis of the 2014 Drought for California Agriculture*. USA, Davis, CA, University of California, Center for Watershed Sciences, 20 pp.

Hue, N.V. & Licudine, D.L. 1999. Amelioration of subsoil acidity through surface application of organic manures. *Journal of environmental quality*, 28: 623-632.

Hugh, B. & Ravenscroft, P. 2009. Arsenic in groundwater: a threat to sustainable agriculture in South and South-east Asia. *Environment International*, 35(3): 647-654.

Husnain, H., Masunaga, T. & Wakatsuki, T. 2010. Field assessment of nutrient balance under intensive rice-farming systems, and its effects on the sustainability of rice production in Java Island, Indonesia. *Journal of Food and Environmental Sciences*, 4: 1–11.

Imhoff, S., Da Saliva, A.P. & Fallow, D. 2004. Susceptibility to compaction, load support capacity and soil compressibility of Hapludox. *Soil Sci. Soc. Am. J.*, 68: 17-24.

Intergovenmiental Panel on Climate Change (IPCC). 2000. Special Report on Emissions Scenarios. Cambridge University Press, UK.

Intergovernmental Panel on Climate Change (IPCC). 1995. *Climate Change* 1995. Working Group 1. IPPC, Cambridge, Cambridge University Press.

Jansson, C., Wullschleger, S.D., Kalluri, U.C. & Tuskan, G.A. 2010. Phytosequestration: Carbon Biosequestration by Plants and the Prospects of Genetic Engineering. *BioScience*, 60(9): 685-696.

Jenny, H. 1930. *A study on the influence of climate upon nitrogen and organic matter content of soil*. Misouri Agr. Exp. Sta. Res. Bulletin No. 52. Columbia, Mo, University of Missouri.

Jjemba, P.K. 2002. The potential impact of veterinary and human therapeutic agents in manure and biosolids on plants grown on arable land: a review. *Agric. Ecosyst. Environ*, 93(1-3): 267–278

Jobbágy, E.G. & Jackson, R.B. 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecological Applications*, 10: 423-436.

Johnson, D.W., Turner, J. & Kelly, J.M. 1982. The effects of acid rain on forest nutrient status. *Water resources research*, 18: 449-461.

Johnson, L.C. 1987. Soil loss tolerance: Fact or myth. *Journal of Soil and Water Conservation*, 42(3): 155-160.

Jones, M.C. & Graves, B. 2010. *State experiences with emerging contaminants – recommendations for Federal Action*. Environmental council of the States (ECOS) Green Report. 42 pp.

Jouve, P. & Oussible, M. 1979. Conséquence du t,assement du sol sur l'environement et la production de plantation de canne à sucre dans le Gharb: Hommes, Terres et Eaux. Bull. de l'Association Nationale pour l'Aménagement Foncier, l'Irrigation et le Drainage. Rabat, Morocco. 9(32): 69-83.

Kaup, E. & Burgess, J.S. 2002. Surface and subsurface flows of nutrients in natural and human impacted lake catchments on Broknes, Larsemann Hills, Antarctica. *Antarctic Science*, 14: 343–352.

Kendy, E., Molden, D. J., Steenhuis, T. S. & Liu, C. 2003. *Policies drain the North China Plain: agricultural policy and ground water depletion in Luancheng County*, 1949–2000. IWMI Research Paper 71. Sri Lanka, Colombo, IWMI.

Kern, J.S. 1994. Spatial Patterns of Soil Organic Carbon in the Contiguous United States. *Soil Sci. Soc. Am. J.*, 58: 439–455.

Kerr, Y.H., Waldteufel, P., Wigneron, J.P., Martinuzzi, J.M., Font, J. & Berger, M. 2001. Soil moisture retrieval from space: the Soil Moisture and Ocean Salinity (SMOS) mission. IEEE Trans. *Geosci. Remote Senss.*, 39: 1729–1735.

Khai, N.M., Ha, P.Q. & Öborn, I. 2007. Nutrient flows in small-scale peri-urban vegetable farming systems in Southeast Asia—A case study in Hanoi. *Agriculture, Ecosystems & Environment*, 122: 192–202.

Kho, R.M., Yacouba, B., Yayé, M., Katkoré, B., Moussa, A., Iktam, A. & Mayaki, A. 2001. Separating the effects of trees on crops: the case of Faidherbia albida and millet in Niger. *Agroforestry Systems*, 52: 219–238.

Kibblewhite, M.G., Ritz, K. & Swift, M.J. 2008. Soil health in agricultural systems. *Philos. Trans. R. Soc. London B Biol. Sci.*, 363(1492): 685-701.

Kirschbaum, M.U.F. 1995. The temperature dependence of soil organic matter decomposition, and the effect of global warming on soil organic storage. *Soil Biology and Biochemistry*, 27: 753-760.

Kirschbaum, M.U.F. 2000. Will changes in soil organic carbon act as a positive or negative feedback on global warming? *Biogeochemistry*, 48: 21-51.

Kirschbaum, M.U.F. 2006. The temperature dependence of organic-matter decomposition – still a topic of debate. *Soil Biology and Biochemistry*, 38: 2510-2518.

Koocheki, A. 2000. The potential of saltbush (Atriplex spp.) as a fodder shrub for the arid lands of Iran. In G. Gintzburger, M. Bounejmate & A. Nefzaoui, eds. *Fodder Shrub Development in Arid and Semiarid Zones*. Vol. 1, pp. 178-183. Syria, Aleppo, ICARDA.

Kopáček, J., Hardekopf, D., Majer, V., Šenáková, P., Tuchlík, E. & Veselý, J. 2004. Response of alpine lakes and soils to changes in acid deposition: the MAGIC model applied to the Tatra Mountain region, Slovakia-Poland. *Journal of Limnology*, 63(1): 143-156.

Kottek, M., Grieser, J., Beck, C., Rudolf, B. & Rubel, F. 2006. World map of the Köppen-Geiger climate classification updated. *Meteorol.* Z., 15: 259-263.

Krmenec, A.J. 2000. *Vehicle traffic and soil compaction*. Poster at Midwest Farm Progress Show, IL.

Kroulik, M., Kumhala, F., Hula, J. & Honzik, I. 2009. The evaluation of agricultural machines field trafficking intensity for different soil tillage technologies. *Soil* Till.Res., 105: 171-175.

Lal, R. 1987. Management the soils of sub Saharan Africa. *Science*, 236: 1069-1076.

Lal, R. 1999. Soil Management and restoration for C sequestration to mitigate the accelerated greenhouse effect. *Progress in Environmental Science*, 1(4): 307-326.

Lal, R. 2004. Soil carbon sequestration impacts on global change and food security. Science, 304: 1623–1627.

Lal, R., Griffin, M., Apt, J., Lave, L., & Morgan, M.G. 2004. Managing soil carbon. Science, 304(5669): 393.

Larney, F.J., Leys, J.F., Muller, J.F. & McTainsh, G.H. 1999. Dust and endosulfan deposition in cotton-growing area of northern New South Wales, Australia. *Journal of Environmental Quality*, 28: 692–701.

Ledwith, T. 2011. *Crisis in the Horn of Africa: Rethinking the humanitarian response*. October 5, 2011. (Also available at *http://www2.unicef.org:60090/infobycountry/kenya_59997.html*).

Lenton, T.M. & Huntingford, C. 2003. Global terrestrial carbon storage and uncertainties in its temperature sensitivity examined with a simple model. *Global Change and Biology*, 9(10): 1333-1352.

Li, J., Okin, G.S., Alvarez, L. & Epstein, H. 2007. Quantitative effects of vegetation cover on wind erosion and soil nutrient loss in a desert grassland of southern New Mexico, USA. *Biogeochemistry*, 85: 317–332.

Li, P., Feng, X.B., Qiu, G.L., Shang, L.H., Li, Z.G. 2009. Mercury pollution in Asia: a review of the contaminated sites. *Journal of Hazardous Materials*, 168: 591-601.

Liiri, M., Setälä, H., Haimi, J., Pennanen, T. & Fritze, H. 2002. Soil Processes are not influenced by the functional complexity of soil decomposer food webs under disturbance. *Soil Biology and Biochemistry*, 34: 1009-1020.

Lin, C. & Melville, M.D. 1994. Acid sulphate soil-landscape relationships in the Pearl River Delta, southern China. *Catena*, 22: 105-120.

Lin, X., Yin, C. & Xu., D. 1996. Input and output of soil nutrients in high-yield paddy fields in South China. *In Proceedings of the international symposium on maximizing Rice Yields through Improved Soil and Environmental Management*, pp. 93–97. Thailand, Khon Kaen.

Liski, J., Ilvesniemi, H., Mäkelä, A. & Westman, C.J. 1999. CO_2 emissions from soil in response to climate warming are overestimated: the decomposition of old soil organic matter is tolerant of temperature. *Ambio*, 28: 171-174.

Liu, J. & Hue, N.V. 2001. Amending subsoil acidity by surface applications of gypsum, lime, and composts. *Communications in soil science and plant analysis*, 32: 2117-2132.

Lockwood, P., Wilson, B., Daniel, H. & Jones, M. 2003. *Soil acidification and natural resource management: directions for the future*. University of New England, Armidale.

Lofts, S., Chapman, P.M., Dwyer, R.,; Mclaughlin, M.J., Schoeters, I., Sheppard, S.C., Adams, W.J., Alloway, B.J., Antunes, P.M.C., Campbell, P.G.C., Davies, B., Degryse, F., De Vries, W., Farley, K.J., Garrett, R.G., Green, A., Jan, G.B., Hale, B., Harrass, M., Hendershot, W.H., Keller, A., Lanno, R., Tao, L., Liu, W.-X., Yibing, M., Menzie, C., Moolenaar, S.W., Piatkiewicz, W., Reimann, C., Rieuwerts, J.S., Santore, R.C., Sauve, S. Schuetze, G. Schlekat, C., Skeaff, J., Smolders, E., Shu, T., Wilkins, J. & Zhao, F.-J. 2007. Critical loads of metals and other trace elements to terrestrial environments. *Environmental Science and Technology*, 41(18): 6326-6331.

Lowdermilk, W.C. 1942. *Conquest of the land through seven thousand years*. Agric. Bull. Vol. 99. Washington, DC, USA Dept. of Agric, Soil Conservation Service.

Luo, Y., Wu, L., Liu, L., Han, C. & Li, Z. 2009. *Heavy Metal Contamination and Remediation in Asian Agricultural Land*. National Institute of Agro-Environmental Sciences, MARCO Symposium, Japan, October 5-7.

MacDonald, G.K., Bennett, E.M., Potter, P.A. & Ramankutty, N. 2011. Agronomic phosphorus imbalances across the world's croplands. *Proceedings of the National Academy of Sciences of the United States of America*, 108: 3086–3091.

Mackenzie, F.T., Ver, L.M. & Lerman, A. 2008. Coupled biogeochemical cycles of carbon, nitrogen, phosphorus and sulphur in the land-ocean atmosphere system. *In J.N. Galloway & J.M. Melillo, eds. Asian Change in the Context of Global Climate Change*, pp. 42–100. New York, Cambridge Univ. Press.

Mackey, B., Colin Prentice, I. C., Steffen, W., House, J.I., Lindenmayer, D., Keith, H. & Berry, S. 2013. Untangling the confusion around land carbon science and climate change mitigation policy. *Nature Climate Change*, 3: 552–557.

Mahowald, N., Jickells, T.D., Baker, A.R., Artaxo, P., Benitez-Nelson, C.R., Bergametti, G., Bond, T.C., Chen, Y., Cohen, D.D., Herut, B., Kubilay, N., Losno, R., Luo, C., Maenhaut, W., McGee, K.A., Okin, G.S., Siefert, R.L. & Tsukuda, S. 2008. Global distribution of atmospheric phosphorus sources, concentrations and deposition rates, and anthropogenic impacts. *Global Biogeochemical Cycles*, 22: GB4026.

Marchand, C., Lallier-Vergès, E., Baltzer, F., Albéric, P., Cossa, D. & Baillif, P. 2006. Heavy metals distribution in mangrove sediments along the mobile coastline of French Guiana. *Marine Chemistry*, 98: 1-17.

Marschner, B. & Noble, A.D. 2000. Chemical and biological processes leading to the neutralisation of acidity in soil incubated with litter materials. *Soil biology and biochemistry*, 32: 805-813.

Materechera, S.A. 2008 Tillage and tractor traffic effects on soil compaction in horticultural fields used for peri-urban agriculture in a semi-arid environment of the North West Province, South Africa. *Soil* Till.Res., 103: 11–15.

Mattern, S., Fasbender, D. & Vanclooster, M. 2009. Discriminating sources of nitrate pollution in an unconfined sandy aquifer. *Journal of Hydrology*, 376: 275–284.

McGarry, D. & Sharp, G. 2003. A rapid, immediate, farmer-usable method of assessing soil structure condition to support conservation agriculture. In L. García-Torres, J. Benites, A. Martínez-Vilela & A. Holgado-Cabrera, eds. *Conservation agriculture: environment, farmers experiences, innovations, socio-economy, policy*. 375 pp.

McNally, A., Funk, C., Husak, G., Michaelsen, J., Cappelaere, B., Demarty, J., Pellarin, T., Young, T.P., Caylor, K.K., Riginos, C. & Veblen, K.E. 2013. Estimating Sahelian and East African soil moisture using the Normalized Difference Vegetation Index. *Hydrology and Earth System Sciences Discussions*, 10: 1–35.

McNutt, M. 2014. The drought you can't see. *Science*, 345(6204): 1543.

Menz, F.C. & Seip, H.M. 2004. Acid rain in Europe and the United States: an update. *Environmental Science & Policy*, 7: 253-265.

Meyfroidt, P. & Lambin, E.F. 2011. Global Forest Transition: Prospects for an End to Deforestation. *Annual Review of Environment and Resources*, 36(1).

Middleton, N. & Thomas D. 1997. *World Atlas of Desertification*, *2nd edition*. London, Arnold Publication. 182 pp.

Miller, R.D. 1980. Freezing phenomena in soils. In D. Hillel, ed. *Applications of soil physics*. pp. 254-299. New York, NY, Academic Press.

Miransari, M., Bahrami, H.A., Rejali, F. & Malakouti, M.J. 2008. Using arbuscular mycorrhiza to alleviate the stress of soil compaction on wheat (Triticum aestivum L.) growth. *Soil Biology and Biochemistry*, 40: 1197–1206.

Monfreda, C., Ramankutty, N. & Foley, J.A. 2008. Farming the planet: 2. Geographic distribution of crop areas, yields, physiological types and net primary production in the year 2000. *Global Biogeochem. Cycles*, 22: GB 1022.

Montgomery, D.R. 2007. Soil erosion and agricultural sustainability. *Proceedings of the National Academy of Sciences of the United States of America*, 104: 13268-13272.

Moormann, F.R. 1963. Acid sulphate soils (cat-clays) of tropics. *Soil science*, 95: 271-275.

Moser, M. & Hohensinn, F. 1983. Geotechnical aspects of soil slips in Alpine regions. *Engineering Geology*, 19: 185-211.

Mubarak, A.R. & Nortcliff, S. 2010. Calcium Carbonate Solubilization through H-Proton Release from some Legumes Grown in Calcareous Saline-Sodic Soil. *Land Degrad. Develop.*, 21: 29-39.

Muhs, D.R., Budahn, J.R., Avila, A., Skipp, G., Freeman, J., Patterson, D., Ravi, S., D'Odorico, P., Breshears, D.D., Field, J.P., Goudie, A.S. & Huxman, T.E. 2010. The role of African dust in the formation of Quaternary soils on Mallorca, Spain and implications for the genesis of Red Mediterranean soils. *Quaternary Science Reviews*, 29: 2518-2543.

Muhs, D.R., Budahn, J.R., Prospero, J.M., Skipp, G. & Herwitz, S.R. 2012. Soil genesis on the island of Bermuda in the Quaternary: The importance of African dust transport and deposition. *Journal of Geophysical Research*, 117: F 03025.

Munson, S.M., Belnap, J. & Okin, G.S. 2011. Responses of wind erosion to climate-induced vegetation changes on the Colorado Plateau. *Proceedings of the National Academy of Sciences*, 108(10): 3854–3859.

Mutert, E.W. 1996. Plant nutrient balances in the Asian and Pacific region- the consequences for agricultural production. Food & Fertilizer Technology Center for the Asian and Pacific Region. 14 pp

Mylona, S. 1996. Sulphur dioxide emissions in Europe 1880-1991 and their effect on sulphur concentrations and depositions. *Tellus B*, 48: 662-689.

National Atmospheric Deposition Program (NADP). 2014. *National Trends Network*. NADP. (Also available at *http://nadp.sws.uiuc.edu/ntn/*)

Nejad, A.T. & Koocheki, A. 2000. Economic aspect of fourwing saltbush (Atriplex canescens) in Iran. In G.Gintzburger, M. Bounejmate & A. Nefzaoui, eds. *Fodder Shrub Development in Arid and Semi-arid Zones*. Vol. 1, pp. 184-186. Proceedings of the Workshop on Native and Exotic Fodder Shrubs in Arid and Semi-arid Zones, October 27- November 2, 1996, Tunisia, Hammamet. Syria, Aleppo, ICARDA.

Ng, H. 2010. *Regional Assessment: Soil Contamination Policy and Practice in Asia*. Asian Environmental Compliance and Enforcement Network (AECEN).

Nielsen, U.N., Ayres, E., Wall, D.H. & Bardgett, R.D. 2011. Soil biodiversity and carbon cycling: a review and synthesis of studies examining diversity-function relationships. *Global Change Biology*, 62: 105-116.

NLWRA. 2001. *Australian agricultural assessment* 2001. Vol. 1. Canberra, National Land and Water Resources Audit.

Nyssen, J., Poesen, J., Moeyersons, J., Haile, M. & Deckers, J. 2008. Dynamics of soil erosion rates and controlling factors in the Northern Ethiopian Highlands - towards a sediment budget. *Earth Surface Processes and Landforms*, 33(5): 695-711.

Oenema, O. 2004. Governmental policies and measures regulating nitrogen and phosphorus from animal manure in European agriculture. *J Anim Sci*, 82; 196–206.

Okin, G.S., Gillette, D.A. & Herrick, J.E. 2006. Multi-scale controls on and consequences of aeolian processes in landscape change in arid and semi-arid environments. *Journal of Arid Environments*, 65: 253–275.

Okin, G.S., Parsons, A.J., Wainwright, J., Herrick, J.E., Bestelmeyer, B.T., Peters, D.C. & Fredrickson, E.L. 2009. Do changes in connectivity explain desertification? *BioScience*, 59(3): 237–244.

Oldeman, I.R. 1992. Global extent of soil degradation. *In ISRIC Bi-annual Report*, pp. 19-36. The Netherlands, Wageningen.

Oldeman, L.R., Hakkeling, R.T.A. & Sombroek, W.G. 1991. *World map of the status of human-induced soil degradation*. An exploratory note (2nd revised ed.). The Netherlands, Wageningen, International Soil Reference and Information Center (ISRIC). 35 pp.

Onduru, D.D. & Preez, C.C. 2007. Ecological and agro-economic study of small farms in sub-Saharan Africa. *Agronomy for Sustainable Development*, 27: 197–208.

Orgiazzi, A., Dunbar, M.B., Panagos, P., de Groot, G.A. & Lemanceau, P. 2015. Soil biodiversity and DNA barcodes: opportunities and challenges. *Soil Biology and Biochemistry*, 80: 244-250.

Orndorff, Z.W. & Daniels, W.L. 2004. Evaluation of acid-producing sulfidic materials in Virginia highway corridors. *Environ. Geol.*, 46: 209–216.

Osman, A.A., Rice, C. & Codling, E. 2008. Occurrence of antibiotics and hormones in a major agricultural watershed. *Desalination*, 226: 121-133.

Ouimet, R., Arp, P.A., Watmough, S.A., Aherne, J. & DeMerchant, I. 2006. Determination and mapping of critical loads and exceedances for upland forests in eastern Canada. *Water, Air and Soil Pollution*, 172: 57–66.

Oussible, M., Crookstone, P.K. & Larson, W.E. 1992. Sub-surface Compaction Reduces the Root and Shoot Growth and Grain of Wheat. *Agron*. J., 84: 34-38.

Panagos, P., Hiederer, R., Van Liedekerke, M. & Bampa, F. 2013. Estimating soil organic carbon in Europe based on data collected through an European Network. *Ecological Indicators*, 24: 439-450.

Pannell, D.J., Marshall, G.R., Barr, N., Curtis, A., Vanclay, F. & Wilkinson, R. 2006. Understanding and promoting adoption of conservation practices by rural landholders. *Animal Production Science*, 46(11): 1407-1424.

Pannier, F. 1979. Mangroves impacted by human-induced disturbance: a case study of the Orinoco Delta mangrove system. *Environmental management*, 3: 205-216.

Pansak, W., Hilger, T. H., Dercon, G., Kongkaew, T. & Cadisch, G. 2008. Changes in the relationship between soil erosion and N loss pathways after establishing soil conservation systems in uplands of Northeast Thailand. *Agriculture, Ecosystems & Environmen*t, 128(3): 167-176.

Paoletti, M.G., Favretto, M.R., Marchiorato, A., Bressan, M. & Babetto, M. 1993. Biodiversità in pescheti forlivesi. In M.G., Paoletti, ed. *Biodiversità negli Agroecosistemi Osservatorio Agroambientale*, pp. 20-56. Italy, Forli, Centrale Ortofrutticola. 159 pp.

Pons, L.J., van Breemen, N. & Driessen, P.M. 1982. Physiography of coastal sediments and development of potential acidity, part 1. In J.A. Kittrick, D.S. Fanning & L.R. Hossner, eds. *Acid SulfateWeathering*, pp. 1-18. Special Publication No 10. US, Madison, WI, *Soil Science Society of America*.

Post, W.M., Emanuel, W.R., Zinke, P.J. & Stangenberger, A.G. 1982. Soil carbon pools and world life zones. *Nature*, 298: 156-159.

Posthumus, H. & Stroosnijder, L. 2010. To terrace or not: the short-term impact of bench terraces on soil properties and crop response in the Peruvian Andes. *Environment, development and sustainability*, 12(2): 263-276.

Potere, D., Schneider, A., Schlomo, A. & Civco, D. 2009. Mapping urban areas on a global scale: which of the eight maps now available is more accurate? Int. J. *Remote Sens*., 24: 6531–6558.

Powell, J.M. & Valentin, C. 1998. Effects of livestock on soil fertility in West Africa. In G. Renard, A. Neef, K. Becker, M. & von Oppen, eds. *Soil fertility management in West African land use systems*, p. 319–338. Germany, Weikersheim, Margraf Verlag.

Prospero, J.M., Ginoux, P., Torres, O., Nicholson, S.E. & Gill, T.E. 2002. Environmental characterization of global sources of atmospheric soil dust identified with the Nimbus 7 Total Ozone Mapping Spectrometer (TOMS) absorbing aerosol product. *Reviews of Geophysics*, 40(1): 1002.

Prudencio, C.Y. 1993. Ring management of soils and crops in the west African semi-arid tropics: The case of the mossi farming system in Burkina Faso. *Agriculture, Ecosystems & Environment*, 47: 237–264.

Qadir, M., Quillérou, E., Nangia, V., Murtaza, G., Singh, M., Thomas, R.J., Drechsel, P. & Noble, A.D. 2014 Economics of Salt-induced Land Degradation and Restoration. *Natural Resources Forum*, 38(4): 282–295.

Qadir, M., Qureshi, A.S. & Cheraghi, M.S. 2007. Extent and characterization of Salt-prone land resources in Iran and strategies for amelioration and management. *Land Degradation and Development*, 19: 214-227.

Quang, D.V., Schreinemachers, P. & Berger, T. 2014. Ex-ante assessment of soil conservation methods in the uplands of Vietnam: An agent-based modelling approach. *Agricultural Systems*, 123: 108-119.

Quinn, M.L. 1989. Early smelter sites: a neglected chapter in the history and geography of acid rain in the United States. *Atmospheric Environment* (1967), 23: 1281-1292.

Quinton, J.N., Govers, G., Van Oost, K. & Bardgett, R.D. 2010. The impact of agricultural soil erosion on biogeochemical cycling. *Nature Geoscience*, 3(5): 311-314.

Qureshi, A.S., McCornick, P.G., Qadir, M. & Aslam, Z. 2008. Managing salinity and waterlogging in the Indus Basin of Pakistan. *Agricultural Water Management*, 95: 1–10.

Ramankutty, N., Evan, A., Monfreda, C. & Foley, J. 2008. Farming the planet: 1. Geographic distribution of global agricultural lands in the year 2000. *Global Biochemical Cycles*, 22: GB 1003.

Ramisch, J.J. 2005. Inequality, agro-pastoral exchanges, and soil fertility gradients in southern Mali. *Agriculture, Ecosystems & Environment*, 105: 353–372.

Rasmussen, K.J. 1985. Jordpakning ved forkelling belating. Summary: Soil compaction with different surface pressure. *Tidsskrift för Planteavdelning*, 89: 31-45.

Ravi, S., D'Odorico, P., Breshears, D.D., Field, J.P., Goudie, A.S., Huxman, T.E., Li, J., Okin, G.S., Swap, R.J., Thomas, A.D., Van Pelt, R., Whicker, J.J. & Zobeck, T.M. 2011. Aeolian processes and the biosphere. *Reviews of Geophysics*, 449(3): RG3001.

Ravi, S., D'Odorico, P., Huxman, T.E. & Collins, S.L. 2010. Interactions Between Soil Erosion Processes and Fires: Implications for the Dynamics of Fertility Islands. *Rangeland Ecology & Management*, 63: 267–274.

Ravi, S., Zobeck, T.M., Over, T.M., Okin, G.S. & D'Odorico P. 2006. On the effect of moisture bonding forces in air-dry soils on threshold friction velocity of wind erosion. *Sedimentology*, 53: 597–609.

Reheis, M. 1997. Dust deposition downwind of Owens (dry) Lake, 1991–1994: Preliminary findings. *Journal of Geophysical Research*, 102: 25999–26008.

Reuss, J.O. & Johnson, D.W. 1986. *Acid deposition and acidification of soils and waters.* Springer Science &

Business Media, LLC. 119 pp.

Reynolds, R., Neff, J., Reheis, M. & Lamothe, P. 2006. Atmospheric dust in modern soil on aeolian sandstone, Colorado Plateau (USA): Variation with landscape position and contribution to potential plant nutrients. *Geoderma*, 130: 108–123.

Robock, A., Vinnikov, K.Y., Srinivasan, G., Entin, J.K., Hollinger, S.E., Speranskaya, N.A., Liu, S. & Namkhai, A. 2000. The global soil moisture data bank. *Bull. Am. Meteorol*. Soc., 81: 1281–1299.

Robson, A.D. & Abbott, L.K. 1989. The effect of soil acidity on microbial activity in soils. In A. Robson, ed. *Soil acidity and plant growth. Academic Press Australia*.

Rodell, M., Beaudoing, H.K., L'Ecuyer, T.S., Olson, W.S., Famiglietti, J.S., Houser, P.R., Adler, R., Bosilovich, M.G., Clayson, C.A., Chambers, D., Clark, E., Fetzer, E.J., Gao, X., Gu, G., Hilburn, K., Huffman, G.J., Lettenmaier, D.P., Liu, W.T., Robertson, F.R., Schlosser, C.A., Sheffield, J. & Wood, E.F. 2015 The Observed State of the Water Cycle in the Early 21st Century. *Submitted to Journal of Climate* (in press).

Roy, R.N., Misra, R.V, Lesschen, J.P. & Smaling, E.M.A. 2003. *Assessment of Soil Nutrient Balance: Approaches and Methodologies*. Rome, FAO.

Rubey, W.W. 1951. Geologic history of sea water – an attempt to state the problem. *Geol. Soc. Am. Bull.*, 62: 1111–1147.

Ruesch, A. & Gibbs, H. 2008. *New Global Biomass Carbon Map for the Year 2000 Based on IPCC Tier-1 Methodology. US, Oak Ridge, TN, Oak Ridge National Laboratory, Carbon Dioxide Information Analysis Center.*

Ruser, R., Flessa, H., Schilling, R., Steindl, H. & Beese, F. 1998. Soil compaction and fertilization effects on nitrous oxide and methane fluxes in potato fields. *Soil Science Society of America Journal*, 62: 1587-1595.

Samaké, O., Smaling, E.M.A., Kropff, M.J., Stomph, T.J. & Kodio, A. 2005. Effects of cultivation practices on spatial variation of soil fertility and millet yields in the Sahel of Mali. *Agriculture, Ecosystems & Environment*, 109: 335–345.

Sangare, S.K., Compaore, E., Buerkert, A., Vanclooster, M., Sedogo, M.P. & Bielders, C.L. 2012. Field-scale analysis of water and nutrient use efficiency for vegetable production in a West African urban agricultural system. *Nutrient Cycling in Agroecosystems*, 92: 207–224.

Scharlemann, J.P.W., Tanner, E.V.J., Hiederer, R. & Kapos, V. 2014. Global soil carbon: understanding and managing the largest terrestrial carbon pool. *Carbon Management*, 5(1): 81-91.

Schimel, D.S. 1995. Terrestrial ecosystems and the carbon cycle. *Global Change Biology*, 1: 77-91.

Schindler, D.W., Hesslein, R.H., Wagemann, R. & Broecker, W.S. 1980. Effects of acidification on mobilization of heavy metals and radionuclides from the sediments of a freshwater lake. *Canadian journal of fisheries and aquatic sciences*, 37: 373-377.

Schlecht, E., Hiernaux, P., Achard, F. & Turner, M.D. 2004. Livestock related nutrient budgets within village territories in western Niger. *Nutrient Cycling in Agroecosystems*, 68: 199–211.

Schmidt, M.W.I., Torn, M.S., Abiven, S., Dittmar, T., Guggenberger, G., Janssens, I., Kleber, M., Kögel-Knabner, I., Lehmann, J., Manning, D.A.C., Nannipieri, P., Rasse, D.P., Weiner, S. & Trumbore, S.E. 2011. Persistence of soil organic matter as an ecosystem property. *Nature*, 478: 49-56.

Schmitter, P., Dercon, G., Hilger, T., Hertel, M., Treffner, J., Lam, N., Vien, T.D., & Cadisch, G. 2011. Linking spatio-temporal variation of crop response with sediment deposition along paddy rice terraces. *Agric. Ecosyst. Environ.*, 140(1-2): 34-45.

Setälä, H. & McLean, M.A. 2004. Decomposition rates of organic substrates in relation to the species diversity of soil saprophytic fungi. *Oecologia*, 139: 98-107.

Setälä, H., Berg.M. & Jones, T.H. 2005: Trophic Structure and Functional redundancy in Soil Communities.

In. Bardgett, R.D., Hopkins , D.W., and Usher, M, (Eds) Biological Diversity and Function in Soils. Cambridge University Press, Cambridge UK , pp.236-249.

Shamshuddin, J., Azuza, E.A., Shazana, M.A.R.S., Fauziah, C.I., Panhwar, Q.A. & Naher, U.A. 2014. Properties and management of acid sulfate soils in Southeast Asia for sustainable cultivation of rice, oil palm and cocoa. *Advances in Agronomy*, 124: 91-142.

Shao, Y. 2000. *Physics and Modelling of Wind Erosion*. Dordrecht, Kluwer Academic Publ.

Shao, Y., Wyrwoll, K.-H., Chappell, A., Huang, J., Lin, Z., McTainsh, G.H., Mikami, M., Tanaka, T.Y., Wang, X. & Yoon, S. 2011. Dust cycle: an e merging core theme in Earth system science. *Aeolian Research*, 2: 181–204

Sheffield, J. & Wood, E.F. 2007. Characteristics of global and regional drought, 1950-2000: Analysis of soil moisture data from off-line simulation of the terrestrial hydrologic cycle. J. *Geophys. Res.*, 112(D 17115).

Sheffield, J. & Wood, E.F. 2008. Global Trends and Variability in Soil Moisture and Drought Characteristics, 1950–2000, from Observation-Driven Simulations of the Terrestrial Hydrologic Cycle. *J. Climate*, 21: 432–458

Sheffield, J. & Wood, E.F. 2011. *Drought: Past Problems and Future Scenarios*. UK, Earthscan. 192 pp.

Sheffield, J., Wood, E.F., Chaney, N., Guan, K., Sadri, S., Yuan, X., Olang, L., Amani, A., Ali, A. & Demuth, S. 2014. A Drought Monitoring and Forecasting System for Sub-Sahara African Water Resources and Food Security. *Bull. Am. Met. Soc*., 95: 861–882.

Shiel, R.S., Adey, M.A. & Lodder, M. 1988. The effect of successive wet/dry cycles on aggregate size distribution in a clay texture soil. J. Soil Sci., 39: 71-80.

Shift Soil Remediation (SSR). 2010. *Soil Contamination in West Africa*, *Shift Soil Remediation*. (Also available at *http://www.scribd.com/doc/71599035/Soil-Contamination-in-West-Africa*.)

Sidhu, D. & Duiker, S.W. 2006. Soil compaction in conservation tillage: crop impacts. *Agronomy Journal*, 98: 1257-1264.

Simpson, A.J., Song, G.X., Smith, E., Lam, B., Novotny, E.H. & Hayes, M.H.B., 2007. Unraveling the structural components of soil humin by use of solution-state nuclear magnetic resonance spectroscopy. *Environmental Science and Technology*, 41 (3): 876--883.

Sinclair Knight Merz (SKM). 2013. *Management of Contaminated Soils in South Australia*. SKM.

Singer, A., Zobeck, T., Poberezsky, L. & Argaman, E. 2003. The PM 10 and PM 2.5 dust generation potential of soils/sediments in the southern Aral Sea Basin, Uzbekistan. *Journal of Arid Environments*, 54(4): 705–728.

Slattery, B. & Hollier, C. 2002. *The impact of acid soils in Victoria. State of Victoria.* Department of Natural Resources and Environment.

Smaling, E.M.A. & Dixon, J. 2006. Adding a soil fertility dimension to the global farming systems approach, with cases from Africa. *Agriculture, Ecosystems & Environment*, 116: 15–26.

Smaling, E.M.A., Stoorvogel, J.J. & Windmeijer, P.N. 1993. Calculating soil nutrient balances in Africa at different scales. 2. *District scale. Fertilizer Research*, 35: 237–250.

Smedley, P.L. 2003. Arsenic in groundwater—south and east Asia. In The World Bank, ed. *Arsenic in ground water*, pp. 179-209. US, Springer.

Smil, V. 2000. Phosphorus in the environment: Natural Flows and Human Interferences. *Annual Review of Energy and the Environment*, 25: 53–88.

Smith, K.A., McTaggart, I.P. & Tsuruta, H. 1997. Emissions of N_2O and NO associated with nitrogen fertilization in intensive agriculture, and the potential for mitigation. *Soil Use and Management*, 13(4): 296-304.

Smith, P. 2012. Soils and climate change. *Current Opinion in Environmental Sustainability*, 4: 1–6.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H.H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, R.J., Sirotenko, O., Howden, M., McAllister, T., Pan, G., Romanenkov, V., Schneider, U., Towprayoon, S., Wattenbach, M. & Smith, J.U. 2008. Greenhouse gas mitigation in agriculture. *Philos. Transact*. R. Soc., B., 363: 789–813.

Smith, S.J., Pitcher, H. & Wigley, T.M.L. 2001. Global and regional anthropogenic sulphur dioxide emissions. *Global and Planetary Change*, 29: 99-119.

Snyder, B.A., Baas, B. & Hendrix, F.P. 2009. Competition between invasive earthworms (Amynthas corticis, Megascolecidae) and native North American millipedes (Pseudopolydesmus erasus, Polydesmidae): Effects on carbon cycling and soil structure. *Soil Biology and Biochemistry*, 41: 1442-1440.

Soane, B.D. & van Ouwerkerk, C. 1994. Soil compaction problems in world agriculture. In B.D. Soane & C. van Ouwerkerk, eds. *Soil Compaction in Crop Production*, pp.1-21. The Netherlands, Amsterdam, Elsevier.

Squires, V.R. & Glenn, E.P. 2011. Salination, Desertification and Soil Erosion. In *The role of food, agriculture, forestry and fisheries in human nutrition*. Australia, EOLSS.

SSSA. 2008. *Glossary of Soil Science Terms.* Madison, WI, Soil Science Society of America.

Steegen, A., Govers, G., Takken, I., Nachtergaele, J., Poesen, J. & Merckx, R. 2001. Factors controlling sediment and phosphorus export from two Belgian agricultural catchments. *Journal of Environmental Quality*, 30(4): 1249-1258.

Steiner, C., Teixeira, W.G., Lehmann, J., Nehls, T., de Macêdo, J.L.V., Blum, W.E.H. & Zech, W. 2007. Long term effects of manure, charcoal and mineral fertilization on crop production and fertility on a highly weathered Central Amazonian upland soil. *Plant Soil*, 291: 275-290.

Stigliani, W.M., Doelman, P., Salomon, W., Schulin, R., Smidt, G.R.B. & Van der Zee, S.E.A.T.M. 1991. Chemical time bombs. *Environmental Science and Policy for Sustainable Development*, 33: 4-30.

Stoorvogel, J. & Smaling, E. 1990. *Assessment of soil nutrient depletion in Sub-Saharan Africa : 1983-2000*. Report 28. The Netherlands, Wageningen.

Stout, J.E., Warren, A. & Gill, T.E. 2009. Publication trends in aeolian research: An analysis of the Bibliography of Aeolian Research. *Geomorphology*, 105: 6–17.

Sumner, M.E. & Noble, A.D. 2003. Soil acidification: the world story. In Z. Rengel, ed. *Handbook of soil acidity*, pp. 1-28. USA, New York, Marcel Dekker Inc.

Sverdrup, H. & Warfvinge, P. 1993. *The effects of soil acidification on the growth of trees, grass and herbs as expressed by the (Ca+Mg+K)/Al ratio.* Reports in Ecology and Environmental Engineering 2. Sweden, Lund, Lund University, Department of Chemical Engineering II

Tan, Z.X., Lal, R. & Wiebe, K.D. 2005. Global Soil Nutrient Depletion and Yield Reduction. *Journal of Sustainable Agriculture*, 26: 123–146.

Tang, C. 2004. *Causes and management of subsoil acidity.* SuperSoil 2004, 3rd Australian New Zealand Soils Conference, December 5-9, 2004. Australia, University of Sidney. 6 pp.

Tarnocai, C., Canadell, J.G., Schuur, E.A.G., Kuhry, P., Mazhitova, G. & Zimov, S. 2009. Soil organic carbon pools in the northern circumpolar permafrost region. *Global Biogeochemical Cycles*, 23(2): GB 2023.

Tedersoo, L., Bahram, M., Polme, S., Koljalg, U., Yorou, N.S., Wijesundera, R., Ruiz, L.V., Vasco-Palacios, A.M., Thu, P.Q., Suija, A., Smith, M.E., Sharp, C., Saluveer, E., Saitta, A., Rosas, M., Riit, T., Ratkowsky, D., Pritsch, K., Poldmaa, K., Piepenbring, M., Phosri, C., Peterson, M., Parts, K., Partel, K., Otsing, E., Nouhra, E., Njouonkou, A.L., Nilsson, R.H., Morgado, L.N., Mayor, J., May, T.W., Majuakim, L., Lodge,

D.J., Lee, S.S., Larsson, K.-H., Kohout, P., Hosaka, K., Hiiesalu, I., Henkel, T.W., Harend, H., Guo, L-D., Greslebin, A., Grelet, G., Geml, J., Gates, G., Dunstan, W., Dunk, C., Drenkhan, R., Dearnaley, J., De Kesel, A., Dang, T., Chen, X., Buegger, F., Brearley, F.Q., Bonito, G., Anslan, S., Abell, S. & Abarenkov, K. 2014. Global diversity and geography of soil fungi. *Science*, 346(6213): 1078-1088.

Thomas, D.S.G. & Wiggs, G.F.S. 2008. Aeolian system responses to global change: Challenges of scale, process and temporal integration. *Earth Surface Processes & Landforms*, 33: 1396–1418.

Thomas, G.W., Haszler, G.R. & Blevins, R.L. 1996. The effects of organic matter and tillage on maximum compactability of soils using the proctor test. *Soil Sci.*, 161: 502–508.

Tittonell, P. 2008. *Targeting resources within diverse, heterogenous and dynamic farming systems of East Africa*. Wageningen University. 320 pp. (PhD Thesis)

Todd-Brown, K.E.O., Randerson, J.T., Post, W.M., Hoffman, F.M., Tarnocai, C., Schuur, E.A.G. & Allison, S.D. 2013. Causes of variation in soil carbon simulations from CMIP 5 Earth system models and comparison with observations. *Biogeosciences*, 10: 1717–1736.

Tote, C., Govers, G., Van Kerckhoven, S., Filiberto, I., Verstraeten, G. & Eerens, H. 2011. Effect of ENSO events on sediment production in a large coastal basin in northern Peru. *Earth Surface Processes and Landforms*, 36: 1776-1788.

Tóth, J.A., Lajtha, K., Kotroczó, Z., Krakomperger, Caldwell, B., Bowden, R. & Papp, M. 2007. The effect of climate change on soil organic matter decomposition. *Acta Silvatica & Lingaria Hungarica*, 3: 1-11.

Tracy, B.F. & Zhang, Y. 2008. Soil compaction, corn yield response and soil nutrient pool dynamics within an integrated crop-livestock system in Illinois. *Crop Science Society of America*, 48: 1211-1218.

Treasury Board of Canada Secretariat. 2014. *Contaminants and Media*. Canada. (Also available at *http://www.tbs-sct.gc.ca/fcsi-rscf/cm-eng.aspx?qid=1200518*.)

Tsiafouli, M., Thébault, E., Sgardelis, S., de Ruiter, P., van der Putten, W.H., Birkhofer, K., Hemerik, L., de Vries, F.T., Bardgett, R., Brady, M., Bjørnlund, L., Jørgensen, H., Christensen, S., D'Hertefeldt, T., Hotes, S., Hol, W.H.G., Frouz, J., Liiri, M., Mortimer, S., Setälä, H., Tzanopoulos, J., Uteseny, K., Karoline, Pižl, V., Stary, J., Wolters, V. & Hedlund K. 2015. Intensive agriculture reduces soil biodiversity across Europe. *Global Change Biology*, 21: 973-985.

Tuller, M. & Or, D. 2003. Retention of water in soil and the soil water characteristic curve. *In Soil water characteristic*, p. 1. Figure 1. USA. (Also available at *http://www.engr.uconn.edu/environ/envphys/pdf/vadose_pdf/SWC_revised* 01.pdf)

U.S. Department of Agriculture (USDA). 2009. Summary Report, 2007. National Resources Inventory. Washington, DC, Natural Resources Conservation Service and Ames, Iowa, Center for Survey Statistics and Methodology, Iowa State University. 123 pp.

UNEP DEWA/GRID-Europe. 2005. E-waste, the hidden side of IT equipment's manufacturing and use. Chapter 5 – Early warning and Emerging Environmental. (Also available at *http://www.grid.unep.ch/products/3_Reports/ew_ewaste.en.pdf*)

UNEP. 1992. Proceedings of the Ad-hoc Expert Group Meeting to Discuss Global Soil Databases and Appraisal of GLASOD/SOTER, February 24-28. Nairobi, UNEP.

United Nations – UN News Center. 2011. *UN declares famine in another three areas of Somalia*. (Also available at *http://www.un.org/apps/news/story.asp?NewsID=39225*).

United Nations Environment Programme (UNEP). 2010. *Latin America and the Caribbean: Environment Outlook*. UNEP.

United Nations. 2008. *World Urbanization Prospects: the 2007 Revision*. Department of Economic and Social Affairs, Population Division.

USGS. 2008. *Mineral Commodity Summaries*. USA, Reston, VA, USGS.

van der Heijden, M.G.A., Klironomos, J.N., Ursic, M., Moutoglis, P., Streitwolf-Engel, R., Boller, T., Wiemken, A. & Sanders, I.R. 1998. Mycorrhizal fungal diversity determines plant biodiversity, ecosystem variability and productivity. *Nature*, 396: 69-72.

Van der Putten, W.H., Anderson, J.M., Bardgett, R.D., Behan-Pelletier, V., Bignell, D.E., Brown, G.G., Brown, V.K., Brussaard, L., Hunt, H.W., Ineson, P., Jones, T.H., Lavelle, P., Paul, E.A., St John, M., Wardle, D.A., **Wojtowicz, T. & Wall, D.H**. 2004. *The sustainable services provided by soil biota. In D.H. Wall, ed. Sustaining biodiversity and Ecosystem Services in Soils and Sediments*, pp. 15-43. San Francisco, Island Press.

Van Haveren, B.P. 1983. Soil bulk density as influenced by grazing intensity and soil type on a shortgrass prairie site. *Journal of Range*, 36: 586–588.

Van Oost, K., Govers, G., de Alba, S. & Quine, T. A. 2006. Tillage erosion: a review of controlling factors and implications for soil quality. *Progress in Physical Geography*, 30(4): 443-466.

Van Oost, K., Quine, T.A., Govers, G., De Gryze, S., Six, J., Harden, J.W., Ritchie, J.C., McCarty, G.W., Heckrath, G., Kosmas, C., Giraldez, J.V., Marques da Silva, J.R. & Merckx, R. 2007. The impact of agricultural soil erosion on the global carbon cycle. *Science*, 318(5850): 626-629.

Van Pelt, R.S. & Zobeck, T.M. 2007. Chemical constituents of fugitive dust. *Environmental Monitoring and Assessment*, 130: 3–16.

Vanacker, V., von Blanckenburg, F., Govers, G., Molina, A., Poesen, J., Deckers, J. & Kubik, P. 2007. Restoring dense vegetation can slow mountain erosion to near natural benchmark levels. *Geology*, 35(4): 303-306.

Vestreng, V., Myhre, G., Fagerli, H., Reis, S. & Tarrasón, L. 2007. Twenty-five years of continuous sulphur dioxide emission reduction in Europe. *Atmospheric Chemistry and Physics*, 7: 3663-3681.

Vicent, T. (ed.) 2013. Emerging Organic Contaminants in Sludges: Analysis, Fate and Biological Treatment. *Hdb Env Chem*, 24: 1–30.

Vitousek, P.M., Naylor, R., Crews, T., David, M.B., Drinkwater, L.E., Holland, E., Johnes, P.J., Katzenberger, J., Martinelli, L.A., Matson, P.A., Nziguheba, G., Ojima, D., Palm, C.A., Robertson, G.P., Sanchez, P.A., Townsend, A.R. & Zhang, F.S. 2009. Nutrient Imbalances in Agricultural Development. *Science*, 324: 1519–1520.

Vlek, P.L.G., Kuhne, R.F. & Denich, M. 1997. Nutrient resources for crop production in the tropics. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 352: 975–985.

Voegelin, A., Barmettler, K. & Kretzschmar, R. 2003. Heavy metal release from contaminated soils. *Journal of environmental quality*, 32: 865-875.

Von Uexküll, H.R. & Mutert, E. 1995. Global extent, development and economic impact of acid soils. *Plant and Soil*, 171: 1-15.

Voorhees, W.B., Nelson, W.W. & Randall, G.W. 1986. Extent and persistence of subsoil compaction caused by heavy axle loads. *Soil Sci. Soc. Am. J.*, 50: 428-433.

Wagg, C., Bender, S.F., Widmer, F. & van der Heijden, M.G.A. 2014. Soil biodiversity and soil community composition determine ecosystem multifunctionality. *PNAS*, 111: 5266-5270.

Wall, D.H., Bradford, M.A., St. John, M.G., Trofymow, J.A., Behan-Pelletier, V., Bignell, D.E., Dangerfield, J.M., Parton, W.J., Rusek, J., Voigt, W., Wolters, V., Gardel, H.Z., Ayuke, F.O., Bashford, R., Beljakova, O.I., Bohlen, P.J., Brauman, A., Flemming, S., Henschel, J.R., Johnson, D.L., Jones, T.H., Kovarova, M., Kranabetter, J.M., Kutny, L., Lin, K.-C., Maryati, M., Masse, D., Pokarzhevskii, A., Rahman, H.,

Sabará, M.G., Salamon, J.-A., Swift, M.J., Varela, A., Vasconcelos, H.L., White, D. & Zou, X. 2008. Global decomposition experiment shows soil animal impacts on decomposition are climate dependent. *Global Change Biology*, 14(11): 2661-2677.

Wallace, A. 1994. Soil organic matter must be restored to near original levels. *Communications in soil science plant analysis*, 25: 29-35.

Wang, Y.F., Zhang, W.F. & Zhang, F.S. 2010. China's fertilizer management system: situation, problems and prospect. *Xiandai Huagong/Modern Chemical Industry*, 31: 6-13.

Wang, Z., Van Oost, K., Lang, A., Quine, T., Clymans, W., Merckx, R., Notebaert, B. & Govers, G. 2014. The fate of buried organic carbon in colluvial soils: a long-term perspective. *Biogeosciences*, 11(3): 873-883.

Wardle, D.A., Bardgett, R.D., Klironomos, J.N., Setälä, H., Van Der Putten W.H. & Wall, D.H. 2004. Ecological linkages between aboveground and belowground biota. *Science*, 304(5677): 1629-1633.

Warren, S.D., Thurow, T.L., Blackburn, W.H. & Garza, N.E. 1986. The influence of livestock trampling shortgrass prairie site. *Journal of Range Manage*, 36: 586–588.

Webb, N.P. & Strong, C.L. 2011. Soil erodibility dynamics and its representation for wind erosion and dust emission models. *Aeolian Research*, 3: 165–179.

Webb, N.P., Chappell, A., Strong, C.L., Marx, S.K. & McTainsh, G.H. 2012. The significance of carbon-enriched dust for global carbon accounting. *Global Change Biology*, 18: 3275–3278.

We[i], C.Y. & Chen, T.B. 2001. Hyperaccumulators and phytoremediation of heavy metal contaminated soil: a review of studies in China and abroad. *Acta Ecologica Sinica*, 21: 1196-1203.

Wendelin, G. 1646. *Pluvia pupurea Bruxellensis*. Parisiis, Apud Ludivicum de Heuqueville.

Wezel, A., Rajot, J.-L. & Herbrig, C. 2000. Influence of shrubs on soil characteristics and their function in Sahelian agro-ecosystems in semi-arid Niger. *Journal of Arid Environments*, 44: 383–398.

White, R.E. 1987. *Introduction to the principles and practice of soil science.* Blackwell Sci. Publication.

Whitfield, C.J., Aherne, J., Watmough, S.A. & McDonald, M. 2010. Estimating the sensitivity of forest soils to acid deposition in the Athabasca Oil Sands Region, Alberta, Canada. *Journal of Limnology*, 69(1): 201–208.

Wiggs, G.F.S., Baird, A.J. & Atherton, R.J. 2004. The dynamic effects of moisture on the entrainment and transport of sand by wind. *Geomorphology*, 59: 13–30.

Wilke, B.M., Duke, B.J. & Jimoh, W.L.O. 1984. Mineralogy and chemistry of harmattan dust in Northern Nigeria. *Catena*, 11: 91-96.

Willatt, S.T. & Pullar, D.M. 1983. Changes in soil physical properties under grazed pastures. *Aust. J. Soil Res.*, 22: 343-348.

Williams, A., Funk, C., Michaelsen, J., Rauscher, S., Robertson, I., Wils, T., Koprowski, M., Eshetu, Z. & Loader, N. 2011. Recent summer precipitation trends in the Greater Horn of Africa and the emerging role of Indian Ocean sea surface temperature. *Climate Dynamics*, 39(9-10): 2307-2328.

Wittmann, H. & von Blanckenburg, F. 2009. Cosmogenic nuclide budgeting of floodplain sediment transfer. *Geomorphology*, 109(3-4): 246-256.

Wolt, J.D. 1981. Sulfate retention by acid sulphate-polluted soils in the Copper Basin area of Tennessee. *Soil Science Society of America Journal*, 45: 283-287.

Wood, M.K. & Blackburn, W.H. 1984. Vegetation and soil responses to cattle grazing systems in the Texas Rolling Plains. *Journal Range Management*, 37: 303-308.

Xu, R.K., Coventry, D.R., Farhoodi, A. & Schultz, J.E. 2002. Soil acidification as influenced by crop rotations, stubble management, and application of nitrogenous fertiliser, Tarlee, South Australia. *Australian Journal of Soil Research*, 40: 483-496.

Yue, X., Wang, H., Wang, Z. & Fan, K. 2009. Simulation of dust aerosol radiative feedback using the Global Transport Model of Dust: 1. Dust cycle and validation. *Journal of Geophysical Research: Atmospheres* (1984–2012), 114(D 10).

Zentner, R., McConkey, B., Campbell, C., Dyck, F. & Selles, F. 1996. Economics of conservation tillage in the semiarid prairie. *Can. J. Plant Sci.*, 76(4): 697-705.

Zhao, Y., Duan, L., Xing, J., Larssen, T., Nielsen, C.P. & Hao, J. 2009. Soil acidification in China: is controlling SO_2 emissions enough? *Environmental Science and Technology*, 43: 8021-8026.

Zobeck, T.M. & Bilbro, J.D. 2001. Crop productivity and surface soil properties of a severely wind eroded soil. In D.E. Stott, R.H. Mohtar & G.C. Steinhardt, eds. *Sustaining the Global Farm*, pp. 617-622. Selected Papers From the 10th International Soil Conservation Organization Meeting. West Lafayette, IN, International Soil Conservation Organization.

Zobeck, T.M. & Van Pelt, R.S. 2011. Wind erosion. In L. Hatfield & T.J. Sauer, eds. *Soil Management: Building a Stable Base for Agriculture*, pp. 209–227. Madison, WI, Soil Science Society of America, Inc.

Zobeck, T.M. 1991. Soil properties affecting wind erosion. *Journal of Soil & Water Conservation*, 46(2): 112–118.

Zreda, M., Shuttleworth, W.J., Zeng, X., Zweck, C., Desilets, D., Franz, T. & Rosolem, R. 2012. COSMOS: The COsmic-ray Soil Moisture Observing System. *Hydrology and Earth System Science*, 16: 4079-4099.

Soil change: impacts and responses

Coordinating Lead Authors:
Chencho Norbu (Bhutan), David Robinson (United Kingdom), Miguel Taboada (Argentina)

Contributing Authors:
Marta Alfaro (Chile), Richard Bardgett (United Kingdom), Sally Bunning (United Kingdom), Jana Compton (United States), William Critchley (United Kingdom), Warren Dick (United States), Scott Fendorft (United States), Gustavo Ferreira (Uruguay), Tsuyushi Miyazaki (Japan), Carl Obst (Australia), Dani Or (Switzerland), Dan Pennock (ITPS/Canada), Matthew Polizzotto (United States), Dan Richter (United States), Marta Rivera-Ferre (Spain), Sonia Seneviratne (Switzerland), Pete Smith (United Kingdom), Garrison Sposito (United States), Susan Trumbore (United States) and Kazuhiko Watanabe (Japan).

Reviewing Authors:
Dominique Arrouays (ITPS/France), Richard Bardgett (United Kingdom), Marta Camps Arbestain (ITPS/New Zealand), Tandra Fraser (Canada), Ciro Gardi (Italy), Neil McKenzie (ITPS/Australia), Luca Montanarella (ITPS/EC), Dan Pennock (ITPS/Canada) and Diana Wall (United States).

7 | The impact of soil change on ecosystem services

7.1 | Introduction

Soils are now recognized to be in the 'front line' of global environmental change and we need to be able to predict how they will respond to changing climate, vegetation, erosion and pollution. This requires a better understanding of the role of soils in the Earth system to ensure that they continue to provide for humanity and the natural world (Schmidt *et al.*, 2011). Although only a thin layer of material at the Earth's surface, soils like many interfaces play a pivotal role in regulating the flow and transfer of mass and energy between the atmosphere, biosphere, hydrosphere and lithosphere. Moreover, the structure and organization of soils leaves an important imprint on the Earth's surface in terms of how land is used and how ecosystems develop. Soils help regulate the Earth's physical processes such as water and energy balances, and act as the biogeochemical engine at the heart of many of the Earth system cycles and processes on which life depends. Some soil processes contribute directly to the delivery of ecosystem goods and services, while other soil processes influence the delivery of goods and services. This section examines how soil processes affect soil and ecosystem function and the production of goods and services of benefit to humanity.

Humanity has had an indelible impact on the Earth's surface, so much so that it has been proposed that the planet has entered a new geological epoch, the Anthropocene (Crutzen, 2002). A population of ca. 7 billion people that will likely grow to 9.6 billion by 2050 is stressing Earth's resources. Maintaining the planet in an equitable state for human life is perhaps our greatest challenge. Currently, humans have adapted 38 percent of the earth's ice-free land surface to agriculture, crops and pasture (Foley *et al.*, 2011). Agricultural production, driven by the need to produce food for a growing population, has had a tremendous impact on our ecosystems and resources, especially through the abstraction of water and the leaving of residues. Rockström *et al.* (2009) proposed that we need a 'safe operating space for humanity with respect to the Earth system'. They argue that that there exist biophysical planetary boundaries (or thresholds) which it is inadvisable to cross if we are to maintain the needed balance. Vince and Raworth (2012) adapted these concepts to include social goals (1). This presentation underlines the fact that we live in a coupled human earth system. The ecosystem services analytic approach has been developed in order to bridge the science/policy divide. The approach aims to make the concepts clear for all and to set out what needs to be considered in order for humanity to live within sustainable boundaries.

Soils and soil security are at the heart of this effort. Soil security is defined in McBratney, Field and Koch (2014) as "maintaining and improving the world's soil resources to produce food, fibre and freshwater, to contribute to energy and climate sustainability, and to maintain the biodiversity and the overall protection of the ecosystem". Soils perform important ecosystem services (e.g. functions for humanity) including: biomass production; storing, filtering and transforming nutrients and water; maintaining a gene pool; providing a source of raw material for products such as bricks and tiles; regulating climate and hydrology; and providing an archive of cultural heritage. Soils provide ecosystem goods and services directly but some soil processes can have an adverse impact on the delivery of ecosystem goods and services. The ability of soils to function can be threatened by human activity (on this, see the Soil Thematic Strategy, SEC, 2006). A growing population, resource extraction, agricultural production, land use change and climate change all contribute to this threat.

As population increases, food security is becoming more important in the global agenda. Our historical solution to producing more food has been to mechanize, cultivate more land, and increase the application of plant nutrients and water. This has led to an almost linear increase in production over time (Pretty, 2008). However, the rate of increase is likely to plateau, as has already been seen with wheat in Northern Europe and with rice in Korea and China (Cassman, Grassini and Wart, 2010). In addition, agricultural growth comes with environmental costs or externalities, which are costs not accounted for in the cost of production. The degradation caused can adversely affect everyone, and even the production systems themselves - for instance, declines in pollinators can threaten future production (Deguines *et al.*, 2014).

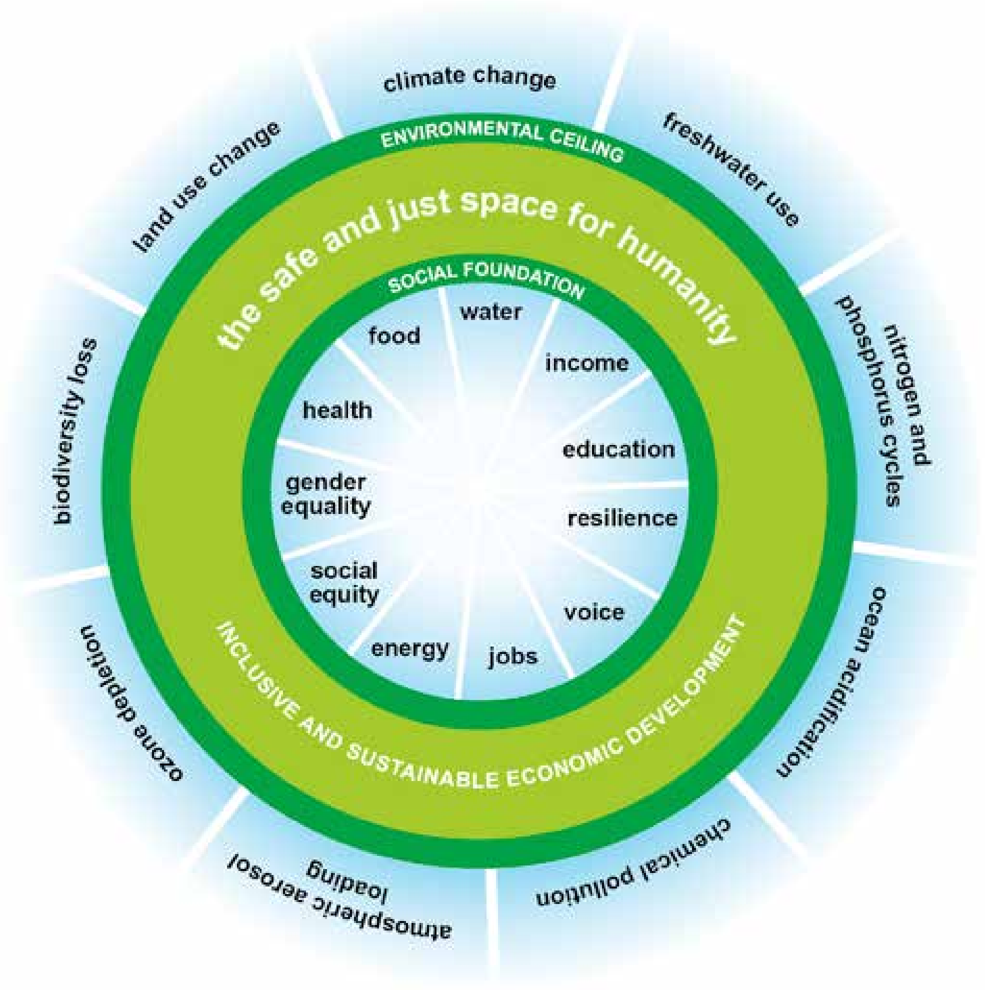


Figure 7.1 | The 11 dimensions of society's 'social foundation' and the nine dimensions of the 'environmental ceiling' of the planet. Source: Vince and Raworth, 2012.

Rockström *et al*. (2009) suggest we are approaching the limits of the planet's cultivatable land, while the addition of nutrients, especially nitrogen, continues to overload many terrestrial-aquatic systems (Diaz and Rosenberg, 2008). At the same time, arable production has seen declines in the carbon content of soils - the largest terrestrial carbon reservoir – and these declines are affecting other soil functions, including water and nutrient retention (Reynolds *et al.*, 2013). Food production systems will need to change to create multifunctional agro-ecosystems capable of maintaining a balance between yields, soil functions and biological diversity. Within the field of ecology, this challenge has led to a rigorous debate concerning the loss of natural species from agricultural lands - often termed, the 'land sparing, land sharing' debate (Green *et al.*, 2005). This debate has now been integrated within the broad ecosystem services discussion whose central ten also focuses on human interaction with ecosystems and their long-term sustainability and continued functionality (MA, 2005).

Conceptually, Foley *et al*. (2005) proposed that a natural ecosystem provides a range of goods and services (2) while on intensively farmed agricultural land, crop production dominates at the expense of all other goods and services. They proposed that an ideal situation would be one of balance, with the system producing a range of goods and services including food – the 'sharing' side of the land sparing/ land sharing debate. Organic agriculture has been seen as a model of this sharing or balance. However, organic agriculture has so far generally failed to maintain productivity levels in either crop or livestock systems (Pretty, 2008). The implication is that organic agriculture does not yet promise balance, because it requires more land and more use of natural capital to maintain production levels. Determining if there are viable 'sharing' systems should continue to be an important research goal but for the moment 'sparing' appears to have the upper hand in the debate (Phalan *et al.*, 2011). But how do we achieve sustainable intensification? While the viability of sharing remains in question, should we focus on a narrow-minded, single service supply management strategy, e.g. arable soils for crop production or peat soils for carbon storage? Sustainable intensification research, which seeks to find ways of optimizing production while blending in new strategies for multifunctional ecosystem service management, is being championed as a way forward (Firbank *et al.*, 2013).

There is no single solution. Foley's conceptual diagram (2) highlights the challenges and possible trade-offs: a natural ecosystem delivers a wide range of ecosystem services but scant production; and an intensive cropland system delivers royally on production but precious little on ecosystem services. A balanced system of cropland with restored ecosystem services would deliver on all services, including production. A recent synthesis and analysis of data from the Countryside Survey, a national survey of Great Britain, suggests that Foley's conceptual diagram of intensive cropland (2) is the current situation. Different services reach optimums at different points along the productivity gradient, but we cannot have everything (3). The ecosystem service indicators alter, often in a non-linear way with the proportion of intensive land use – but with exception of production, they all decline with intensification. 3 b and c go on to show that changes in moisture inputs or moisture regime, or alteration of soil pH would change the service delivery balance. At no point do we get everything, so we will need to choose priorities with our current systems.

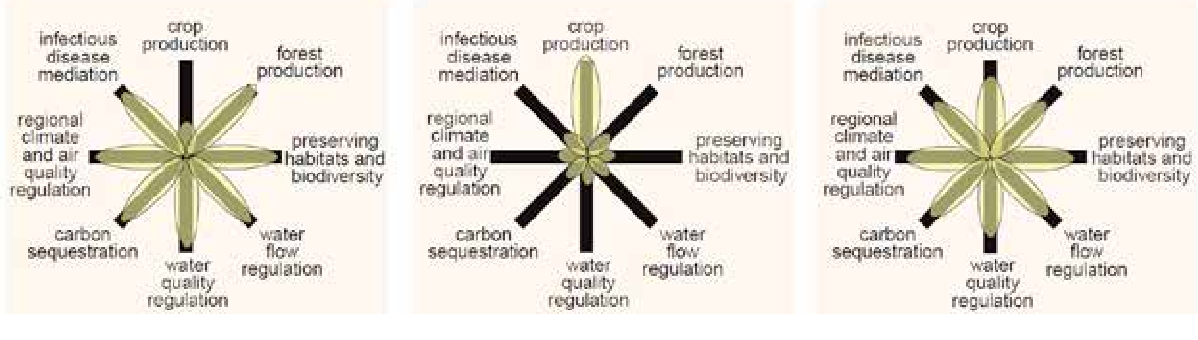


Figure 7.2 | Conceptual framework for comparing land use and trade-offs of ecosystem services.
Source: Foley et al., 2005.

The land sparing, land sharing approach can also been framed in terms of resilience (sharing) and efficiency (sparing). Efficient systems by their very nature will prioritise the performance of one function over that of others. The degree to which others are affected will depend on whether they perform well under similar management or not. Currently, the data in 3 indicate that the choice lies between efficiency and redundancy. We can have an efficient carbon storage system, e.g. peat development, which may also perform well as a climate thermal buffer because the conditions for peat accumulation require lots of water, but it will not be productive for crops in that state, nor will the arable system have high biodiversity as this is inefficient.

Choices need to be made as to what types of systems we wish to promote. In light of this, the focus of this chapter is to assess the global scientific literature and understand how soil change discussed in Chapters 5 and 6 is likely to impact soil functions and the likely consequences for ecosystem service delivery. Each section of this chapter outlines key soil processes involved with the delivery of goods and services and how these are changing or - where evidence permits - may change. Each section then reviews how this change impacts soil function and affects ecosystem service delivery. Some soil change does not produce an ecosystem service, but does impact it; these impacts are considered when assessed as important and adverse. The focus is on the local, regional and global scales and follows the general reporting categories of the MA (2005) modified by TEEB (2014) to provisioning, regulating and cultural services. Towards the end of the section there is a focus on the links with policy, institutions and management.

7.2 | Soil change and food security

Keating *et al*. (2014) provide a useful frame for examining the main roles of soils in food supply through their development of the food wedge concept. The food wedge is the triangular area between the level of food demand in 2010 and the upper bound of food demand in 2050 (suggested by Keating and Carberry (2010) as a wedge equal to approximately 127 x 1015 kcal). The food wedge presented by Keating *et al.* (2014) assumes that food supply and demand were broadly in balance in 2010. Increases in food supply (through, for example, the strategies suggested by Foley *et al.*, 2011) would increase the supply to meet the rising demand for food.

Either the incremental loss of productivity from current agricultural land or the total loss of agricultural land due to degradation in the future would cause the lower boundary of the wedge to decrease and hence increase the gap between food supply and demand (Figure 7.4). This decrease (or total loss) could occur if the services for plant production supplied by the soil decreased due to a significant impairment of one or more of the soil functions. Alternatively the restoration of productivity to previously degraded land would increase plant production in addition to addressing the yield gap or increases in food delivery. Therefore a key soil-focused strategy is to reduce future productivity loss from agricultural soils due to degradation to a minimum and to restore productivity to soils that have previously experienced productivity losses.

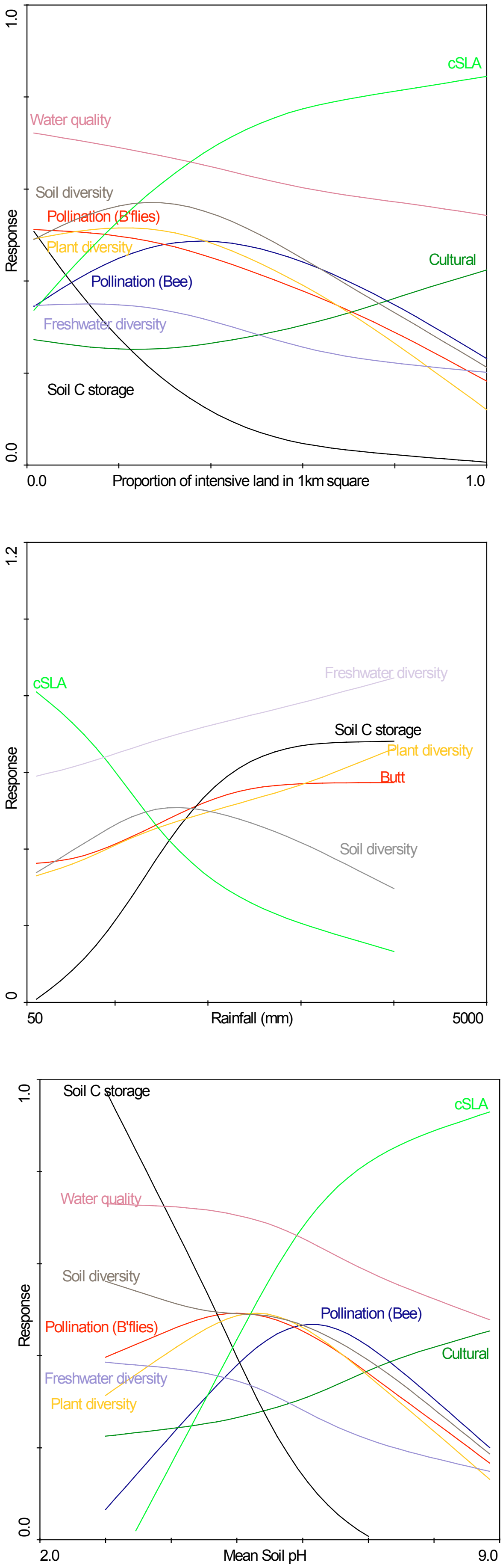


Figure 7.3 | Response curves of mean ecosystem service indicators per 1-km² across Great Britain.
Source: Maskell et al., 2013.

The curves are fitted using generalized additive models to ordination axes constrained by; (a) proportion of intensive land (arable and improved grassland habitats) within each 1-km square from CS field survey data; (b) mean long-term annual average rainfall (1978–2005); and (c) mean soil pH from five random sampling locations in each 1-km square. All X axes are scaled to the units of each constraining variable

The restoration of productivity on degraded soils can be complex insofar as soils may have been degraded to the point where they cannot readily respond to fertility-improving management techniques. These complex interactions among inherent soil properties, management history and the response to inputs is well illustrated in the work of Rusinamhodzi *et al.* (2013) on maize production intensification on smallholder farms in Zimbabwe. In this region two major controls of productivity exist – significant differences in yield between sandy and clay soils (e.g. inherent soil properties); and pronounced fertility gradients between more productive fields close to the homestead and more degraded soils in outfields further from the homestead (a management-induced fertility gradient common in many areas of Africa). The sandy soils required long-term additions of manure to restore soil functions before the benefit of the mineral fertilizer additions could begin to be realized; however even after nine years of substantial organic inputs, the highly degraded sandy outfields did not recover their productivity. The authors speculate that the initial soil organic carbon levels in the sandy outfields were too low for yields to recover. Moreover at the village scale, the overall amount of manure produced is insufficient to apply the required amounts of manure in all fields.

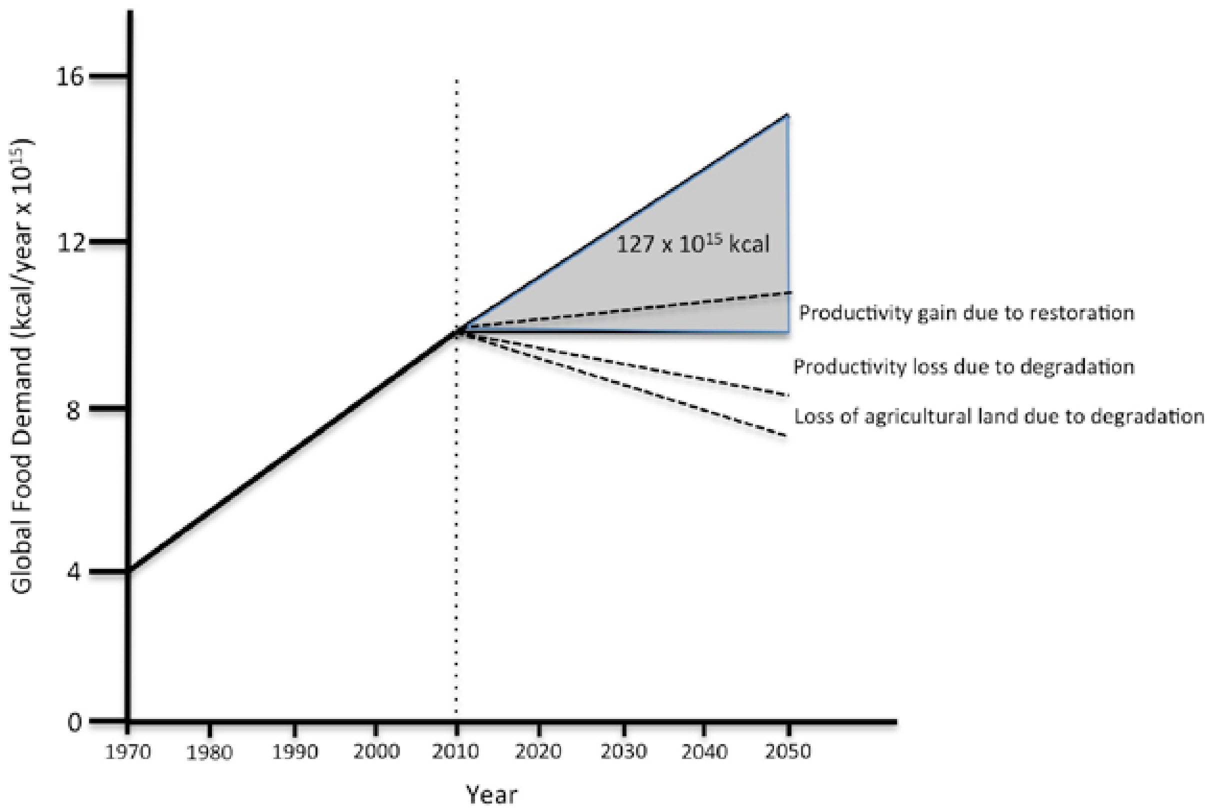


Figure 7.4 | The food wedge and the effect of soil change on the area of the wedge.
Source: Keating et al., 2014.

The relative sizes of the effects of soil change on the food wedge are not drawn to scale.

One approach to maintaining soil health is 'conservation agriculture', which comprises a range of agricultural practices that include reduced tillage and no-till, greater retention of crop residues, and crop rotations. However, the lack of organic inputs which constrained productivity in the Zimbabwe example above also limits the ability of conservation agriculture to restore fertility in sub-Saharan Africa generally. Palm *et al*. (2014) found that the greatest obstacle to improving soil functions and other ecosystem services in Sub-Saharan Africa region is the lack of residues produced due to the low productivity of the soils. The limited supply of crop residues also highlights the need to make optimum use of all sources of organic inputs, such as animal manure and properly processed human wastes.

These studies emphasize the inability of mineral fertilizers alone to significantly increase food production in regions where the yield gap is greatest. Removing the nutrient limitations through additions of mineral fertilizers alone will also exacerbate the range of environmental issues (e.g. N_2O emission from N-fertilizer, surface and groundwater contamination) in all food-producing regions unless the efficiency of crop use of agricultural inputs can be increased. Additionally the fraction of P available as mineable phosphate rock is finite. Recent concerns that the world's supply of phosphorus was being rapidly depleted and that 'peak phosphorus' was only a few decades away (Cordell and White, 2010) have been dispelled, due to recent upward revisions of world phosphate rock reserves and resources (Van Kauwenbergh, 2010). However, the world supply of phosphorus is limited, and rising prices and market volatility are inevitable. More efficient use of phosphorus is therefore essential. This overall issue is termed the 'Goldilocks' problem by Foley *et al*. (2011) – there are many regions with too much or too little fertilizer but few that are 'just right'.

A final strategy is to minimize diversion of agricultural soils to production of non-food crops. Recent large-scale bioenergy production on land previously used for food production has driven a significant land use change and represents a major shift of agricultural soils away from food production. Demand for soybean, maize and oil palm for biofuel has been a driver of agricultural land conversion in recent years particularly in Latin America. Conversion of existing cropland or the development of new cropland for bioethanol and biodiesel production competes with food production and carbon returns to the soil (Foley *et al*., 2011) and thus constitutes a threat to soil and food security. Biofuels produced from crops using conventional agricultural practices will exacerbate stresses on water supplies, water quality and land use. In any case, biofuels are not expected to mitigate the impact of climate change as compared with petroleum (Delucchi, 2011).

Threats to the food security dimension 'availability' are mainly (but not only) caused by soil and land degradation and associated water resources (Khan *et al*., 2009). This is particularly the situation when the degradation is irreversible or very hard to reverse. This may, for example, be the case with severe topsoil losses caused by wind or water erosion, terrain deformation by gully erosion or mass movement, acidification, alkalinization/salinization, soil sealing, or contamination with toxic substances (Scherr, 1999; Palm *et al*., 2007; Mullan, 2013). The resulting loss of productivity will reduce yields from a site, leading to reduced returns to producers and, in some cases, abandonment of production at the site. Productivity may be restored, but economic considerations may limit the adoption of restorative measures.

The impact of each threat on specific soil functions relevant to crop production has been covered in Chapter 6 and is summarized in Figure 7.5. The present chapter will focus on the implications for food security of the trends in each threat.

7.2.1 | Soil erosion

A summary by den Biggelaar *et al*. (2003) suggests that global mean rates of erosion are between 12 to 15 tonnes ha^{-1} yr^{-1} (Table 7.1). The mid-point of this range yields a soil loss of 0.9 mm yr^{-1} (see Table 7.1), very similar to the mean soil loss of 0.95 mm yr^{-1} calculated by Montgomery (2007). Overall these rates are substantially higher than rates of soil formation, and hence pose a long-term global threat to soils (Montgomery, 2007; see also Section 6.1 above).

Our understanding of the rates for the three erosion agents (wind, water and tillage) is uneven. Erosion rates due to water erosion remain very high (> ca. 20 tonnes ha^{-1} yr^{-1}) in cropland in many agricultural regions (Figure 7.2); essentially any cropped area with hilly land and sufficient precipitation is at risk. No reliable global estimates for current wind erosion rates exist, and the estimates of the human contribution to current dust emissions range from only 8 percent in North Africa to approximately 75 percent in Australia (see also Section 6.1 above). Tillage erosion primarily results in in-field redistribution of soil, and decreases the productivity of soils in convex slope elements and near-upslope field or terrace borders. Global-scale summaries also require consideration of the fate of eroded soil – in some regions deposition of eroded soil in river floodplains and deltas creates areas of very high and enduring fertility.

The effect of soil erosion on individual soil properties related to crop production is well documented, but the aggregate effect of soil loss on crop yields themselves is less firmly established. The four integrative studies summarized in Table 7.1 are based on data sources which range from experimental plot data to re-interpretation of GLASOD data. The range of estimates of annual crop loss due to erosion ranges from 0.1 percent to 0.4 percent, with two studies estimating 0.3 percent yield reduction.

If the median value of 0.3 percent annual crop loss is valid for the period from 2015 to 2050, a total reduction of 10.25 percent could be projected to 2050 (assuming no other changes such as the adoption of additional conservation measures by farmers). Foley *et al.* (2011) cite a value of 1.53 billion ha for cropland globally; the loss of 10.25 percent of yield due to erosion would be equivalent to the removal of 150 million ha from crop production or 4.5 million ha per year.

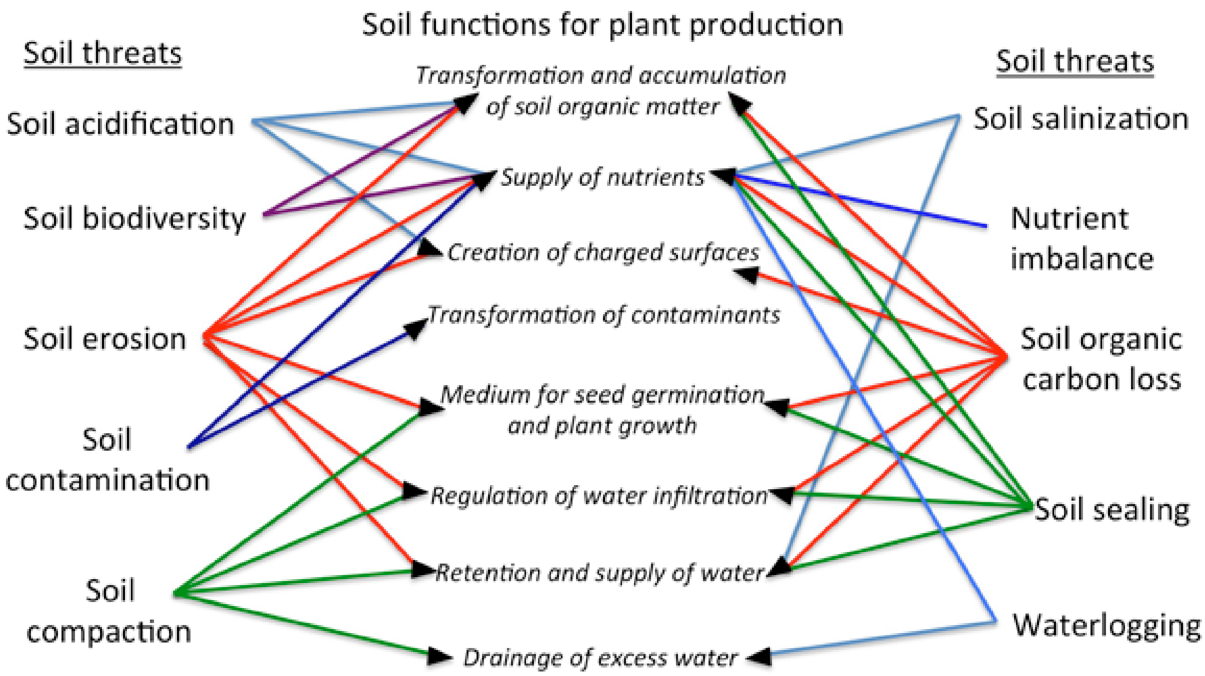


Figure 7.5 | Direct impacts of soil threats on specific soil functions of relevance to plant production.

Table 7.1 | Erosion and crop yield reduction estimates from post-2000 review articles

1 *Calculated using average bulk density of 1.5 tonnes m^{-3} (den Biggelaar et al., 2003) and average erosion rate of 13.5 tonnes ha^{-1} yr^{-1} (mid-point of den Biggelaar et al., 2003 range)*

Author	Database Used	Extent	Estimates
Den Biggelaar *et al.* (2003)	Erosion: 179 plot-level studies Crop yield-erosion: 362	Global (37 countries)	Erosion: Average rates between 12 – 15 t ha^{-1} yr^{-1} (0.8 to 1.0 mm per year)[1] Relative annual crop yield reduction due to erosion: 0.3 percent per year (for six major crops)
Bakker, Govers and Rounsevell (2004)	Erosion-yield: 24 experimental studies	Primarily North America + Europe	Yield reductions of approximately 4 percent per 10 cm soil loss (= 0.36 percent per year)[1]
Scherr (2003)	28 regional studies and 54 national or sub-national studies on soil degradation (many GLASOD based, primarily soil erosion)	Global	Productivity losses since WWII of about 0.3 percent per year for cropland and 0.1—0.2 percent for pasture.
Crosson (2003)	Re-analysis of GLASOD and Dregne and Chou (1992)	Global	Cumulative loss of 5 percent of productivity on 4.7 billion ha of cropland and permanent pasture in 1945-1990 period; average annual rate of loss of 0.1 percent

The regional differences in crop response to erosion are, however, major. There are great disparities in the sensitivity of soils to erosion – soils with growth-limiting sub-soil layers (e.g. shallow soils over bedrock, soils with high sodium and/or dense B horizons) are inherently more susceptible to yield reductions due to soil loss (Bakker *et al.*, 2007). In a study modelling the impact of erosion in Europe over the next century, Bakker *et al.* (2007) predicted yield reductions on the order of 6 to 12 percent in southern Europe and reductions of only 0 to 1 percent in much of northern Europe. The overall impact on European food production is, however, relatively small as the yields from southern Europe are lower to begin with. In addition, increases in climatic extremes associated with human-induced climate change may lead to enhanced levels of wind and water erosion, but the impact of these changes will differ greatly among regions.

Finally, the crop yield/soil erosion relationship may be a less critical reason to reduce soil erosion than the off-site impacts of erosion, especially the transport of agricultural inputs such as N and P to waterways (Steffen *et al.*, 2015).

7.2.2 | Soil sealing

Soil sealing is most commonly associated with the expansion of urban areas and leads to a permanent, non-reversible loss of agricultural land. Yields are eliminated, not just reduced and the soil, if completely sealed, becomes effectively non-soil. Urbanization of agricultural land should thus be considered as a threat to future food production, not only for the loss of good quality agricultural land but also because of the risk of soil pollution through waste disposal and acid deposition from urban air pollution (Chen, 2007; Hubacek *et al.*, 2009; Clavero, Villero and Brotons, 2011). Blum and Nortcliff (2013) provide a very rough estimate of daily losses of soil due to sealing at the global scale of 250-300 km^2, and suggest this rate could increase due to continuing migration of rural dwellers to urban areas. Thus, new policies that favour sustainable rural development, oriented to avoid rural-urban migration as well as to support the return to rural areas of people living in the cities, could avoid soil degradation and promote food security.

7.2.3 | Soil contamination

Soil contamination reduces food security both by reducing yields of crops due to toxic levels of contaminants and by causing the crops that are produced to be unsafe to consume. As summarized in Chapter 6 (Section 6.3), there are worldwide tens of thousands of known contaminated sites due to local or point-source contamination. In regions with a long-standing industrial base, the expansion of contamination is limited, but in countries undergoing rapid industrialization or resource development the potential for the further spread of contamination is great. The tremendous expansion of industry in China is one example of this: 20 million ha of China's farmland (approximately one fifth of China's total farmland) is estimated to be contaminated by heavy metals, and this may lead to a significant reduction in food availability (see also Section 6.3 above). Contamination is also severe due to point sources such as Cs pollution from the Fukushima Dai-ichi nuclear power plant and the Chernobyl disaster of 1983. Diffuse soil contamination occurs in many regions (Blum and Nortcliff, 2013), but is more commonly linked with concerns about food safety rather than significant decreases in crop yields.

7.2.4 | Acidification

Acidification of agricultural soils is primarily associated with the net removal of base cations (e.g. product removal without replacement with ameliorants such as lime) and the direct addition of acidifying inputs (e.g. ammonium-based N fertilizer) to inherently low-pH soils, which have a low capacity to buffer added acidity. It is most prevalent on ancient, highly weathered soils. Acidification is a significant regional threat in countries such as Australia and Vietnam (see Chapters 10 and 15). Liming is an effective response to control acidity of surface horizons, but rates of lime addition lag behind required levels even in developed countries like Australia (SOE, 2011) and continuing loss of yield occurs.

7.2.5 | Salinization

Salinization in a soil progressively reduces crop yields; beyond a certain crop-specific threshold, growth of a given crop may fail entirely. The regional summaries in Chapters 9 to 16 illustrate how difficult it often is to separate the causes of salinization: whether the saline soils are naturally occurring (primary salinization) or the salinization has been caused by inappropriate management, which is often the case with poorly executed irrigation programmes (secondary salinization). Estimates from the 1990s place the land area affected by primary salinization at approximately one billion ha, and the area of land with secondary salinization at 77 million ha (Ghassemi, Jakeman and Nix, 1995).

Salinization is typically associated with arid and semi-arid areas, and may be exacerbated by climate change (see also Section 6.5). An increase in irrigated land is commonly suggested as a means to increase food production, but poorly designed and implemented irrigation schemes can readily cause an increase in salinization. Safe design and operation of irrigation systems requires a high level of managerial expertise. Irrigation expansion can contribute to increases in food production but great care is needed in planning and design to avoid negative effects such as salinization.

7.2.6 | Compaction

Compaction impairs soil functions by impeding root penetration and limiting water and gas exchange. In soils where it occurs, it can reduce crop yields but it rarely eliminates plant growth entirely. The susceptibility of different crops to compaction differs greatly (see also Section 6.9 above). Good soil management requires care in minimizing soil compaction and the adoption of management practices which alleviate existing compaction. The effect of soil compaction on output and hence on food security is, however, difficult to assess, especially in tropical areas (Lal, 2003).

7.2.7 | Nutrient imbalance

The problems associated with under-supply of nutrients in regions such as Sub-Saharan Africa will be discussed in Chapter 8 in the context of closing the yield gap. Foley *et al*. (2011) and Steffen *et al*. (2015) clearly indicates the regions where over-supply of nutrients is occurring: mid-west United States, western Europe, northern India, and the coastal areas of China. Foley *et al*. (2011) emphasize the need to address the economic and environmental issues in nutrient over-supply by increasing the efficiency of nutrient uptake by plants. This, coupled with reductions in transport of nutrients to waterways by minimizing erosion, would substantially reduce eutrophication. It would also allow the redistribution of N and P to areas of nutrient-poor soils without exceeding the planetary boundaries for the elements (Steffen *et al*., 2015).

7.2.8 | Changes to soil organic carbon and soil biodiversity

Soil organic carbon (SOC) and soil biodiversity are commonly linked to three dimensions of food security: increases in food availability, restoration of productivity in degraded soils, and the resilience of food production systems.

Soil C is not itself a direct control on food production but is a proxy for soil organic matter (SOM), which is one of the key attributes associated with many soil functions. Soil microbial C is normally included in aggregate measures of SOC, and soil microbes are a component of the soil organic matter; hence in terms of mass, SOC/SOM and soil microorganisms are directly related. The focus on SOC, rather than SOM, occurs because of the ease of measurement of C as a proxy for SOM, and because of the direct connection between SOC and atmospheric C.

The roles of SOC and soil biodiversity in increasing food availability are also inextricably bound together. Increases in SOC and in soil biodiversity are believed to be beneficial for crop production, and decreases in both are equally believed to be deleterious for crops; however providing evidence for these qualitative statements and establishing predictive relationships has been difficult (Naeem *et al*., 2009; Bommarco, Kleijn and Potts, 2013; Palm *et al*., 2014).

The more readily understood relationship between soil C storage and atmospheric C levels has driven much of the work in the past 15 years on soil carbon dynamics, but the secondary benefit of increasing SOC levels for crop production is commonly cited, if rarely quantified. Efforts to determine a threshold SOC value for maximum crop production in temperate soils have not been successful as it depends on management and on other factors such as soil limitations and precipitation (Loveland and Webb, 2003). Lal (2006) estimates yield gains associated with a 1 Mg ha^{-1} gain in SOC in the tropics and sub-tropics ranging from 20-70 kg ha^{-1} yr^{-1} for wheat to 30-300 kg ha^{-1} yr^{-1} for maize. However, the study acknowledges that the data are meagre and that functional relationships between SOC pool and crop yield are not available, especially for degraded soils in the tropics and subtropics.

Research in tropical and semi-tropical lands has established that inputs of organic material through the return of residues and manure to the soil are essential for fertility restoration in degraded soils, but that low residue production and competing uses for residues and manure limit the adoption of these SOC-aggrading approaches (e.g. Lal, 2006; Rusinamhodzi *et al.*, 2013; Palm *et al.* 2014). Sustainable soil management that increases SOM levels will assist in maintaining productivity, but the specific measures taken to increase SOM must be locally developed.

Establishing a direct, quantitative link between soil biodiversity and increasing food production is even more elusive. Sylvain and Wall (2011) observe that "the total invertebrates found in a soil will interact to provide many services and participate in several ecosystem functions, but it is unlikely that a single species will influence all services and functions that influence plant growth or composition at the same time or in the same manner". Biodiversity beyond the soil plays an important role in regulating services such as biological pest control and crop pollination (Bommarco, Kleijn and Potts, 2013), and public concerns about the effects of pesticides on key species continues to grow.

A final role for SOC enhancement and maintenance of soil biodiversity is to increase the resilience of the soil for food production, especially its ability to withstand disruption due to human-induced climate change. SOC buffers the impact of climate extremes on soils and crops by: (i) regulating water supply by reducing runoff and increasing soil-water holding capacity; (ii) reducing erosion through runoff reductions and improved aggregation; and (iii) providing sites for nutrient retention and release (Loveland and Webb, 2003; Lal, 2006). The combined role of soil organic matter and biodiversity in nutrient cycling ensures a continuing supply of nutrients for crop growth. It is difficult to quantify this relationship, especially in the light of the uncertainties associated with human-induced climate change, but the existing qualitative understanding is sufficient to establish the importance of SOC and biodiversity in sustainable soil management.

Summary

The importance of soil degradation and soil rehabilitation are highlighted in principles eight and nine of the proposed World Soil Charter:

> Soil degradation inherently reduces or eliminates soil functions and their ability to support ecosystem services essential for human well-being. Minimizing or eliminating significant soil degradation is essential to maintain the services provided by all soils and is substantially more cost-effective than rehabilitating soils after degradation has occurred.
>
> Soils that have experienced degradation can, in some cases, have their core functions and their contributions to ecosystem services restored through the application of appropriate rehabilitation techniques. This increases the area available for the provision of services without necessitating land use conversion.

Our ability to predict the effect of soil degradation on food security is very limited for two main reasons. First, there is a lack of up-to-date knowledge both on the area affected by degradation and on the linkages between degradation and soil functions (and ultimately plant production). The research community continues to cite research summaries on the effects of soil degradation on crop yields from the 1990s based on data gathered in the 1980s. Yet crop production in many regions has undergone profound change since the 1980s – for example, the widespread adoption of conservation tillage in many regions occurred during the 1990s and 2000s. There is a pressing need for meta-analyses on all of the soil threats discussed here. This in-depth review of existing work needs to be complemented by new research to address major information gaps, and in particular to prove more conclusively the functional relationships between soil attributes and plant production.

The second limitation to predictions is that farmers are not simply passive observers of inexorable degradation processes – farmers in all regions, including the tropics, are willing to invest in the future to protect soils and the essential services that they provide (Stocking, 2003). For example, the adoption of conservation tillage in heavily mechanized systems such as those in North America has substantially lowered erosion rates (Montgomery, 2007). The general applicability of conservation tillage in other regions may be limited (Palm *et al.*, 2014) but the principle that farmers are active participants in soil change is essential to recognize and encourage.

7.3 | Soil change and climate regulation

Soils play a fundamental role in the maintenance of a climate favourable to life. A range of soil processes helps regulate climate, including the thermal and moisture balance, greenhouse gases (H_2O, CO_2, CH_4 and N_2O) and particulates in the atmosphere. Soils can also adversely impact the maintenance of air quality.

7.3.1 | Soil carbon

Although it is hard to estimate quantities, it is certain that soils contain vast reserves of carbon. Recent estimates range between 1200 and 3000 Pg C depending on the depth to which estimates extend, and on the way in which wetland soils are counted (Hiederer and Köchy, 2012). Roughly 1670 Pg of C is stored in peatlands and permafrost soils in high northern latitudes (Tarnocai *et al.*, 2008). Hence soil organic matter is a large pool. Consequently, only small changes in soil C storage can have a large effect on atmospheric CO_2. Soils also contain approx. 950 PgC in the form of pedogenic carbonates to 2 m depth (Batjes, 1996).

Carbon respired from soils and derived from decomposition of organic matter in soils approximately balances annual net primary production of carbon by biomass. Carbon dioxide derived from plant roots and their symbionts below ground adds to the total flux of CO_2 from soils to the atmosphere, which in total is ~10 times larger than the current release of CO_2 to the atmosphere by fossil fuel burning (Schimel, 1995). Hence relatively small changes in the cycling of soil C can lead to large changes in atmospheric CO_2.

Management that changes C inputs or tillage that alters the stability of soil organic matter through changes in soil aeration or structure measurably alter soil C storage. Historically, the expansion of agriculture has led to losses of soil C to the atmosphere, estimated globally to be of order 40-90 PgC, some of which has remained in the atmosphere (Smith, 2004, 2012).

In terms of climate change, most projections suggest soil carbon changes driven by future climate change will range from small losses to moderate gains, but these global trends show considerable regional variation (Smith, 2012). The response of soil C in future will be determined by two factors: (i) the impacts of increased temperature and altered soil moisture on decomposition rates; and (ii) the balance between increases in C losses resulting from accelerated decomposition and predicted C gains through enhanced productivity under elevated CO_2 and nutrient deposition (Smith, 2012).

Soil organic matter (SOM) is considered dynamic and has importance beyond its climate role. Plant residues added to soils provide energy for a cascade of heterotrophic organisms. A key outcome of organic matter (OM) breakdown is the release of essential nutrients into the soil. If the breakdown of OM exceeds the supply of O_2, e.g. under high moisture conditions, the degradation of OM using other electron acceptors drives the production and consumption of other important greenhouse gases such as methane and nitrous oxide. The degradation of OM also indirectly affects greenhouse gases like troposphere ozone by altering the emission of reactive trace gases. In addition to climate effects through regulation of greenhouse gases, SOM determines properties such as nutrient retention, water retention, and the structure and size of the microbial community in soils.

SOM feedbacks to climate change (Figure 7.6) include direct responses such as: (i) the alteration of microbial activity with temperature (Conant *et al.*, 2011); and (ii) moisture-related changes in the supply of O_2 relative to other electron acceptors that reflect precipitation change. In this context, probably the biggest concern is the thawing of large stores of C in permafrost at high northern latitudes, which will make organic C that has been frozen for millennia available for decomposition. This response is predicted to create a significant positive feedback to climate change (Schuur *et al.*, 2008).

Another direct response of soil organic carbon pools is predicted in response to elevated CO_2 and its effect on ecosystem productivity. Free Air Carbon Dioxide Enrichment (FACE) studies have shown that belowground productivity can be strongly affected, with cascading and mixed consequences for SOM storage. However indirect effects are such as altered stabilization of older C associated with the increased inputs of fresh plant inputs ('priming') add uncertainty to the prediction of future soil C responses. As with CO_2 fertilization, increased deposition of reactive N associated with regional air pollution affects production, quality and spatial distribution of plant inputs (e.g. above- versus belowground) and can alter the decomposition rates through changes in the soil microbial community (Berg and Matzner, 1997). Hence the net effects on soil C storage are difficult to predict, though the combined effects of climate change and fertilization are expected to result in net losses of soil C overall from temperate forest soils (Hopkins, Torn and Trumbore, 2012)

Many of the processes affecting SOM over the past century have been dominated by human management of vegetation, which in turn affects the inputs and status of SOM. Changes in vegetation cover, including those occurring in response to climate as well as to land use or management, influence soil organic matter by altering the rates, quality and location of plant litter inputs to soils. In turn, litter inputs influence the amount and composition of the decomposer organisms, including soil fauna, as well as the soil microbial community. Studies of a number of vegetation transitions – for example the replacement of forests with agriculture or pasture – have shown that these transitions have led to a loss of soil C to the atmosphere. However, the trajectory of vegetation change in response to climate, and the consequences for atmospheric CO_2, are not well known, as soils will in turn determine what kind of vegetation will take over. For example, C from thawing permafrost soils may eventually be sequestered in the biomass of forests that can grow in the warmer climate. Evidently the time lags required for these transitions are an important part of understanding the net effect of soil C on the carbon cycle.

In addition to direct effects of changes in plant litter addition to soils, management or vegetation change also alters the chemical and physical framework of soil and thereby the organisms inhabiting it. For example, ploughing can break up soil aggregates and make organic matter that was previously protected available to decomposers. Changes in evapotranspiration can change local and regional water resources. Addition of fertilizers increases plant productivity but also alters soil microbial communities and can stimulate production of reactive N gases and N_2O.

Large-scale soil erosion is thought to slow decomposition of buried, eroded organic matter, while growth of vegetation on the remaining soil will tend to increase soil C storage. However, these effects have been shown to be relatively small (Van Oost *et al.*, 2007). By removing topsoil that is generally high in organic matter, erosion can have profound effects on physical and chemical soil properties such as water retention and cation exchange capacity.

Global increases in carbon stocks have a large, cost-competitive potential for climate change mitigation (Smith *et al.*, 2008). Mechanisms include reduced soil disturbance, improved rotations and residue/organic input management, and restoration of degraded soils. Nevertheless, limitations on soil C sequestration include time limitation, non-permanence, displacement and difficulties in verification (Smith, 2012). Despite these limitations, soil C sequestration can be useful to meet short- to medium-term targets. In addition, soil C sequestration confers a number of co-benefits on soils. It is thus a viable option for reducing the atmospheric

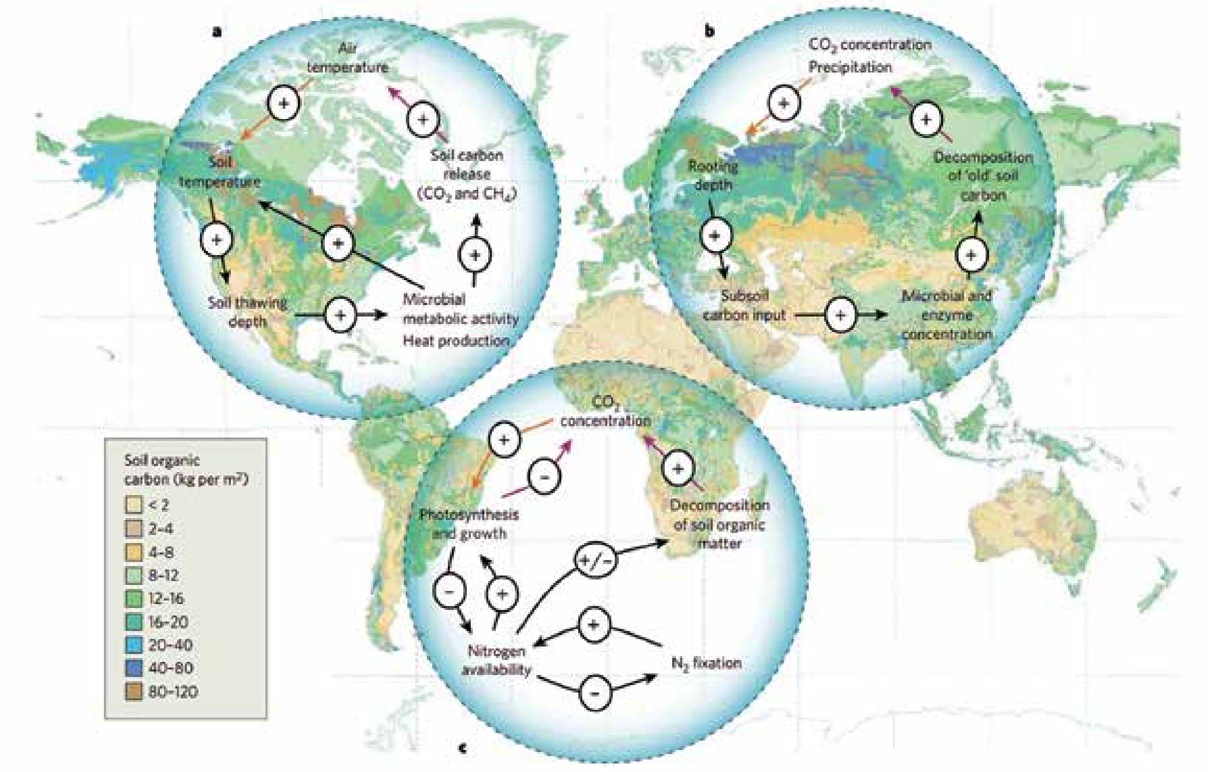


Figure 7.6 | Some soil-related feedbacks to global climate change to illustrate the complexity and potential number of response pathways. Source: Heimann and Reichstein, 2008.

CO_2 concentration in the shorter term, buying time to develop longer term emission reduction solutions across all sectors of the global economy (Smith, 2012).

Just as reductions in soil C stocks are associated with negative consequences for soil function, increased soil carbon stocks are associated with increased soil fertility, workability, water holding capacity, reductions in greenhouse gas emissions and reduced erosion risk (Lal, 2004). Increasing soil carbon stocks can thus reduce the vulnerability of managed soils to future global warming (Smith and Olesen, 2010). Management practices effective in increasing SOC stocks include: (i) improved plant productivity through nutrient management, rotations and improved farming practices; (ii) reduced or conservation tillage and residue management; (iii) more effective use of organic amendments; (iv) land use change, for example from crops to grass or trees; (v) set-aside; (vi) agroforestry; (vii) optimizing livestock densities; and (viii) planting legumes or improving the crop mix (Smith *et al.*, 2008). While these measures have the technical potential to increase SOC stocks by about 1 – 1.3 Pg C yr^{-1} (Smith *et al.*, 2007a, 2008), they are dependent on economics: the economic potential for SOC sequestration was estimated to be 0.4, 0.6 and 0.7 Pg C yr^{-1} at carbon prices of up to US$20, $50 and $100 per tonnes CO_2-eq. yr–1, respectively (Smith *et al.*, 2008). In addition, the size of the potential sequestration is relatively small in comparison to the threats: only a small loss of C from permafrost or peatlands could offset this potential sequestration (Joosten *et al.*, 2014). However, an increase in SOC through improved management is expected to also reduce vulnerability of the soils to future SOC loss under global warming. As such, soil carbon sequestration can, in many respects, be regarded as a 'win-win' and a 'no regrets' option (Smith *et al.*, 2007b).

7.3.2 | Nitrous oxide emissions

Soils emit nitrous oxide (N_2O), a greenhouse gas that is around 300 times more potent for radiative forcing (climate warming) over 100 years than CO_2. Of the approximately 16 Tg N_2O-N yr^{-1} emitted globally in the 1990s, between 40 and 50 percent was a result of human activities (Reay *et al.*, 2012). The main sources were agriculture, industry, biomass burning and indirect emissions from reactive nitrogen, such as leaching, runoff and atmospheric deposition (Reay *et al.*, 2012). Of these sources, agricultural soils are the dominant source, contributing over 80 percent of global anthropogenic N_2O emissions during the 1990s (Smith *et al.*, 2007a). N_2O emissions from agricultural soils have increased from just under 4 Tg N_2O-N yr^{-1} in 1990, to over 4 Tg N_2O-N yr^{-1} in 2010. Emissions are projected to increase to over 5 Tg N_2O-N yr^{-1} by 2030 (Reay *et al.*, 2012).

Nitrous oxide is emitted from soils through two processes, nitrification and denitrification. Any mineral N available in the soil is subject to loss through one of these processes. The processes depend on soil environmental conditions such as the availability of mineral N, soil temperature and soil water content, soil pH, organic matter content and soil type. Nitrification tends to be favoured under aerobic conditions and denitrification under anaerobic conditions (Galloway *et al.*, 2003). Subject to mineral N being available, any soil can emit N_2O through mineralisation of soil organic matter. However, the majority of emissions are driven by sources of N added to the soil as fertiliser, either as synthetic fertilizer, or as organic amendments (e.g. manures, slurries, composts). So close is the relationship between N addition and emission, that N_2O emissions are often calculated as a direct function of N added to the soil (Reay *et al.*, 2012). Emissions of N_2O from agricultural soils driven by addition of synthetic fertilizers have increased from 67 $MtCO_2$-eq. yr^{-1} in 1961, to 683 $MtCO_2$-eq. yr^{-1} in 2010 (Tubiello *et al.*, 2013).

Given the close association between N inputs and N_2O emissions, soil management strategies to reduce N_2O emissions, and thereby improve this aspect of their climate regulation function, are mostly centred on removing surplus N in the soil. This is mainly accomplished by improving N-use efficiency to reduce the N surplus, either by reducing inputs or by better matching applications (timing and amount) to plant demand (Snyder *et al.*, 2014). In a recent review, Snyder *et al.* (2014) noted that soil N_2O emissions can be reduced by selecting the right source, rate, time and place of N application and that new technologies and greater farmer/adviser skills can improve N input management. They estimate that crop N recovery could be increased by >20 percent, reducing risks of N_2O emissions by >20–30 percent (Snyder *et al.*, 2014).

Beyond these technical measures, N_2O emissions could also be reduced through demand-side management, for example through reduced food waste. Another demand-side measure could be to encourage dietary change away from less efficiently produced food products such as meat and other livestock products, or foods with very high energy inputs, such as heated glasshouses during winter (Reay *et al.*, 2012).

In summary, managed soils can play a key role in climate regulation via N_2O emissions, and a number of options exist to improve the soil's delivery of its climate regulation service both by enhanced N management and by wider systemic changes in agriculture (Flynn and Smith, 2010; Reay *et al.*, 2012; Snyder *et al.*, 2014).

7.3.3 | Methane emissions

Methane (CH_4) is a greenhouse gas that is around 20–35 times more potent for radiative forcing (climate warming) over 100 years than CO_2. Soils often emit methane through methanogenesis when decomposition of organic matter occurs in anaerobic soil layers. Methane is also oxidised by methanotrophy in aerobic layers, so the emission is a balance between methanogenesis and methanotrophy (Le Mer and Roger, 2001). About 30 percent of total global CH_4 emissions are natural (including the natural wetland flux), and around 70 percent anthropogenic (Le Mer and Roger, 2001). Given that methanogenesis occurs under anaerobic conditions, waterlogged soils, particularly wetlands, peatlands and rice paddies, are the largest source of methane emissions (Le Mer and Roger, 2001). Since much of the methane flux from wetland and peatland soils occurs on largely unmanaged areas, the emissions are not considered anthropogenic, so are not routinely included in greenhouse gas inventories. This means that the quantification of soil methane emissions over time from peatlands and wetlands is not as well documented as for N_2O. Nonetheless, some global estimates of CH_4 emissions from wetlands do exist: in 1998, total global emissions of CH_4 from wetlands were estimated to be 145 Tg yr^{-1}, of which 92 Tg yr^{-1} came from natural wetlands and 53 Tg yr^{-1} from rice paddies (Cao, Gregson and Marshall, 1998), with some estimates a little higher (Le Mer and Roger, 2001). Emissions from rice paddies, however, are included in inventories: CH_4 emissions from rice paddies were estimated to have increased from 366 $MtCO_2$-eq. yr^{-1} in 1961 to 499 $MtCO_2$-eq. yr^{-1} in 2010 (Tubiello *et al.*, 2013).

By contrast, aerobic soils tend to act as sinks for CH_4, thereby having a positive impact on climate regulation. Temperate and tropical aerobic soils that are exposed to atmospheric concentrations of CH_4 usually exhibit low levels of atmospheric CH_4 oxidation but, since they cover large areas, they are estimated to consume ~10 percent of the atmospheric CH_4 (Le Mer and Roger, 2001). Forest soils are the strongest CH_4 sink, followed by grasslands, with the sink capacity of cultivated land much lower than that of undisturbed soils (Steudler *et al.*, 1996; Priemé *et al.*, 1997). Atmospheric CH_4 oxidation also occurs in extreme environments such as deserts and glaciers, in the floodwater of submerged soils and in river waters (Le Mer and Roger, 2001). Potter, Davidson and Verchot (1996) estimated global soil CH_4 consumption to be 17–23 Tg yr^{-1}.

Soil management strategies to reduce CH_4 emissions or enhance CH_4 uptake can improve this aspect of the soil's climate regulation function. Enhancing uptake in managed soils is difficult, so most mitigation options occur for CH_4 emission reduction, and since wetlands or often unmanaged, most mitigation options have been developed for rice paddies. These include draining the wetland rice once or several times during the growing season, selection of rice cultivars with low exudation rates, off-rice season water management, fertilizer management and the timing and composting of organic residue additions (Smith *et al.*, 2008). For managed peatlands and wetlands (e.g. those used for forestry or agriculture), methane emissions can be reduced by fertilizer, water and tillage management (Le Mer and Roger, 2001). Rewetting of drained or cultivated peatlands to restore wetland function and maintain carbon stocks is likely to increase CH_4 emissions, but the overall impact on climate will vary between systems and depending on the time horizon considered (Joosten *et al.*, 2014).

7.3.4 | Heat and moisture transfer

Soils play an essential role in storage of water. Soil moisture strongly affects water, energy and carbon exchanges, leading to major forcings and feedbacks within the climate system (Seneviratne *et al.*, 2010). Soil moisture generally refers to the amount of water stored in the unsaturated soil zone. The most important soil moisture storage is that affecting plant transpiration, e.g. the water available within the root zone. Land evapotranspiration is an essential component of the continental water cycle, since it returns as much as 60 percent of precipitated land water back to the atmosphere (e.g. Dirmeyer *et al.*, 2006; Oki and Kanae, 2006; van der Ent *et al.*, 2010). Soil moisture is the main water source for this process, through plant transpiration and bare soil evaporation. Plant transpiration contributes about 60 percent of all land evapotranspiration (Schlesinger and Jasechko, 2014).

Evapotranspiration is itself a function of soil moisture (Koster *et al.*, 2004; Seneviratne *et al.*, 2010). This dependency is conceptually illustrated in Figure 7.7, which builds upon the classical Budyko framework (Budyko, 1956, 1974). It shows that three main soil moisture regimes can be distinguished: (i) a wet soil moisture regime in which evapotranspiration is solely limited by the availability of energy; (ii) a transitional soil moisture regime in which evapotranspiration is strongly sensitive to the availability of soil moisture; and (iii) a dry soil moisture regime in which soil moisture is at or below the wilting point and for which evapotranspiration is negligible.

The geographical distribution of these soil moisture and evapotranspiration regimes can be estimated with various methods, as discussed in Seneviratne *et al.* (2010). As an illustration, Figure 7.8 displays the correlation of annual mean evapotranspiration with radiation and precipitation in an observation-driven land surface model using a two-dimensional colour map. This analysis illustrates the existence of distinct evapotranspiration regimes, with most regions clearly displaying either the characteristics of a soil moisture- or energy-limited evapotranspiration regime.

One should note that the relationship displayed in Figure 7.7 is qualitative, and is affected (both in space and time) by variations in soil parameters, land cover characteristics, and other factors (e.g. Teuling *et al.*, 2010; Koster and Mahanama, 2012; Guillod *et al.*, 2013).

The water and energy balances of land are tightly connected through the process of evapotranspiration. It follows that the soil moisture effects on evapotranspiration (illustrated in Figure 7.8) are also highly relevant for land energy exchanges at the land surface. This link makes soil moisture a strong control of temperature variability and temperature extremes on land (e.g. Seneviratne *et al.*, 2006; Fischer *et al.*, 2007; Vautard *et al.*, 2007; Mueller and Seneviratne, 2012). Modelling estimates suggest that soil moisture feedbacks affect about 60 percent of temperature variability in the present Mediterranean climate in summer (Seneviratne *et al.*, 2006) and that they induced additional temperature anomalies of the order of 2°C in Central Europe during the 2003 European summer heat wave (Fischer *et al.*, 2007). Observation-based analyses also confirm the existence of strong correlations between the occurrence of hot extremes in regional hottest months and prior precipitation deficits in regions with soil moisture-limited evapotranspiration regimes (Hirschi *et al.*, 2011; Quesada *et al.*, 2012; Mueller and Seneviratne, 2012). The example of the European summer heat wave shows, moreover, that these feedbacks can be relevant in extreme years even in regions like Central Europe which have a dominant energy-limited evapotranspiration regime under the present climate.

For present climate conditions, the relationship between soil moisture deficits and hot extremes implies that information on soil moisture deficits could be used for improved forecasting of temperature mean and

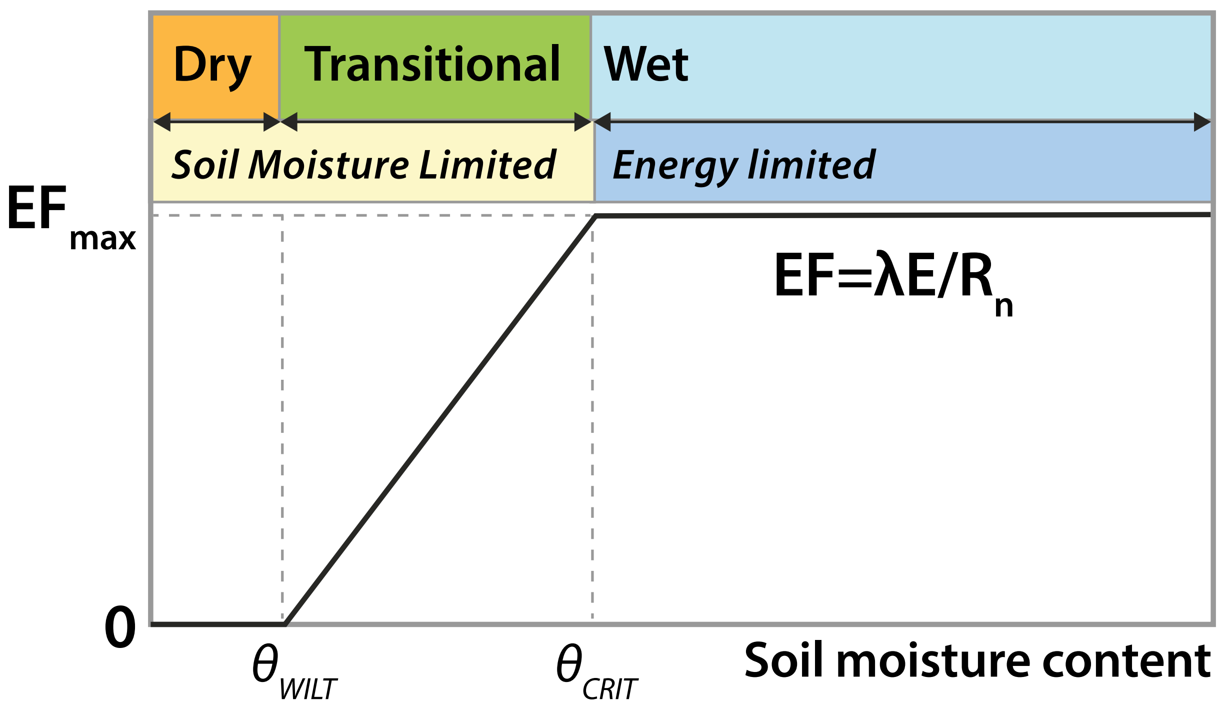


Figure 7.7 | Definition of soil moisture regimes and corresponding evapotranspiration regimes.
Source: Seneviratne *et al.*, 2010.

EF denotes the evaporative fraction, and EFmax its maximal value.

extremes several weeks in advance (e.g. Koster *et al.*, 2010a; Mueller and Seneviratne, 2010). Such early soil moisture information can be either provided by an offline land surface model driven with observation-based forcing (e.g. Dirmeyer *et al.*, 2006), by remote sensing products (e.g. Wagner *et al.*, 2007; De Jeu *et al.*, 2008), or by the assimilation of remote sensing products in land surface models (e.g. Reichle, 2008). However, the scarcity of precipitation and soil moisture observations still limits the derivation of reliable soil moisture estimates and the evaluation of satellite approaches on most continents (e.g. Koster *et al.*, 2010b; Dorigo *et al.*, 2013).

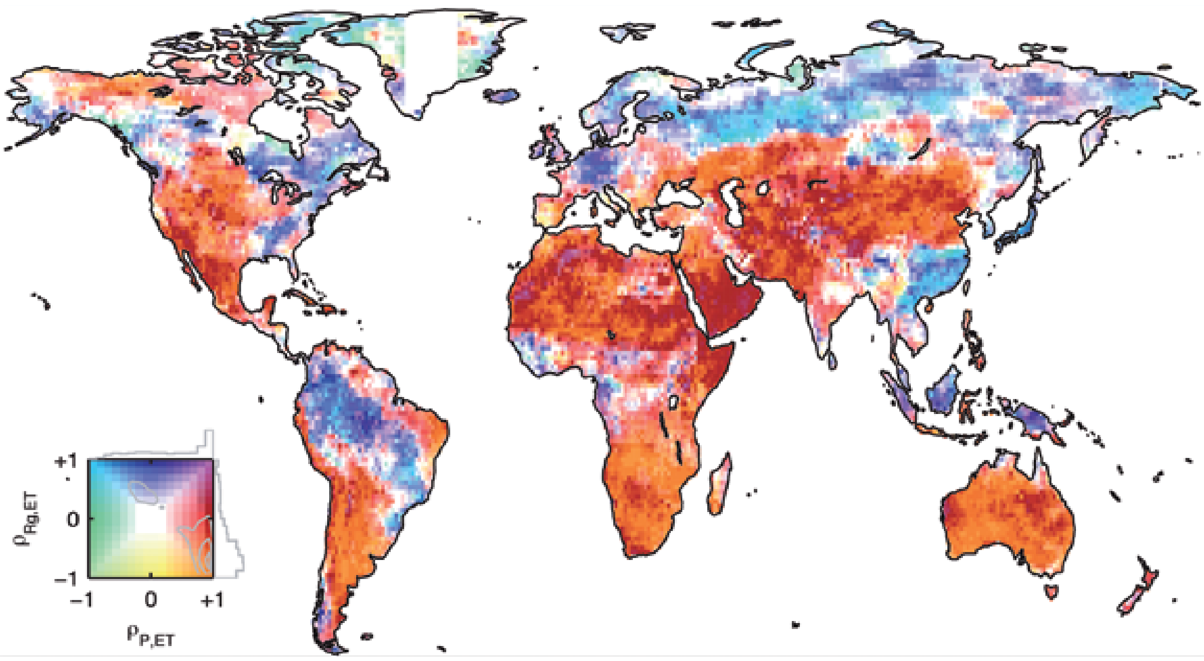


Figure 7.8 | Estimation of evapotranspiration drivers (moisture and radiation) based on observation-driven land surface model simulation.
Source: Seneviratne et al., 2010.

The figure displays yearly correlations of evapotranspiration with global radiation Rg and precipitation P in simulations from the 2nd phase of the Global Soil Wetness Project (GSWP, Dirmeyer et al., 2006) using a two-dimensional color map, based on Teuling et al. 2009, redrawn for the whole globe. (Seneviratne et al., 2010)

Climate models project that several regions will be affected by more frequent drought conditions in the future as a consequence of enhanced greenhouse gas concentrations (e.g. Wang, 2005; Sheffield and Wood, 2007; Seneviratne *et al.*, 2012). This implies shifts in climate and soil moisture regimes, with important impacts on temperature projections (e.g. Seneviratne *et al.*, 2006; Dirmeyer *et al.*, 2012), in particular for temperature extremes (Seneviratne *et al.*, 2013).

Another feedback of soil moisture on climate is the possible impact of droughts on plant carbon uptake and a resulting decreased sink for CO_2 emissions (Ciais *et al.*, 2005; Friedlingstein *et al.*, 2006; Sitch *et al.*, 2008; Reichstein *et al.*, 2013). One particularly important region for this feedback is the Amazon rainforest, which is projected in some models to dry substantially (e.g. Mahli *et al.*, 2008). However, these projections are associated with high uncertainty in current climate models (Orlowsky and Seneviratne, 2013), and the resulting effects on carbon uptake could also be affected by the representation of plant physiology in the land surface schemes (Huntingford *et al.*, 2012).

Finally, the combined effects of soil moisture on near-surface humidity and temperature are also relevant for boundary layer development and precipitation occurrence (e.g. Betts, 2004; Koster *et al.*, 2004; Taylor *et al.*, 2012). More details on these feedbacks are provided in Sections 7.5 and 7.6 below.

According to the World Health Organisation[1], air pollution is "contamination of the indoor or outdoor environment by any chemical, physical or biological agent that modifies the natural characteristics of the atmosphere". The status of air pollution is often referred to as air quality (Monks *et al.*, 2009). Air quality affects human health through exposure to toxic inorganic compounds (e.g. HBr, elemental Hg vapour), toxic organic compounds (e.g. organic pesticides), and particulate matter (PM). Air quality also affects the climate system through changes in greenhouse gas concentrations (CO_2, CH_4, N_2O) – as discussed in Section 7.3 – and through aerosols (e.g. mineral particles, black carbon or 'BC'). After deposition of atmospheric pollutants (e.g. N and S compounds, or compounds containing trace elements) on land or water, acidification, eutrophication, and contamination might occur (see Section 4.4), which can have harmful effects on ecosystem function and structure, particularly where deposition exceeds the 'critical load' that a particular soil can buffer (Nilsson and Grennfelt, 1988). Specific compounds in the atmosphere, such as ammonia (NH_3), can result in a host of environmental problems (e.g. impacts on human health, odour, climate change, soil acidification, eutrophication, biodiversity). The magnitude of the problems would depend on interactions with other compounds (Aneja, Schlesinger and Erisman, 2009).

7.4.2 | Ammonia emissions

Agriculture accounts for 80–99 percent of all NH_3 emissions (FAO, 2014). In Europe, agriculture accounts for 94 percent (EEA, 2012). These emissions mainly come from animal manure and fertiliser application (Olivier *et al.*, 1998). In the United States, NH_3 reductions are voluntary and there are neither federal nor national regulations controlling its emission (Aneja, Schlesinger and Erisman, 2009; Greaver *et al.*, 2012). In Europe, however, NH_3 emissions have been an important policy issue (van der Hoek, 1998) and regulation has led to an overall reduction in NH_3 emissions. Between 1990 and 2010, NH_3 emissions decreased in the EU-27 by 28 percent (EEA, 2012), with especially large reductions in Poland, the Netherlands and Germany. Ammonia emission reductions have been associated with a reduction in the number of livestock (especially cattle), improvement of manure management, and the lower input of nitrogenous fertilisers to soils (EEA, 2011, 2012). The effectiveness of manure injection to decrease emissions is under debate, as a result of its effect on pollutant swapping, as there may be a reduction in NH_3 but an increase in N_2O emissions and/or NO_3 leaching (Erisman *et al.*, 2008). A better understanding is needed on the contribution of NH_3 as a precursor of PM concentrations, both emissions of primary PM 10 (particulate matter with a size < 10 µm) and secondary formation of PM 2.5 (particulate matter with a size < 2.5 µm) (Aneja, Schlesinger and Erisman, 2009). It is worth mentioning that interactions occur with other compounds in the atmosphere, to the extent that reductions in SO_2 and NO_x are only effective in the reduction of PM 2.5 if carried out simultaneously with NH_3 reductions (Erisman and Schaap, 2004).

7.4.3 | Aerosols

Mineral dust, sulphate aerosols, and organic C and black C (BC) aerosols from fossil fuel and biomass burning have a significant effect on radiative forcing (Forster *et al.*, 2007). Mineral dust is mainly emitted from deep and extensive alluvial flood deposits emplaced during the Pleistocene, for example in the Sahara, East Asia, the Arabian deserts, and Central Australia (Prospero *et al.*, 2002). The largest sources are located in the Northern Hemisphere, in the so-called 'global dust belt' that extends from the west coast of North Africa, through the Middle East, into Central Asia. Outside this belt, areas with remarkable persistent dust activity include the Great Basin in south-western North America, the Lake Eyre Basinin Australia, some areas of South America (predominantly in Argentina), and southern Africa (Prospero *et al.*, 2002) (Figure 6.2). The 'Red Dawn' dust storm that affected Sydney, Australia in September 2009 is described in Chapter 15. Mineral dust originating in the Sahel has been reported to be regularly carried over large areas of the Atlantic and the Caribbean; the

1 http://www.who.int/topics/air_pollution/en/

largest export occurs during years of low rainfall in the source region (Prospero and Lamb, 2003). Although this process might have been exacerbated by anthropogenic activities (Prospero and Nees, 1978), recent evidence indicates that vegetation cover in the region has not changed substantially in the past 20 years and that, on a global scale, dust mobilisation is probably mostly driven by natural events (Prospero *et al.*, 2002).

The direct effect of aerosols on the climatic system is mainly through the reflection and absorption of solar radiation (Miller and Tegen, 1998). The indirect effect involves the modification of cloud properties (Kaufman, Tanre and Boucher, 2002). Greenhouse gases, in contrast, reduce the outgoing thermal radiation to space. Differences in lifetime and spatial distribution between greenhouse gases and aerosols are also considerable: greenhouse gases have a lifetime of more than 100 years and a homogeneous distribution (Forster *et al.*, 2007), whereas aerosols have a lifetime of about a week and a rather heterogeneous distribution (Andreae *et al.*, 1986). Soil dust aerosols have also been reported to modify the lifetime of some greenhouse gases (Dentener *et al.*, 1996). They also provide essential nutrients to ocean ecosystems that may increase the efficiency of the ocean's biological pump and help sequester CO_2 in the deep ocean (Martin, 1990). This is specially the case of iron, which is an important micronutrient for phytoplankton (Falkowski, Barber and Smetacek, 1998).

Most aerosols are highly reflective, thus raising the albedo of our planet and having a cooling effect. However, aerosols containing BC are dark and strongly absorb the incoming sunlight (Kaufman, Tanre and Boucher, 2002). This warms the atmosphere and cools the Earth's surface before a redistribution of the energy occurs in the atmosphere column (Ramanathan and Carmichael, 2008). Black C alters the radiative forcing through different processes: (i) the presence of BC in the atmosphere above surfaces with high albedo such as snow or clouds may cause a significant positive radiative forcing (Ramaswamy *et al.*, 2001); (ii) BC aerosols deposited on snow may promote melting (Warren and Wiscombe, 1980; Hansen and Nazarenko, 2004); and (iii) BC influences evaporation and cloud formation by modifying the atmosphere's vertical temperature gradient (Ackerman *et al.*, 2000; Raufman and Fraser, 1997). However, the exact radiative forcing depends on how BC is mixed with other aerosol constituents (Jacobson, 2001).

Carbonaceous aerosol emission inventories suggest that approximately 34-38 percent of these emissions come from biomass burning sources, the remainder from fossil fuel burning sources (Forster *et al.*, 2007). Fossil-fuel-dominated BC emissions are approximately 100 percent more efficient warming agents than biomass-burning-dominated plumes (Ramana *et al.*, 2010). The type of smoke is also largely influenced by the type of biomass being burned (Takemura *et al.*, 2002). In savannah ecosystems, about 85 percent of the biomass (mostly grasses) is consumed by flaming during fire events. In forest fires this value decreases to 50 percent or less, as the flaming stage is followed by a long, cooler smouldering stage in which the thicker wood, not completely consumed, emits smoke composed of organic particles without BC (Takemura *et al.*, 2002). Black C is thus mostly emitted during the hot, flaming stage of the fire (Kaufman *et al.*, 2002). The intense surface heating caused by fires can further cause a rapid uplift of heated air, known as pyro-convection, which can considerably disturb the chemical conditions in the free and upper troposphere and, in some cases, in the stratosphere (Monks *et al.*, 2009). Aerosols from fires are more likely to be injected at higher altitudes and are likely to experience long-range transport. Aerosol emissions from large boreal fires in Alaska and Russia have been shown to be transported very efficiently over long distances (Damoah *et al.*, 2006; Petzold *et al.*, 2007).

7.5 | Soil change and water quality regulation

Soils provide a biogeochemically activated filtration and cleaning service that transforms or retains materials deposited at the land surface. These materials include not only nitrogen and phosphorous, elements from grey water used for irrigation, and acidic compounds, but also inorganic and organic toxins. If the capacity of the soil to retain, transform or filter these materials is exceeded, there can be severe environmental consequences for water quality. Soils also adversely impact the provision of clean water through erosion into water courses, through salinization and through redox cycling and the release of metals such as arsenic.

7.5.1 | Nitrogen and phosphorous retention and transformation

By increasing fertilizer production and crop N fixation, human activities have doubled nitrogen (N) fixation from the atmosphere during the last century. Half (210 Tg N yr^{-1}) of global nitrogen fixation (413 Tg N yr^{-1}) is human-driven (Fowler *et al.*, 2013). Mining and erosion have increased the phosphorus (P) flow from land into the ocean by at least ten-fold (preindustrial value of 1 to current estimate of 9-32 Tg P yr^{-1}; Carpenter and Bennett, 2011). A recent inventory indicates that approximately 60 percent of the nitrogen fixed by human activities is released back into the environment without being incorporated into food or products (Houlton *et al.*, 2013). Increases in the release of reactive nitrogen (N) and phosphorus (P) to the environment are associated with many significant environmental concerns, including surface water contamination, harmful algal blooms, hypoxia, air pollution, nitrogen saturation in forests, drinking water contamination, stratospheric ozone depletion and climate change (Bennett, Carpenter and Caraco, 2001; Sutton *et al.*, 2011; Davidson *et al.*, 2012).

Soils serve as an important regulator of the leakage of this anthropogenic N and P back into the air or to surface and ground water, since much of the release occurs from fertilizers or atmospheric deposition. Soil is the largest pool of N and P within terrestrial ecosystems (Cole and Rapp, 1981), illustrating the magnitude and stability of soil N and P storage. Review of 15N tracer studies reinforces that idea that soils are the strongest sink for nitrogen in the short and medium term (Fenn *et al.*, 1998; Templer *et al.*, 2012). Flows of N through the landscape and the consequences of excess N can be represented by the N cascade (Galloway *et al.*, 2003). Nitrogen and phosphorus removal occurs through plant or microbial uptake, storage in soil organic matter, by complexation, and sorption or exchange. Nitrogen is cycled biologically through plant uptake, litterfall and microbial cycling, and is stored in organic forms except in areas with substantial rock-derived N (Morford, Houlton and Dahlgren, 2011). By contrast, soil P is mainly found in an inorganic form, sorbed or complexed by soil minerals and the exchanger. Organic P is a smaller pool in most soils, found in a review of global soil P to range from 5-40 percent (Yang and Post, 2011). For N, there are also significant gaseous losses via NO_x or NH_3 and through denitrification as N_2 or N_2O. Storage in soils or perennial plants and conversion into other inert forms (N_2 for N or stable inorganic complexes for P) represent stable sinks that remove N and P from flowpaths and the N cascade for a period of time determined by the residence time of those sinks.

An important service provided by soils is to remove N and P along flowpaths, preventing mobile nitrate and phosphate from moving from terrestrial ecosystems into surface waters and groundwater. Global models indicate that soils are responsible for the largest portion of landscape N removal - 22 percent of global N removal as denitrification - second only to coastal ocean sediments (Seitzinger *et al.*, 2006). Riparian soils or wetlands can remove N that has leaked from forests, farms, rangelands or the built environment (Peterjohn and Correll, 1984), as long as riparian zones are downgradient of the N source (Weller and Baker, 2014). One study indicates that replacement of 10 percent of historical riparian buffers could substantially reduce N loading to the Gulf of Mexico (Mitsch *et al.*, 2001).

Phosphorus cycling has important distinctions from N cycling. In particular, the dominant inorganic form of phosphorus, orthophosphate, binds strongly to soil particles via sorption or complexation as inorganic P, in contrast to nitrate, which is quite mobile. Phosphorus can be displaced under reducing conditions, and thus efforts to target N removal may in fact cause unanticipated increases in dissolved P concentrations (Ardón *et al.*, 2010). While we do not have a parallel conceptual P cascade, P availability can drive the formation of harmful algal blooms, and recent work indicates that joint management of N and P is critical (Conley *et al.*, 2009). In efforts to reduce effects on ecosystems and water quality, it is important to consider the soil processes involved in removal of both elements and their interactions.

Perturbations that increase the mobility of N and P may saturate the retention capacity of soils such that the ability to remove these elements declines as inputs increase. Disturbances that affect soil structure, rooting patterns and organic matter also decrease N and P retention capacity. At the ecosystem scale, N removal capacity declines as N loads increase above a point where N can be taken up by plants and soil processes (Aber *et al.*, 1989). While studies illustrate that the rate of N removal does generally decline with increasing N inputs (e.g. Perakis, Compton and Hedin, 2005), there are still questions about the ability of soils to retain N over time. The saturation point may vary by ecosystem and soil type. For example, wetland ecosystems have a tremendous capacity to retain N – a recent meta-analysis indicates that wetland N removal is linear with N loading, removing about 47 percent of N inputs even at very high loads (Jordan, Stoffer and Nestlerode, 2011). However, recent work on agricultural soils found that N_2O production increases with increased N loading (Shcherbak, Millar and Robertson, 2014). This reinforces the pattern of decline in capacity of soils to serve as a stable N sink under high N inputs, and suggests that efforts to reduce N_2O production should target areas of high N loads where larger benefits will be seen per unit N.

The connection between ecosystem services and soil processes is sometimes distant. The benefit of N uptake in a riparian soil in Iowa might be most appreciated in distant coastal fisheries. In addition, ecosystem services do not turn on or off with the flick of a switch; for example, it may take decades to recover water quality after a widespread land use change (Hart, 2003; Howden *et al.*, 2010). Our perspectives about soils and ecosystem services should include these distant connections and time lags.

Removal of N from the cascade has implications for many aspects of human health and well-being (Figure 1; Brauman *et al.*, 2007; Compton *et al.*, 2011), and an increasing number of studies are including soil processes in ecosystem service assessments and valuation frameworks (De Groot, Wilson and Boumans, 2002; Robinson *et al.*, 2013). Soil N and P removal is generally seen as an intermediate service or a supporting or regulating service in current ecosystem services classification schemes, as it affects a number of final ecosystem goods and services (Boyd and Banzhaf, 2007).

Impacts of nitrogen on ecosystem services (ES), on the economy and on human well-being have been examined in a number of studies (Birch *et al.*, 2010; Compton *et al.*, 2011; van Grinsven *et al.*, 2013). Soil N and P storage could have implications for many benefits, including the following: (i) avoidance of consequences to ecosystem services provided by freshwater, groundwater and coastal waters from reduced quality for swimming, drinking, recreation or fishing; (ii) avoidance of air quality problems associated with N such as those affecting human respiratory health or visibility (NO_x, NHy); (iii) avoidance of damage from climate change and stratospheric ozone depletion (N_2O); and (iv) maintenance of soil fertility and ecosystem production (both N and P). Eutrophication of coastal areas and associated hypoxia can result in physiological and behavioural impacts on important coastal organisms, populations and ecosystems that result in lowered fitness and productivity. However, there is a good deal of uncertainty about the economic damages associated with coastal eutrophication in many areas (Rabotyagov *et al.*, 2014). Efforts to inform policy should bring together ecologists and economists to study the impacts of N and P on ecosystem services all along the cascade.

7.5.2 | Acidification buffering

Soil acidity is controlled by both biota (plant roots and microorganisms) and particles (soil minerals and organic matter). Production of carbon dioxide, organic matter decomposition, and the excretion of acidic compounds by biota increase soil acidity, while binding of acidic compounds to root and particle surfaces, as well as mineral weathering, decrease it (Sposito, 2008). Over periods ranging from centuries to millennia, while most of the less resistant minerals become depleted through weathering reactions with rainwater and subsequent leaching, highly acidic soils are produced naturally. They now occupy about one-third of the ice-free land area on Earth (Guo *et al.*, 2010), mainly in the humid tropics and in the forested regions of temperate zones.

Industrial effluents (for example, sulphur and nitrogen oxide gases dissolved in atmospheric precipitation or transformed to particles, or acidic wastewaters) and nitrogenous fertilizers, such as urea, are typical anthropogenic inputs of acidity to soils. If these two acidic inputs exceed about 15 percent of the capacity of soil to neutralize them, acidification increases markedly, with a variety of serious problems arising for both plant and microbial growth. The potential for generating polluted runoff or drainage water also increases markedly. Over a 20 year period Guo *et al.* (2010) documented such increases of acidity in Chinese topsoils, caused by nitrogen fertilization and acidic deposition. The topsoils investigated showed an average pH decrease of 0.50, which is quite serious. Other long-term studies document decadal changes in soil acidity that are even larger (Richter and Markewitz, 2001). Acidic deposition is an important problem in China, but the acidification caused by nitrogen fertilization was found to be 10 to 100 times greater than that caused by acid rain. In the principal double-cropping cereal systems of China (wheat-maize, rice-wheat, and rice-rice), nitrogen fertilizer use efficiencies are only 30 to 50 percent. The progressive acidification of topsoil – as well as nitrogen pollution of agricultural runoff and drainage – will remain unchecked as long as this low nitrogen use efficiency is not addressed. Guo *et al.* (2010) noted that optimal nutrient-management strategies can significantly reduce nitrogen fertilization rates without decreasing crop yield, thus providing benefits to both agriculture and water quality.

7.5.3 | Filtering of reused grey water

Nearly 80 percent of urban 'blue water' becomes wastewater. At about 100 $m3yr^{-1}$ per household in the developed world, wastewater thus represents a rapidly expanding environmental and health challenge, particularly in urban centres. The ecological footprint of untreated wastewater is unsustainable even in regions where water is plentiful (e.g. South East Asia), as it may either increase nutrient loads in rivers and coastal regions or represent a direct hazard to human health. By contrast, arid regions increasingly rely on treated wastewater for irrigation, often practiced with little consideration of long-term impacts on the soil, hydrology and ecology of the producing area. The sustainability of this coupled agro-urban hydrological cycle hinges on proper management to mitigate adverse impacts of long-term wastewater use and avoid potential collapse of soil ecological functions. Various studies (e.g. Bond, 1998; Assouline and Narkis, 2013) have shown that, over the long term, even irrigation with wastewater results in significantly increased soil ESP that can adversely impact soil structure and hydraulic properties. In the absence of proper regulation, irrigation with wastewater may pose a range of human health and other ecological risks associated with introduction of pathogenic microorganism into the soil and crop (del Mar *et al.*, 2012). The sustainable management of wastewater irrigation requires new management strategies including water source mixing, proper selection and rotation of crops, and avoidance of sensitive soils.

7.5.4 | Processes impacting service provision

Trace elements

Elevated concentrations of potentially toxic trace elements can affect provision of the services that depend on soils. Trace elements – such as arsenic, cadmium, chromium, lead, mercury, and selenium – naturally occur in low quantities within soils. They may also be introduced and concentrated through anthropogenic activities like waste disposal, fertilizer and pesticide application, and atmospheric particulate emission and deposition

(Sparks, 2003; Pierzynski, Vance and Sims, 2005). Even when at low concentrations in soils, they can have pronounced impacts on water quality. This is particularly the case where the capacity of soils to store trace elements is exceeded or where there are changes in the soil chemical, physical and/or biological environment that influence the partitioning of trace elements between the solid and aqueous phases.

The concept of the critical load of a specific trace element enables a precautionary assessment of the risks its input causes to food quality and of the eco-toxicological effects on organisms in soils and surface waters (Lofts *et al.*, 2007; de Vries *et al.*, 2013b). The critical load of trace elements is defined as "the load resulting at steady state in a concentration in a compartment (e.g. soil solution, plant, fish) that equals the critical limit for that compartment" (Lofts *et al.*, 2007; de Vries *et al.*, 2013b). The critical limit is a receptor-specific concentration below which significant effect on the receptor is assumed not to occur (Lofts *et al.*, 2007). The concept of critical loads – specifically the critical loads of acidity – was key in gaining acceptance of the need for reduction of atmospheric deposition of N and S (Section 4.4.1 above). However, the usefulness of the concept of critical loads of trace elements in international negotiations aimed at reducing trace element deposition is not equally evident. This is mainly owing to two factors that distinguish trace elements from the case of acidity and acid rain: (i) the time needed for a specific trace element in a specific scenario to attain steady state is much longer than for N and S; and (ii) other changes in the environment, notably acidification, may have a greater influence on the exposure and effects of a specific trace element than the particular amount entering the system (de Vries *et al.*, 2013b). In fact, problems associated with trace elements in soils are commonly exacerbated by changes in land use that alter environmental conditions and increase the potential for exposure to trace elements through food and water consumption. Because of this, in addition to applying the concept of critical loads, the assessment of the future risks of trace elements needs to employ dynamic models (de Vries *et al.*, 2013b).

Salinity

Salinization of soil and water resources remains a chronic problem in many parts of the world, mostly in arid regions where evapotranspiration exceeds rainfall. The increased frequency of extreme climate events (droughts, intense rainfall events) together with the expansion of irrigated agriculture are expected to increase the range of soils affected by salinity.

In addition to the effects of hotter and drier climate patterns, the primary causes of salinity risk include: (i) increasing salt loads due to use of marginal water sources such as waste water; (ii) over exploitation of coastal aquifers and related sea water intrusion (Várallyay, 1994); (iii) overpumping and degradation of slowly replenishing inland aquifers (Ogallala); (iv) sea level rise impacting coastal wetlands (e.g. Mexico pacific coastline); (v) mismanagement of rapidly expanding irrigation in arid regions, particularly inadequate leaching and drainage and (vi) clearing of perennial vegetation in landscapes with significant salt stores in soils and deeper regolith.

One solution is to reduce the salt content of irrigation water through desalination. Recent advances in desalination techniques have resulted in a dramatic reduction in costs. Irrigation experiments with desalinated water show substantial increase in yield with less water used and less salt leaching to groundwater resources. However, the use of desalinated water requires careful management to avoid soil and ecological damage (e.g. clay dispersion) due to irrigation with extremely pure water (Yermiyahu *et al.*, 2007; Tal, 2006).

Erosion

Intensification of agriculture, changes in rainfall patterns with more intense rain events, and potentially more compacted soil surfaces may all contribute to increased rates of surface soil erosion. In addition to the removal of the top layer of productive soil and the incision of stream channels, the potential increase in soil transport to surface water may cause a cascade of adverse effects downstream. Pimentel *et al.* (1995) list impacts on stream and lake ecology, dam siltation and effects on waterways, and of course, potential for enhanced pollution by agrochemicals and colloid-facilitated transport of phosphorous and carbon. Soil erosion is also linked to climate change as it mobilizes large amounts of soil organic carbon (SOC). Since the industrial revolution and associated land use changes, SOC has been estimated to contribute 78±12 Gt of C to the atmosphere, of which about one-third is due to accelerated erosion and two-thirds to mineralization (WMO, 2005).

The WMO (2005) report estimates that 25 percent of African soils are prone to risk of water erosion (excluding deserts that comprise about 46 percent of the African land surface), and that 50 percent of cropland in Australia is susceptible to water erosion. Drier conditions associated with future climate extremes (droughts) may limit rates of soil carbon accumulation and reduce soil aggregation, thereby enhancing vulnerability to wind erosion. WMO (2005) estimate that about 22 percent of the African land surface is prone to wind erosion, and 15 percent of the cropland in Australia. A host of soil conservation strategies for combating land degradation due to soil erosion also offer co-benefits such as enhanced water storage in the soil profile (Pimentel *et al.*, 1995; Troeh, Hobbs and Donahue, 1991). Eroded landscapes may take centuries to millennia before their abilities to provide quality ecosystem services are restored.

7.6 | Soil change and water quantity regulation

Soil moisture regulation of precipitation

Soil moisture acts as a buffer for precipitation anomalies. As long as the soil is not saturated, it can reduce the direct impact of flooding. Similarly, soil moisture acts as a buffer against dry anomalies in the onset of meteorological droughts, before soil moisture or streamflow droughts are noticeable. However, if pre-event soil moisture is anomalously wet or dry, these same properties can also lead to significant flooding and droughts even where precipitation is not abnormally high or low. For these reasons, the monitoring of soil moisture conditions (as well as of snow and groundwater) is valuable for the forecasting of floods and droughts (e.g. Koster *et al.*, 2010b; Fundel, Jörg-Hess and Zappa, 2013; Orth and Seneviratne, 2013; Reager, Thomas and Famiglietti, 2014).

In addition to effects related to the buffering or persistence of soil moisture, several studies suggest that soil moisture also affects the regional water cycle through impacts of evapotranspiration on precipitation (e.g. Beljaars *et al.*, 1996; Koster *et al.*, 2004; Seneviratne *et al.*, 2010; Taylor *et al.*, 2012). However, the underlying feedbacks, including their sign, are strongly model-dependent (e.g. Koster *et al.*, 2004; Hohenegger *et al.*, 2008). Also observational studies diverge with respect to inferred soil moisture-precipitation feedbacks. Some suggest the presence of positive (temporal) feedbacks while others identify mostly negative (spatial) feedbacks (Findell *et al.*, 2011; Taylor *et al.*, 2012). In addition, causality is very difficult to establish based on observations (e.g. Salvucci, Saleem and Kaufmann, 2002). Precipitation persistence could, for example, lead to some confounding effects (Guillod *et al.*, 2014). Overall, effects of soil moisture on precipitation are still uncertain.

Human land and water use strongly affects soil moisture variations and the resulting land water balance, for instance through irrigation (Wisser *et al.*, 2010; Wei *et al.*, 2013) or other changes in agricultural practices (Davin *et al.*, 2014; Jeong *et al.*, 2014). These effects are generally not considered in present day climate models, although they could substantially affect soil moisture and hydrological drought projections, including feedbacks to the atmosphere.

7.6.2 | Precipitation interception by soils

Together with vegetation, soils help to regulate water quantity by intercepting water, reducing floods and maintaining the soil moisture buffer. Precipitation arriving at the Earth's surface can be intercepted by vegetation canopies and returned directly to the atmosphere through evaporation, never reaching the soil moisture pool. Typically, trees can intercept 25-50 percent of precipitation and shrubs 10-25 percent, while interception by grass is significantly less (Calder, 1999). The rest of the precipitation arrives at the soil surface, the characteristics of which control the partitioning between what infiltrates and what runs off into surface water. In a recent meta-analysis, Jarvis *et al.* (2013) have shown that K is largely dependent on bulk density, organic carbon content and land use. This has important consequences for ecosystem service delivery by soils, as it indicates that management and land-use change will affect the soil infiltration service temporally as well as spatially. This analysis by Jarvis *et al.* (2013) corresponds to an increasing number of studies that show the importance of vegetation in determining soil K values on similar soils.

Because of their large root systems, trees in particular create conduits for conducting water into soil. Both dead and living roots can create flow networks. Beven and Germann (1982) cited work suggesting that as much as 35 percent of the volume of a forest soil may contain macropores formed by roots. Chandler and Chappell (2008) demonstrated that K was highest near the trunk of single oak trees and decreased toward the edge of the canopy. The ratio of K geometric mean values under the tree at 3 metres from the trunk to the adjacent pasture was 3.4 times higher, similar to results compiled from the literature in the same paper. Gonzalez-Sosa *et al.* (2010) presented conductivity data for a range of land use types in France, with trees being generally higher, and crops and pasture lower for the same soil. In the tropics, deforestation results in a major reduction in infiltration, whether the forest is recently cleared or has been turned into pasture (Zimmermann, Elsenbeer and De Moraes, 2006).

Soil macrofauna - worms, ants and termites etc. - also play an important role in determining infiltration at local scales (Beven and Germann, 1982; Lal, 1988), and perhaps also regionally and globally given the prevalence of these organisms. There are typically two modes of macrofauna action impacting hydraulic properties. The first is the creation of burrows forming macropores; the other is the turnover of soil and aggregation which impacts infiltration and water retention, generally increasing both. Soils might offer potential for slowing water movement across landscapes under certain precipitation conditions (Marshall *et al.*, 2009). However, once runoff is generated and large quantities of precipitation fall, the role of soils is likely to be less important. Above a certain threshold, massive floods can occur in almost any landscape

Although often cited as an important ecosystem service, the impact of land use on altering flood risk remains hard to quantify with any precision (Pattison and Lane, 2011). The link between land management and flood risk is complex and scale dependent as conceptualized by Bloschl *et al.* (2007). Many studies have demonstrated how land or soil management impact infiltration and runoff generation at the plot to hillslope scale (Wheater and Evans, 2009). These tend to be local effects in temperate zones, but can be large scale in the tropics. Beven *et al.* (2008) found a distinct land use signal hard to detect, and also pointed out that "adequate information about past land management changes and soil conditions is not readily available but will need to be collected and made available in future for different land use categories if improved understanding of the links between runoff and land management is to be gained and used at catchment scales."

7.6.3 | Surface water regulation

Soils provide a maintenance service that contributes to the regulation of base flow and water supply in rivers. Groundwater, lakes and soil drainage all play a role in setting base flow in surface waters (Price, 2011). Groundwater dominates in the lowlands, but soil drainage dominates upland catchments. Changes to the hydraulic characteristics of upland catchments and to the quantity of water stored by soils will have distinct implications for water supply downstream. Again, the soil water retention characteristics and hydraulic conductivity play a crucial role in the regulation of drainage.

7.7 | Soil change and natural hazard regulation

Soil and its characteristics (depth, hydro-mechanical properties, mineralogy, ecological function, and position in the landscape) play an important role in several natural hazards including: landslides, debris flows, floods, dam failure, droughts, shrink and swell damage to roads and infrastructure, and more. The United Nations International Strategy for Disaster Reduction (UNISDR, 2009) defines a natural hazard as a "natural process or phenomenon that may cause loss of life, injury or other health impacts, property damage, loss of livelihoods and services, social and economic disruption, or environmental damage". Projected human population expansion, agricultural intensification, and greater human presence and infrastructure in mountainous regions combined with projected changes in climate extremes (IPCC, 2012) are expected to jointly contribute to enhanced vulnerability to soil-mediated natural hazards (Figure 7.9). The extent of the

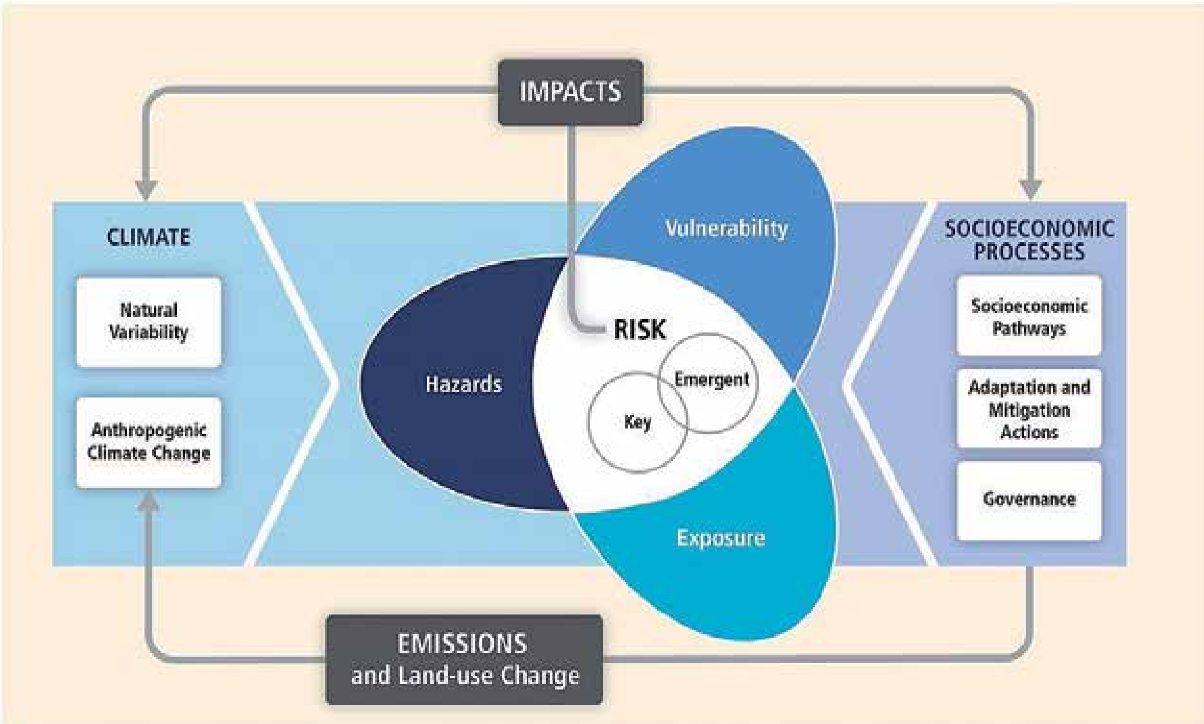


Figure 7.9 | A conceptual sketch of how vulnerability, exposure and external events (climate, weather, geophysical) contribute to the risk of a natural hazard.
Source: IPCC, 2012.

vulnerability and exposure to a particular type of hazard vary considerably among regions (ESPON, 2013). For example, floods may increase in flat terrains with increasing mean precipitation or rapid snowmelt, and landslides may become more common in mountainous areas with changes in the seasonality and intensity of rainfall (Huggel, Clague and Korup, 2012).

The past few decades have been marked by an increase in the frequency and magnitude of damages caused by soil-climate related hazards such as landslides (Figure 7.10, FAO 2011). In part this increase may be simply attributed to more timely and accurate reporting, and also to deeper human penetration into soil-hazard prone regions, facilitated by increases in mobility and personal wealth (Keiler, 2013; Papathoma-Köhle *et al.*, 2015). The reports of EM-DAT (*http://www.emdat.be/publications*) provide a global perspective of all aspects of natural disasters and their human and economic impacts. The 2013 EM-DAT[2] report estimates global damages by natural hazard attributed to hydrological and geophysical causes (most closely related to soil) in excess of US$ 60 billion, with impacts on the lives of 40 million people in 2013 alone. It is instructive to place the various natural hazards in their soil-human-climate context to enable general inferences and detection of future trends with global change (population growth, land use, and climate change).

2 http://www.emdat.be/publications

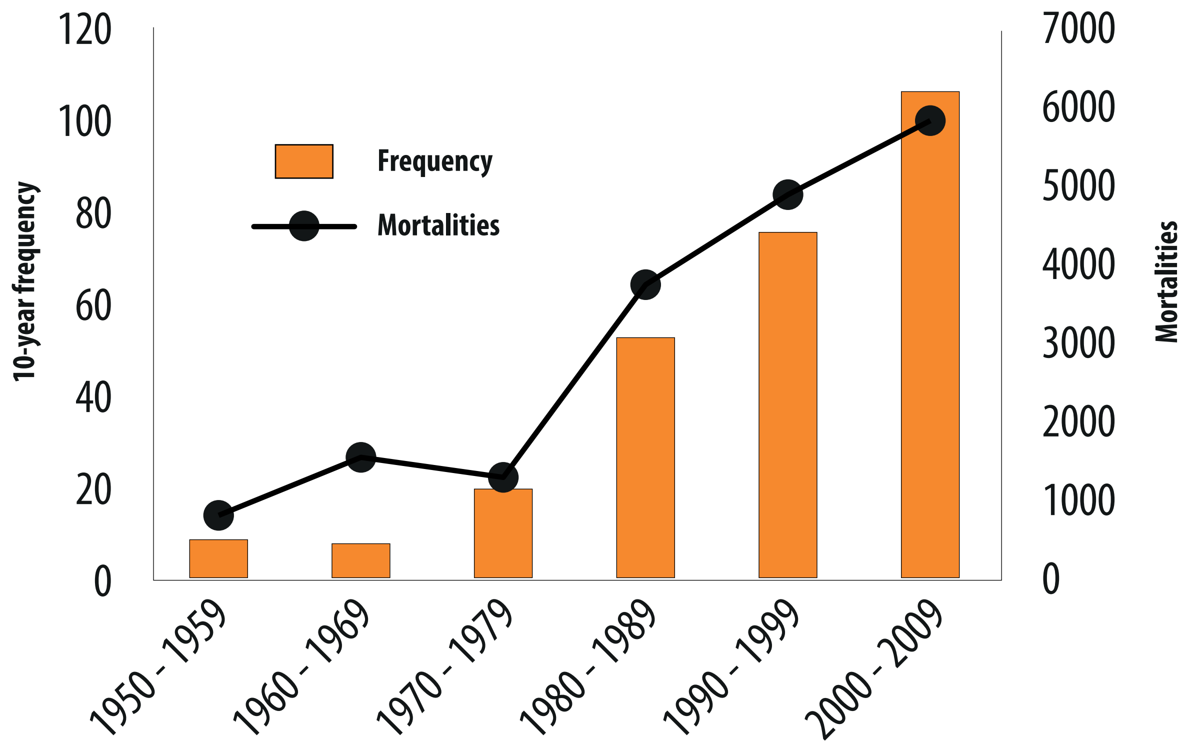


Figure 7.10 Trends in landslide frequency and mortality on Asia.
Source: FAO, 2011; EM-DAT, 2010.

7.7.1 | Soil landslide hazard

The depth of the soil mantle forming over mountainous topography reflects a natural balance between soil production and soil erosion processes (Trustrum and De Rose, 1988; Heimsath *et al.*, 1997). The primary soil removal process in mountainous regions is landsliding, driven by the topographic relief and triggered by climatic forcing such as rainfall or snowmelt (Iverson, 2000; Larsen, Montgomery and Korup, 2010; Kawagoe, Kazama and Sarukkalige, 2009) or by earthquakes (Huang and Fan, 2013). Landslide damage is costly: Sidle and Ochiai (2006) estimated the direct costs associated with rebuilding or replacing infrastructure at several billion dollars per year, even without considering indirect costs related to construction and temporary loss of site functionality. Similar estimates have been made just for Europe (Papathoma-Köhle *et al.*, 2015).

Rainfall is the most common trigger for shallow landslides (Iverson, 2000). The strong relationship between rainfall intensity-duration and landslide triggering conditions has prompted the use of rainfall characteristics for early warning (Guzzetti *et al.*, 2008; von Ruette, Lehmann and Or, 2014). The observed increase in precipitation variability and in extreme events attributed to climate change has been linked to the observed increase in landslide frequency in mountainous regions (Huggel, Clague and Korup, 2012). The recent IPCC report (IPCC, 2012) lists evidence for the contiguous United States confirming statistically significant increases in heavy (upper 5 percent) and very heavy (upper 1 percent) precipitation of 14 and 20 percent, respectively. Moreover, evidence from Europe and the United States suggests that the relative increase in precipitation extremes is larger than the increase in mean precipitation.

Schmidt and Dikau (2004) found that climatic scenarios representing unstable conditions of transition from more humid to a dryer climate produced the highest slope instabilities. Soil hydraulic properties play an important in imparting mechanical sensitivity. Indeed, the soil plays multiple roles in the landslide hazard, not only as the mass that slides down the slope, but also through its own mechanical strength and through its modulation of local hydrology via infiltration capacity, base flow, macropore flow and ground cover (Iverson, 2000; Sidle and Ochiai, 2006; Lehmann and Or, 2012). The partitioning of precipitation between infiltration, overland flows and base flows is critical to the loading of the soil and to the ultimate soil failure. The mechanical reinforcement by plant roots helps to stabilize the soil mantle (Abe and Ziemer, 1991; Schwarz, Cohen and Or, 2012), and bulk soil mechanical and hydraulic properties affect the susceptibility to failure.

Recent widespread drought-induced forest die-offs highlight how climate change could accelerate forest mortality. This has potential consequences for the carbon cycle and for ecosystem services (Anderegg *et al.*, 2013). Through loss of root reinforcement, die-off may also increase landslide hazard. Rapid landslide processes have also been observed in Southeast Asia and the Western Pacific where large tropical cyclones induce numerous landslides and remove significant amounts of soil and particular carbon through the river systems to the ocean (Hilton *et al.*, 2008). These extreme tropical precipitation events are likely to increase in frequency and magnitude (Huang *et al.*, 2013).

7.7.2 | Soil hazard due to earthquakes

Keefer (2002) provides a historical overview of the study of earthquake-induced landslides. These are often extensive in their size and occurrence and cause more significant damage than hydrologically-induced shallow landslides. For example, the 2008 Wenchuan earthquake in Sichuan province in China triggered more than 60 000 landslides over an area of 35 000 km^2 causing about one-third of the total number of fatalities in the earthquake disaster (Huang and Fan, 2013). In addition to the direct damages, the Wenchuan earthquake induced an unprecedented number of secondary geohazards such as heightened subsequent landslide frequency, causing river damming and consequent floods as well as debris flows. The links between seismic activity and landslide characteristics were systematically investigated by Malamud *et al.* (2004) based on landslide inventory data of landslide size-frequency distribution in the affected landscape. These analyses are useful for deriving large-scale soil erosion rates enhanced by seismic activity. Erosion rates in active subduction zones are around 0.2–7 mm yr^{-1}. Hazard schemes often classify earthquake-induced landslides as 'geophysical' or 'dry' events to indicate they do not require water for mass movement initiation, unlike hydrological 'wet' landslides.

On March 11, 2011, a seaquake followed by an enormous tsunami and by the destruction of the Fukushima Atomic Power Plant, Japan, brought about additional soil changes such as liquefaction, tsunami sedimentation and radio isotope contamination, all of which affected the local population. Liquefaction brought about by the earthquake occurred mainly on soil-banked lands or soil-dressed lands, causing extreme damage to housing and structural facilities. The tsunami carried massive deposits from the bottom of the sea onto farmlands along the seashore. This sedimentation contained considerable quantities of arsenic (Kozak and Niedzielski, 2013). The explosion of the atomic power plant resulted in soil contamination (mainly with Cesium 137) of an area as large as 800 square kilometres (Steinhauser, 2014; Itoha *et al.*, 2014). Cleaning these contaminants is vital before the population can return. More broadly, although a variety of soil hazard regulation techniques have been developed (Gasso *et al.*, 2013; Delgado *et al.*, 2011; Esteves *et al.*, 2012) there is a need for both more research and more regulation related to soil hazards than hitherto.

7.7.3 | Soil and drought hazard

Droughts limit primary production and thus the accumulation rates of organic matter. Reduced accumulation rates contribute to soil vulnerability to water and wind erosion. Recent meso-scale strategies for combating drought damage and reducing risk in agro ecosystems have proposed landscape-scale vegetation management. This can, for example, take the form of patches or bands of perennial vegetation to promote feedbacks that are conducive to recycling of water vapour, soil moisture and nutrients (Ryan, McAlpine and Ludwig, 2010). An often ignored consequence of prolonged drought and soil water depletion is soil subsidence and related damage to buildings and infrastructure (Corti *et al.*, 2011). Corti *et al.* (2011) presented a systematic study of damage costs from drought-induced soil subsidence applicable across different climate regimes. The primary variables include drought severity, soil type (shrink/swell properties), land use, and vegetation.

Prolonged droughts and drier climate patterns accentuate damages due to soil shrink/swell properties.

The insurance industry reports that damage to infrastructure often peaks following extreme drought events, especially in the densely built up regions of Europe and United States. Dry climate also induces other phenomena such as the onset of massive dust storms. Dust storms can arise either from destabilization of vulnerable surface soils (the Dust Bowl), or from the drying of lake beds, or from desertification and loss of vegetation and similar soil destabilizing activities over large scales. The rates of wind erosion associated with sand storms may exceed 100 mm topsoil yr^{-1} in sensitive regions in the Sahel. Prolonged exposure is known to pose respiratory health hazards to human population.

7.7.4 | Soil and flood hazard

Agricultural intensification has been linked to alteration of runoff mechanisms and to increased risk and burden of floods (Marshall *et al.*, 2014). Some of the primary changes in land management documented in the United Kingdom and elsewhere that affect soils include: heavy traffic contributing to soil compaction, tillage operation and consequent loss of soil structure, the formation of larger fields, choice of cover crops in rainy seasons, and increased livestock densities (O'Connell *et al.*, 2007). However, establishing rigorous causal links between changes in land management practices, local runoff generation and catchment scale flood behaviour remains a challenge (Ewen *et al.*, 2013). Nevertheless, mounting evidence suggests that soil and land management contribution to flood risk is not limited to management of lowland agricultural regions. Management of upland soils and related impacts on runoff generation mechanisms cascade and also have impacts on flood risk downstream (Wheater and Evans, 2009; Marshall *et al.*, 2009). A recent review by Hall *et al.* (2014) on flood trends in Europe (including climatic effects) confirms the important role of land use changes (urbanization, afforestation, etc.) as key factors in modifying large scale flood risk. Some of the strategies for reducing flood risk include afforestation in upland catchments (Ewen *et al.*, 2013), creation of retention basins, and adding floodplains by lowering levees (Hall *et al.*, 2014).

7.7.5 | Hazards induced by thawing of permafrost soil

Permafrost is perennially frozen soil remaining at or below 0°C for at least two consecutive years (Brown *et al.*, 1998). Permafrost regions occupy about 24 percent of the exposed land area in the Northern Hemisphere and in some high mountainous regions (UNEP, 2012). Expected thawing of permafrost is projected to induce alterations in soil hydrology and biological activity, and to have an impact on the global carbon cycle (Schuur *et al.*, 2008). In addition, the thawing of permafrost is expected to change vegetation species and reshape many ecosystem functions. The mechanical weakening of the previously frozen soil is likely to result in foundation settling, with damage to buildings, roads, pipelines, railways and power lines (Nelson, Anisimov and Shiklomonov, 2001; Jorgenson, Shur and Pullman, 2006). Estimates of infrastructure repair in Alaska up to 2030 are in the range of US$ 6 billion (UNEP, 2012). Changes in mean temperature and snow cover also affect sensitive permafrost in high mountains, and contribute to a higher risk of landslides and avalanches (Gruber and Haeberli, 2007; Harris *et al.*, 2009). Schoeneich *et al.* (2011) present an extensive report and case studies, largely from the European Alps, on various slope movement hazards (landslides, rock fall, and debris flow initiation) associated with degrading permafrost. Evidence suggests accelerated erosion rates of the thawed permafrost, especially along coastlines and rivers banks as documented by Schreiner, Bianchi and Rosenheim (2014) and Vonk *et al.* (2012), with subsequent transport of the carbon-rich sediment through river systems to the ocean.

7.8 | Soil biota regulation

Soil biodiversity is vulnerable to many anthropogenic disturbances, including land use and climate change, nitrogen enrichment, soil pollution, invasive species and the sealing of soil. A recent sensitivity analysis revealed that increasing land use intensity and associated soil organic matter loss are placing the greatest pressure on soil biodiversity (Gardi, Jeffery and Saltelli, 2013). Numerous studies report soil biodiversity declines as result of the conversion of natural lands to agriculture (Bloemers *et al.*, 1997; Eggleton *et al.*, 2002; Dlamini and Haynes, 2004), and as a result of agricultural intensification (Mulder *et al.*, 2005; Postma-Blaauw *et al.*, 2010; De Vries *et al.*, 2013a). In particular, studies show larger bodied soil animals, such as earthworms and termites are especially vulnerable, but intensive land use can also reduce the abundance and variety of species of nematodes, mites and collembolans.

Climate change also poses a considerable threat to soil biodiversity through direct effects of warming and altered precipitation (e.g. drought and flooding) on the availability of moisture in soil (Bardgett *et al.*, 2008). Indirect climate change effects of warming and elevated atmospheric carbon dioxide may also have an impact on the quantity and quality of organic matter in soil (Blankinship, Niklaus and Hungate, 2011; van Groenigen *et al.*, 2014). Although poorly understood, predicted increases in the frequency of erosive rainfall events (Nearing *et al.*, 2005) and climate-induced shifts in land use (Mullan, 2013) could pose a considerable future threat to soil biodiversity. Other threats to soil biodiversity include nitrogen enrichment, which negatively impacts soil fungi (Treseder, 2008), soil sealing, which effectively stops the natural functioning of soil (Gardi, Jeffery and Saltelli, 2013), and invasive species, which affect native soil biodiversity through a range of mechanisms, including altered resource supply, competitive interactions and predation, and physical and chemical modification of the soil environment (Wardle *et al.*, 2011).

Although it is well known that soil organisms play key roles in many ecosystem processes, our understanding of the functional consequences of belowground diversity loss is limited, at least compared to what is known about aboveground losses (Cardinale *et al.*, 2012). Recent synthesis of experimental studies on soil diversity-function relationships indicate that diversity effects on processes of nutrient and carbon cycling are highly variable, but effects of species loss are most pronounced at the low end of the diversity spectrum (Nielsen *et al.*, 2011). There is also a general consensus that changes in the functional composition of belowground communities, rather than species diversity per se, are of most importance for ecosystem functioning (Nielsen *et al.*, 2011). Consistent with this, laboratory studies with low numbers of species have shown the functional composition of soil macrofauna communities to be a better predictor of litter decomposition than species richness (Heemsbergen *et al.*, 2004). The selective removal of different groups of soil organisms has been shown to impair soil functioning (Wagg *et al.*, 2014). Likewise, a recent cross-biome field experiment showed that the loss of key components of the decomposer communities consistently slowed rates of litter decomposition and carbon and nitrogen cycling, indicating negative effects of diversity loss on soil functions (Handa *et al.*, 2014). A field-based study of different sites across Europe also showed that changes in soil food web composition resulting from intensive agriculture consistently strongly affected processes of carbon and nitrogen cycling (De Vries *et al.*, 2013a). At one site, high intensity management reduced the resistance and resilience of the soil food web to drought, increasing soil carbon and nitrogen loss as greenhouse gases and in leachates (De Vries *et al.*, 2012a, 2012b, 2012c).

Changes in soil biodiversity can also modify vegetation dynamics, both directly through associations of symbionts and pathogens with plant roots, and indirectly, by modifying nutrient availability to plants (van der Putten *et al.*, 2013). For example, mycorrhizal fungi, which form symbiotic associations with roots of most plant species and are very vulnerable to soil disturbances, can enhance plant species diversity by relaxing plant competition intensity and promoting more equitable distribution of resources within the plant community (van der Heijden, Bardgett and van Straalen, 2008). Also, plant diversity and productivity have been shown, in some situations, to be positively related to arbuscular mycorrhizal fungal diversity due to more efficient use of soil phosphorus (van der Heijden *et al.*, 1998). Soil pathogens, which cause considerable problems for agricultural crops, have also been shown to impact vegetation dynamics in natural settings, by suppressing the growth of their host plant species more than their neighbours, thereby contributing to vegetation change (Bever, Westover and Antonovics, 1997; Packer and Clay, 2000; Klironomos, 2002). The spread of invasive plant species has also been linked to release from their natural soil enemies in their new territories, giving the invasive plant a competitive edge over native species. This often leads to declines in plant diversity and to shifts in the functioning of the soil (Wardle *et al.*, 2011; van der Putten *et al.*, 2013).

Although poorly explored, diversity changes in soil are likely to impact soil physical properties, with consequences for ecosystem services related to soil formation and water regulation. Diversity effects on soil physical properties have not been explicitly studied, but they are likely to be important given the potential for different groups of soil organisms to differentially impact soil structure through different routes. For example, fungi promote soil aggregate stability through the physical enmeshment of soil particles by their extensive networks of mycelia, whereas bacteria produce metabolic products, mainly polysaccharides, which bind soil particles together (Hallett *et al.*, 2009). Mycorrhizal infection can also influence soil aggregate stability through physical enmeshment of soil particles by their extensive networks of mycelium, but also through the binding of soil particles via the production of extracellular polysaccharides and proteins, including the protein glomalin, which alters the wetting behaviour of soil (Rillig and Mulley, 2006). Finally, soil animals, especially ecosystem engineers such as earthworms and termites, impact soil structure by creating macropores and channels, thereby improving water movement through soil (Bardgett, 2005).

While evidence is mounting that shifts in soil biodiversity resulting from human activities have significant consequences for ecosystem functions and the services that they underpin, there is still much to be learned. The mechanisms by which soil biodiversity change can impact ecosystem are enormous, involving a range of ecological and evolutionary processes at different spatial and temporal scales, and links between aboveground and soil communities. Moreover, impacts of soil biodiversity change on soil functions are likely to be context dependent, varying with soil abiotic properties and vegetation type. Unravelling this complexity in order to make better predictions about the consequences of soil biodiversity change for the services that ecosystems provide is a major challenge.

7.9 | Soils and human health regulation

The linkage between soils and human health is increasingly being recognized (Abrahams, 2006, 2013; Baumgardner, 2012; Brevik and Burgess, 2013; Jeffrey and van der Putten, 2011; Oliver, 1997). A central understanding is that soils form an integral link in a holistic view of human health that includes physical, mental and social dimensions. The soil acts as a natural filter, it can kill off pathogens, it can biodegrade organics and, in general, it does a wonderful job of protecting us from human health threats. However, soil is not able to protect itself against all the insults it is subject to on a regular basis.

Soils aid in the regulation of human health. They do this by keeping in check, or balancing, the beneficial versus deleterious concentrations of elements and moderating disease-causing organisms. For example, soils regulate human health by impacting the nutrient quality or nutrient density of foods. Too little of an essential nutrient in soil can lead to human diseases such as Keshan disease caused by selenium deficiencies in the human diet (Chen, 2012). Conversely, health problems can be caused by an excess amount of organics or trace elements such as the arsenic released by soils into the drinking and irrigation waters of Bangladesh (Khan, Hamra and Mu, 2009) (Section 7.3).

Soil is a natural source of radiation that can adversely affect human health, and soil can also affect human health by directly interacting with people. One example is the disease of podoconiosis or Mossy Foot disease (Mossy Foot Project, 2014). Mossy Foot disease affects about 5 percent of the population in highland tropical areas with volcanic soils and lots of rainfall. These soils are rich in silicates that can penetrate the skin of susceptible people as they go barefoot about their daily business. Soils can also act as a reservoir of all kinds of introduced materials that can impact human health. The dioxin at Love Canal in New York, United States is a classic example (Silkworth, Culter and Sack, 1989). There are large quantities of industrial and agricultural products and by-products added to soil every year that have the potential to impact human health.

Vast numbers of people, primarily women, infants and children, are afflicted with trace element deficiencies (notably Fe, I, Se, and Zn), mostly in the resource-poor countries of the developing world. A diet with low boron (B) has been found to lead to a number of general health problems and to increase cancer risk. The most common symptoms of B deficiency include arthritis, memory loss, osteoporosis, degenerative and soft cartilage diseases, hormonal disequilibria and a drop in libido (Scoreⁱ and Popa, 2010). According to one hypothesis, the low cervical cancer incidence in Turkey correlates with its B-enriched soil (Simsek *et al.*, 2003). Indeed, the ingestion of B via drinking water prevents cervical cancer risk (Korkmaz *et al.*, 2007). In a survey in northern France, exposure to high levels of boron (>0.3 mg L^{-1}) in the drinking water was associated with a significantly lower mortality rate as compared to that of a low-boron reference area (Yazbeck *et al.*, 2005).

Silicon is the second most abundant element in the Earth's crust. Dietary silicon intake is positively associated with bone mineral density in men and premenopausal women of the Framingham Offspring cohort (Jugdaohsingh *et al.*, 2004). Silicon is bound to glycosaminoglycans and has an important role in the formation of cross-links between collagen and proteoglycans (Carlisle, 1976). In vitro studies have demonstrated that silicon stimulates type 1 collagen synthesis and osteoblast differentiation (Reffitt *et al.*, 2003).

Many physicians have believed that zinc deficiency is a rare occurrence in Japan. Nevertheless, One study found many zinc-deficient patients at a clinic in Japan since 2002 (Kurasawa, Kubori and Okuizumi, 2010). Their complaints were anorexia, general fatigue, impaired sense of taste, burning mouth, various types of skin lesion, delayed wound healing and emotional instability.

Based on dietary intake recommendations, subclinical or marginal Mg deficiency (50 percent to <100 percent of requirement) commonly occurs throughout the world (Nielsen, 2010). Yet, pathological conditions attributed specifically to dietary Mg deficiency alone are considered rare. However, epidemiological and correlation studies indicate that a low Mg status is linked to numerous pathological conditions associated with aging, including atherosclerosis and hypertension (Ma *et al.*, 1995), osteoporosis (Rude, Singer and Gruber, 2009), diabetes mellitus (Barbagallo *et al.*, 2003), and some cancers (Dai *et al.*, 2007). Magnesium (Mg) deficiency increases genomic instability and Mg intake has been reported to be inversely associated with a risk of colorectal cancer (CRC). An experiment designed to determine whether Mg in drinking water suppresses inflammation-associated colon carcinogenesis in mice showed Mg at all doses caused a significant inhibition of CRC development (Kuno *et al.*, 2012).

The role of soil in contributing to human health is considerable. Soils rich in biodiversity produce healthier and more nutritious foods and control the proliferation of any pathogenic microorganisms that affect both plant and human health. Biodiversity is a result of a highly functioning, high quality soil with a good balance of nutrients and good water infiltration and aeration.

An example of how imbalance in a soil, for example improper soil water balance, causes human and animal disease comes from Australia (Hampton *et al.*, 2011; Creswell, 2012). Birds and people were being infected and some died due to a disease, melioidiosis, caused by the bacteria Burkholderia pseudomalle. This microorganism is normally found only at low levels in soil, mostly in subsoils. However, after heavy rains, pools or puddles of water provide a suitable habitat for the bacteria to proliferate. It enters the body via a cut or graze or through the lungs by inhalation. Soil that is properly drained and aerated regulates the prevalence of this bacteria and keep it in check.

Soils also serve as a source of many medicines. For example, soil microorganisms still account for many of the current clinically relevant antibiotics (D'Costa *et al.*, 2006; Pepper *et al.*, 2009). It is therefore important to maintain the vast diversity of microorganisms in soil in order preserve the untapped potential for future discoveries important to human health.

We are just now beginning to understand how the chemical, physical and biological properties of soil can affect the health of humans and animals and entire ecosystems. The same techniques that are available to map the human microbiome can also be applied to map the soil microbiome. We are thus on the verge of understanding what constitutes a healthy soil microbiome and how a degraded or unhealthy soil microbiome may affect our food production and overall human health.

7.10 | Soil and cultural services

The soil is one of the main sources of information on the prehistoric culture of humankind. Indeed, soil is an excellent medium for preserving artefacts. Different soil types have particular characteristics to preserve remains. For instance, in permanently or seasonally wet soils, the lack of oxygen slows down the decomposition of organic matters. Sometimes the remains of animals can be found with hunting marks from arrows or spears. Well-preserved human bodies have been excavated from moors and bogs. The anaerobic conditions preserve the bodies very well and several thousands of years later they are excavated with skin, flesh and clothes still present. Wooden constructions, such as poles for bridges, boats and wooden tools may also be preserved, giving us valuable information on the level of technology at that time.

Past farming practices can also be recognized in the soil profile, particularly in Anthrosols. For example, in northwest Europe, notably in the Netherlands and Germany, a human-made soil type, known as plaggen soil, has developed as a result of a specialized agricultural system. On the strongly leached, acid sandy outwash plains and moraines, Podzols developed underneath a vegetation cover of heather. Farmers used the heather and the uppermost level of the soil as bedding in the stables. The droppings from the animals, mixed with the bedding, were later used as manure on the nearby fields, slowly building up a thick soil layer rich in organic matter and high in nutrients and with a good soil water retention. These fields provide a relatively high and stable crop production compared to the surrounding land (European Soil Bureau Network, 2005).

In the Amazon basin, the Terra Preta soil owes its name to its very high charcoal content. It was created by the addition of a mixture of charcoal, bone and manure to the otherwise relatively infertile Amazonian soil. Terra Preta soils were created by indigenous peoples in the pre-Columbian era between 450 BC and AD 950 (Sombroek *et al.*, 2002). Technosols are modern examples of soils that store artefacts or are strongly influenced by (modern) humankind. They include soils from wastes (landfills, sludge, cinders, mine spoils and ashes), pavements with their underlying unconsolidated materials, and constructed soils in human-made materials (FAO, 2014).

Soils provide aesthetic and recreational value through the landscape, particularly in Globally Important Agricultural Heritage Systems (Koohafkan and Altieri, 2011). They have also been used as an aesthetic approach to raise soil awareness in contemporary art (Toland and Wessolek, 2014). Churchman and Landa (2014) provide a comprehensive treatment of the topic.

References

Abe, K. & Ziemer, R.R. 1991. Effects of tree roots on a shear zone: Modelling reinforced shear stress. *Can. J. For. Res.*, 21: 1012–1019..

Aber, J.D., Nadelhoffer, K. J., Steudler, P. & Melillo, J.M. 1989. Nitrogen saturation in northern forest ecosystems. *BioScience*, 39: 378-286.

Abrahams, P.W. 2006. Soil, geography and human disease: A critical review of the importance of medical cartography. *Progress in Physical Geography*, 30: 490-512.

Abrahams, P.W. 2013. Soils: Their implication to human health. *Sci. Total Environ.*, 291: 1-32.

Ackerman, A.S., Toon, O.B., Stevens, D.E., Heymsfield, A.J., Ramanathan, V. & Welton, E.J. 2000. Reduction of tropical cloudiness by soot. *Science*, 288: 1042-1047.

Anderegg, W.R.L., Plavcova, L., Anderegg, L.D.L., Hacke, U.G., Berry, J.A. & Field, C.B. 2013. Drought's legacy: multi year hydraulic deterioration underlies widespread aspen forest die-off and portends increased future risk. *Glob Change Biol.*, 19: 1188–1196.

Andreae, M.O., Charlson, R.J., Bruynsells, F., Storms, H., Van Grieken, R. & Maenhaut, W.1986. Internal mixture of sea salt, silicates, and excess sulfate in marine aerosols. *Science*, 232: 1620-1623.

Aneja, V.P., Schlesinger, W.H. & Erisman, J.W. 2009. Effects of agriculture upon the air quality and climate: research, policy and regulations. *Environ. Sci. Technol.*, 43: 4234-4240.

Ardón, M., Morse, J.L., Doyle, M.W. & Bernhardt, E.S. 2010. The water quality consequences of restoring wetland hydrology to a large agricultural watershed in the southeastern coastal plain. *Ecosystems*, 13:1060-1078.

Assouline, S. & Narkis, K. 2013. Effect of long-term irrigation with treated wastewater on the root zone environment. *Vadose Zone Journal*, 12(2).

Bakker, M.M., Govers, G. & Rounsevell, M.D.A. 2004. The crop productivity-erosion relationship: an analysis based on experimental work. *Catena*, 57(1): 55-76.

Bakker, M.M., Govers, G., Jones, R.A. & Rounsevell, M.D.A. 2007. The effect of soil erosion on Europe's crop yields. *Ecosystems*, 10(7): 1209-1219.

Barbagallo, M., Dominguez, L.J., Galioto, A., Ferlisi, A., Cani, C., Malfa, L., Pineo, A., Busardo, A. & Paolisso, G. 2003: Role of magnesium in insulin action, diabetes and cardio-metabolic syndrome X. *Mol. Aspects Med.*, 24: 39-52.

Bardgett, R.D. 2005. *The Biology of Soil: A Community and Ecosystem Approach*. UK, Oxford, Oxford University Press.

Bardgett, R.D., Manning, P., Morrien, E. & De Vries, F.T. 2008. Hierarchical responses of plant–soil interactions to climate change: consequences for the global carbon cycle. *Journal of Ecology*, 101: 334- 343.

Batjes, N.H. 1996. Total carbon and nitrogen in the soils of the world. *European Journal of Soil Science*, 47: 151–163.

Baumgardner, M.D. 2012. Soil-related bacterial and fungal infections. *J. Am. Board Fam. Med.*, 25: 734-744.

Beljaars, A.C.M., Viterbo, P., Miller, M.J. & Betts, A.K. 1996. The anomalous rainfall over the United States during July 1993: Sensitivity to land surface parameterization and soil moisture. *Mon. Weather Rev.*, 124(3): 362-383.

Bennett, E.A., Carpenter, S.R. & Caraco, N.F. 2001. Human impact on erodable phosphorus and eutrophication: A global perspective. *Bioscience*, 51: 1-8.

Berg, B. & Matzner, E. 1997. Effect of N deposition on decomposition of plant litter and soil organic matter in forest systems. *Environmental Review*, 5: 1–25.

Betts, A.K. 2004. Understanding hydrometeorology using global models. *B. Am. Meteorol. Soc.*, 85(11): 1673-1688.

Beven, K. & Germann, P. 1982. Macropores and water flow in soils. *Water resources research*, 18(5): 1311-1325.

Beven, K.J., Young, P., Romanowicz, R., O'Connell, P.E., Ewen, J., O'Donnell, G.M.O., Homan, I., Posthumus, H., Morris, J., Hollis, J., Rose, S., Lamb, R. & Archer, D. 2008. Analysis of Historical Data Sets to Look for Impacts of Land Use and Management Change on Flood Generation. London, Department for Environment, Food and Rural Affairs.

Bever, J.D., Westover, K.M. & Antonovics, J. 1997. Incorporating the soil community into plant population dynamics: the utility of the feedback approach. *Journal of Ecology*, 85: 561–573.

Birch, M.B., Gramig, B.M., Moomaw, W.R., Doering III, O.C. & Reeling, C.J. 2010. Why Metrics Matter: Evaluating Policy Choices for Reactive Nitrogen in the Chesapeake Bay Watershed. *Environmental Science & Technology*, 45: 168-174.

Blankinship, J., Niklaus, P. & Hungate, B. 2011. A meta-analysis of responses of soil biota to global change. *Oecologia*, 165: 553-565.

Bloemers, G.F., Hodda, M., Lambshead, P.J.D., Lawton, J.H. & Wanless, F.R. 1997. The effects of forest disturbance on diversity of tropical soil nematodes. *Oecologia*, 111: 575 -582.

Bloschl G, Ardoin-Bardin S, Bonnell M, Dorninger M, Goodrich D, Gutknecht D, Matamoros, D., Merz, B., Shand, P. & Szolgay, J. 2007. At what scales do climate variability and land cover change impact on flooding and low flows? *Hydrological Processes*, 21: 1241–1247.

Blum, W.E.H. & Nortcliff, S. 2013. Soils and food security. In E.C. Brevik, & L.C. Burgess, eds. *Soils and Human Health*. pp. 299-321. USA, Boca Raton, FL, CRC Press.

Bommarco, R., Kleijn, D. & Potts, S.G. 2013. Ecological intensification: harnessing ecosystem services for food security. *Trends in Ecology and Evolution*, 28: 230-238.

Bond, W.J. 1998. Effluent irrigation—an environmental challenge for soil science. *Aust. J. Soil Res*. 36: 543–555.

Boyd, J. & Banzhaf, S. 2007. What are ecosystem services? The need for standardized environmental accounting units. *Ecological Economics*, 63: 616-626.

Brauman, K.A., Daily, G.C., Duarte, T.K.E. & Mooney, H.A. 2007. The nature and value of ecosystem services: an overview highlighting hydrologic services. *Ann. Rev. Environ.* Res., 32: 67-98.

Brevik, E.C. & Burgess, L.C. (eds.) 2013. *Soils and Human Health*. USA, Boca Raton, FL, CRC Press.

Brown, J., Ferrians, O.J.Jr**., Heginbottom, J.A. & Melnikov, E.S.** 1998. *Circum-Arctic map of permafrost and ground-ice conditions*. USA, Boulder, CO, National Snow and Ice Data Center/World Data Center for Glaciology, Digital Media.

Budyko, M.I. 1956. *Heat balance of the Earth's Surface*. Leningrad, Gidrometeoizdat. 255 pp. [in Russian]

Budyko, M.I. 1974. *Climate and Life*. Academic Press. 508 pp. [in Russian].

Calder, I.R. 1999. *The blue revolution: land use and integrated water resources management*. Earthscan.

Cao, M., Gregson, K. & Marshall, S. 1998. Global methane emission from wetlands and its sensitivity to climate change. *Atmospheric Environment*, 19: 3293-3299.

Cardinale, B.J., Duffy, J.E., Gonzalez, A., Hooper, D.U., Perrings, C., Venail, P., Narwani, A., Nace, G.M., Tilman, D., Wardle, D.A., Kinzig, A.P., Dailey, G.C., Loreau, M., Grace, J.B., Larigauderie, A, Srivastave, D.S. &.Naeem, S. 2012. Biodiversity loss and its impact on humanity. *Nature*, 486: 59-67.

Carlisle, E.M. 1976: In vivo requirement for silicon in articular cartilage and connective tissue formation in the chick. *J Nutr.*, 106: 478-484.

Carpenter, S.R. & Bennett, E.M. 2011. Reconsideration of the planetary boundary for phosphorus. *Environmental Research Letters*, 6: 014009.

Cassman, K.G., Grassini P. & van Wart, J. 2010. Crop yield potential, yield trends, and global food security in a changing climate. In D. Hillel & C. Rosenzweig, eds. *Handbook of Climate Change and Agroecosystems: Impacts, Adaptation, and Mitigation*. pp. 37-51. Imperial College Press. 453 pp.

Ciais P., Reichstein, M., Viovy, N., Granier, N.A., Ogée, J., Allard, V., Buchmann, N., Aubinet, M., Bernhofer, C., Carrara, A., Chevallier, F., De Noblet, N., Friend, A., Friedlingstein, P., Grünwald, T., Heinesch, B., Keronen, P., Knohl, A., Krinner, G., Loustau, D., Manca, G., Matteucci, G., Miglietta, F., Ourcival, J.M., Pilegaard, K., Rambal, S., Seufert, G., Soussana, J.F., Sanz, M.J., Schulze, E.D., Vesala, T. & Valentini, R. 2005. Unprecedented European-level Reduction in Primary Productivity caused by the 2003 Heat and Drought. *Nature*, 437: 529-533.

Clavero, M., Villero, D. & Brotons, L. 2011. Climate Change or Land Use Dynamics: Do we know what climate change indicators indicate? *PlosOne*, 6(4): e 18581.

Coates, J. 2013. Build it back better: Deconstructing food security for improved measurement and action. *Global Food Security*, 2: 188-194.

Cole, D.W. & Rapp, M. 1981. Elemental cycling in forest ecosystems. In D.E. Reichle, ed., *Dynamic properties of forest ecosystems*. pp 341–409. UK, Cambridge, Cambridge University Press.

Compton, J.E., Harrison, J.A., Dennis, R.L., Greaver, T.L., Hill, B.H., Jordan, S.J., Walker, H. & Campbell, H.V. 2011. Ecosystem services altered by human changes in the nitrogen cycle: a new perspective for United States decision making. *Ecology Letters*, 14: 804-815.

Conant, R.T., Ryan, M.G., Ågren, G.I., Birge, H.E., Bradford, M.A., Davidson, E.A., Eliasson, P.E., Evans, S.E., Frey, S.D., Giardina, C.P., Hopkins, F.M., Hyvönen, R., Kirschbaum, M.U.F., Lavallee, J.M., Leifeld, J., Parton, W.J., Steinweg, J.M., Wallenstein, M.D. & Wetterstedt, J.A.M. 2011. Temperature and soil carbon decomposition – synthesis of current knowledge and a way forward. *Global Change Biology*, 17: 3392-3404.

Conley, D.J., Paerl, H.W., Howarth, R.W., Boesch, D.F., Seitzinger, S.P., Havens, K.E., Lancelot, C. & Likens, G.E. 2009. Controlling eutrophication: nitrogen and phosphorus. *Science*, 323: 1014-1015.

Cordell, D. & White, S. 2010. Securing a sustainable phosphorus future for Australia. *Farm Policy Journal*, 7: 1–17.

Corti, T., Wuest, M., Bresch, D. & Seneviratne, S.I. 2011. Drought-induced building damages from simulations at regional scale. *Nat. Hazards Earth Syst. Sci.*, 11: 3335–3342.

Creswell, A. 2012. Warning as three die from soil disease. The Australian. Article of February 15, 2012. (Also available at *http://www.theaustralian.com.au/news/health-science/warning-as-three-die-from-soil-disease/story-e6frg8y* 6-1226271161811)

Crosson, P. 2003. Global consequences of land degradation: An economic perspective. In K. Wiebe, ed. *Land Quality, Agricultural Productivity, and Food Security*. pp. 36-46. UK, Cheltenham, Edward Elgar.

Crutzen, P.J. 2002. Geology of mankind. *Nature*, 415: 23-23.

Chandler, K.R. & Chappell, N.A. 2008. Influence of individual oak (Quercus robur) trees on saturated hydraulic conductivity. *Forest Ecology and Management*, 256(5): 1222-1229.

Chen, J. 2007. Rapid urbanization in China: A real challenge to soil protection and food security. *Catena*, 69: 1-15.

Chen, J. 2012. An original discovery: Selenium deficiency and Keshan disease (an epidemic heart disease). *Asia Pac. J. Clin. Nutr.*, 21: 320-326.

Churchman, G.J. & Landa, E.R. (eds.). 2014. *The Soil Underfoot: Infinite Possibilities for a Finite Resource*. CRC Press.

D'Costa, V.M., McGrann, K.M., Hughes, D.W. & Wright, G.D. 2006. Sampling the antibiotic resistome. *Science*, 311: 374-377.

Dai, Q., Shrubsole, M.J., Ness, R.M., Schlundt, D., Cai, Q., Smalley, W.E., Li, M., Shyr, Y. & Zheng, W. 2007:

The relation of magnesium and calcium intakes and a genetic polymorphism in the magnesium transporter to colorectal neoplasia risk. *Am. J. Clin. Nutr.*, 86: 743-751.

Damoah, R., Spichtinger, N., Servranckx, R., Fromm, M., Eloranta, E.W., Razenkov, I.A., James, P., Shulski, M., Forster, C. & Stohl, A. 2006. A case study of pyro-convection using transport model and remote sensing data. *Atmospheric Chemistry and Physics*, 6: 173-185.

Davidson, E.A., David, M.B., Galloway, J.N., Goodale, C.L., Haeuber, R., Harrison, J. A., Howarth, R.W., Jaynes, D.B., Lowrance, R.R., Nolan, B.T., Peel, J.L., Pinder, R.W., Porter, E., Snyder, C.S., Townsend, A.R. & Ward, M.H. 2012. Excess Nitrogen in the U.S. Environment: Trends, Risks, and Solutions. *Ecological Society of America, Issues in Ecology*, 15: 1-16.

Davin, E.L., Seneviratne, S.I., Ciais, P., Olioso, A., & Wang, T. 2014. Preferential cooling of hot extremes from cropland albedo management. *Proc. Natl. Acad. Sci*., 111(27): 9757-9761.

De Groot, R.S., Wilson, M.A. & Boumans, R.M.J. 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics*, 41: 393-408.

De Jeu, R., Wagner, W., Homes, T.R.H., Dolman, A.J., van de Giesen, N.C. & Friesen, J. 2008. Global soil moisture patterns observed by space borne microwave radiometers and scatterometers. *Surv. Geophys*., 29(4-5): 399-420.

De Vries, F.T. & Bardgett, R.D. 2012a. Plant-microbial linkages and ecosystem N retention: lessons for sustainable agriculture. *Frontiers in Ecology and the Environment*, 10: 425-432.

De Vries, F.T., Bloem, J., Quirk, H., Stevens, C.J., Bol, R., & Bardgett, R.D. 2012b. Extensive Management Promotes Plant and Microbial Nitrogen Retention in Temperate Grassland. *PLoS ONE*, 7: e 51201.

De Vries, F.T., Liiri, M., Bjørnlund, L., Bowker, M., Christensen, S., Setälä, H., & Bardgett, R.D. 2012c. Land use alters the resistance and resilience of soil food webs to drought. *Nature Climate Change*, 2: 276-280.

De Vries, F.T., Thébault, E., Liiri, M., Birkhofer, K., Tsiafouli, M.A., Bjørnlund, L., Jørgensen, H.B., Brady, M.V., Christensen, S., de Ruiter, P.C., d'Hertefeldt, T., Frouz, J., Hedlund, K., Hemerik, L., Hol, W.H.G., Hotes, S., Mortimer, S.R., Setälä, H., Sgardelis, S.P., Uteseny, K., van der Putten, W.H., Wolters, V. & Bardgett, R.D. 2013a. Soil food web properties explain ecosystem services across European land use systems. *Proceedings of the National Academy of Sciences*, 110: 14296–14301.

De Vries, W., Groenenberg, J.E., Lofts, S., Tipping, E. & Posch, M. 2013b. Critical loads of heavy metals for soils. *In* B.J. Alloway, ed. *Heavy metals in soils: trace metals and metalloids in soils and their bioavailability*. 3rd edition. pp. 211-237. The Netherlands, Dordrecht, Springer. 587 pp.

Deguines, N., Jono, C., Baude, M., Henry, M., Julliard, R. & Fontaine, C. 2014. Large- scale trade- off between agricultural intensification and crop pollination services. *Frontiers in Ecology and the Environment*, 12: 212-217.

Del Mar A.M., Torrecillas, E., Torres, P., Garcı´a-Orenes, F. & Roldan, A. 2012. Long-Term Effects of Irrigation with Waste Water on Soil AM Fungi Diversity and Microbial Activities: The Implications for Agro-Ecosystem Resilience. *PLoS ONE*, 7(10): e 47680.

Delgado, J.A., Groffman, P.M., Nearing, M.A., Goddard, T., Reicosky, D., Lal, R., Kitchen, N.R., Rice, C.W., Towery, D. & Salon, P. 2011.Conservation practices to mitigate and adapt to climate change. *Journal of Soil and Water Conservation*, 66(4): 118-129.

Delucchi, M.A. 2011. A conceptual framework for estimating the climate impacts of land-use change due to energy crop programs. *Biomass and Bioenergy*, 35(1): 2337-2360.

Den Biggelaar, C., Lal, R., Eswaran, H., Breneman, V.E. & Reich, P.F. 2003. Crop losses to soil erosion at regional and global scales: evidence from plot-level and GIS data. In K. Wiebe, ed. *Land Quality, Agricultural Productivity, and Food Security*. pp. 262-279. UK, Cheltenham, Edward Elgar.

Dentener, F.J., Carmichael, G.R., Zhang, Y., Lelieved, J. & Crutzen, P.J. 1996. Role of mineral aerosol as a reactive surface in the global atmosphere. *Journal of Geophysical Research*, 101: 22869-22890.

Diaz, R.J. & Rosenberg, R. 2008. Spreading dead zones and consequences for marine ecosystems. *Science*, 321: 926-929.

Dirmeyer, P.A., Cash, B.A., Kinter III, J.L., Stan, C., Jung, T., Marx, L., Towers, P., Wedi, N., Adams, J.M., Altshuler, E.L., Huang, B., Jin, E.K., & Manganello, J. 2012. Evidence for enhanced land-atmosphere feedback in a warming climate. *J. Hydrometeorology*, 13: 981-995.

Dirmeyer, P.A., Gao, X., Zhao, M., Guo, Z., Oki, T. & Hanasaki, N. 2006. GSWP-2: Multimodel analysis and implications for our perception of the land surface. *B. Am. Meteorol. Soc.*, 87: 1381—1397.

Dlamini, T.C. & Haynes, R.J. 2004. Influence of agricultural land use on the size and composition of earthworm communities in northern KwaZulu-Natal, South Africa. *Applied Soil Ecology*, 27: 77-88.

Dorigo, W.A., Xaver, A., Vreugdenhil, M., Gruber, A., Hegyova, A., Sanchis-Dufau, A.D., Zamojski, D., Cordes, C., Wagner, W. & Drusch, M. 2013: Global automated quality control of in situ soil moisture data from the International Soil Moisture Network. *Vadose Zone Journal*, 12.

EEA. 2011. *Ammonia (NH_3) emissions (APE 003)*. Copenhagen, EEA. (Also available at *http://www.eea.europa.eu/data-and-maps/indicators/eea-32-ammonia-nh 3-emissions-1/assessment-1*).

EEA. 2012. *Long-range Transboundary Air Pollution (LRTAP)*. EEA Technical report No 8/2012.

Eggleton, P., Bignell, D.E., Hauser, S., Dibog, L., Norgrove, L. & Madong, B. 2002. Termite diversity across an anthropogenic disturbance gradient in the humid forest zone of West Africa. *Agriculture, Ecosystems and Environment*, 90: 189-202.

EM-DAT. 2010. *The international disasters database*. CRED. (Also available at *http://www.emdat.be*)

Erisman, J.W. & Schaap, M. 2004. The need for ammonia abatement with respect to secondary PM reductions in Europe. *Environmental Pollution*, 129: 159-163.

Erisman, J.W., Bleeker, A., Hensen, A. & Vermeulen, A. 2008. Agricultural air quality in Europe and the future perspective. *Atmospheric Environment*, 42: 3209-3217.

ESPON. 2013. Territorial Dynamics in Europe - Natural Hazards and Climate Change in European Regions. Territorial Observation No. 7. Luxembourg, ESPON. 27 pp.

Esteves, T.C.J., Kirkby, M.J., Shakesby, R.A., Ferreira, A.J.D., Soares, J.A.A., Irvine, B.J., Ferreira, C.S.S., Coelho, C.O.A., Bento, C.P.M. & Carreiras, M.M. 2012. Mitigating land degradation caused by wildfire: Application of the PESERA model to fire-affected sites in central Portugal. *Geoderma*, 191: 40–50.

European Soil Bureau Network. 2005. *Soil Atlas of Europe*. Luxembourg, Office for Official Publications of the European Communities. 128 pp.

Ewen, J., O'Donnell, G., Bulygina, N., Ballard, C. & O'Connell, E. 2013. Towards understanding links between rural land management and the catchment flood hydrograph. *Q. J. R. Meteorol. Soc.*, 139: 350–357.

Falkowski, P.G., Barber, R.T. & Smetacek, V. 1998. Biogeochemical controls and feedbacks on ocean primary production. *Science*, 281: 200-206.

FAO. 1996. *Rome declaration on world food security*. World Food Summit. Rome, FAO.

FAO. 2011. *Forests and Landslides*. Bangkok, FAO, RAP. (available at *http://www.fao.org/docrep/016/ba0126e/ba0126e 00.htm*).

FAO. 2014. *IUSS Working Group WRB. 2014. World Reference Base for Soil Resources 2014. International soil* *classification system for naming soils and creating legends for soil maps. World Soil Resources Reports No. 106*. Rome, FAO.

Fenn, M.E., Poth, M.A., Aber, J.D., Baron, J.S., Bormann, B.T., Johnson, D.W., Lemly, A.D., McNulty, S.G., Ryan, D.F. & Stottlemyer, R. 1998. Nitrogen excess in North American ecosystems: predisposing factors, ecosystem responses, and management strategies. *Ecological Applications*, 8(3): 706-733.

Findell, K.L., Gentine, P., Lintner, B.R. & Kerr, C. 2011. Probability of afternoon precipitation in eastern United States and Mexico enhanced by high evaporation. *Nature Geoscience*, 4: 434-439.

Firbank, L, Bradbury, R.B., McCracken, D.I. & Stoate, C. 2013. Delivering multiple ecosystem services from Enclosed Farmland in the UK. *Agriculture Ecosystems and Environment*, 166: 65–75.

Fischer, E.M., Seneviratne, S.I., Vidale, P.L., Lüthi, D. & Schär, C. 2007. Soil moisture - atmosphere interactions during the 2003 European summer heatwave. *J. Climate*, 20: 5081-5099.

Flynn H.C. & Smith P. 2010. *Greenhouse Gas Budgets of Crop Production and the Mitigation Potential of Nutrient Management*. UK, York, International Fertiliser Society.

Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E.A., Kucharik, C.J., Monfreda, C., Patz, J.A., Prentice, I.C., Ramankutty, N. & Snyder, P.K. 2005. Global consequences of land use. *Science*, 309: 570-574.

Foley, J.A., Ramankutty, N., Brauman, K.A., Cassidy, E.S., Gerber, J.S., Johnston, M., Mueller, N.D., O'Connell, C., Ray, D.K., West, P.C., Balzer, C., Bennett, E.M., Carpenter, S. R., Hill, J., Monfreda, C., Polasky, S., Rockström, J., Sheehan, J., Siebert, S., Tilman, D. & Zaks, D.P.M. 2011. Solutions for a cultivated planet. *Nature*, 478: 337-342.

Forster, P., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R., Fahey, D.W., Haywood, J., Lean, J., Lowe, D.C., Myhre, G., Nganga, J., Prinn, R., Raga, G., Schulz, M. & Van Dorland, R. 2007. Changes in Atmospheric Constituents and in Radiative Forcing. *In*: S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller, eds. *Climate Change 2007: The Physical Science Basis*. pp. 133–134. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press.

Fowler, D., Coyle, M., Skiba, U., Sutton, M.A., Cape, J.N., Reis, S., Sheppard, L.J., Jenkins, A., Grizzetti, B., Galloway, J.N., Vitousek, P., Leach, A., Bouwman, A.F., Butterbach-Bahl, K., Dentener, F., Stevenson, D., Amann, M. & Voss, M. 2013. The global nitrogen cycle in the twenty-first century. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 368(1621): 20130164.

Friedlingstein, P., Cox, P., Betts, R., Bopp, L., Von Bloh, W., Brovkin, V., Cadule, P., Doney, S., Eby, M., Fung, I., Bala, G., John, J., Jones, C., Joos, F., Kato, T., Kawamiya, M., Knorr, W., Lindsay, K., Matthews, H.D., Raddatz, T., Rayner, P., Reick, C., Roeckner, E., Schnitzler, K.G., Schnur, R., Strassmann, K., Weaver, A.J., Yoshikawa, C. & Zeng, N. 2006. Climate–Carbon Cycle Feedback Analysis: Results from the C 4MIP Model Intercomparison. *J. Climate*, 19: 3337—3353.

Fundel, F., Jörg-Hess, S. & Zappa, M. 2013. Monthly hydrometeorological ensemble prediction of streamflow droughts and corresponding drought indices. *Hydrological and Earth System Sciences*, 17(1-1): 395–407.

Galloway, J.N., Aber, J.D., Erisman, J.W., Seitzinger, S.P., Howarth, R.W., Cowling, E.B. & Cosby, B.J. 2003: The nitrogen cascade. *Bioscience*, 53: 341-356.

Gardi, C., Jeffery, S. & Saltelli, A. 2013. An estimate of potential threats levels to soil biodiversity in EU. *Global Change Biology*, 19: 1538-1548.

Gasso, V., Sörensen, C.A.G., Oudshoorn, F.W. & Green, O. 2013. Review, Controlled traffic farming: A review of the environmental impacts. *Europ. J. Agronomy*, 48: 66–73.

Ghassemi, F., Jakeman, A.J. & Nix, H.A. 1995. *Salinisation of land and water resources: human causes, extent,* *management and case studies*. Australia, Canberra, The Australian National University & UK, Oxfordshire, Wallingford, CAB International.

Greaver, T.L., Sullivan, T.J., Herrick, J.D., Barber, M.C., Baron, J.S., Cosby, B.J., Deerhake, M.E., Dennis, R.L., Dubois, J.-J. B., Goodale, C.L., Herlihy, A.T., Lawrence, G.B., Liu, L., Lynch, J.A. & Novak, K.J. 2012. Ecological effects of nitrogen and sulfur air pollution in the US: what do we know? *Front. Ecol. Environ.*, 10: 365-372.

Green, R.E., Cornell, S.J., Scharlemann, J.P. & Balmford, A. 2005. Farming and the fate of wild nature. *Science*, 307(5709): 550-555.

Gruber, S. & Haeberli, W. 2007. Permafrost in steep bedrock slopes and its temperature related destabilization following climate change. *Journal of Geophysical Research*, 112: F 02S 18.

Guillod, B., Davin, E.L., Kündig, C., Smiatek, G. & Seneviratne, S.I. 2013: Impact of soil map specifications for European climate simulations. *Climate Dynamics*, 40: 123-141.

Guillod, B.P., Orlowsky, B., Miralles, D., Teuling, A.J., Blanken, P.D., Buchmann, N., Ciais, P., Ek, M., Findell, K.L., Gentine, P., Lintner, B.R., Scott, R.L., van den Hurk, B. & Seneviratne, S.I. 2014. Land surface controls on afternoon precipitation diagnosed from observational data: uncertainties and confounding factors. *Atmos. Chem. Phys.*, 14(16): 8343-8367.

Guo, J.H., Liu, X.J., Zhang, Y., Shen, J.L., Han, W.X., Zhang, W.F., Christie, P., Goulding, K.W.T., Vitousek, P.M. & Zhang, F.S. 2010. Significant acidification in major Chinese croplands. *Science*, 327: 1008-1010.

Guzzetti, F., Peruccacci, S., Rossi, M. & Stark, C. 2008. The rainfall intensity-duration control of shallow landslides and debris flows: an update. *Landslides*, 5: 3-17.

Hall, J., Arheimer, B., Borga, M., Brázdil, R., Claps, P., Kiss, A., Kjeldsen, T.R., Kriaučiūnienė, J., Kundzewicz, Z.W., Lang, M., Llasat, M.C., Macdonald, N., McIntyre, N., Mediero, L., Merz, B., Merz, R., Molnar, P., Montanari, A., Neuhold, C., Parajka, J., Perdigão, R.A.P., Plavcová, L., Rogger, M., Salinas, J. L., Sauquet, E., Schär, C., Szolgay, J., Viglione, A. & Blöschl, G. 2014. Understanding flood regime changes in Europe: a state-of-the-art assessment. *Hydrol. Earth Syst. Sci.*, 18: 2735-2772.

Hallett, P.D., Feeney, D.S., Bengough, A.G., Rillig, M.C., Scrimgeour, C.M. & Young, I.M. 2009. Disentangling the impact of AM fungi versus roots on soil structure and water transport. *Plant and* Soil, 314: 183-196.

Hampton, V., Kaestli, M., Mayo, M., Choy, J.L., Harrington, G., Richardson, L., Benedict, B., Noske, R., Garnett, S.T., Godoy, D., Spratt, B.G. & Currie, B.J. 2011. Melioidosis in birds and *Burkholderia pseudomalle*[i] dispersal, Australia. *Emerg. Infect. Dis.*, 17: 1310-1312.

Handa, I.T., Aerts, R., Berendse, F., Berg, M.P., Bruder, A., Butenschoen, O., Chauvet, E., Gessner, M.O., Jabiol, J., Makkonen, M., McKie, B.G., Malmqvist, B., Peeters, E.T.H.M., Scheu, S., Schmid, B., van Ruijven, J., Vos, V.C.A. & Hattenschwiler, S. 2014. Consequences of biodiversity loss for litter decomposition across biomes. *Nature*, 509: 218-221.

Hansen, J. & Nazarenko, L. 2004. Soot climate forcing via snow and ice albedo. *Proceedings of the National Academy of Science*, 101: 423-428.

Harris, C., Arenson, L.U., Christiansen, H.H., Etzelmüller, B., Frauenfelder, R., Gruber, S., Haeberli, W., Hauck, C., Hölzle, M., Humlum, O., Isaksen, K., Kääb, A., Kern-Lütschg, Lehning, M.A., Matsuoka, M., Murton, N., Nötzli, J.B., Phillips, J., Ross, M., Seppälä, N., Springman, M., Vonder, S.M. & Mühll, D. 2009. Permafrost and climate in Europe: Monitoring and modelling thermal, geomorphological and geotechnical responses. *Earth Science Reviews*, 92: 117–171.

Hart, R. 2003. Dynamic pollution control—time lags and optimal restoration of marine ecosystems. *Ecological Economics*, 47: 79–93.

Heemsbergen, D.A., Berg, M.P., Loreau, M., van Haj, J.R., Faber, J.H. & Verhoef, H.A. 2004. Biodiversity effects on soil processes explained by interspecific functional dissimilarity. *Science*, 306: 1019-1020.

Heimann, M. & Reichstein, M. 2008. Terrestrial ecosystem carbon dynamics and climate feedbacks. *Nature*, 451: 289-292.

Heimsath, A., Dietrich, W., Nishiizumi, K. & Finkel, R. 1997. The soil production function and landscape equilibrium. *Nature*, 388: 358-361

Hiederer, R. & Köchy, M. 2012. *Global Soil Organic Carbon Estimates and the Harmonized World Soil Database*. EUR 25225 EN. Publications Office of the European Union.79 pp.

Hilton, R., Galy, A., Hovius, N., Chen, M.C., Horng, M.J. & Chen, H. 2008. Tropical-cyclone-driven erosion of the terrestrial biosphere from mountains. *Nature Geosci*., 1: 759-762.

Hirschi, M., Seneviratne, S.I., Alexandrov, V., Boberg, F., Boroneant, C., Christensen, O.B., Formayer, H., Orlowsky, B. & Stepanek, P. 2011. Observational evidence for soil-moisture impact on hot extremes in southeastern Europe. *Nature Geoscience*, 4: 17-21.

Hohenegger, C., Brockhaus, P., Bretherton, C. & Schär, C. 2008. The soil moisture-precipitation feedback in simulations with explicit and parameterized convection. *J. Climate*, 22(19): 5003-5020.

Hopkins, F.M., Torn, M.S. & Trumbore, S.E. 2012. Warming accelerates decomposition of decadal-cycling soil carbon. *Proceedings of the National Academy of Sciences*, 109: 1753-1761.

Houlton, B.Z., Boyer, E., Finzi, A., Galloway, J., Leach, A., Liptzin, D., Melillo, J., Rosenstock, T.S., Sobota, D. & Townsend, A.R. 2013. Intentional versus unintentional nitrogen use in the United States: Trends, efficiency and implications. *Biogeochemistry*, 114: 11-23.

Howden, N.J.K., Burt, T.P., Worrall, F., Whelan, M.J. & Bieroza, M. 2010. Nitrate concentrations and fluxes in the River Thames over 140 years (1868–2008): are increases irreversible? *Hydrological Processes*, 24: 2657-2662.

Huang, P., Xie, S.-P., Hu, K., Huang, G. & Huang, R. 2013. Patterns of the seasonal response of tropical rainfall to global warming. *Nat. Geosci*., 6(5): 357–361.

Huang, R. & Fan, X. 2013. The landslide story. *Nature Geosci*., 6: 325–326.

Hubacek, K., Guan, D., Barrett, J. & Wiedmann, T. 2009. Environmental implications of urbanization and lifestyle change in China: Ecological and Water Footprints. *Journal of Cleaner Production*, 17(14): 1241-1248.

Huggel, C., Clague, J.J. & Korup, O. 2012. Is climate change responsible for changing landslide activity in high mountains? *Earth Surf. Process. Landf*., 37(1): 77–91.

Huntingford, C., Zelazowski, P., Galbraith, D., Mercado, L.M., Sitch, S., Fisher, R., Lomas, M., Walker, A.P., Jones, C.D., Booth, B.B.B., Malhi, Y., Hemming, D., Kay, G., Good, P., Lewis, S.L., Phillips, O.L., Atkin, O.K., Lloyd, J., Gloor, E., Zaragoza-Castells, J., Meir, P., Betts, R., Harris, P.P., Nobre, C., Marengo, J. & Cox, P.M. 2012. Simulated resilience of tropical rainforests to CO_2-induced climate change. *Nature Geoscience*, 6: 268-273.

IPCC. 2012. Managing the Risks of Extreme Events and Disasters to Advance Climate Change Adaptation. A Special Report of Working Groups I and II of the Intergovernmental Panel on Climate Change. UK, Cambridge, Cambridge University Press & USA, NY, New York. 582 pp.

Itoha, S., Katao, N., Eguchib, T., Katao, N. & Takahashia, S. 2014. Radioactive particles in soil, plant, and dust samples after the Fukushima nuclear accident. *Soil Science and Plant Nutrition* , 60: 540–550.

Iverson, R.M. 2000. Landslide triggering by rain infiltration. *Water Resour. Res*., 36(7): 1897–1910.

Jacobson, M.Z. 2001. Strong radiative heating due to the mixing state of black carbon in atmospheric aerosols. *Nature*, 409: 695-697.

Jarvis, N., Koestel, J., Messing, I., Moeys, J. & Lindahl, A. 2013. Influence of soil, land use and climatic factors on the hydraulic conductivity of soil. *Hydrology and Earth System Sciences*, 17: 5185-5195.

Jeffrey S. & van der Putten, W.H. 2011. Soil borne human diseases. European Commission, Joint Research Centre Scientific and Technical Report. Luxembourg, Publications Office of the European Union.

Jeong, S.-J., Ho, C.-H., Piao, S., Kim, J., Ciais, P., Lee, Y.-B., Jhun, J.-G. & Park, S.K. 2014. Effects of double cropping on summer climate of the North China Plain and neighbouring regions. *Nature Climate Change*, 4(7): 615-619..

Joosten, H., Sirin, A., Couwenberg, J., Laine, J. & Smith, P. 2014. The role of peatlands in climate regulation. In A. Bonn, T. Allott, M. Evans, H. Joosten, & R. Stoneman, eds. *Peatland restoration and ecosystem services*. UK, Cambridge, Cambridge University Press.

Jordan, S., Stoffer, J. & Nestlerode, J. 2011. Wetlands as sinks for reactive nitrogen at continental and global scales: A meta-analysis. *Ecosystems*, 14: 144-155.

Jorgenson, M.T., Shur, Y.L. & Pullman, E.R. 2006. Abrupt increase in permafrost degradation in Arctic Alaska. *Geophys. Res. Lett.*, 33: L 02503.

Jugdaohsingh, R., Tucker, K.L., Qiao, N., Cupples, L.A., Kiel, D.P. & Powell, J.J. 2004: Dietary Silicon Intake Is Positively Associated With Bone Mineral Density in Men and Premenopausal Women of the Framingham Offspring Cohort. *J. Bone Miner. Res.*, 19: 297-307.

Kaufman, Y.J., Tanre D. & Boucher, O. 2002. A satellite view of aerosols in the climate system. *Nature*, 419: 215-223.

Kawagoe, S., Kazama, S. & Sarukkalige, P.R. 2009. Assessment of snowmelt triggered landslide hazard and risk in Japan. *Cold Reg. Sci. Technol.*, 58: 120–129.

Keating, B.A. & Carberry, P.S. 2010. Sustainable production, food security and supply chain implications. *Asp. Appl. Biol*. 102: 7–20.

Keating, B.A., Herrero, M., Carberry, P.S., Gardner, J. & Cole, M.B. 2014. Food wedges: Framing the global food demand and supply challenge towards 2050. *Global Food Security*, 3: 125-132.

Keefer, D.K. 2002. Investigating landslides caused by earthquakes:a historical review. *Surv. Geophys.*, 23: 473–510.

Keiler, M. 2013. World-wide trends in natural disasters. *In* P. Bobrowsky, ed., *Encyclopaedia of Natural Hazards*. pp. 1111-1114. Springer.

Khan, M.A., Islam, M.R., Panaullah, G.M., Duxbury, J.M., Jahiruddin, M. & Loeppert, R.H. 2009. Fate of irrigating-water arsenic in rice soils or Bangladesh. *Plant Soil*, 322: 263-277.

Khan, S., Hamra, R.A. & Mu, J. 2009. Water management and crop production for food security in China: a review. *Agricultural Water Management*, 96(3): 349-360.

Klironomos, J.N. 2002. Feedback with soil biota contributes to plant rarity and invasiveness in communities. *Nature*, 417: 67–70.

Koohafkan, P. & Altieri, M.A. 2011. *Globally important agricultural heritage systems, a legacy for the future*. Rome, FAO.

Korkmaz, M., Uzgo, E., Bakirdere, S., Aydin, F. & Ataman, Y. 2007. Effects of dietary boron on cervical cytopathology and on micronucleus frequency in exfoliated buccal cells. *Environ. Toxicol.*, 22: 17-25.

Koster, R.D. & Mahanama, S.P.P. 2012. Land surface controls on hydroclimatological means and variability. *J. Hydrometeorology*, 13: 1604-1620.

Koster, R.D., Dirmeyer, P.A., Guo, Z.C., Bonan, G., Chan, E., Cox, P., Gordon, C.T., Kanae, S., Kowalczyk, E., Lawrence, D., Liu, P., Lu, C.H., Malyshev, S., McAvaney, B., Mitchell, K., Mocko, D., Oki T., Oleson, K., Pitman, A., Sud, Y.C., Taylor, C.M., Verseghy, D., Vasic, R., Xue, Y.K. & Yamada, T. 2004. Regions of strong coupling between soil moisture and precipitation. *Science*, 305: 1138–1140.

Koster, R.D., Mahanama, S., Yamada, T.J., Balsamo, G., Boisserie, M., Dirmeyer, P., Doblas-Reyes, F., Gordon, C.T., Guo, Z., Jeong, J.H., Lawrence, D., Li, Z., Luo, L., Malyshev, S., Merryfield, W., Seneviratne, S.I., Stanelle, T., van den Hurk, B., Vitart, F. & Wood, E.F. 2010a. The contribution of land initialization to subseasonal forecast skill: First results from the GLACE-2 Project. *Geophys. Res. Lett.*, 37: L 02402.

Koster, R.D., Mahanama, S.P.P., Livneh, B., Lettenmeier, D.P. & Reichle, R.H. 2010b. Skill in streamflow forecasts derived from large-scale estimates of soil moisture and snow. *Nature Geoscience*, 3: 613-616.

Kozak, L. & Niedzielski, P. 2013.The evolution of December 2004 tsunami deposits: Temporal and spatial distribution of potentially toxic metalloids. *Chemosphere*, 93: 1856–1865.

Kuno, T., Hatano, Y., Tomita, H., Hara, A., Hirose, Y., Hirata, A., Mori, H., Terasaki, M., Masuda, S. & Tanaka, T. 2012. Organomagnesium suppresses inflammation-associated colon carcinogenesis in male Crj: CD-1mice. *Carcinogenesis*, 34: 361-369.

Kurasawa, R., Kubori, S., & Okuizumi, H. 2010. The Symptoms and Pathogenesis of Zinc Deficiency. *Biomed. Res. Trace Elem.*, 21: 1-12.

Lal, R. 1988. Effects of macrofauna on soil properties in tropical ecosystems. *Agriculture, ecosystems & environment*, 24(1): 101-116.

Lal, R. 2003. Soil degradation and global food security: A soil science perspective. In K. Wiebe, ed. *Land Quality, Agricultural Productivity, and Food Security*. pp. 16-35. UK, Cheltenham, Edward Elgar.

Lal, R. 2004. Soil carbon sequestration impacts on global climate change and food security. *Science* 304: 1623-1627.

Lal, R. 2006. Enhancing crop yields in the developing countries through restoration of the soil organic carbon pool in agricultural lands. *Land Degradation & Development*, 17: 197-209.

Larsen, I. J., Montgomery, D.M. & Korup, O. 2010. Landslide erosion controlled by hillslope material. *Nature Geosci.*, 3: 247-251.

Le Mer, J. & Roger, P. 2001. Production, oxidation, emission and consumption of methane by soils: A review. *Eur. J. Soil Biol.*, 37: 25–50.

Lehmann, P. & Or, D. 2012. Hydromechanical triggering of landslides: From progressive local failures to mass release. *Water Resour. Res.*, 48: W 03535.

Lofts, S., Chapman, P.M., Dwyer, R.,; Mclaughlin, M.J., Schoeters, I., Sheppard, S.C., Adams, W.J., Alloway, B.J., Antunes, P.M.C., Campbell, P.G.C., Davies, B., Degryse, F., De Vries, W., Farley, K.J., Garrett, R.G., Green, A., Jan, G.B., Hale, B., Harrass, M., Hendershot, W.H., Keller, A., Lanno, R., Tao, L., Liu, W.-X., Yibing, M., Menzie, C., Moolenaar, S.W., Piatkiewicz, W., Reimann, C., Rieuwerts, J.S., Santore, R.C., Sauve, S. Schuetze, G. Schlekat, C., Skeaff, J., Smolders, E., Shu, T., Wilkins, J. & Zhao, F.-J. 2007. Critical loads of metals and other trace elements to terrestrial environments. *Environmental Science and Technology*, 41(18): 6326-6331.

Loveland, P. & Webb, J. 2003. Is there a critical level of organic matter in the agricul- tural soils of temperate regions? A review. *Soil Till. Res.*, 70: 1–18.

Ma, J., Folsom, A.R., Melnick, S.L., Eckfeldt, J.H., Sharrett, A.R., Nabulsi, A.A., Hutchinson, R.G., Metcalf, P.A. 1995: Associations of serum and dietary magnesium with cardiovascular disease, hypertension, diabetes, insulin, and carotid arterial wall thickness: the ARIC study. Atherosclerosis Risk in Communities *Study. J. Clin. Epidemiol*, 48(7): 927-940.

MA. 2005. *Millennium Ecosystem Assessment*. Ecosystems and Human Well-being: Synthesis. Washington, DC, Island Press.

Mahli, Y., Roberts, J.T., Betts, R.A., Killeen, T.J., Li, W. & Nobre, C.A. 2008. Climate change, deforestation, and the fate of the Amazon. *Science*, 319: 169-172.

Malamud, B.D., Turcotte, D.L., Guzzetti, F. & Reichenbach, P. 2004. Landslides, earthquakes and erosion. *Earth and Planetary Science Letters*, 229: 45–59.

Marshall, M.R., Ballard, C.E., Frogbrook, Z.L., Solloway, I., McIntyre, N., Reynolds, B. & Wheater, H.S. 2014. The impact of rural land management changes on soil hydraulic properties and runoff processes: Results from experimental plots in upland UK. *Hydrol. Process*, 28: 2617–2629.

Marshall, M.R., Francis, O.J., Frogbrook, Z.L., Jackson, B.M., McIntyre, N., Reynolds, B., Solloway, I., Wheater, H.S. & Chell, J. 2009. The impact of upland land management on flooding: results from an improved pasture hillslope. *Hydrological Processes*, 23(3): 464-475.

Martin, J.H. 1990. Glacial-interglacial CO_2 exchange: the iron hypothesis. *Paleoceanography*, 5: 1-13.

Maskell, L.C., Crowe, A., Dunbar, M.J., Emmett, B., Henrys, P., Keith, A.M., Norton, L.R., Scholefield, P., Clark, D.B., Simpson, I.C. & Smart, S.M. 2013. Exploring the ecological constraints to multiple ecosystem service delivery and biodiversity. *Journal of Applied Ecology*, 50: 561-571.

McBratney, A., Field, D. J. & Koch, A. 2014. The dimensions of soil security. *Geoderma*, 213: 203-213.

Miller, R. & Tegen, I. 1998. Climate response to soil dust aerosols. *American Meteorological Society*, 11: 3247-3267.

Mitsch, W.J., Day, J.W., Gilliam, J.W., Groffman, P.M., Hey, D.L., Randall, G.W. & Wang, N. 2001. Reducing nitrogen loading to the Gulf of Mexico from the Mississippi River Basin: Strategies to counter a persistent ecological problem. *BioScience*, 51: 373-388.

Monks, P.S., Granier, C., Fuzzi, S., Stohl, A., Williams, M.L., Akimoto, H., Amann, M., Baklanov, A., Baltensperger, U., Bey, I., Blake, N., Blake, R.S., Carslaw, K., Cooper, O.R., Dentener, F., Fowler, D., Fragkou, E., Frost, G.J., Generoso, S., Ginoux, P., Grewe, V., Guenther, A., Hanson, H.C., Henne, S., Hjorth, J., Hofzumahaus, A., Huntrieser, H., Isaksen, I.S.A., Jenkin, M.E., Kaiser, J., Kanakidou, M., Klimont, Z., Kulmala, M., Laj, P., Lawrence, M.G., Lee, J.D., Liousse, C., Maione, M., McFiggans, G., Metzger, A., Mieville, A., Moussiopoulos, N., Orlando, J.J., O'Dowd, C.D., Palmer, P.I., Parrish, D.D., Petzold, A., Platt, U., Poschl, U., Prevot, A.S.H., Reeves, C.E., Reimann, S., Rudich, Y., Sellegri, K., Steinbrecher, R., Simpson, D., ten Brink, H., Theloke, J., van der Werf, G.R., Vautard, R., Vestreng, V., Vlachokostas, Ch. & von Glasow, R. 2009. Atmospheric composition change – global and regional air quality. Atmospheric Environment, 43(33): 5268-5350.

Montgomery, D.R. 2007. Soil erosion and agricultural sustainability. *Proceedings of the National Academy of Sciences of the United States of America*, 104: 13268-13272.

Morford, S.L., Houlton, B.Z., & Dahlgren, R.A. 2011. Increased forest ecosystem carbon and nitrogen storage from nitrogen rich bedrock. *Nature*, 477(7362): 78-81.

Mossy Foot Project. 2014. (Also available at *http://mossyfoot.com/podoconiosis/*).

Mueller, B. & Seneviratne, S.I. 2012. Hot days induced by precipitation deficits at the global scale. *Proc. Natl Acad. Sci.*, 109(31): 12398-12403.

Mulder, C., Cohen, J.E., Setala, H., Bloem, J. & Breure, A.M. 2005. Bacterial traits, organism mass, and numerical abundance in the detrital soil food web of Dutch agricultural grasslands. *Ecology Letters*, 8: 80-90.

Mullan, D. 2013. Soil erosion under the impacts of future climate change: Assessing the statistical significance of future changes and the potential on-site and off-site problems. *Catena*, 109: 234-246.

Naeem, S., Bunker, D.E., Hector, A., Loreau, M. & Perrings, C. 2009. Introduction: the ecological and social implications of changing biodiversity. An overview of a decade of biodiversity and ecosystem functioning research. In S. Naeem, D.E. Bunker, A. Hector, M. Loreau, Perrings, eds., *Biodiversity, Ecosystem Functioning, and Human Wellbeing*. pp. 3-13. UK, Oxford, Oxford University Press.

Nearing, M.A., Jetten, V., Cerdan, O., Couturier, A., Hernandez, M., Bissonnais, Y.Le**, Nicholas, M.H., Nunes, J.P., Renschler, C.S., Souchere, V. & Van Oost, K**. 2005. Modeling response of soil erosion and runoff to changes in precipitation and cover. *Catena*, 61: 131-154.

Nelson, F.E., Anisimov, O.E. & Shiklomonov, O.I. 2001. Subsidence risk from thawing permafrost. *Nature*, 410: 889–890.

Nielsen, F.H. 2010. Magnesium, inflammation, and obesity in chronic disease. *Nutr. Rev*., 68: 333-340.

Nielsen, U., Ayres, E., Wall, D. & Bardgett, R.D. 2011. Soil biodiversity and carbon cycling: a review and synthesis of studies examining diversity-function relationships. *European Journal of Soil Science*, 62: 105-116.

Nilsson, J. & Grennfelt, P. 1988. *Critical loads for sulfur and nitrogen*. Report of the Skokloster workshop. Miljørapport 15. Copenhagen, Nordic Council of Ministers.

O'Connell, E., Ewen, J., O'Donnell, G. & Quinn, P. 2007. Is there a link between agricultural land-use management and flooding? *Hydrology and Earth System Science*, 11: 96–107.

Oki, T. & Kanae, S. 2006. Global hydrological cycles and world water resources. *Science*, 313: 1068–1072.

Oliver, M.A. 1997. Soil and human health. A Review. *European J. Soil Sci*., 48: 573-592.

Olivier, J.G.J., Bouwman, A.F., Van der Hoek, K.W. & Berdowski, J.J.M. 1998. Global air emission inventories for anthropogenic sources of NO_x, NH_3 and N_2O in 1990. *Environmental Pollution*, 102: 135-148.

Orlowsky, B. & Seneviratne, S.I. 2013, Elusive drought: Uncertainty in observed trends and short- and long-term CMIP 5 projections. *Hydr. Earth Syst. Sci*., 17: 1765-1781.

Orth, R., & Seneviratne, S.I. 2013. Predictability of soil moisture and streamflow on sub-seasonal timescales: A case study. *J. Geophysical Res. - Atmospheres*, 118: 10963-10979.

Packer, A. & Clay, K. 2000. Soil pathogens and spatial patterns of seedling mortality in a temperate tree. *Nature*, 404: 278–281.

Palm, C., Blanco-Canqui, H., DeClerck, F. & Gatere, L. 2014. Conservation agriculture and ecosystem services: An overview. *Agriculture, Ecosystems, and Environment*, 187: 87-105.

Palm, C., Sanchez, P., Ahamed, S. & Awiti, A. 2007. Soils: A contemporary perspective. *Annu. Rev. Environ. Resour.*, 32: 99–129.

Papathoma-Köhle, M., Zischg, A.P., Fuchs, S., Glade, T. & Keiler, M. 2015. Loss estimation for landslides in mountain areas – An integrated toolbox for vulnerability assessment and damage documentation. *Environmental Modelling & Software*, 63: 156-169.

Pattison, I. & Lane, S.N. 2011. The link between land-use management and fluvial flood risk: A chaotic conception?. *Progress in Physical Geography*, 36(1): 72-92.

Pepper, I.L., Gerba, C.P., Newby, D.T. & Rice, C.W. 2009. Soil: A public health threat or savior? *Crit. Rev. Environ. Sci. Technol.*, 39: 416-432.

Perakis, S.S., Compton, J.E. & Hedin, L.O. 2005. Nitrogen retention across a gradient of 15N additions to an unpolluted temperate forest soil in Chile. *Ecology*, 86: 96-105.

Peterjohn, W.T. & Correll, D.L. 1984. Nutrient dynamics in an agricultural watershed: Observations on the role of a riparian forest. *Ecology*, 65: 1466-1475.

Petzold, A., Weinzierl, B., Huntrieser, H., Stohl, A., Real, E., Cozic, J., Fiebig, M., Hendricks, J., Lauer, A. & Law, K. 2007. Perturbation of the European free troposphere aerosol by North American forest fire plumes during the ICARTT-ITOP Experiment in summer 2004. *Atmospheric Chemistry and Physics Discussions, European Geo-sciences Union (EGU)*, 7(2): 4925-4979.

Phalan, B., Onial, M., Balmford, A. & Green, R.E. 2011. Reconciling Food Production and Biodiversity Conservation: Land Sharing and Land Sparing Compared. *Science*, 333: 1289-1291.

Pierzynski, G.M., Vance, G.F. & Sims, J.T. 2005. *Soils and Environmental Quality, 3rd Ed. US*, Boca Raton, Taylor & Francis. 569 pp.

Pimentel, D., Harvey, C., Resosudarmo, P., Sinclair, K., Kurz, D., McNair, M., Crist, S., Shpritz, L., Fitton, L. & Saffouri, R. 1995. Environmental and economic costs of soil erosion and conservation benefits. *Science-AAAS-Weekly Paper Edition*, 267: 1117-1122.

Postma-Blaauw, M.B., de Goede, R.G.M., Bloem, J., Faber, J.H. & Brussaard, L. 2010. Soil biota community structure and abundance under agricultural intensification and extensification. *Ecology*, 91: 460–473.

Potter, C.S., Davidson, E.A & Verchot, L.V. 1996. Estimation of global biogeochemical controls and seasonality in soil methane consumption. *Chemosphere*, 32: 2219–2246.

Pretty, J. 2008. Agricultural sustainability: concepts, principles and evidence. *Philosophical Transactions of the Royal Society B-Biological Sciences*, 363: 447-465.

Price, K. 2011. Effects of watershed topography, soils, land use, and climate on baseflow hydrology in humid regions: A review. *Progress in Physical Geography*, 35(4): 465-492.

Priemé, A., Christensen, S., Dobbie, K.E. & Smith, K.A. 1997. Slow increase in rate of methane oxidation in soils with time following land use change from arable agriculture to woodland. *Soil Biol. Biochem.*, 29: 1269–1273.

Prospero, J.M. & Lamb, P.J. 2003. African droughts and dust transport to the Caribbean: climate change implications. *Science*, 302: 1024-1027.

Prospero, J.M. & Nees, R.T. 1978. Dust concentration in the atmosphere of the equatorial North Atlantic: possible relationship to the Sahelian Drought. *Science*, 196: 1196-1198.

Prospero, J.M., Ginoux, P., Torres, O., Nicholson, S.E. & Gill, T.E. 2002. Environmental characterisation of global sources of atmospheric soil dust identified with the NIMBUS 7 total ozone mapping spectrometer (TOMS) absorbing aerosol product. *Reviews of Geophysics*, 40: 1-31.

Quesada, B., Vautard, R., Yiou, P., Hirschi, M. & Seneviratne, S.I. 2012. Asymmetric European summer heat predictability from wet and dry Southern winter/springs. *Nature Climate Change*, 2: 736-741.

Rabotyagov, S.S., Kling, C.L., Gassman, P.W., Rabalais, N.N. & Turner, R.E. 2014. The economics of dead Zones: Causes, impacts, policy challenges, and a model of the Gulf of Mexico hypoxic zone. *Review of Environmental Economics and Policy*, 8: 58-79.

Ramana, M.V., Ramanathan, V., Feng, Y., Yoon, S-C., Kim, S-W. & Carmichael, G.R. 2010. Warming influenced by the ratio of black carbon to sulphate and the black-carbon source. *Nature Geoscience*, 3(8): 542-545.

Ramanathan, V. & Carmichael, G. 2008. Global and regional climate changes due to black carbon. *Nature Geoscience*, 1: 221-227.

Ramaswamy, V., Leovy, C., Rodhe, H., Shine, K., Wang, W.-C., Wuebbles, D., Ding, M., Edmonds, J.A., Fraser, P., Grant, K., Johnson, C., Lashof, D., Leggett, J., Lelieveld, J., McCormick, M.P., Oort, A., Schwarzkopf, M.D., Sutera, A., Warrilow, D.A. & Wigley, T. 2001. Radiative forcing of climate change. *In* J.T. Houghton, Y. Ding, D.J. Griggs, M. Noguer, P.J. van der Linden, X. Dai, K. Maskell & C.A. Johnson, eds. *Climate Change 2001: The Scientific Basis. Contribution of Working Group I to the Third Assessment Report of the*

Intergovernmental Panel on Climate Change. pp. 349–416. UK, Cambridge, Cambridge University Press & USA, NY, New York. 881 pp.

Raufman, Y.J. & Fraser, R.S. 1997. Confirmation of the smoke particles effect on clouds and climate. *Science*, 277: 1636-1639.

Reager, J.T, Thomas, B.F. & Famiglietti, J.S. 2014. River basin flood potential inferred using GRACE gravity observations at several months lead time. *Nature Geoscience*, 7: 588-592.

Reay, D., Davidson, E.A., Smith, K., Smith, P., Melillo, J.M., Dentener, F. & Crutzen, P.J. 2012. Global agriculture and nitrous oxide emissions. *Nature Climate Change*, 2(6): 410-416.

Reffitt, D.M., Ogston N., Jugdaohsingh R., Cheung, H.F., Evans, B.A., Thompson, R.P., Powell, J.J. & Hampson, G.N. 2003. Orthosilicic acid stimulates collagen type 1 synthesis and osteoblastic differentiation in human osteoblast-like cells in vitro. *Bone*, 32: 127-135.

Reichle, R.H. 2008. Data assimilation methods in the Earth Sciences. *Adv. Water Resour.*, 31(11): 1411-1418

Reichstein, M., Bahn, M., Ciais, P., Frank, D., Mahecha, M.D., Seneviratne, S.I., Zscheischler, J., Beer, C., Buchmann, N., Frank, D.C., Papale, D., Rammig, A., Smith, P., Thonicke, K., van der Velde, M., Vicca, S., Walz, A. & Wattenbach, M. 2013. Climate extremes and the carbon cycle. *Nature*, 500: 287-295.

Reynolds, B., Chamberlain, P.M., Poskitt, J., Woods, C., Scott, W.A., Rowe, E.C., Robinson, D.A., Frogbrook, Z.L., Keith, A.M., Henrys, P.A., Black, H.I.J. & Emmett, B.A. 2013. Countryside Survey: National "Soil Change" 1978-2007 for Topsoils in Great Britain-Acidity, Carbon, and Total Nitrogen Status. *Vadose Zone Journal*, 12(2).

Richter, D.D. & Markewitz, D. 2001. *Understanding Soil Change*. Cambridge Univ. Press.

Rillig, M.C. & Mulley, D.L. 2006. Mycorrhizas and soil structure. *New Phytol.*, 171: 41-53.

Robinson, D.A., Hockley, N., Cooper, D.M., Emmett, B.A., Keith, A.M., Lebron, I., Reynolds, B., Tipping, E., Tye, A.M., Watts, C.W., Whalley, W.R., Black, H.I.J., Warren, G.P. & Robinson, J.S. 2013. Natural capital and ecosystem services, developing an appropriate soils framework as a basis for valuation. *Soil Biology and Biochemistry*, 57: 1023-1033.

Rockström, J., Steffen, W., Noone, K., Persson, A., Chapin, F.S., Lambin, E.F., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., de Wit, C.A., Hughes, T., van der Leeuw, S., Rodhe, H., Sorlin, S., Snyder, P.K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R.W., Fabry, V.J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P. & Foley, J.A. 2009. A safe operating space for humanity. *Nature*, 461: 472-475.

Rude, R.K., Singer, F.R. & Gruber, H.E. 2009. Skeletal and hormonal effects of magnesium deficiency. *J. Am. Coll. Nutr.*, 28: 131-141.

Rusinamhodzi, L., Corbeels, M., Zingore, S., Nyamangara, J. & Giller, K.E. 2013. Pushing the envelope? Maize production intensification and the role of cattle manure in recovery of degraded soils in smallholder farming areas of Zimbabwe. *Field Crops Research*, 147: 40-53.

Ryan, J.G., McAlpine, C.A. & Ludwig, J.A. 2010. Integrated vegetation designs for enhancing water retention and recycling in agroecosystems. *Landscape Ecology.*, 25(8): 1277-1288.

Salvucci, G.D., Saleem, J.A. & Kaufmann, R. 2002. Investigating soil moisture feedbacks on precipitation with tests of Granger causality. Adv. Water Resour., 25(8-12): 1305-1312.

Scorei, R. & Popa, R. 2010. Boron-containing compounds as preventive and chemotherapeutic agents for cancer. *Anticancer Agents Med. Chem.*, 10: 346-351.

Scherr, S.J. 1999. *Soil degradation. A threat to developing country food security by 2020? Food Agriculture, and the Environment Discussion Paper 27.* Washington, DC, International Food Policy Research Institute.

Scherr, S.J. 2003. Productivity-related economic impacts of soil degradation in developing countries: an evaluation of regional experience. *In* K. Wiebe, ed. *Land Quality, Agricultural Productivity, and Food Security*. pp. 223-261. UK, Cheltenham, Edward Elgar.

Schimel, D.S. 1995. Terrestrial ecosystems and the carbon cycle. *Global Change Biology*, 1(1): 77–91.

Schlesinger, W.H. & Jasechko, S. 2014. Transpiration in the global water cycle. *Agricultural and Forest meteorology*, 189-190: 115-117.

Schmidt, J. & Dikau, R. 2004. Modeling historical climate variability and slope stability. *Geomorphology*, 60: 433–447.

Schmidt, M.W.I., Torn, M.S., Abiven, S., Dittmar, T., Guggenberger, G., Janssens, I.A., Kleber, M., Kogel-Knabner, I., Lehmann, J., Manning, D.A.C., Nannipieri, P., Rasse, D.P., Weiner, S. & Trumbore, S.E. 2011. Persistence of soil organic matter as an ecosystem property. *Nature*, 478: 49-56.

Schoeneich, P., Dall'Amico M., Deline P. & Zischg A. (eds). 2011. *Hazards related to permafrost and to permafrost degradation*. *PermaNET project, state-of-the-art report* 6.2. On-line publication.

Schreiner, K.M., Bianchi, T.S. & Rosenheim, B.E. 2014. Evidence for permafrost thaw and transport from an Alaskan North Slope watershed. *Geophys. Res. Lett*., 41: 3117–3126.

Schuur, E.A.G., Bockenheim, J.,Canadell, J.P., Euskirchen, E., Field, C.B., Goryachkin, S.V., Hagemann, S., Kuhry, P., Lafleur, P.M., Lee, H., Mazhitova, G., Nelson, F.E., Rinke, A., Romanovsky, V.E., Shiklomanov, N., Tarnocai,C., Venevsky, S., Vogel, J.G., and Zimov, S.A., 2008. Vulnerability of permafrost carbon to climate change: Implications for the global carbon cycle. *Bioscience*, 58(8): 701-714

Schwarz, M., Cohen, D. & Or, D. 2012. Spatial characterization of root reinforcement at stand scale: Theory and case study. *Geomorphology*, 171–172: 190–200.

SEC. 2006. *Impact assessment of the thematic strategy on soil protection. Document accompanying, Thematic Strategy for Soil Protection*. Communication from the commission to the Council, the European Parliament, the European economic and social committee and the committee of the regions. Brussels, SEC.

Seitzinger, S., Harrison, J.A., Bohlke, J.K., Bouwman, A.F., Lowrance, R., Peterson, B., Tobias, C. & Van Drecht, G. 2006. Denitrification across landscapes and waterscapes: A synthesis. *Ecological Applications*, 16: 2064-2090.

Seneviratne, S.I., Corti, T., Davin, E.L., Hirschi, M., Jaeger, E.B., Lehner, I., Orlowsky, B. & Teuling, A.J. 2010. Investigating soil moisture-climate interactions in a changing climate: A review. *Earth-Science Reviews*, 99(3-4): 125-161.

Seneviratne, S.I., Lüthi, D., Litschi, M. & Schär, C. 2006. Land-atmosphere coupling and climate change in Europe. *Nature*, 443: 205-209.

Seneviratne, S.I., Nicholls, N., Easterling, D., Goodess, C.M., Kanae, S., Kossin, J., Luo, Y., Marengo, J., McInnes, K., Rahimi, M., Reichstein, M., Sorteberg, A., Vera, C. & Zhang, X. 2012. Changes in climate extremes and their impacts on the natural physical environment. In C.B. Field, V. Barros, T.F. Stocker, D. Qin, D.J. Dokken, K.L. Ebi, M.D. Mastrandrea, K.J. Mach, G.-K. Plattner, S.K. Allen, M. Tignor, & P.M. Midgley, eds. *Managing the Risks of Extreme Events and Disasters to Advance Climate Change Adaptation*. pp. 109-230. A Special Report of Working Groups I and II of the Intergovernmental Panel on Climate Change.

Seneviratne, S.I., Wilhelm, M., Stanelle, T., van den Hurk, B.J.J.M., Hagemann, S., Berg, A., Cheruy, F., Higgins, M.E., Meier, A., Brovkin, V., Claussen, M., Ducharne, A., Dufresne, J.-L., Findell, K.L., Ghattas, J., Lawrence, D.M., Malyshev, S., Rummukainen, M. & Smith, B. 2013. Impact of soil moisture-climate feedbacks on CMIP 5 projections: First results from the GLACE-CMIP 5 experiment. *Geophys. Res. Lett*., 40(19): 5212-5217.

Shcherbak, I., Millar, N. & Robertson, G.P. 2014. Global meta-analysis of the nonlinear response of soil nitrous oxide (N_2O) emissions to fertilizer nitrogen. *Proceedings of the National Academy of Sciences*, 111: 9199-9204.

Sheffield, J. & Wood, E.F. 2007. Projected changes in drought occurrence under future global warming from multi-model, multi-scenario, IPCC AR 4 simulations. *Climate Dynamics*, 31: 79-105.

Sidle, R.C. & Ochiai, H. 2006. *Landslides: Processes, Prediction and Land Use*. Washington, DC, American Geophysical Union.

Silkworth, J.B., Culter, D.S. & Sack, G. 1989. Immunotoxicity of 2,3,7,8-tetracholordibenzo-p-dioxin in a complex environmental mixture from the Love Canal. *Toxicol. Sci.*, 12:303-312.

Simsek, A., Velioglu, S.Y., Coskun, L.A. & Sayli, B.S. 2003. Boron concentrations in selected foods from borate-producing regions in Turkey. *J. Sci. Food Agric.*, 83: 586-592.

Sitch, S., Huntingford, C., Gedney, N., Levy, P.E., Lomas, M., Piao, S.L., Betts, R., Ciais, P., Cox, P., Friedlingstein, P., Jones, C.D., Prentice, I.C. & Woodward, F.I. 2008. Evaluation of the terrestrial carbon cycle, future plant geography and climate-carbon cycle feedbacks using five Dynamic Global Vegetation Models (DGVMs). *Global Change Biology*, 14: 2015–2039.

Smith, P. & Olesen, J.E. 2010. Synergies between mitigation of, and adaptation to, climate change in agriculture. *Journal of Agricultural Science*, 148: 543-552.

Smith, P. 2004. Soils as carbon sinks - the global context. *Soil Use and Management*, 20: 212-218.

Smith, P. 2012. Soils and climate change. *Current Opinion in Environmental Sustainability*, 4: 539–544.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H.H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, R.J., Sirotenko, O., Howden, M., McAllister, T., Pan, G., Romanenkov, V., Schneider, U., Towprayoon, S., Wattenbach, M. & Smith, J.U. 2008. Greenhouse gas mitigation in agriculture. *Philosophical Transactions of the Royal Society*, B., 363: 789-813.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H.H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, R.J., Sirotenko, O., Howden, M., McAllister, T., Pan, G., Romanenkov, V., Rose, S., Schneider, U. & Towprayoon, S. 2007a. Agriculture. In B. Metz, O.R. Davidson, P.R. Bosch, R. Dave & L.A. Meyer, eds. *Climate change 2007: Mitigation. Contribution of Working group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Chapter. 8. pp. 497-540. UK, Cambridge, Cambridge University Press and New York, USA.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H.H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, R.J., Sirotenko, O., Howden, M., McAllister, T., Pan, G., Romanenkov, V., Schneider, U. & Towprayoon, S. 2007b. Policy and technological constraints to implementation of greenhouse gas mitigation options in agriculture. *Agriculture, Ecosystems & Environment*, 118: 6-28.

Snyder, C.S., Davidson, E.A., Smith, P. & Venterea, R.T. 2014. Agriculture: sustainable crop and animal production to help mitigate nitrous oxide emissions. *Current Opinion in Environmental Sustainability*, 9–10: 46–54.

Sombroek, W., Kern, D., Rodrigues, T., Cravo, M., Jarbas, T., Woods, W. & Glaser, B. 2002. *Terra Preta and Terra Mulata: pre Columbian Amazon kitchen middens and agricultural fields, their sustainability and their replication*. Proceedings 17th World Congress of Soil Science. Thailand, Bangkok.

Sparks, D.L. 2003. *Environmental Soil Chemistry, 2nd Ed. US*, San Diego, Academic Press. 352 pp.

Sposito, G. 2008. *The Chemistry of Soils, 2nd Ed*. Chap. 11. US, New York, Oxford Univ. Press.

Steffen, W., Richardson, K., Rockström, J., Cornell, S.E., Fetzer, I., Bennett, E. M., Biggs, R., Carpenter, S.R., de Vries, W., de Wit, C.A., Folke, C., Gerten, D., Heinke, J., Mace, G.M., Persson, L.M., Ramanathan,

V., Reyers, B. & Sörlin, S. 2015. Planetary boundaries: Guiding human development on a changing planet. *Sciencexpress* 10.1126/science.1259855.

Steinhauser, G. 2014. Fukushima's Forgotten Radionuclides: A Review of the Understudied Radioactive Emissions. *Environ. Sci. Technol.*, 48: 4649–4663.

Steudler, P.A., Jones, R.D., Castro, M.S., Melillo, J.M. & Lewis, D.L. 1996. Microbial controls of methane oxidation in temperate forest and agricultural soils. In J.C. Murrell & D.P. Kelly, eds. *Microbiology of Atmospheric Trace Gases*, pp. 69–84. Springer.

Stocking, M.A. 2003. Tropical soils and food security: the next 50 years. *Science*, 302: 1356-1359.

Sutton, M.A., Oenema, O., Erisman, J.W., Leip, A., van Grinsven, H. & Winiwarter, W. 2011. Too much of a good thing. *Nature*, 472: 159-161.

Sylvain, Z.A. & Wall D.H. 2011. Linking soil biodiversity and vegetation: implications for a changing planet. *Am. J. Bot.*, 98(3): 517-27.

Takemura, T., Nakajima,T., Bubovik,O., Holben, B. & Kinne S. 2002. Single scattering albedo and radiative forcing of various aero- sol species with a global three dimensional model. *J. Climat*e, 15: 333–352.

Tal, A. 2006. Seeking sustainability. Israel's Evolving Water Management Strategy. *Science*, 313: 1081 – 1084.

Tarnocai, C., Canadell, J.G., Schuur, E.A.G., Kuhry, P., Mazhitova, G. & Zimov, S. 2009. Soil organic carbon pools in the northern circumpolar permafrost region. *Global Biogeochem. Cycles*, 23: GB 2023.

Taylor, C.M., de Jeu, R.A.M., Guichard, F., Harris, P.P. & Dorigo, W.A. 2012. Afternoon rain more likely over drier soils. *Nature*, 489: 423-426.

Templer, P.H., Mack M.C., Chapin III F.S., Christenson L.M., Compton J.E., Crook H.D., Currie W.S., Curtis, C., Dail, B., D'Antonio, C.M., Emmett, B.A., Epstein, H., Goodale, C.L., Gundersen, P., Hobbie, S.E., Holland, K., Hooper, D.U., Hungate, B.A., Lamontagne, S., Nadelhoffer, K.J., Osenberg, C.W., Perakis, S.S., Schleppi, P., Schimel, J., Schmidt, I.K., Sommerkorn, M., Spoelstra, J., Tietema, A., Wessel, W.W. & Zak, D.R. 2012. Sinks for nitrogen inputs in terrestrial ecosystems: a meta-analysis of 15N tracer field studies. Ecology, 93(8): 1816-1829.

Teuling, A.J., Hirschi, M., Ohmura, A., Wild, M., Reichstein, M., Ciais, P., Buchmann, N., Ammann, C., Montagnani, L., Richardson, A.D., Wohlfahrt, G. & Seneviratne, S.I. 2009. A regional perspective on trends in continental evaporation. *Geophys. Res. Lett.*, 36: L 02404.

Teuling, A.J., Seneviratne, S.I., Stöckli, R., Reichstein, M., Moors, E., Ciais, P., Luyssaert, S., van den Hurk, B., Ammann, C., Bernhofer, C., Dellwik, E., Gianelle, D., Gielen, B., Grünwald, T., Klumpp, K., Montagnani, L., Moureaux, C., Sottocornola, M. & Wohlfahrt, G. 2010. Contrasting response of European forest and grassland energy exchange to heatwaves. *Nature Geoscience*, 3: 722-727.

The economics of ecosystems and biodiversity (TEEB). 2014. (Also available at http://www.teebweb.org/)

Toland, A.R. & Wessolek, G. 2014. Picturing soils aesthetic approaches to raising soil awareness in Contemporary Art. In G.J. Churchman & E.R. Landa, ed., Th*e Soil Underfoot: Infinite Possibilities for a Finite Resource*. USA, Boca Raton, FL, CRC Press.

Treseder, K.K. 2008. Nitrogen additions and microbial biomass: a meta-analysis of ecosystem studies. *Ecology Letters*, 11: 1111-1120.

Troeh, F.R., Hobbs, J.A. & Donahue, R.L. 1991. *Soil and Water Conservation*. USA, Englewood Cliffs, NJ, Prentice Hall.

Trustrum, N.A. & De Rose, R.C. 1988. Soil depth-age relationship of landslides on deforested hillslopes, Taranaki, New Zealand. *Geomorphology*, 1: 143-160.

Tubiello, F.N., Salvatore, M., Rossi, S., Ferrara, A., Fitton, N. & Smith, P. 2013. The FAOSTAT database of greenhouse gas emissions from agriculture. *Environmental Research Letters*, 8: 015009.

UNEP. 2012. *Policy Implications of Warming Permafrost*. UNEP.

UNISDR. 2009. *UNISDR terminology on disaster risk reduction*. UNISDR.

Van der Ent, R.J., Savenije, H.H.G., Schaefli, B. & Steele-Dunne, S.C. 2010. Origin and fate of atmospheric moisture over continents. *WRR*, 46: W 09525.

Van der Heijden, M., Bardgett, R. & van Straalen, N. 2008. The unseen majority: soil microbes as drivers of plant diversity and productivity in terrestrial ecosystems. *Ecology Letters*, 11(3): 296-310.

Van der Heijden, M.G.A., Klironomos, J.N., Ursic, M., Moutoglis, P., Streitwolf-Engel, R., Boller, T., Wiemken, A. & Sanders, I.R. 1998. Mycorrhizal fungal diversity determines plant biodiversity, ecosystem variability and productivity. *Nature*, 396: 72–75.

Van der Hoek, K.W. 1998. Estimating ammonia emission factors in Europe: summary of the work of the UNECE ammonia expert panel. Atmospheric. *Environment*, 32: 315-316.

Van der Putten, W.H., Bardgett, R.D., Bever, J.D., Bezemer, T.M., Casper, B.B., Fukami, T., Kardol, P., Klironomos, J.N., Kulmatiski, A., Schweitzer, J.A., Suding, K.N., Van de Voorde, T.F.J. & Wardle, D.A. 2013. Plant–soil feedbacks: the past, the present and future challenges. *Journal of Ecology*, 101: 265-276.

Van Grinsven, H.J., Holland, M., Jacobsen, B.H., Klimont, Z., Sutton, M.A., & Willems, W.J. 2013. Costs and benefits of nitrogen for Europe and implications for mitigation. *Environmental Science & Technology*, 47: 3571-3579.

Van Groenigen, K.J., Qi, X., Osenberg, C.W., Luo, Y. & Hungate, B.A. 2014. Faster decomposition under increased atmospheric CO_2 limits soil carbon storage. *Science*, 344: 508-509.

Van Kauwenbergh, S. 2010. *World phosphate rock reserves and resources*. USA, Alabama, Muscle Shoals, International Fertilizer Development Center.

Van Oost, K., Quine, T.A., Govers, G., De Gryze, S., Six, J., Harden, J.W., Ritchie, J.C., McCarty, G.W., Heckrath, G., Kosmas, C., Giraldez, J.V., da Silva, M.J.R. & Merckx, R. 2007. The impact of agricultural soil erosion on the global carbon cycle. *Science*, 318(5850) : 626-629.

Várallyay, G. 1994. Climate change, soil salinity and alkalinity. In M.D.A. Sounsevell & R.J. Loveland, eds. Soil Responses to Climate Change. NATO ASI Series. Vol. 23, pp. 3–11. Springer-Verlag, London.

Vautard, R., Yiou, P., D'Andrea, F., de Noblet, N., Viovy, N., Cassou, C., Polcher, J., Ciais, P., Kageyama, M. & Fan, Y. 2007. Summertime European heat and drought waves induced by wintertime Mediterranean rainfall deficit. *Geophys. Res. Lett.*, 34 (7): L 07711.

Vince, G. & Raworth, K. 2012. Living in the doughnut. *Nature Climate Change*, 2: 225-226.

Von Ruette, J., Lehmann, P. & Or, D. 2014. Effects of rainfall spatial variability and intermittency on shallow landslide triggering patterns at a catchment scale. *Water Resour. Res.*, 50(10): 7780-7799.

Vonk, J.E., Sánchez-García, L., van Dongen, B.E., Alling, V., Kosmach, D., Charkin, A., Semiletov, I.P., Dudarev, O.V., Shakhova, N., Roos, P., Eglinton, T.I., Andersson, A. & Gustafsson, Ö. 2012. Activation of old carbon by erosion of coastal and subsea permafrost in Arctic Siberia. *Nature*, 489: 137–140.

Wagg, C., Bender, S.F., Widmer, F. & van der Heijden, M.G.A. 2014. Soil biodiversity and soil community composition determine ecosystem multifunctionality. *Proceedings of the National Academy of Sciences*, 111: 5266-5270.

Wagner, W., Bloschl, G., Pampaloni, P., Calvet, J.C., Bizzarri, B., Wigneron, J.P. & Kerr, Y. 2007. Operational readiness of microwave remote sensing of soil moisture for hydrological applications. *Nord. Hydrol.*, 38: 1-20.

Wang, G.L. 2005. Agricultural drought in a future climate: results from 15 global climate models participating in the IPCC 4th assessment. *Clim. Dynam.*, 25: 739-753.

Wardle, D.A., Bardgett, R.D., Callaway, R.M. & van der Putten, W.H. 2011. Terrestrial ecosystem responses to species loss and gain. *Science*, 332: 1273-1277.

Warren, S.G. & Wiscombe, W.J. 1980. A model for the spectral albedo of snow. II. Snow containing atmospheric aerosols. *Journal of Atmospheric Sciences*, 37: 2734-2745.

Wei, J., Dirmeyer, P.A., Wisser, D., Bosilovich, M.G. & Mocko, D.M. 2013. Where Does the Irrigation Water Go? An Estimate of the Contribution of Irrigation to Precipitation Using MERRA. *J. Hydrometeorology*, 14: 275-289.

Weller, D.E. & Baker, M.E. 2014. Cropland Riparian Buffers Throughout Chesapeake Bay Watershed: Spatial patterns and effects on nitrate loads delivered to streams. *Journal of the American Water Resources Association*, 50: 696-712.

Wheater, H. & Evans, E. 2009. Land use, water management and future flood risk. *Land Use Policy*, 26: 251-264.

Wisser, D., Fekete, B.M., Vörösmarty, C.J. & Schumann, A.H. 2010. Reconstructing 20th century global hydrography: A contribution to the Global Terrestrial Network–Hydrology (GTN–H). *Hydrol. Earth Syst. Sci.*, 14: 1–24.

WMO. 2005. *Climate and Land Degradation*. World Meteorological Organization. ISBN 92-63-10989-3 No. 989

Yang, X., & Post, W.M. 2011. Phosphorus transformations as a function of pedogenesis: A synthesis of soil phosphorus data using Hedley fractionation method. *Biogeosciences*, 8(10): 2907-2916.

Yazbeck, C., Kloppmann, W., Cottier, R., Sahuquillo, J., Debotte, G. & Huel, G. 2005. Health impact evaluation of boron in drinking water: a geographical risk assessment in Northern France. *Environ. Geochem. Health*, 27: 419-427.

Yermiyahu, U., Tal, A., Ben-Gal, A., Bar-Tal, A., Tarchitzky, J. & Lahav, O. 2007. Rethinking desalinated water quality and agriculture. *Science*, 318: 920-921.

Zimmermann, B., Elsenbeer, H., & De Moraes, J.M. 2006. The influence of land-use changes on soil hydraulic properties: implications for runoff generation. *Forest ecology and management*, 222(1): 29-38.

8 | Governance and policy responses to soil change

8.1 | Introduction

This chapter provides an overview of policy and governance responses to soil change. While most attention is given to issues at the global, regional and national levels, it is emphasized that effective responses nearly always have a basis in local action by individual land managers. Indeed, understanding the interconnectedness and the consequences of actions at each level is central to effective governance and policy.

This book, and in particular the regional assessments of soil change (Chapters 9 to 16), demonstrate that at the global scale there is a qualitative appreciation of the pressures on soil resources but limited consistent evidence on their condition and trajectories of change. These assessments reveal that some of the world's soil management challenges are immediate, obvious and serious – they arise partly because of the nature of soils in different regions and their associated history of land management. Other problems are more subtle but equally important in the long term – they require vigilance and a sustained policy response over decades. At present, few countries have effective policies to deal with these problems. In short, the world's soils need to support at least a 70 percent increase in food production by 2050 (FAO, 2011) but there are some fundamental uncertainties. For example:

Is there enough arable land with suitable soils to feed the world in coming decades?

Are soil constraints partly responsible for the apparent yield plateau for major crops?

Can changes to soil management have a significant impact on the seemingly unsustainable global demand for nutrients?

Can changes to soil management have a significant impact on atmospheric concentrations of greenhouse gases without jeopardising other functions such as food and fibre production?

Will the extent and rate of soil degradation threaten food security and the provision of ecosystem services in the coming decades?

Can water-use efficiency be improved through better soil management in key regions facing water scarcity?

How will climate change interact with the distribution of soils to produce new patterns of land use?

A comprehensive global view is needed to respond to these questions. A comprehensive view is also needed to deal with the trans-national aspects of food security and soil degradation. Through trade, most urbanised people are protected from local resource depletion. The area of land and water used to support a global citizen is scattered all over the planet. As a consequence, soil degradation and loss of production are not just local or national issues – they are genuinely international.

The consideration of soil in policy formulation has been weak in most parts of the world. Reasons for this weakness include the following.

Lack of ready access to the evidence needed for policy action.

The challenge of dealing with a natural resource that is often privately owned but is at the same time an important public good.

The long-time scales involved in soil change – some of the most important changes take place over decades and they can be difficult to detect. As a result, communities and institutions may not respond until critical and irreversible thresholds have been exceeded.

Perhaps even more significant for policy makers is the disconnection between our increasingly urbanized human societies and the soil. The task of developing effective policies to ensure sustainable soil management is neither simple to articulate nor easy to implement. This is true regardless of a country's stage of development, its natural endowment of soil resources, or the threats to its soil function.

8.2 | Soils as part of global natural resources management

In setting the stage, it is useful to examine the major drivers, pressures and institutional responses to land use and then set these within the broader international sustainable development agenda (see Table 8.1).

8.2.1 | Historical context

The 'Great Dust Bowl' of the 1930s in the United States of America was pivotal because it triggered widespread public concern about land use, degradation and the need for sustainable management. Severe wind erosion resulted from the opening up of vast areas for cereal production through mechanisation, with associated loss of protective vegetation cover. In response, the Soil Conservation Service of the USDA was established in 1935. This served as a model for many other countries facing similar issues (Young, 1994). In 1937, the United States President Franklin D. Roosevelt famously stated 'The nation that destroys its soil destroys itself.' This is perhaps the most succinct and sharpest challenge for policy makers and it remains an all-too-real contemporary challenge for policy makers in many countries.

After World War Two, many countries experienced food shortages and governments responded by increasing their investments in agricultural research. Understandably, most of this research focussed on increasing crop yields and food production. During this period there was also a large investment in soil and land resource surveys, particularly in Africa and Asia. In the following decades, soil science was strongly supported, with diverse institutional responses emerging in different regions. In some countries, there was close integration with other aspects of natural resource management while in others, separate soil agencies were established. The FAO played an important role in developing influential technical standards (e.g. FAO, 1976) and supporting within-country programs that aimed to establish sustainable soil and land management. The production of the FAO-UNESCO (1980) Soil Map of the World was a landmark achievement.

The success of the Green Revolution (Borlaug, 2000) along with large increases in crop yields in North America and Europe eventually led to less investment by public agencies in agricultural science and related activities. The emphasis shifted to environmental issues, a transition which occurred during the 1970s and 1980s, particularly in developed western countries. During the 1990s and 2000s, disinvestment in soil science was widespread and many soil departments in universities or governments were either closed or incorporated into natural resource or environmental units. The UN commitment to soil resources through the FAO and related agencies was also scaled back dramatically.

The food price rises in 2007 and 2008 shocked many policymakers out of the belief that stable or declining food prices and assured supplies could be taken for granted (Beddington *et al.*, 2012). This period also marked the start of a critical re-examination of the capacity of the world's soil resources to support sustainable agriculture, assist with climate regulation, and safeguard ecosystem services and biodiversity. Before exploring this topic in more detail, it is useful to review some of the key global agreements relating to soils that emerged from the 1980s onwards.

8.2.2 | Global agreements relating to soils

In 1982, the FAO adopted the World Soil Charter and UNEP published the World Soils Policy (FAO, 1982; UNEP, 1982). It has been difficult to assess the practical impact of these initiatives. Nevertheless, the principles and definitions provided useful guidance for national governments that pursued actions on sustainable soil management.

The first United Nations Conference on Environment and Development (UNCED, 1992, also known as the 'Earth Summit') launched the global environmental agenda (Table 8.1). The UN Convention to Combat Desertification (UNCCD) addressed issues of desertification, land degradation and drought; the UN Framework to Combat Climate Change (UNFCCC) was to tackle climate change; and the Convention on Biological Diversity (CBD) dealt with the challenges of biodiversity conservation and sustainable use (CBD). Supported by the Global Environment Facility (GEF), these conventions have raised awareness and mobilised increased efforts by countries and partners to generate global environmental benefits. These conventions also cover, albeit with less prominence, issues of soil conservation, sustainable land management and land use change, taking into account human as well as ecological perspectives (Hurni *et al.*, 2006).

The ecosystem approach promoted by the CBD between 1998 and 2004 (CBD, 2014), recognised that human management is central to biodiversity conservation and sustainable use. This ecosystem approach was further developed in the Millennium Ecosystem Assessment of 2005. This paved the way amongst international agencies and donor funds for more integrated ecosystem approaches in agriculture. These approaches emphasized the need for sectoral integration, with increased attention given to the benefits of mixed agroforestry and agro-silvo-pastoral systems. Similar approaches had already been developed in many countries. Globally, there was a trend towards the use of incentive measures to encourage land users to adopt sustainable practices which not only enhance production but also maintain biodiversity and ecosystem services (FAO, 2007; MA, 2005; UNEP, 2004). Soils came to be seen in relation to the services they provide for human well-being and poverty reduction. However, compared to other functions, soil-related matters did not feature prominently in policy or programmes.

Following the food crisis in 2008, policy makers at the international level began to appreciate that soils were finite and an important factor that had to be considered in the debate on food security. Within the framework of UNCCD's 'Zero Net Land Degradation', discussions were initiated about the need for quantitative targets and indicators to measure soil degradation (UNCCD, 2012). Concerns over food insecurity, water scarcity, climate change and increasing pressures on limited land and water resources led to much greater dialogue, advocacy and partnerships supporting integrated approaches to this complex set of issues (Beddington *et al.*, 2012; Steffen *et al.*, 2015).

The UN Conference on Sustainable Development (Rio+20), took place in June 2012, two decades after the Earth Summit. In the resulting document, The Future We Want, the international community agreed on the need to achieve a land degradation neutral world in the context of sustainable development (UN, 2012). The conference also initiated the process of developing universal Sustainable Development Goals (SDGs).

All of the above developments relating to soils and land degradation are framed by the broader issue of climate change. Again, there is a long institutional history but it is useful to start with the establishment of the Intergovernmental Panel on Climate Change (IPCC) in 1988 by the UN Environment Programme (UNEP) and the World Meteorological Organization (WMO). The IPCC provides the world with scientific and technical information on climate change and its socio-economic impacts. The next major development was adoption of the Kyoto Protocol in 1997 by the UNFCCC. The Protocol, which entered into force in 2005, committed industrialized countries to stabilize greenhouse gas emissions, in particular carbon dioxide (CO_2). The Protocol started as a non-binding agreement but later progressed to legally binding agreements on emission reduction targets. The Protocol is of great importance for soils and land management because soils are important carbon sinks. The Protocol recognized opportunities for better management of carbon stores and for the enhancement of carbon sequestration in forestry and agriculture. There was thus clear recognition that soil management can be a vehicle to achieve climate goals – and conversely, that soils can be managed to avoid the loss of carbon through land degradation. Because of the climate system's sensitivity to soil processes, soil-related issues are set to attract increasing attention in future climate agreements.

In recent years, FAO and its member countries have made significant progress in supporting strategies and policies to improve global governance of soil resources. In order to meet the need for a multilateral agreement focusing specifically on soil challenges, and to advocate for sustainable soil and land management at global level, the Global Soil Partnership1 (GSP) was proposed by FAO and the EU and then established in September 2011. The GSP strives to raise awareness among decision makers on the role of soil resources in relation to food security, climate change, and the provision of ecosystem services (Montanarella and Vargas, 2012). Technical and scientific guidance is provided by the Intergovernmental Technical Panel on Soils (ITPS). The ITPS complements related scientific advisory panels including the Intergovernmental Panel on Climate Change (IPCC), the Intergovernmental Panel on Biodiversity and Ecosystem Services (IPBES), and the UNCCD's Science-Policy Interface (SPI).

The ITPS has been key to the development of the Plans of Action for the five pillars of the Global Soil Partnership (Table 8.2). It has also been engaged in the development of the Sustainable Development Goals and the initiation of formal reporting mechanisms, including the present book. An indication of the emerging priority accorded to soils and a measure of the impact of the GSP was the declaration by the United Nations General Assembly of 2015 as the International Year of Soils.

1 www.fao.org/globalsoilpartnership

Table 8.1 | Recent Milestones in soil governance and sustainable development

Year	
1982	FAO World Soil Charter
1988	Intergovernmental Panel on Climate Change (IPCC)
1992	UN Conference on Environment and Development
	Rio Declaration
	Agenda 21
	Global Environmental Facility
	UN Convention to Combat Desertification (UNCCD)
	UN Framework to Combat Climate Change (UNFCCC)
	Convention on Biological Diversity (CBD)
1997	Kyoto Protocol
2000	Millennium Development Goals (MDGs)
2005	Millennium Ecosystem Assessment
2008	UNCCD's Zero Net Land Degradation
2011	Global Soil Partnership initiated (FAO/EU)
2012	Rio+20
	Sustainable Development Goals (SDGs) and Post-2015 Development Agenda
2013	Intergovernmental Technical Panel on Soils (ITPS) of the GSP
	Updated FAO World Soil Charter
	Land and Soils integrated in the Open Working Group of the Sustainable Development Goals
	Regional Soil Partnerships of the GSP
2015	International Year of Soils declared by the UN General Assembly

Table 8.2 | The 5 Pillars of Action of the Global Soil Partnership.

Pillar No.	Action
1	Promote sustainable management of soil resources for soil protection, conservation and sustainable productivity
2	Encourage investment, technical cooperation, policy, education awareness and extension in soil
3	Promote targeted soil research and development focusing on identified gaps and priorities and on synergies with related productive, environmental and social development actions
4	Enhance the quantity and quality of soil data and information: data collection, analysis, validation, reporting, monitoring and integration with other disciplines
5	Harmonize methods, measurements and indicators for the sustainable management and protection of soil resources

8.3.1 | Sustainable soil management – criteria and supporting practices

International agreements on soil and land resources are helpful but they are all to no avail unless there are complementary policies and coordinated activities at regional, national, district and local levels. Appropriate and effective policies need to reflect the local context in terms of the natural resource issues, culturally acceptability and economic feasibility. However, a unifying scientific narrative is also needed. In broad terms, the criteria for determining whether a landscape is functioning effectively and whether soils are being managed sustainably are as follows.

Leakage of nutrients is low.

Biological production is high relative to the potential limits set by climate and water availability.

Levels of biodiversity within and above the soil are relatively high.

Rainfall is efficiently captured and held within the root zone.

Rates of soil erosion and deposition are low, with only small quantities being transferred out of the system.

Contaminants are not introduced into the landscape and existing contaminants are not concentrated to levels that cause harm.

Systems for producing food and fibre for human consumption do not rely on large net inputs of energy

Net emissions of Greenhouse Gases are zero or less.

We can manage what we can measure, so the task is to ensure that the above criteria can be measured against locally appropriate benchmarks. Without this information, policy makers and land managers do not have indicators of whether they are moving towards sustainability or going backwards. Policy makers also require an appreciation of how soil and land management practices can be applied to achieve desired outcomes. Regardless of the level of mechanization and technological sophistication, farming practices in general need to (FAO, 2013):

Minimize soil disturbance by avoiding mechanical tillage in order to maintain soil organic matter, soil structure and overall soil function.

Enhance and maintain a protective organic cover on the soil surface, using cover crops and crop residues, in order to protect the soil surface, conserve water and nutrients and promote soil biological activity.

Cultivate a wide range of plant species – both annuals and perennials – in associations, sequences and rotations that include trees, shrubs, pastures and crops, in order to enhance crop nutrition and improve system resilience.

Use well-adapted varieties with resistance to biotic and abiotic stresses and with improved nutritional quality, and to plant them at an appropriate time, seedling age and spacing.

Enhance crop nutrition and soil function through crop rotations and judicious use of organic and inorganic fertilizer.

Ensure integrated management of pests, diseases and weeds using appropriate practices, biodiversity and selective, low-risk pesticides when needed.

Manage water efficiently.

Control machines and field traffic to avoid soil compaction.

Thousands of different soil and land management practices have been developed around the world in response to local biophysical, social and cultural settings (e.g. WOCAT, 2007). Most cultures have deep connections with the land, and soil is venerated in diverse ways (Churchman and Land, 2014). In many regions, traditional knowledge still plays an important role in determining land management. However, most traditional systems have been disrupted or modified for a wide range of reasons. The two most common reasons have been the loss of access to land (e.g. invasion and displacement; increasing population densities causing shorter fallow periods on smaller areas; loss of access to grazing lands) and the arrival of new technologies.

8.3.2 | Education about soil and land use

Regardless of the culture or landscape setting, knowledge of soil and land resources is the foundation for achieving sustainable soil management (Dalal-Clayton and Dent, 2001). Spreading knowledge about soils requires formal education, preferably at all levels of schooling. Some countries are developing comprehensive and imaginative curricula that use an understanding of soils as a basis for teaching a wide range of cultural, social, scientific and economic subjects. At a more advanced level, training is needed in a range of soil science sub-disciplines (e.g. soil physics, soil chemistry, soil biology and pedology), Training in soil science needs to be linked to related disciplines including geology, ecology, forestry, agronomy, hydrology and other environmental sciences. Mechanisms for outreach, vocational training and extension are also needed.

Policy makers need to ensure that education systems provide sufficient understanding and training for a nation to achieve sustainable soil management. In particular, farmers and others directly involved in soil management require sufficient knowledge to manage their soils profitably and sustainably.

8.3.3 | Soil research, development and extension

The second key area where policy makers have responsibility is in relation to research, development and extension. The pioneering work of the Soil Conservation Service in the United States and the technical innovations of the Green Revolution are two examples that demonstrate the power of agricultural science and technology. The Green Revolution also highlights how trade-offs are required when there is a focus on a single ecosystem service (food production) at the expense of others (e.g. water quality). Contemporary science policy often focuses on impact and public benefit. In this regard, soil research is often considered simply as a means to an end. Although soil science is vital to several important ends, notably agriculture, environment, water management and climate change, it is often overlooked in priority setting exercises. More formal recognition of soil resources as a cross-cutting issue in science policy is necessary to ensure it receives sufficient support. The recent Australian initiative to achieve a more integrated view of soil research, development and extension is instructive in this regard (Australian Government, 2014).

8.3.4 | Private benefits, public goods and payments for ecosystem services

The amount of regulation on land use and management varies substantially between countries depending largely on the degree of government intervention. Effective regulations on land use and management require a good information base for setting critical limits, implementing various zoning schemes and monitoring compliance. In practice, regulating soil management practices (e.g. application of manure, moderating or increasing fertilizer use, control of dryland salinity) and implementing zoning systems (e.g. to protect the best agricultural soils) involves complex technical, institutional and policy challenges.

Countries that rely less on regulation often opt for incentive schemes to achieve outcomes. Incentives can range from subsidy systems (e.g. for fertilizer in poor countries or for equipment for conservation tillage in developed countries) through to various forms of certification for the adoption of specified soil management practices (e.g. organic farming). Some of these systems have strong economic drivers because they are mandatory for market access (e.g. participation in supply chains to supermarkets).

Implementing effective policies requires organized systems for monitoring soil conditions and an understanding of the relationship between soils and land management. Without this basic information, policy makers have no way of knowing whether regulations and incentive schemes are achieving the desired result.

8.3.5 | Intergenerational equity

Ensuring intergenerational equity is becoming more difficult as human pressures on soil resources reach critical limits. Most traditional cultures and systems of family farming have strong cultural norms that ensure tribal lands or family farms are passed to the next generation in the same or better condition than when they were inherited. However, dramatic changes to land management associated with intensive agriculture, the adoption of Green Revolution technologies, and intensification of land use more generally, are having a major impact on soil resources. The area of arable land per capita is decreasing sharply (0.45 ha in 1961, 0.25 ha in 2000 and a forecast of 0.19 ha in 2050). Future generations will inherit a radically modified land and soil resource.

Many countries have sophisticated reporting systems for assessing issues relating to intergenerational equity (e.g. long-term forecasts to determine the viability of pension and health systems; decadal plans for critical infrastructure). Scenario analysis and futures forecasting are essential to national preparedness and long-term sustainability. There is now an imperative for policy makers to assess the current trends in soil condition and natural resource scarcity summarised in this book and to factor in the consequences to scenario analysis and futures forecasting.

8.3.6 | Land degradation and conflict

Land degradation and resource scarcity can play a role in the rise of conflicts, but these conflicts are rarely purely resource driven. Where tensions about access and use of natural resources do exist, they depend on a variety of factors – the outcomes of which may sometimes cascade from tension into violent conflict, but certainly not always. More often than not, natural resource degradation is a result of conflict rather than a cause. The existence of land degradation can also lead people to seek cooperative solutions. Policy makers and others involved in land management can not only act to resolve resource conflicts but also help to prevent them and to find peaceful mutually acceptable solutions (Frerks *et al.*, 2014; Bernauer, Böhmelt and Koubi, 2012).

8.4 | Regional soil policies

8.4.1 | Africa

Africa has a diverse range of soils and land use systems. However, very large areas, particularly in West Africa, are infertile or of low fertility, and unsustainable systems of land use are widespread. A leading cause of low fertility is nutrient depletion (Smaling, 1993; Stoorvogel, Smaling and Janssen, 1993). This is considered to be the chief biophysical factor limiting small-scale farm production (Drechsel, Giordano and Gyiele, 2004) although other factors including limited organic matter and erosion are significant as well (Bossio, Geheb and Critchley, 2010). Mounting concern over these issues contributed to the creation of the New Partnership for Africa's Development (NEPAD). This is a vision and policy framework produced by the African Union (AU) that aims to provide member countries with guidance over their development agenda. Within NEPAD, the Comprehensive Africa Agriculture Development Programme (CAADP) sets out an agenda targeting annual growth of 6 percent in agricultural production. The Abuja Declaration on Fertilizers, agreed in 2006, laid out the vision for an African Green Revolution. Central to this was the aim of increasing the level of fertilizer application from 8 kg ha^{-1} to 50 kg ha^{-1}. However, only slow progress has been made in implementing this agenda at regional and country level (NEPAD, 2012).

Food policy and agricultural development in Africa pose challenges beyond the scope of this book. However, there are some promising developments even for countries facing the most daunting difficulties owing to rapid population growth, very low incomes, weathered and infertile landscapes, low levels of literacy, vulnerability to climate variability and change, disease and significant potential for social unrest. Two of these promising developments have been supported by the Bill and Melinda Gates Foundation.

First is the AGRA Soil Health Programme which aims to increase income and food security by promoting the wide adoption of integrated soil fertility management (ISFM) practices among smallholder farmers and creating an enabling environment for wide adoption of these improved practices across sub-Saharan Africa. The objective is to improve supply and access to appropriate fertilizers, as well as access to knowledge on IFSM for over four million smallholders and to strengthen extension and advisory capacity. The Programme also seeks to influence national policy in favour of investment in fertilizer and ISFM. Some 1.8 million smallholders are reported to be using ISFM, including fertilizer micro-dosing, manure and legumes in crop rotations, with yields in the Sahel up three to fourfold in good seasons.

The second promising initiative is the AgWaterSolutions Project. The project concept builds on the existence of sizable untapped groundwater systems in the region and on the recent availability of small affordable motorized water pumps. The project promotes small-scale distributed irrigation systems that rely primarily on groundwater. In these systems, the access point for water, the distribution system and the irrigated crop all occur at or near the same location. These systems are typically privately owned and managed by individuals or small groups. The potential in countries such as Burkina Faso is large. This initiative is helping to shift the attention of policy makers and planners away from large scale irrigation developments.

There are many other significant soil policy issues facing the region. Examples include: the costs and benefits of subsidy schemes for fertilizers; the growing pressure on land resources and the consequent shortening of fallow periods; the challenge of making inputs affordable and ensuring market access in areas where poverty is prevalent; and addressing urban and peri-urban planning so that more intensive and safe food production systems can develop in and around the rapidly growing African cities.

8.4.2 | Asia

In regions of rapid development in Asia, urbanization, industrialization and intensive land use lead to unbalanced use of agro-chemicals, poor waste management and acid deposition caused by urban air pollution. These factors have contributed to increasing soil contamination and acidification. In China, for example, a soil pollution survey found that 6.4 million square kilometres of arable land are contaminated and that this represents an alarming threat to human health (Yue, 2014). In consequence, China's Environmental Protection Law was revised and strengthened in 2014. However, in China and all across the region the greatest environmental challenges arise from the gap between legislation and implementation (Mu *et al.*, 2014).

In recent years government policy responses across the region have encouraged improved land use practices that increased tree cover for carbon sequestration. Carbon-financing schemes have been implemented. However, government policies have been less effective in dealing with the issue of foreign investment in agricultural land. In some countries, foreign companies have begun a variety of contractual arrangements with local farmers, resulting in some cases in the loss of land for smallholders (Fox *et al.*, 2011).

8.4.3 | Europe

Europe has well-established and strong formal governance mechanisms to address environmental issues at regional, national and sub-national levels. European Union (EU) environmental policies are agreed at central level but legislated and implemented at the national level. However, the experience with soils policy has been more complex and only a handful of member states have specific legislation on soil protection. With the objective of protecting soils across Europe, the European Commission adopted a Soil Thematic Strategy in 2006 which consists of a communication, a proposal for a framework directive (under European Union legislation) and an impact assessment (EC, 2006). The proposal for a Soil Framework Directive would require member states to adopt a systematic approach to identifying and combating soil degradation. However, this could not be agreed by the required majority in the European Council and the draft Directive was consequently withdrawn by the European Commission at the end of 2014. The failure to adopt the directive was largely due to concerns about subsidiarity, with some member states maintaining that soil was not a matter to be negotiated at the European level. Others felt that the cost of the directive would be too high, and that the burden of implementation would be too heavy. However, the Seventh EU Environment Action Plan, which entered into force in 2014, recognises the severe challenge of soil degradation. It provides that by 2020 land in the EU should be managed sustainably, soil should be adequately protected, and the remediation of contaminated sites should be well underway. Furthermore it commits the EU and its member states to increasing efforts to reduce soil erosion, to increase soil organic matter and to remediate contaminated sites (EC, 2013).

8.4.4 | Eurasia

Eurasian countries have well-developed environmental policies and regulations. However following the break-up of the Soviet Union, the system of environmental monitoring and conservation collapsed and has only recently been partially restored. Countries all across the region have maintained and even improved environmental and soil conservation legislation in recent years, but in most countries the mechanisms for quality control and environmental monitoring have been weakened. For example, only Belarus and Uzbekistan maintain their soil survey institutes, and even there soil monitoring has been discontinued.

Ukraine, Russia and Kazakhstan are the countries with the largest under- or unused agricultural lands in the world. The World Bank (2011) states that these countries have the capacity to meet the growing global demand for food. In Russia in 2002 the area of abandoned land reached 70 million ha. Since then there has been a slow decrease in the area of unused land (Nefedova, 2013). However, it should be noted that most land abandonment occurred in badlands, wetlands, steep slopes and areas with an unfavourable climate, while in areas with fertile soils the investment in land management increased. In countries such as Ukraine and Georgia, where land tenure legislation allows land ownership by non-residents, foreign capital is being invested in farmland. Non-transparent land grabs on a large scale are expected to increase, and might have far reaching consequences for the livelihoods of the rural population (Visser and Spoor, 2010).

8.4.5 | Latin America and the Caribbean (LAC)

This region is one of the richest in the world in terms of natural resources. However, rapid exploitation and export of these resources (minerals, gas, forests, and pastures) is occurring with associated dramatic land use changes and widespread land degradation. Nonetheless, some countries in the region have developed and implemented good policies and approaches to mitigate land degradation. These policies, implemented at national and sub-national levels, are good practice examples that could be replicated in other countries in the region (UNEP, 2012).

Uruguay provides a good example of soil and land conservation policies: here the soil conservation policy was designed by the Ministry of Livestock, Agriculture and Fisheries (MGAP) within a programme promoting agricultural intensification, with the objective of implementing a sustainable intensification model. Under this policy, crop producers must submit soil management plans and state the rotation sequence on each plot. They must stay within the maximum tolerable soil erosion amount based on local soil characteristics (Hill, Mondelli and Carrazzone, 2014).

Another example is Cuba's National Environmental Strategy of 2011/2015 which characterizes soil degradation as one of the fundamental environmental challenges in the country. The Cuban government has also implemented action plans to fight desertification and, since 2001, has undertaken programmes for soil conservation (CITMA, 2011). Brazil's Forest Code was updated in 2012: it establishes general standards for protection of forests and other native resources, including soil and water resources. The Forest Code also integrates legal and economic incentives to promote sustainable production activities. However, closer analysis of the updated Forest Code suggests that it may in fact allow more deforestation than the previous version, in response to the demands of agricultural intensification (Soares-Filho *et al.*, 2014).

8.4.6 | The Near East and North Africa (NENA)

This region is considered as the most water scarce and arid region in the world. Moreover, given the scarcity of land and water resources, this region is particularly vulnerable to the impacts of climate change, increasing drought, declining soil fertility and consequently declining agricultural production (Wingkvist and Drakenberg, 2010; Drine, 2011). There are government programs to improve land management in several countries, especially countries that are party to international agreements and are in receipt of donor support. Most actions promoting sustainable land management have been to combat desertification under the framework of the UNCCD (UNCCD, 2012).

Despite significant improvements in the region in tackling the root cause of land degradation, there are still challenges in enforcing environmental regulations and implementing environmental conservation policies. The main implementation constraints are: the weakness of institutions at all levels; the difficulty of coordinating action across sectors, themes, donors and stakeholders; the lack of participation of the local communities; and tenure insecurity.

MENARID (Integrated Natural Resources Management in the Middle East and North Africa) is a partnership working for improvement of the governance of natural resources, including water. MENARID supports restoration of natural resources. In particular, the programme aims to improve the livelihoods of target communities through the restoration of degraded natural resources, including land and soils. It offers a platform for coordination between stakeholders and information sharing in the countries (ICARDA, 2013).

The NENA region is endowed with oil and gas reserves. In areas of rapid urbanization and oil production, soil pollution and soil sealing are associated challenges. Parts of the region are extremely sensitive to political conflicts, and peace and post-conflict are the main focus. Land degradation issues become more pronounced, but inevitably they have to take second place to other concerns.

8.4.7 | North America

In the United States, federal policies favour market-based instruments within an overall environmental governance framework, and these instruments have superseded traditional regulatory instruments. Land use is a priority issue on the political agenda, due to its contribution to GDP through forestry and agriculture. Governments diminish environmental impacts by paying land managers to implement sustainable land management practices and soil conservation. Taxes and incentives encourage land and farmland preservation programmes through payment for ecosystem services (UNEP, 2012).

The United States Conservation Reserve Program (CRP) pays farmers to remove land from agricultural production in order to prevent soil erosion and improve ecosystem functions. This set-aside generates economic benefits of around US$1.3 billion per year (Hellerstein, 2010). However, high prices have made agriculture more profitable and the rates of payment from CRP have not risen so fast. The amount of land enrolled in the programme is therefore expected to decline (Wu and Weber, 2012). The Environmental Quality Incentives Program and the Observation Security Program of 2002 are other programmes that reward farmers for applying sustainable land management practices. It has been estimated that soil erosion could be reduced by 17 percent, saving around 36 million tonnes of soil annually. Valued at US$2 per tonne, the cost of conservation would thus be US$34 million annually, compared to the cost of restoring the soils, estimated at up to US$332 million (Hellerstein, 2010).

In Canada land-use planning is a provincial responsibility and legislation differs widely among provinces. British Columbia has a long-standing Agricultural Land Reserve Program that prohibits development on approximately 4.7 million ha of agricultural land throughout the province. In the early 2000s Ontario created a Greenbelt that protects 0.7 million ha of agricultural and natural lands in the most populated region surrounding Toronto. Generally in Canada the implementation of Payment for Ecosystem Services needs still to be complemented with land use planning frameworks in order to become more effective at all levels of government (Calbick, Day and Gunton, 2003).

8.4.8 | Southwest Pacific

The scale of land degradation across the countries of the Southwest Pacific has given rise to a range of significant policy responses all with a strong emphasis on participative engagement and local action. Perhaps most significant has been the rise of the Landcare Movement in Australia. It began with an unlikely alliance between traditional opponents (conservationists and farmers) and grew into a movement with thousands of groups in Australia and in other countries. The activities of Landcare Groups transformed many landscapes with large areas being revegetated and restored. Youl *et al.* (2006) provide a good outline of the history and factors that were important for success. They conclude that the strength of Australian Landcare is that community groups and networks, with government and corporate support, conceive their own visions and set goals for local and regional environmental action. Working from the ground up to achieve these goals creates freedom and flexibility, giving communities a great sense of purpose.

The Secretariat of the Pacific Community (SPC) is a regional intergovernmental organisation whose membership includes both nations and territories in the Pacific Ocean and their metropolitan powers. The Land Resources Division assists the Pacific Community to improve food, nutritional and income security and sustainable management and development of land, agriculture and forestry resources.

In New Zealand, there are few regulatory instruments directly related to soil. Where they exist, they focus on soil conservation. However there is an increasing number of national policy instruments that legislate against the impacts of unwise soil use. The Resource Management Act is given effect at regional level and regulates activities not outcomes (through regional policy statements, plans and resource consents). These regulatory instruments typically focus on ensuring soil intactness. However, new initiatives are increasingly looking at the consenting of land use according to soil capability. New Zealand has also used non-regulatory approaches to achieve good soil management. These approaches include direct payments, support to the development of industry codes of practice, and certification schemes to ensure market access.

8.5 | Information systems, accounting and forecasting

The distribution and characteristics of the soils in any district or nation are neither obvious nor easy to monitor. As a consequence, understanding whether a land use is well-matched to the qualities of the soil requires some form of diagnostic system to identify the most appropriate form of management and to monitor how the soil is functioning. Important components of the diagnostic system necessary for sustainable land use and management are:

an understanding of spatial variations in soil function (e.g. maps and spatial information)

an ability to detect and interpret soil change with time (e.g. via monitoring sites, long-term experiments, environmental proxies)

a capacity to forecast the likely state of soils under specified systems of land management and climates (e.g. through the use of simulation models)

an understanding of the edaphic requirements of plants

Preparation of this book was severely constrained by the lack of relevant information. Soil map coverages are variable and, in some regions, out-of-date. The capacity to monitor and forecast soil change is also rudimentary. All nations require coordinated soil information systems that parallel those that exist in many countries for economic data, weather and water resources. Action on soil information systems is enshrined in the World Soil Charter's guidelines for action for governments (Sections VIII and IX) and international organizations (Sections I and II). However, creating appropriate institutional systems for soil information gathering and dissemination is challenging for the following reasons:

All levels of government need reliable information on soil resources but often no single level of government or department has responsibility for collecting this information on behalf of other public sector agencies.

Public and private interests in soil are large and overlapping – mechanisms for co-investment by public and private agencies are therefore needed.

Market failure in relation to the supply and demand of soil information is a significant and widespread problem. Simply stated, beneficiaries of soil information do not usually pay for its collection and this reduces the pool of investment for new survey, monitoring and experimental programmes.

Partly as a result of the above, soil-information gathering activities in many countries are currently funded through short-term government programmes, private companies or individuals or are produced in response to specific regulatory requirements. This piecemeal approach does not result in the kind of enduring, accessible and broadly applicable information systems that are needed to meet the requirements of stakeholders.

The following sections outline some specific requirements that policy makers have of soil information systems.

8.5.1 | Soil information for markets

The various types of markets regulated by governments and other institutions need to be sufficiently informed to ensure economic efficiency and the desired allocation of resources.

These markets include:

- traditional real-estate markets where information is needed on the capital value of soil resources (e.g. the nutrient status of a farm, presence of contaminants, and options for improved soil management)
- carbon trading schemes
- cap-and-trade systems for nutrient loading or other pollutants
- forecasting of within-season production of agricultural commodities
- insurance (e.g. crop insurance, disaster insurance, risk analysis of supply chains).

Oversight and regulation of market activities is a central function of governments in most countries. A key responsibility for policy makers is to ensure the availability of reliable soil information.

8.5.2 | Environmental accounting

A closely related area where policy makers are starting to need better information is environmental accounting. Globally, national accounts of economic activity are recorded and indicators such as gross domestic product (GDP) are widely used in government and policy to assess economic activity and progress. However, indicators such as GDP measure mainly market-based transactions and are not a good indicator of welfare; GDP ignores social costs, environmental impacts and income inequality (Costanza *et al.*, 2014). GDP also does not deduct the direct cost of the depletion of natural resources on national income nor does it take into account the impact that our resource extraction and use of nature has on the continued functioning of the Earth system for life support.

In light of these limitations of the current national economic accounting system, the ecosystem services approach seeks to include nature in our accounting and acknowledge that it has value and its use is not simply free and limitless (Westman, 1977; Daily, 1997; Costanza *et al.*, 1997; MA, 2005; Robinson *et al.*, 2014). In this context, soils make an important contribution to the supply of ecosystem services (Daily *et al.*, 1997; Wall, 2004; Robinson, Lebron and Vereecken, 2009; Dominati, Patterson and Mackay, 2010; Robinson *et al.*, 2013).

One proposal to address the deficiency of the current national accounts is to have a set of complementary accounts. Since the early 1990s, the international official statistics community has been developing such a set of accounts, named the System of Environmental Economic Accounting (SEEA). The over-arching objective of the SEEA approach is to develop an accounting structure that integrates environmental information with the standard national accounts and hence to mainstream environmental information in economic and development policy discussion.

The SEEA accounts are presented in two volumes. First, the SEEA Central Framework (UN *et al.*, 2014) which was adopted as an international statistical standard in 2012, and second, SEEA Experimental Ecosystem Accounting (UN *et al.*, 2014) which was endorsed in 2013. The SEEA Central Framework deals with individual environmental assets (minerals, timber, fish, water, soil, etc.), the flows of mass and energy between the environment and the economy, and the space in which this occurs (Obst and Vardon, 2014). SEEA Experimental Ecosystem Accounting is focused on the function of ecosystems and the generation of ecosystem services which is dependent on ecosystem extent, condition and quality.

The SEEA Central Framework identifies seven individual components of the environment as environmental assets; mineral and energy resources, land, soil resources, timber resources, aquatic resources, other biological resources (excluding timber and aquatic resources, for example, livestock, orchards, wild plants for medicine, wild animals that are hunted), and water resources.

SEEA Experimental Ecosystem Accounting uses the same definition of environmental assets but rather than considering individual components as assets, it seeks to consider the way in which these components function jointly as ecosystems. To apply this logic it defines spatial areas, such as different vegetation habitats (forests, wetlands, agricultural land etc.) as ecosystem accounting units. In this approach soil is considered a component within a broader ecosystem rather than being considered as a distinct ecosystem.

Soils form an important part of the Central Framework, being recognized as an environmental asset in their own right. An important distinction is made between land and soil resources. Land is considered in terms of space and location often referred to as Ricardian land (Daly and Farley, 2011). Soil resources are the volume of biologically active topsoil, and its composition in the form of nutrients, soil water and organic matter. The accounts are structured to recognize, and distinguish between, the use of an asset (e.g. soil volume and area within the asset accounts); and the use of the soil resource or elements of the soil resource (e.g. carbon, nutrients and soil moisture in the physical flow accounts). Fundamental to the accounting process is the measurement of change for both the environmental and ecosystem accounts, which is underpinned by the availability of good quality data (Obst, Edens and Hein, 2013). The major aspects of soil of interest for the environmental accounts are: the volume of soil moved or extracted; the area of soil under different land uses; carbon, nutrient and moisture stocks; and changes in these three aspects. Hence the understanding and quantification of soil change is central to environmental accounting (Robinson, 2015).There is still no agreed set of soil indicators, although soil carbon content is widely seen as being perhaps the main indicator. There is still much work to do to synthesize soil quality work into the SEEA framework for the creation of useful, informative accounts, and to encourage countries to adopt this unified approach.

8.5.3 | Assessments of the soil resource

It is essential to have some form of regular reporting on the rate and extent of soil change along with the likely consequences for society at local, national and global scales. Some countries now have various forms of audits and state-of-the-environment reports. However, most countries do not produce regular assessments showing where land management systems can operate sustainably within the constraints set by changing climate, weather and soils. These are necessary given the economic and environmental significance of soil resources.

Regular reporting forces policy makers to impose an operational discipline on the management of soil information. Systems for collecting and analysing data can be progressively improved and a body of knowledge will be developed over several cycles of reporting. The assessments need to adopt a highly participative mode of engagement so that all stakeholders are represented and then empowered to make the necessary changes to land management.

The World Soil Charter addresses this issue directly. It encourages governments to develop a national institutional framework for monitoring implementation of sustainable soil management and overall state of soil resources. International organizations are encouraged to facilitate the compilation and dissemination of authoritative reports on the state of the global soil resources and sustainable soil management protocols.

This book is a sign that progress is being made in relation to regular assessment and reporting. Further progress will depend on successful implementation of Pillar Four of the Global Soil Partnership - *Enhance the quantity and quality of soil data and information* – and of Pillar Five - *Harmonize methods, measurements and indicators for the sustainable management and protection of soil resources*.

References

Australian Government. 2014. *The National Soil Research, Development and Extension Strategy, Securing*

Australia's Soil for Profitable Industries and Healthy Landscapes. CC BY 3.0 (also available at daff.gov.au/natural-resources/soils).

Beddington, J., Asaduzzaman, M., Clark, M., Fernández, A., Guillou, M., Jahn, M., Erda, L., Mamo, T., Van Bo, N., Nobre, C.A., Scholes, R., Sharma, R. & Wakhungu, J. 2012. Achieving food security in the face of climate change: Final report from the Commission on Sustainable Agriculture and Climate Change. Denmark, Copenhagen, CGIAR Research Program on Climate Change, Agriculture and Food Security (CCAFS).

Bernauer, T., Böhmelt, T. & Koubi, V. 2012. Environmental changes and violent conflict. *Environmental Research Letters*, 7(1): 015601.

Borlaug, N.E. 2000. Ending world hunger: the promise of biotechnology and the threat of antiscience zealotry. *Plant Physiology*, 124(2): 487-490.

Bossio, D., Geheb, K. & W. Critchley. 2010. Managing water by managing land: addressing land degradation to improve water productivity and rural livelihoods. *Agricultural Water Management*, 97 (4): 536-542.

Calbick, K.S., Day, J.C. & Gunton, T.I. 2003. Land use planning implementation: a 'best practices' assessment. *Environments* 31: 69–82.

CBD. 2014. *Decisions of the Conference of the Parties on the ecosystem approach from decision IV/1 in 1998 to decision IX/7 in 2008*. (Available at http://www.cbd.int/ecosystem/decisions.shtml)

Churchman, G.J. & Land, E.R. 2014. The Soil Underfoot: Infinite possibilities for a Finite Resource. CRC Press Book. 472 pp.

CITMA. 2011. *Proyecto Estrategia Ambiental Nacional 2011/2015*. Versión 1.10. 23 de mayo 2011.

Costanza, R., dArge, R., deGroot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., Oneill, R. V., Paruelo, J., Raskin, R. G., Sutton, P. & van den Belt, M. 1997. The value of the world's ecosystem services and natural capital. *Nature*, 387: 253-260.

Costanza, R., Kubiszewski, I., Giovannini, E., Lovins, H., McGlade, J., Pickett, K. E., Ragnarsdottir, K. V., Roberts, D., De Vogli, R. & Wilkinson, R. 2014. Time to leave GDP behind. *Nature*, 505: 283-285.

Daily, G. 1997. *Natures services: societal dependence on natural ecosystems*. Washington, DC, Island Press.

Daily, G., Matson, P. & Vitousek, P. 1997. Ecosystem services supplied by soils. *In* G. Daily, ed., *Nature's services: Societal dependence on natural ecosystems*. pp 113–142. Washington, DC, Island Press.

Dalal-Clayton, B. & Dent, D. 2001. *Knowledge of the land: land resources information and its use in rural development*. Oxford University Press.

Daly, H.E. & Farley, J. 2011. *Ecological economics, principles and applications*. Washington, DC, Island Press.

Dominati, E.J., Patterson, M. & Mackay, A. 2010. A framework for classifying and quantifying the natural capital and ecosystem services of soils. *Ecol. Econ*., 69: 1858–1868.

Drechsel, P., Giordano, M. & Gyiele, L. 2004. Valuing Nutrients in Soil and Water Concepts and Techniques with Examples from IWMI Studies in the Developing World. Research Report #82. Colombo, Sri Lanka, IWMI.

Drine, I. 2011. *Climate Variability and Agricultural Productivity in MENA region*. Working Paper No. 2011/96. December 2011.United Nations University.

EC. 2006. *Proposal from the Commission to the Council, the European Parliament, the European Economic and Social Committee and the Committee of the Regions for a Directive of the European Parliament and of the Council establishing a framework for the protection of soil and amending*. Directive 2004/35/EC. COM (2006) 232 final. Brussels, European Commission.

EC. 2013. *Decision No 1386/2013/EU of the European Parliament and of the Council of 20 November 2013 on a General Union Environment Action Programme to 2020 'Living well, within the limits of our planet' Text with EEA relevance*. (available at: http://eur-lex.europa.eu/legal content/EN/TXT/?uri=CELEX:32013D 1386).

FAO. 1976. *A framework for land evaluation*. Soils Bulletin no 32. Rome, FAO.

FAO. 1982. *World Soil Charter*. Rome, FAO.

FAO. 2007. *The state of food and agriculture. Paying farmers for environmental services*. Rome, FAO.

FAO. 2011. *The state of the world's land and water resources for food and agriculture- Managing systems at risk (SOLAW).* Rome, FAO.

FAO. 2013. *Policy Support Guidelines for the Promotion of Sustainable Production Intensification and Ecosystem Services*. Vol 19. *Integrated Crop Management*. Rome, FAO.

FAO-UNESCO. 1980. *FAO-UNESCO soil map of the world. Vol 1. Legend*. Paris, UNESCO.

Fox, J., Castella, J.C. & Ziegler, A.D. 2011. Swidden, rubber and carbon: Can REDD+ work for people and the environment in Montane Mainland Southeast Asia? *Global Environmental Change*, 29: 318-326.

Frerks, G., Dietz, T. & van der Zaag, P. 2014. Conflict and cooperation on natural resources: Justifying the CoCooN programme. Chapter 2. In M, Bavinck, L. Pellegrini, E. Mostert, eds. *Conflicts over Natural Resources in the Global South–Conceptual* Approaches.Balkema, Taylor & Francis.

Hellerstein, H. 2010. Challenges facing USDA's Conservation Reserve Program. *AmberWave*,s 8.

Hill, M., Mondelli, M. & Carrazzone, S.E.M. 2014. *Soil conservation in Uruguay: soil management plans as a national policy*. Regional Consultation on SAI in LAC, CGIAR Consortium – August 2014. Ministry of Africulture and Fisheries of Uruguay (MGAP).

Hurni H., Giger M. & Meyer K. (eds.). 2006. *Soils on the global agenda. Developing international mechanisms for sustainable land management*. Prepared with the support of an international group of specialists of the IASUS Working Group of the International Union of Soil Sciences (IUSS). Bern, Centre for Development and Environment. 64 pp.

ICARDA. 2013. *MENARID Gateway- Strengthening and scaling up Integrated Natural Resource Management across MENA*. International Center for Agricultural Research in the Dry Areas. (available at https://menarid.icarda.org/Pages/Welcome%20Page.aspx).

MA. 2005. *Millennium Ecosystem Assessment, Ecosystems and Human Well-being: Synthesis*. Washington, DC, Island Press.

Montanarella, L & Vargas, R. 2012. Global governance of soil resources as a necessary condition for sustainable development. *Curr Opin Environ* Sust, 4: 1–6.

Mu, Z., Bu, S. & Xue, B. 2014. Environmental Legislation in China: Achievements, Challenges and Trends. *Sustainability*, 6(12): 8967-8979.

Nefedova, T.G. 2013. *Ten topical issues about rural Russia: A geographer's viewpoint*. Moscow, LENAND. [In Russian]

NEPAD. 2012. New Partnership For Africa's Development. (Available at http://www.nepad.org/about).

Obst, C. & Vardon, M. 2014. Recording environmental assets in the national accounts. *Oxford Review of Economic Policy*, 30(1): 126-144.

Obst, C., Edens, B. & Hein, L. 2013. Ecosystem services: Accounting standards. *Science*, 342: 420-420.

Robinson, D. 2015. Moving toward data on soil change. *Science*, 347(6218): 140.

Robinson, D.A., Fraser, I., Dominati, E.J., Davíðsdóttir, B., Jónsson, J.O.G. Jones, L., Jones, S.B., Tuller, M., Lebron, I., Bristow, K.L., Souza, D.M., Banwart S. & Clothier, B.E. 2014. On the value of soil resources in the context of natural capital and ecosystem service delivery. *Soil Science Society of America Journal*, 78(3): 685-700.

Robinson, D.A., Jackson, B.M., Clothier, B.E., Dominati, E.J., Marchant, S.C., Cooper, D.M. & Bristow, K.L. 2013. Advances in soil ecosystem services: Concepts, models and applications for earth system life support. *Vadose Zone* J., 12.

Robinson, D.A., Lebron, I. & Vereecken, H. 2009. On the definition of the natural capital of soils: A framework for description, evaluation, and monitoring. *Soil Sci. Soc. Am.* J., 73: 1904–1911.

Smaling E.M.A. 1993. The soil nutrient balance: an indicator of sustainable agriculture in sub-Saharan agriculture, pp. 18. *In* Proceedings of the Fertilizer Society, 340.

Soares-Filho, B., Rajão, R., Macedo, M., Carneiro, A., Costa, W., Coe, M., Rodrigues, H. & Alencar, A. 2014. Land Use. Cracking Brazil's Forest Code. *Science*, 344(6182): 363-364.

Steffen, W., Richardson, K., Rockström, J., Cornell, S.E., Fetzer, I., Bennett, E.M., R. Biggs, R., Carpenter, S.R., de Vries, W., de Wit, C.A., Folke, C., Gerten, D., Heinke, J., Mace, G.M., Persson, L.M., Ramanathan, V., Reyers, B. & Sörlin, S. 2015. Planetary boundaries: Guiding human development on a changing planet. *Science*, 347(6223): 1259855.

Stoorvogel, J.J., Smaling, E.M.A. & Janssen, B.H. 1993. Calculating soil nutrient balances at different scale. I. Supra-national scale. *Fert. Res.*, 35: 227-235

UN, EU, FAO, IMF, OECD, World Bank. 2014. *System of Environmental-Economic Accounting 2012. Central Framework*. New York, United Nations. (Available at. http://unstats.un.org/unsd/envaccounting/seeaRev/SEEA_CF_Final_en.pdf)

UN. 2012. *The Future We Want: Outcome document adopted at Rio+20*.
(Available at http://www.uncsd 2012.org/content/ocuments/727The%20Future%20We%20Want%2019%20June%201230pm.pdf).

UNCCD. 2012. Offical webpage. (Available at http://www.unccd.int/en/regional-access/Asia/Pages/asia.aspx).

UNCED. 1992. UN Conference on Environment and Development.
(Available at http://www.sidsnet.org/about-sids/unced).

UNEP. 1982. *World Soils Policy*. Nairobi, United Nations Environmental Programme.

UNEP. 2004. *UNEP's Strategy on Land Use Management and Soil Conservation. A Strengthened Functional Approach*. United Nations Environment Programme Policy Series.

UNEP. 2012. *Global Environmental Outlook: Fifth Edition*. Nairobi & New York, UNEP.
(Available at: http://www.unep.org/geo/geo5.asp (accessed February 2015).

Visser, O. & Spoor, M. 2011. Land grabbing in post-Soviet Eurasia: the worlds largest agricultural land reserves at stake. *Journal of Peasant Studies*, 38(2): 299-323. doi: http://dx.doi.org/10.1080/03066150.2011.559010. (Last accessed: March 2015).

Wall, D. 2004. *Sustaining biodiversity and ecosystem services in soils and sediments*. Washington, DC, Island Press.

Westman, W.E. 1977. How much are nature's services worth. *Science*, 197: 960–964.

Wingqvist, G.O. & Drakenberg, O. 2010. Environmental and Climate Change Analysis. SIDA working paper.

WOCAT. 2007. *Where the land is greener - case studies and analysis of soil and water conservation initiatives worldwide*. H.P. Liniger,& W. Critchley, eds. CDE Berne Co-published by CTA, UNEP, FAO & CDE.

World Bank. 2011. *Rising Global Interest in Farmland. Can it yield sustainable and equitable benefits?* WB.

Wu, J.J. & Weber, B. 2012. Implications of a Reduced Conservation Reserve Program. *In The Conservation Crossroads in Agriculture: Insight from Leading Economists*. The Council on Food, Agriculture and Natural

Resources, August, 2012.

Youl, R., Marriott, S. & Nabben, T. 2006. *Landcare in Australia: founded on local action*. (available at www.agriculture.gov.au/SiteCollectionDocuments/natural-resources/landcare/communiques/landcare_in_australiaJune 08.pdf)

Young G.L. 1994. Soil conservation service. *In* W.P Cunningham, ed. *Environmental* Encyclopedia.USA, Detroit, Gale Research Inc. 774 pp.

Yue, W. 2014. *Almost one-fifth of our arable land is polluted*. Web blog post. Chinadialogue. 17-04-2014. (Available at https://www.chinadialogue.net/blog/6921-Almost-one-fifth-of-our-arable-land-is-polluted-admit-Chinese-officials/en).

9 | Regional Assessment of Soil Changes in Africa South of the Sahara

Regional Coordinator: Victor Chude (ITPS/Nigeria)

Regional Lead Author: Ayoade Ogunkunle (Nigeria)

Contributing Authors: Victor Chude (ITPS/Nigeria), Isaurinda Dos Santos (ITPS/Cape Verde), Tekalign Mamo (ITPS/Ethiopia), Garry Paterson (South Africa), Ndaye Soumare (Senegal), Liesl Wiese (South Africa), and Martin Yemefack (ITPS/Cameroon).

9.1 | Introduction

Land degradation in sub-Saharan Africa (SSA) is believed to be expanding at an alarming rate, accompanied by the lowest agriculture and livestock yields of any region in the world. While cereal production has increased marginally over the past two decades, more than 70 percent of this growth is due to area expansion rather than yield increases. The region also suffers from the world's highest rate of deforestation, with some countries having lost more than 10 percent of their forest cover in the five years up to 2009 (IFAD, 2009) and is most likely continuing at the same rate to this day.

There is a growing and long-standing recognition among both policy-makers and soil specialists that soil degradation is one of the root causes of declining agricultural productivity in sub-Saharan Africa and that, unless the process of degradation is controlled, many parts of the continent will suffer increasingly from food insecurity (e.g. see Lal, 1990; UNEP, 1982). The consequences of allowing the productivity of Africa's soil resources to continue on its present downward spiral will be severe, not only for the economies of individual countries, but for the welfare of the millions of rural households across the continent who are dependent on agriculture (FAO, 1999).

Soil degradation is the decline in soil quality caused its improper use by humans, usually for agricultural, pastoral, industrial or urban purposes. Soil degradation may be exacerbated by climate change and encompasses physical, chemical and biological deterioration. Examples of soil degradation cited by Charman and Murphy (2005) are: loss of organic matter; decline in soil fertility; decline in structural condition; topsoil loss and erosion; adverse changes in salinity, acidity or alkalinity; and the effects of toxic chemicals, pollutants and excessive flooding.

There is no consensus on the exact extent and severity of land degradation or its impacts in SSA as a whole (Reich *et al.*, 2001; GEF, 2006). Lack of information and knowledge is considered to be one of the major obstacles for reducing land degradation, improving agricultural productivity, and facilitating the adoption of sustainable land management (SLM) among smallholder farmers (Liniger *et al.*, 2011). The recent publication of the first Soil Atlas of Africa has provided a first comprehensive overview of the soil resources of Africa (Jones *et al.*, 2013).

Four continental-scale studies have assessed the extent of soil degradation in Africa. A literature review by Dregne (1990) of 33 countries found compelling evidence of serious land degradation in sub-regions of 13 countries: Algeria, Ethiopia, Ghana, Kenya, Lesotho, Mali, Morocco, Nigeria, Swaziland, Tanzania, Tunisia, Uganda, and Zimbabwe. In another literature review, focused on drylands only, Dregne and Chou (1992) estimated that 73 percent of drylands were degraded and 51 percent severely degraded. They concluded that 18 percent of irrigated lands, 61 percent of rainfed lands, and 74 percent of rangelands located in SSA drylands are degraded.

The Global Assessment of Soil Degradation (GLASOD) expert survey found that 65 percent of soils on agricultural lands in Africa had become degraded since the middle of the twentieth century, as had 31 percent of permanent pastures, and 19 percent of woodlands and forests (Oldeman, Hakkeling and Sombroek, 1991). Serious degradation affected 19 percent of agricultural land. A high proportion (72 percent) of degraded land was in drylands. The most widespread cause of degradation was water erosion, followed by wind erosion, chemical degradation (three-quarters from nutrient loss, the rest from salinization), and physical degradation. In terms of causes of degradation, overgrazing was responsible for half of all degradation, followed by agricultural activities, deforestation, and overexploitation.

The Land Degradation Assessment in Drylands project (LADA) started in 2006 with the general purpose of creating the basis for informed policy advice on land degradation at global, national and local levels. This goal is being reached through the assessment of land degradation at different spatial and temporal scales in six countries and through the creation of a baseline at global level for future monitoring (FAO, 2010). Two of the six countries involved (Senegal and South Africa) are within SSA and national results are reported at the end of this chapter.

Lal (1995) calculated continent-wide soil erosion rates from water using data from the mid to late 1980s, and then used these rates to compute cumulative soil erosion for 1970-90. The highest erosion rates occurred in the Maghreb region of Northwest Africa, the East African highlands, eastern Madagascar, and parts of Southern Africa. Excluding the 42.5 percent of arid lands and deserts with no measurable water erosion, Lal found that the land area affected by erosion fell into the following six classes of erosion hazard: none, 8 percent; slight, 49 percent; low, 17 percent; moderate, 7 percent; high, 13 percent; and severe, 6 percent.

Soils host the majority of the world's biodiversity and healthy soils are essential to securing food and fibre production. Soils assure an adequate and clean water supply over the long term, as well as providing cultural functions. Ecosystem services provided by soils are integral to the carbon and water cycles. Major increases in agricultural production have been associated with different kinds of soil degradation, especially since the agricultural growth came in part from extensive clearing of new agricultural lands. Yet, even with this expansion, arable land per capita in Africa declined from just under 0.5 ha in 1950 to just under 0.3 ha in 1990 (FAO, 1993). During this time period, yield increases on land already in production thus contributed far more to the total production. For example, more than 90 percent of the growth in developing country cereal production between 1961 and 1990 came from yield growth (World Bank, 1992). Agricultural expansion and yield growth at such a scale is inevitably associated with some degradation of soil resources. However, the type and extent of degradation vary in the different ecological/farming systems (IFPRI, 1999).

9.2 | Stratification of the Region

The region is diverse in terms of relief, climate, lithology, soils and agricultural systems. A combination of some of these have been used to stratify the region into agro-ecological zones (AEZs) (Fischer *et al.*, 2002; Global HarvestChoice, 2010). Table 9.1 shows the AEZs into which the region has been grouped and some of their characteristics, while Figure 9.1 shows the distribution of the AEZs in the region.

9.2.1 | Arid zone

The arid zone occupies 36 percent of the land area of SSA, most of which is in West and East Africa. Rainfall is low and extremely variable in this zone. The annual rainfall of less than 500 mm, combined with high temperatures and consequent high rates of evapotranspiration, make this zone capable of sustaining plant life for less than 90 plant growth days (or length of growing season).

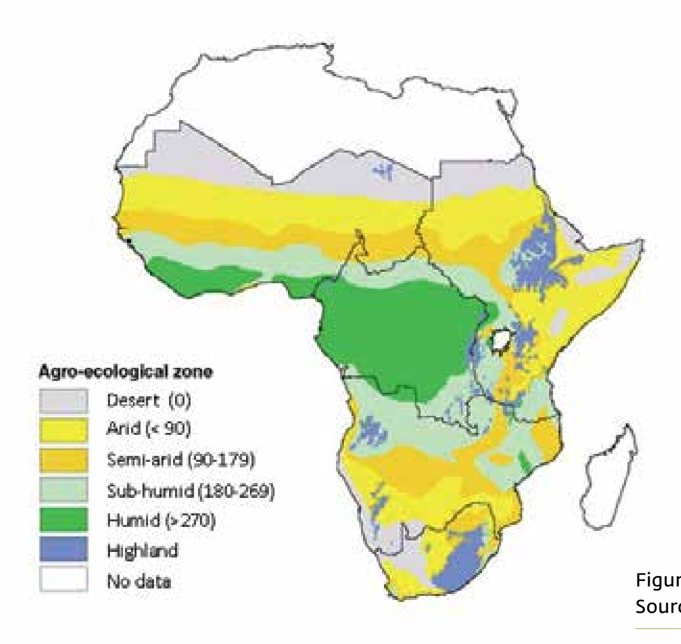


Figure 9.1 | Agro-ecological zones in Africa South of the Sahara. Source: Otte and Chilonda, 2002.

Table 9.1 | Characteristics and distribution of agro-ecological zones in Africa
Source: ILCA, 1987, after Jahnke, 1982.

[a]pgd = plant growth days; [b]Areas with mean daily temperature during the growing period less than 20°C; [c]: n.a. = not available.

Zone	Definition	Rainfall range (mm)	Area (percent)				
			West Africa	Central Africa	East Africa	Southern Africa	Area of zone (percent)
Arid	<90 pgda	0–500	54	1	52	20	36
Semi-arid	90–180 pgd	500–1000	20	7	18	34	18
Sub-humid	180–270 pgd	1000–1500	16	29	16	38	22
Humid	>270 pgd	1500+	10	59	2	7	19
Highlandsb	<20°C	n.a.c	0	4	12	1	5
Total			**100**	**100**	**100**	**100**	**100**
Total area (10^6 km²)			**7.3**	**5.3**	**5.8**	**3.2**	

The arid zone is mostly associated with sandy soils (Arenosols, Psamments) which are weakly differentiated and are often of aeolian origin. Water and air move freely through these soils, which are low in all nutrients.

The accompanying vegetation consists of short annual grasses, legumes, scattered shrubs and trees. Mobile herds of sheep, goats, cattle and camels browse the herbage and shrubs, while farmers use most of the trees and shrubs for fuel. The low rainfall and its erratic distribution make cropping uncertain in most years. Owing to this unreliability, arable farming is mostly restricted to opportunistic cultivation of short-season millets, except in topographically favourable sites such as oases or irrigated areas. Opportunities for livestock development are limited but existing techniques could be improved upon, if not to increase productivity, then at least to sustain it.

9.2.2 | Semi-arid zone

The semi-arid zone receives 500 to 1000 mm of rainfall annually which can be capable of sustaining 90 to 180 plant growing days. This zone occupies 18 percent of the land area of SSA. Semi-arid lands are found in all regions of SSA except central Africa. The low rainfall and the long dry season make the semi-arid zone a relatively healthy environment for humans and livestock. Arenosols (Psamments) and Cambisols (Inceptisols) are widespread and include coarse sandy soils, fine sands, and loamy sandy soils. Water retention is poor and nutrient contents, including N, P and S levels, are generally low. The permeability of the undisturbed soil is good, but algal skins contribute to the formation of surface crusts. The natural vegetation is an open low-tree grassland but this has been severely modified in many regions.

The lower rainfall areas of this zone are used for grazing. Cropping and crop–livestock systems dominate the areas with higher rainfall where farmers commonly grow millet, sorghum, groundnut, maize and cowpeas.

9.2.3 | Sub-humid zone

The sub-humid zone occupies 22 percent of SSA, mainly in southern and central Africa. The zone receives 1 000 to 1 500 mm of rain annually which can sustain plants for 180 to 270 plant growing days. Within the climatic definition, this is a very varied zone in terms of climate, soils and land use. Luvisols (Alfisols) and Cambisols (Inceptisols) occur widely, parent material is often strongly weathered, and the levels of mineral nutrients as well as the clay fraction are low. Cambisols (Inceptisols) have fewer constraints to plant production than the older, more weathered soils, since their high base status provides adequate Ca and eliminates constraints related to low pH levels. The fertility of many soils in this zone is low, especially due to leaching of NO_3^-, accompanied by the loss of cations and P adsorbtion. In addition, structural stability in these soils can be poor, with crusting and hardening occurring when soils are dry.

The natural vegetation is typically medium height or low woodland with understory shrubs and a ground cover of medium to tall, mainly perennial, grasses; Hyparrhenia spp. are common.

Food and cash crops are grown, including cassava, yams, maize, fruits, vegetables, rice, millet, groundnut, cowpeas and cotton. From these crops, products such as cottonseed cakes and the residues of the crops are available as feed for livestock. In some areas of this zone farmers grow soybean and leguminous forage crops.

The humid zone occupies 19 percent of SSA mostly in central and west Africa. The zone receives more than 1 500 mm of rainfall annually which can sustain plants for 270 to 365 plant growing days. The zone is found at low latitudes north and south of the equator. Soils in this zone include Ferralsols (Oxisols), Acrisols (Ultisols) and Luvisols (Alfisols), the last of which are commonly encountered at the forest-savannah boundary. Vegetation consists of rain forest and derived savannahs with natural vegetation dominated by tall, closed forest which may be evergreen or semi-deciduous and which is often floristically rich. The herbaceous vegetation often contains large amounts of the major nutrients.

The soils are strongly weathered and hence have high levels of iron and aluminium oxides and low levels of phosphorous. The organic matter content is therefore generally low and the soils are fragile and easily degraded when the vegetative cover is lost. This zone has limited potential for livestock development, particularly because of the threat of the trypanosomiasis-transmitting tsetse fly.

9.2.5 | Highlands zone

The highland zone represents 5 percent of the land area of SSA, most of which is in eastern Africa, and half in Ethiopia. This zone occupies areas above 1 500 m altitude that have a mean daily temperature of less than 20 ºC. The main highland areas in sub-Saharan Africa (SSA) are in Ethiopia, Kenya, Uganda, Rwanda, Burundi, western Zaire, Tanzania, Angola and Lesotho. There are also many other areas above the 1500 m contour and some of these afford tsetse-free grazing (e.g. Fouta Djallon and Bamenda). The highland areas vary in climate, topography, soils and land use.

Topography varies from gently rolling hills to deeply incised valleys and steep slopes. Soils are sometimes deep and fertile Vertisols and Nitosols, but shallow soils of inherently low fertility are widespread. In many mountain grassland areas, soils only have a very shallow surface horizon that is fertile. Undisturbed upland areas are normally stable, although some soils exhibit 'slumping' even where undisturbed. Cultivating the so-called 'duplex' soils[5] and soils that form a surface crust on slopes results in high run-off. Unless soil conservation measures are taken and soils are sufficiently covered with vegetation, overland flow removes large amounts of soil. The zone receives bimodal rainfall (>1000 mm annually) and there are two growing seasons. Livestock rearing is widespread: farmers grow fodder, and animal traction is of increasing importance. Population pressure is encouraging crop–livestock integration, for which the cool highlands have high potential

9.3 | General soil threats in the region

The various threats to soil health and ecosystem services in SSA include: (1) erosion by water or wind; (2) loss of soil organic matter; (3) soil nutrient depletion; (4) loss of soil biodiversity; (5) soil contamination; (6) soil acidification; (7) salinization and sodification; (8) waterlogging; and (9) compaction, crusting and sealing/capping (Mabogunje, 1995; Oldeman, 1991; Meadows and Hoffman, 2002; World Bank, 1997; IFPRI, 1999).

9.3.1 | Erosion by water and wind

About 77 percent of SSA is affected by erosion, with the most serious erosion areas in the Republic of South Africa, Sierra Leone, Guinea, Ghana, Liberia, Kenya, Zaire, Central African Republic, Ethiopia, Senegal, Mauritania, Nigeria, Niger, Sudan and Somalia.

According to the GLASOD results (ISRIC/UNEP, 1990), about 494 million ha of the land in SSA is affected by one form of degradation or another. Of this, 227 million ha (46 percent) is by water erosion, 187 million ha (38 percent) by wind erosion, 62 million ha (12 percent) by chemical degradation and 18 million ha (4 percent) by physical degradation. The intensity of water erosion has been described as very high to extreme on about 102 million ha (45 percent of the total SSA area affected), moderate on about 67 million ha (30 percent) and slight on about 58 million ha (25 percent) (Oldeman, 1991).

Water erosion: This is the most widespread soil degradation type in SSA. Water erosion increases on slopes where vegetation cover is reduced due to deforestation, overgrazing or cultivation that leaves the soil surface bare. It is further aggravated where there has been a loss of soil structure or infiltration rates have been reduced. The areas particularly affected are humid and sub-humid zones. Almost 70 percent of **Uganda** was degraded by soil erosion and soil nutrient depletion between 1945 and 1990. More than 20 percent of agricultural land and pastures in the country have been irreversibly degraded.

Water erosion poses the greatest threat to soils in **Nigeria**, where it affects over 80 percent of the land (NEST, 1991). Wind, sheet, gully and beach erosion affect different parts of the country at varying intensities, but attention will focus here on the impact of erosion on agricultural land. While wind erosion is confined to the arid north, sheet erosion by water is ubiquitous throughout the country. Areas most prone to sheet erosion are where farming has cleared the original vegetation, and the soils became impoverished scrubland. Gully erosion is by far the most alarming type of erosion, particularly in the Eastern region, because it often threatens settlements and roads. Although it affects a small fraction (less than 0.1 percent) of Nigeria's 924 000 km^2 of landmass, gully erosion claims large amounts of public funds annually for remedial action.

Wind erosion: Wind erosion occurs most frequently in the arid and semi-arid parts of SSA, especially in areas with sandy or loamy soils. Wind erosion leads to loss of topsoil over extended areas causing soil fertility decline. Bielders, Michels and Rajot (1985) stated that wind erosion can remove up to 80 tonnes of soil from 1 ha in a given year. In SSA wind erosion is second in importance to water erosion, constituting 38 percent of the total erosion in the region (ISRIC/UNEP, 1990) and affecting about 186 million ha of land in the region. The intensity of wind erosion is strong on about 9 million ha (5 percent), moderate on 89 million ha (48 percent) and light on 89 million ha (48 percent) (Oldeman, 1991). Over 99 percent of wind erosion in Africa occurs in the dry land zone, with less than 1 percent affecting the humid zone.

Wind erosion is a natural process that commonly occurs in deserts and on coastal sand dunes and beaches. During drought, it can also occur in agricultural regions where vegetation cover is reduced. If the climate becomes drier or windier, wind erosion is likely to increase. Climate change forecasts suggest that wind erosion will increase over the next 30 years due to more droughts and more variable climate. The combination of a changing climate and consequent increase in wind erosion will cause a series of changes affecting soils:

- less rain, which will support less vegetation
- lower soil moisture, which will decrease the ability of soil particles to bind together into larger, heavier aggregates
- increased wind speeds, which will result in more force exerted on the ground surface and more wind erosion (if wind speed doubles, the erosion rate increases eight times)
- large losses of soil and nutrients
- more large dust storms, which will impact soils and the community
- poorer air quality, increased respiratory health risks, and temperature and rainfall changes due to atmospheric pollution (all off-site effects).

9.3.2 | Loss of soil organic matter

Land degradation leads to a release of carbon to the atmosphere through oxidation of soil organic matter (Oldeman, Hakkeling and Sombroek, 1991). Africa's current major negative role in the global carbon cycle can be attributed to the substantial releases of carbon associated with land use conversion from forest or woodlands to agriculture (Smith, 2008). In the 1990s, these releases accounted for approximately 15 percent of the global net flux of carbon from land use changes (Hooper *et al.*, 2006). Land management following conversion also impacts carbon status, soil fertility, and agricultural sustainability – a point underlined by many including Lal (2006), Ringius (2002), Zivin and Lipper (2008) and Tieszen, Tappan and Toure (2004). Soils often continue to lose carbon over time following land conversion (Woomer, Toure and Sall, 2004; Tschakert, Khouma and Sene, 2004; Liu *et al.*, 2014), resulting in further reductions in crop yields and impoverishment of the farming population. However, these carbon stocks can be replenished with combinations of residue retention, manuring, nitrogen (N) fertilization, agroforestry, and conservation practices (Lal, 2006).

In most sub-humid and semi-arid areas, much of the grazing land is burned annually during the dry season to remove the old and coarse vegetation and to encourage the growth of young and more nutritious grasses. Burning causes the loss of soil organic matter (released as CO_2) and thus impairs agricultural productivity.

It exposes the soil to the erosive forces of the wind during the dry season and of the rain during the rainy season. Furthermore, the annual burn of the vegetation severely reduces the return of organic matter to the soil. This results in loss of the benefits of soil organic matter, including fertility, structure, water retention and biodiversity. The soil becomes biologically, chemically and physically poorer (FAO, 2001). Land degradation further leads to a release of carbon to the atmosphere through the oxidation of soil organic matter which results from soil disturbance and from the consequent exposure of new soil surfaces to the weather.

In agricultural land, the challenge has been to produce increasing quantities of food in an economic and institutional context where the means to improve productivity in a sustainable fashion are generally not available (e.g. lack of sustainable technological packages, absence of extension, training or affordable inputs etc.). Pressures to increase output in the absence of these supporting factors has led to: (i) the rapid expansion of agricultural land (over 65 percent in the last three decades); and (ii) the shortening of the fallow periods in traditional, extensive land use systems, which reduced the rehabilitation of soil fertility through natural processes. The increased use of fire as a clearing tool has led to the further loss of nutrients in many systems. Fertilizer consumption has not increased to compensate for the loss of soil nutrients resulting from the intensification of land use. Hence, there has been widespread mining of soil organic matter and nutrients. As a consequence of this poor land management combined with the vulnerable nature of many soils, much of SSA's cropland is now characterized by low organic matter content, often in combination with a low pH and with aluminium toxicity. On degraded soils with low organic matter, inorganic fertilizers are also easily leached, which is likely to have negative long-term effects on agricultural productivity and on the quality of downstream water resources.

9.3.3 | Soil nutrient depletion

Soils in a large part of SSA are strongly weathered and inherently low in organic matter. Because of the increasing pressure on land, natural replenishment of nutrients during fallow periods is now insufficient to maintain soil productivity over the long-term. Insufficient nutrient replacement in agricultural systems on land with poor to moderate potential results in soil degradation. Already soil moisture stress inherently constrains land productivity on 85 percent of soils in Africa (Eswaran, Reich and Beinroth, 1997). Now soil fertility degradation places an additional serious human-induced limitation on productivity.

The low nutrient status of most soils in SSA is further exacerbated by insufficient use of fertilizers and manure and by the practice of mono-cropping. Overall use of inorganic fertilizers in SSA is just 12 kg ha^{-1}, the lowest in the World, and soil nutrient depletion is widespread in croplands. Approximately 25 percent of soils in Africa are acidic, and therefore deficient in phosphorus (P), calcium and magnesium with often toxic levels of aluminium (McCann, 2005). Use of fertilizer in the region involves average applications of less than 9 kg of nitrogen and 6 kg of phosphorus per ha, compared with typical crop requirements of 60 kg of nitrogen and 30 kg of phosphorus per ha. Recent research estimates that on average every country in SSA has a negative soil nutrient balance; in all countries studied, the amount of nitrogen, phosphorus and potassium (K) added as inputs was significantly less than the amount removed as harvest or lost by erosion and leaching (Swift and Shepherd, 2007). Although many farmers have developed soil management strategies to cope with the poor quality of their soil, low inputs of nutrients, including of organic matter, contribute to poor crop growth and to the depletion of soil nutrients.

Stoorvogel, Smaling and Janssen (1993) calculated nutrient balances for arable soils in 38 sub-Saharan countries and for 35 crops for 1982-1983 and made forecasts for 2000. Subtracting values of the output (made up of harvest, removal of residues, leaching, denitrification and erosion) from the values of the input (made up of fertilizers, manures, rain, dust, biological N-fixation and sedimentation), the study reported alarming average nutrient losses for SSA as follows: 1982-1983: 22 kg N, 2.5 kg P and 15 kg K; 2000: 26 kg N, 3 kg P and 19 kg K. This indicated persistent nutrient mining over time (Bationo *et al.*, 2012). Other estimations claim that each year 4 million tonnes of nutrients are harvested annually in SSA against <0.25 million tonnes returned to the soils in the form of fertilizers.

Sub-national studies of nutrient depletion found annual losses of 112 kg per ha of N, 2.5 kg of P, and 70 kg of K in the western Kisii highlands of Kenya. Significantly lower losses were, however, recorded in southern Mali (Smaling, 1993; Smaling, Nandwa and Janssen, 1997). Farm monitoring and modelling of nutrient cycles for the western highlands of Kenya found that more nitrogen (63 kg per ha) was being lost through leaching, nitrification, and volatilization than through removal of crop harvests (43 kg per ha). Depending on the type of farm management practice, net nitrogen balances on cropped land varied between -39 and 110 kg per ha per year, and net phosphorus balances between -7 and 31 kg per ha per year (Shepherd and Soule, 1998).

9.3.4 | Loss of soil biodiversity

Loss of soil biodiversity is considered the fourth major threat in SSA. Biodiversity loss occurs in a number of ways including destruction of habitat, land use change, introduction of new species, and harvesting and hunting of individual wild species. It has been estimated that in the mid-1980s in Sub-Saharan Africa (SSA), 65 percent of the 'original' ecosystems had been converted (Perrings and Lovett, 1999). The most important factors affecting soil biodiversity are: (i) habitat fragmentation; (ii) resource availability – the amount and quality of nutrients and energy sources; (iii) temporal heterogeneity e.g. seasonal effects; (iv) spatial heterogeneity - spatial differences in the soil; (v) climate variability; and (vi) interactions within the biotic community.

Habitat destruction and/or fragmentation remains the primary threat to soil biodiversity in SSA. For instance, the once great equatorial forest that stretched from western Africa into eastern Africa is now fragmented into pockets represented by Lamto forest in Ivory Coast, Mbalmayo forest in Cameroun, Congo forest in Democratic Republic Congo, Kabale, Budongo and Mabira forests in Uganda and Kakamega forest in Kenya. The surrounding communities still rely heavily on these forests for basic needs such as fuelwood, charcoal, timber, poles, and other building materials. Due to human encroachments, the forests are subject to a mosaic of different land uses. There are patches of secondary forest, fallow and arable fields amidst significant remnants of primary vegetation. In the process of conversion and change in land use, soil biota have not been spared. Studies by Okwakol (2000) and Ayuke *et al.* (2011) have shown that up to 50 percent of soil macrofauna species within the forest area have been lost due to habitat destruction or fragmentation.

Other threats to soil biodiversity in SSA include land use and land cover change, mainly through conversion of natural ecosystems, particularly forests and grasslands, to agricultural land and urban areas. In a study conducted across different ecosystems of Eastern (Kenya), Western (Nigeria, Burkina Faso, Ghana, Niger) and Southern Africa (Malawi), Ayuke *et al.* (2011) demonstrated a substantial reduction in the number of species and abundance of soil macrofauna groups such as earthworms and termites because of conversion of native or undisturbed ecosystems into arable systems. Continuous cultivation also exacerbates soil biodiversity loss because of loss of soil organic matter and hence of food resources for the soil organisms (Ayuke *et al.*, 2011). It is likely that land clearing and deforestation will continue, further threatening genetic diversity as more species are lost (IAASTD, 2009). Sub-Saharan Africa suffers the world's highest annual deforestation rate because of overexploitation of forest resources and conversion of forested land to agriculture. Although deforestation occurs throughout the continent, particularly affected areas are the moist forests of Western Africa and the highland forests of the Horn of Africa (FAO, 2007; Hansen *et al.*, 2013).

Mulugeta (2004) reported that in Ethiopia deforestation and subsequent cultivation of the tropical dry Afromontane forest soils endangered the native forest biodiversity not only through the outright loss of habitat but also by impairing the soil seed banks. The results showed that the contribution of woody species to the soil seed bank declined from 5.7 percent after seven years to nil after 53 years of continuous cultivation. However, soil quality and native flora degradation are reversible through reforestation. In fact, reforestation of abandoned farm fields with fast-growing tree species was shown to restore soil quality. Tree plantations established on degraded sites also fostered the recolonization of diverse native forest flora under their canopies. An important result from studying the effects of reforestation is that good silviculture, particularly selection of appropriate tree species, can significantly affect the rate and magnitude of restoration processes for both soil quality and biodiversity.

In many African cultures, harvesting of soil fauna groups such as the termite alates and queens, chafer grubs for food, and the use of earthworms as bait by fishermen can also be a threat to soil biodiversity, and may in the long run contribute to substantial loss of many species of soil fauna.

Harsh climatic conditions and/or climate change may also contribute to changes in soil biodiversity in SSA. For example, a more than average numbers of earthworm and termite taxa are found under relatively warmer, drier conditions (Ayuke *et al.*, 2011). This is contrary to the observation that earthworm and termite diversity increases with increases in rainfall or soil moisture, as generally found in temperate climates (Bohlen *et al.*, 1995; Curry, 2004). However, seasonality of rainfall in the tropical regions means rainfall amounts per season may be more important than the annual total. Lower taxonomic richness among sites in Eastern Africa may be attributable to less favourable conditions arising from high rainfall and low temperatures at higher altitudes (Ayuke *et al.*, 2011).

Intense management practices that include application of pesticides and frequent cultivation affect soil organisms, often altering community composition of soil fauna. Soil biological and physical properties (e.g., temperature, pH, and water-holding characteristics) and microhabitat are altered when natural habitat is converted for agricultural production (Crossley, Mueller and Perdue, 1992). Changes in these soil properties may be reflected in the distribution and diversity of soil meso fauna. Organisms adapted to high levels of physical disturbance become dominant within agricultural communities, thereby reducing the richness and diversity of soil fauna (Paoletti, Foissner and Coleman, 1993).

The extent of soil sterilization and loss of soil biodiversity in SSA has yet to be quantified on a large-scale across the region. However, it is clear that unsustainable soil management practices have depleted soil organic matter, promoted soil degradation and may have caused soil fauna and flora imbalances. This land degradation will continue unless land users in SSA adopt an agro-biological approach to managing their soils (Van der Merwe *et al.*, 2002).

9.3.5 | Soil contamination and pollution

Chemical fertilizers and pesticides have had negative effects on the environment in most SSA countries. However, soil pollution through agrochemical use in SSA has been of less concern compared to other regions of the world, mainly because of the low levels of application. However, with the increasing push towards higher use of fertilizer, pesticide and herbicide to boost productivity, efforts will be needed to reduce the associated negative impacts on soil quality (IAASTD, 2009).

Chemical pollution has emerged as a threat to soil quality. According to a United Nations Environment Programme (UNEP, 2007) environmental assessment in ten communities in Ogoni land in southeastern Nigeria which had been affected by crude oil spills, drinking water, the air and agricultural soils contained over 900 times the permissible levels of hydrocarbon and heavy metals. The report acknowledged that, even if all its recommendations were implemented, recovery might take 30 years. Other published research work suggests that heavy metal pollution is occurring across SSA. Heavy metal (Pb, Cd, Hg, Cu, Co, Zn, Cr, Ni, As) pollution of soils has been reported in Nigeria, Kenya, Ghana and Angola (Fakayode and Onianwa, 2002; UNEP, 2007; Odai et al, 2008).

Change of land use, particularly urbanization, is another factor in soil contamination. National data from South Africa indicate that areas under urban, forestry and mining land uses have all increased over the last decade, whereas the cultivated area has decreased. The urban area has increased from 0.8 percent of total area to 2 percent, forestry from 1.2 percent to 1.6 percent, and mining from 0.1 percent to 0.2 percent, while the cultivated area has decreased from 12.4 percent to 11.9 percent. The increase in the urban and mining areas is a major concern in terms of soil conservation and future use. Urban development involves soil sealing which irreversibly removes soils from other land uses. Mining results in serious chemical and physical soil degradation which subsequently can only be partially restored.

9.3.6 | Soil acidification

In SSA, extremely acid soils, mainly potential or actual acid sulphate soils, occur only in a small area around the Niger delta and sporadically along the coastal plains of West Africa. Other acid soils occupy about 15 percent of the continent and are mainly found in the moist parts of the semi-arid zones and in sub-humid areas. Many of the Acrisols (Ultisols) and some Lixisols (Alfisols) have acid surface and subsurface horizons which, coupled with the moisture stress conditions, makes these soils extremely difficult to manage under low-input conditions. In West Africa, the annual additions of dust from the Sahara brought by the Harmattan winds raise the pH of the surface horizons. The problem is therefore less acute there, although subsoil acidity remains (Eswaran *et al.*, 1996). Another region of acid soils occurs south of the tropic of Capricorn and includes parts of South Africa (Beukes, Stronkhorst and Jezile, 2008a,b) where it poses a serious soil chemical problem and is in fact one of the greatest production-limiting factors.

9.3.7 | Salinization and sodification

Salinization is defined as a change in the salinity status of the soil. This can be caused by improper management of irrigation schemes, particularly in the arid and semi-arid regions. Irrigation-induced soil acidity is aggravated when irrigation is practiced on soils unsuitable for irrigation (Barnard *et al.*, 2002). Salinization can also be caused if sea water intrudes into coastal regions either on the surface or into groundwater. It may also arise in closed basins when there is excessive abstraction of groundwater from aquifers of different salt content. Salinization also takes place where human activities lead to increased evapotranspiration from soils on salt-containing parent material or where saline ground water is being pumped out (Oldeman, 2002).

In the arid and semi-arid parts of Africa, soil salinity and alkalinity are major problems affecting about 24 percent of the continent. Soils with pH>8.5 are designated as alkaline (Eswaran *et al.*, 1996). Soil salinity and sodicity problems are common in arid and semi-arid regions where rainfall is insufficient to leach salts and excess sodium ions out of the rhizosphere. More than 80 million ha of such soils are found in Africa.

Increasing temperatures may result in high evaporative demands that may activate the capillary rise of salts, leading to soil salinization. The results of a study in Sudan showed a significant increase in salinity in the Dongla area in the north, where the annual rainfall is the lowest in the country. This increase is associated with fluctuation and erratic distribution of rainfall, as well as with a rise in temperature (Abdalla *et al.*, 2011).

9.3.8 | Waterlogging

Human intervention in natural drainage systems may lead to waterlogging or flooding by river water. Most waterlogging threats are due to effects of human-induced hydromorphy. Causes include a rising water table (for example, due to construction of reservoirs or irrigation) or increased flooding caused by higher peak flows of rivers. The technology of flooding in paddy fields to provide a proper environment for paddy rice is generally not considered a threat to ecosystem services, although it may increase the emissions of GHG. It is estimated (Oldeman, Hakkeling and Sombroek, 1991) that waterlogging constitutes 1.5 percent of the non-erosion soil degradation threats in Africa.

9.3.9 | Compaction, crusting and sealing

The population of the Sub-Saharan Africa (830 millions) is approximately 12 percent of the world population. SSA population has been growing at a rate of 2.6 percent year during the last decade, although the rate is now declining. The tendency in the region is towards the concentration of growing populations in moderately large cities (rather than mega-cities). Since the early 1970s, several SSA countries have experienced accelerated urban expansion, recording some of the highest urban growth rates in the world of up to 5 percent per year (Todaro, 2000). There are numerous examples of single-city dominance in the region. For instance, in Mozambique, Maputo accounts for 83 percent of the country's urban population, while the figures for Dakar, Lome, Kampala and Harare are 65, 60, 52 and 50 percent respectively (World Bank, 2002). Nigeria and South Africa represent exceptions to this single-city dominance, as they have several large and well distributed urban centres. South Africa and Nigeria are also the countries recording the highest amount of impervious surface area (ISA) in the region, and they have high Urbanization Indexes (the ratio between the total area of the country and the urbanized area) (Figure 9.2).

9.4 | The most important soil threats in Sub-Saharan Africa

Of the threats to soils and related ecosystem functions in SSA listed in Section 9.3, the most critical are soil erosion, loss of soil organic matter and soil nutrient depletion (UNEP, 2013). Loss of soil biodiversity is also a significant threat in SSA. These four threats are interrelated. More is known about the first three and these are discussed in greater detail in this section.

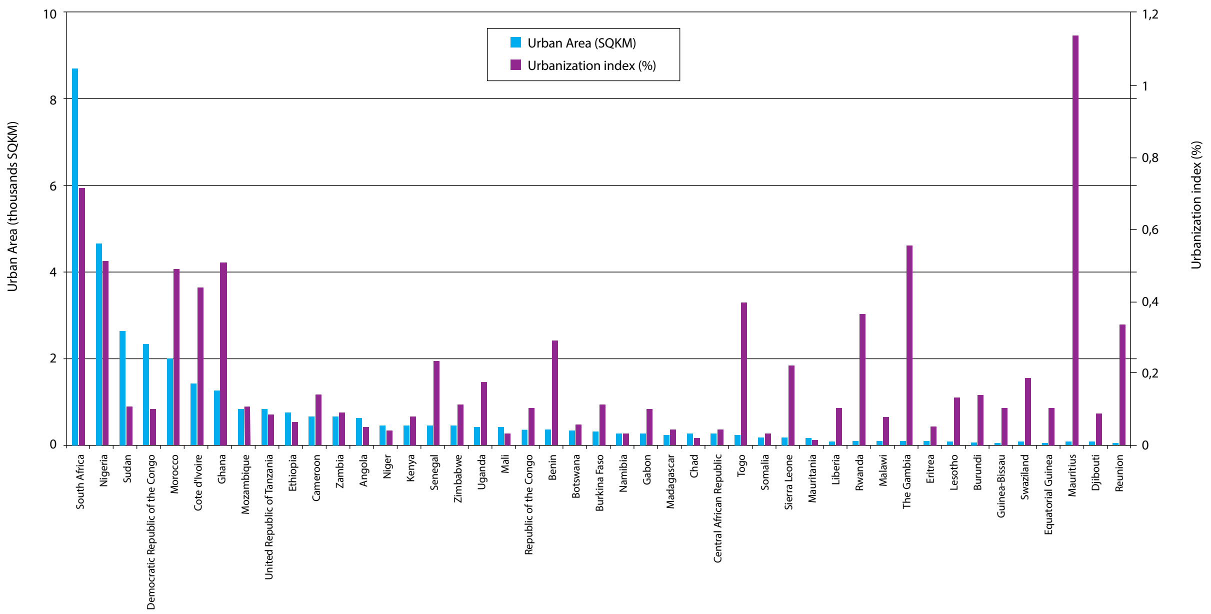


Figure 9.2 | Extent of urban areas and Urbanization Indexes for the Sub-Saharan African countries. Source: Schneider, Friedl and Potere, 2010.

9.4.1 | Erosion by water and wind

Direct causes of soil erosion

Expansion of land for agriculture: Soil erosion can be a natural process but it is also often caused or accelerated by human activities that involve inappropriate land use. As discussed above (Section 9.3), much of the change in land use practices in SSA has been driven by the need to increase production and incomes in an economic and institutional context where the means to improve productivity in a sustainable fashion are generally not available. The processes involved - rapid expansion of agricultural land, shortening of fallow periods, increased use of fire – are discussed in full in Section 9.3.2 above.

As fertilizer consumption did not increase to compensate for the loss of soil nutrients resulting from the intensification of land use, there has been widespread mining of soil nutrients and soil organic matter. As a consequence of the type of soils that occur in the region and because of generally poor land management, many SSA croplands now have low soil organic matter contents and soils that have a low pH and suffer from aluminium toxicity. On degraded soils with low organic matter, inorganic fertilizers are also easily leached, and this process has devastating long-term effects for agricultural productivity. Alternative means of maintaining soil fertility, such as crop rotation with biological nitrogen fixing (BNF) species, application of green manure, agroforestry, composting, rock phosphates, etc., have proved to be highly effective at the local scale. However, these technologies have not been applied widely enough to have an impact at a national let alone continental scale.

Overgrazing: There has been much debate on the impacts in SSA of high grazing pressures on vegetation composition in rangelands. The current understanding is that continued high grazing pressure may affect rangeland productivity, particularly in the long term. Vegetation studies also show that high grazing pressures lead to changes in species composition, which may reduce the resilience of rangelands to drought (Hein and De Ridder, 2006). During a drought, degraded rangelands show a much stronger decline in productivity than non-degraded rangelands. Recent years have seen droughts with severe impacts on livestock and local livelihoods in parts of Niger and in the East African drylands (Uganda and Kenya).

Deforestation: Most forests and woodlands in SSA are experiencing rapid rates of deforestation. Deforestation is driven by a number of processes, in particular: (i) the continued demand for agricultural land; (ii) local use of wood for fuel, charcoal production and construction purposes; (iii) large-scale timber logging, often without effective institutional control of harvest rates and logging methods; and (iv) population movements and resettlement schemes in forested areas. The amount of cropped land in SSA has increased by about 40 million ha in 30 years (1975-2005), most of it at the expense of forests and woodlands (FAO, 2015). Further expansion of cropland would be at the expense of forests or rangeland.

Socio-economic causes of soil erosion

Population expansion: Behind these direct drivers of erosion lies the demographic driver of a continuously growing population (Figure 9.3). The rate of SSA population growth has moderated in recent years and is currently 2.1 percent per year. Nonetheless, in the next 15 years SSA will have to accommodate at least 250 million additional people, a 33 percent increase (UNDP, 2005). With the increase in population comes an increased demand for living space and food which will directly affect soil use in the region.

<u>Poverty</u>: General poverty of the farming population and the low potential of the farming systems characteristic of SSA pose considerable challenges to sustainable agricultural growth and poverty reduction. Poverty is particularly prevalent in the pastoral/agro-pastoral, highland perennial and forest based farming systems which constitute one-third of the total SSA production systems (FAO and World Bank, 2001). From Figure 9.4 it is clear that SSA has many countries where a large percentage of the population is living below the poverty line.

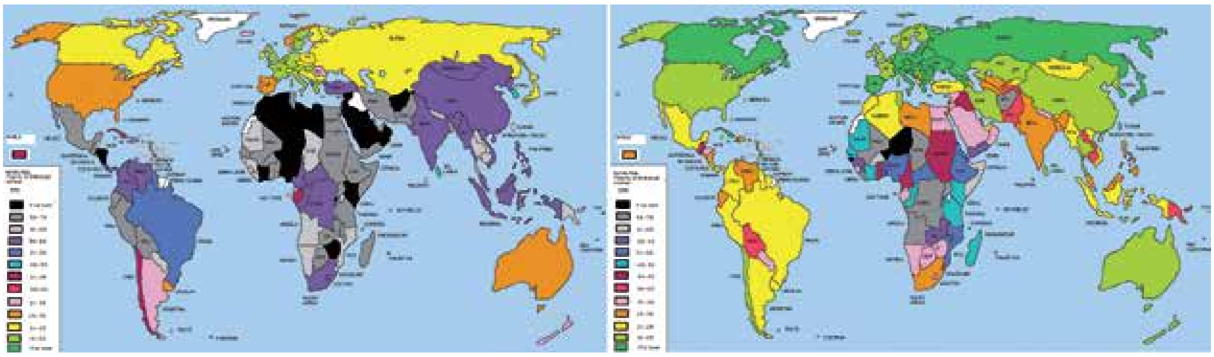


Figure 9.3 | The fertility rate (the number of children a woman is expected to bear during her lifetime) for 1970 and 2005.
Source: Fooddesert.org

<u>Climate Change</u>: Climate change is predicted to affect SSA agro-ecosystems on a significant scale in the coming decades. The continent has a long history of rainfall fluctuations of varying lengths and intensities. Severe droughts affected East and West Africa alike during the 1910s. Drought episodes were generally followed by increasing rainfall levels, but negative trends were observed again from the 1950 onwards, culminating in the droughts of the early 1970s and mid-1980s. These droughts had an impact on the susceptibility of soils to erosion.

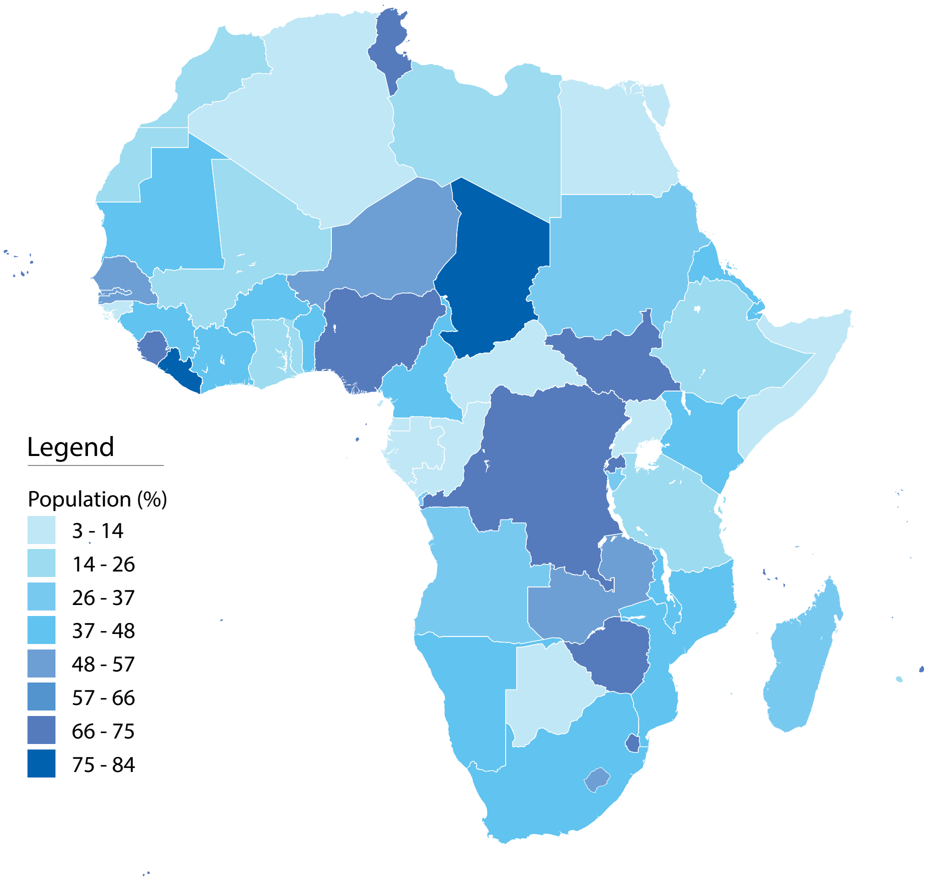


Figure 9.4 | Percentage of population living below the poverty line.
Source: CIA World Factbook, 2012.

Water erosion: its extent and distribution in the region

Water erosion constitutes 46 percent of the land degradation types in SSA, and wind erosion accounts for a further 38 percent (FAO, 2005). The most recent continent-wide assessment shows that 494 million ha, or 22 percent of the agricultural land (including rangelands) in Africa, are affected by water erosion (Oldeman, Hakkeling and Sombroek, 1991). The assessment confirms common field observations that overgrazing is the main cause of soil erosion, followed by inappropriate cultivation techniques on arable land. In this context it is important to note that the number of cattle in Africa almost doubled in the period 1961-1994, while the area of grazing lands hardly increased (FAO, 2015). For the future, the expected intensification of use on currently cultivated lands, expansion of cultivation into more marginal areas, reduction in grazing lands and the increasing numbers of livestock are likely to increase vulnerability to erosion.

As discussed above (9.3.1), severely eroded areas in Africa can be found in South Africa, Sierra Leone, Guinea, Ghana, Liberia, Kenya, Nigeria, Zaire, Central African Republic, Ethiopia, Senegal, Mauritania, Niger, Sudan and Somalia. More than 20 percent of SSA's agricultural land and pasture has been irreversibly degraded, mainly by soil erosion (UNSO/SEED/BDP, 1999).

Erosion has assumed a serious dimension in Nigeria, affecting every part of the country. In the eastern part of the country, erosion has ravaged wide areas. Active and inactive gullies have formed with surface areas ranging from 0.7 km^2 in Ohafia to 1.15 km^2 in Abiriba in Abia State. The width of the gullies ranges between 0.4 km in Ohafia and 2.4 km in Abiriba. A gully with a depth of 120 m has been recorded at Abiriba (Ofomata, 1985). In addition, agricultural practices have contributed to the problems of widespread sheet erosion. Erosion is thus exerting major pressure on soil resources with far-reaching consequences for both the population and the environment (Jimoh, 2000). In the northern areas of Nigeria, erosion is equally serious, especially in places like Shendam and Western Pankshin in Plateau State, as well as at Ankpa and Okene in Kogi State. Gully erosion is also prominent in Efon-Alaaye, Ekiti State in the western part of the country (Adeniran, 1993).

The areas of Nigeria most affected by erosion are the Agulu and Nanka districts of the eastern part of Nigeria, and the Shendam and western Pankshin areas of Plateau State, Nigeria (Udo, 1970; Okigbo, 1977). Elsewhere, the Imo State government has estimated that about 120 000 km^2 of land has been devastated by gully erosion. As a result, eight villages have been destroyed and 30 000 people needed to be resettled. Erosion damage in Imo and Anambra states was estimated to cause the loss of over 20 tonnes of fertile soil per annum, at an economic cost of over 300 million naira per annum. Gullies extended to depths of over 120 m and widths up to 2 km wide (Adeleke and Leong, 1980). In 1994, about 5 000 people were rendered homeless due to erosion in Katsina State, Nigeria. Properties worth over 400 million naira were destroyed and many lives lost. Other areas affected by erosion include Auchi in Edo State, Efon Alaye in Ondo State, Ankpa and Okene in Kogi State, and Gombe in Bauchi State. In many areas, erosion has resulted in a physical loss of available land for cultivation. For example, about 1 000 ha of cultivable land has been lost to erosion at the Agulu-Nanka area of Nigeria. Thus the loss of homes and crops, disruption of communication routes, financial losses and attendant hydrological problems can all stem from erosion problems.

Nearly 90 percent of rangelands and 80 percent of farmlands in the West African Sahel, Sudan, and northeast Ethiopia are seriously affected by land degradation, including soil erosion. More than 25 percent of South Africa is seriously degraded by erosion. Almost 70 percent of Uganda's territory was degraded by soil erosion and soil nutrient depletion between 1945 and 1990. Across SSA, more than 20 percent of agricultural land and pastures has been irreversibly degraded, affecting more than 65 percent of Africa's population (Global HarvestChoice, 2011).

Considering that over 80 percent of South Africa's land surface is covered by natural vegetation, the estimated annual soil loss of 2.5 tonnes soil per ha is excessive. These rates of soil loss far exceed tolerance levels and are almost ten times the estimated rate of soil formation, which has been estimated at 0.31 tonnes ha^{-1} yr^{-1} in the case of a 1 m thick solum of a tropical soil (Van der Merwe, 1995; see Section 6.1). Soil organic matter (SOM) plays a major role in ensuring soil stability. There is a general decline in SOM in South African soils. An estimated 20 percent of the country's total surface area is potentially highly erodible. Bearing in mind the country's geology, rainfall and topographic characteristics in addition to declining SOM, soil erosion is likely to stay a dominant soil degradation process.

Sediment movement by erosion contributes significantly to shifts in soil fertility. Sediment movement is widespread in South Africa as reflected by the annual losses of 3 300 tonnes N, 26 400 tonnes P and 363 000 tonnes K estimated by Du Plessis in 1986 (Van der Merwe, 1995). Periodic floods transport massive amounts of sediment and nutrients within catchments. The Demoina flood in 1984, for instance, deposited as much as 34 million tonnes of sediment in the Mfolozi flats (Scotney and Dijkhuis, 1990). One 1985 study used a siltation approach to estimate the siltation load carried by the Tugela River, finding soil loss from the catchment area as high as 4.4 tonnes ha^{-1} yr^{-1} (De Villiers *et al.*, 2002). It has been estimated that water erosion affects 6.1 million ha of cultivated soils in South Africa. Of this area, 15 percent of soils are seriously affected, 37 percent moderately affected, and the rest slightly affected.

Wind erosion in the region

Wind erosion physically removes the lighter, less dense soil constituents such as organic matter, clays and silts, thus removing the most fertile part of the soil and lowering soil productivity (Lyles, 1975). In SSA, soil erosion by wind occurs mainly in the arid and semiarid regions. The occurrence of wind erosion at any one site is a function of weather events interacting with soil and land management through the effects of weather on soil structure, tilth and vegetation cover. At the southern fringe of the Sahara Desert, a special dry and hot wind, locally termed Harmattan, occurs. These North-easterly or Easterly winds normally blow in the dry winter season under a high atmospheric pressure system. When the wind force of Harmattan is beyond the threshold value, sand particles and dust particles will be blown away from the land surface and transported for several hundreds of kilometres across the land and as far as the Atlantic Ocean (WMO, 2005). Areas in SSA most susceptible to wind erosion are the southern fringe areas of the Sahara, Botswana, Namibia, Zimbabwe, Tanzania and South Africa (Favis-Mortlock, 2005).

It is estimated that 25 percent of South Africa is affected by wind erosion (Laker, 2005), amounting to an estimated 10.9 million ha. Of this area, 7 percent is seriously affected, 29 percent moderately and 64 percent slightly (Barnard *et al.*, 2002). Wind erosion is particularly evident on drift sands in the coastal areas, but also on cultivated land in the Highveld areas. The seriousness of wind erosion can be deduced from the situation in the Eastern Cape Province where there are over 14 000 ha of drift sand (Barnard *et al.*, 2002). Most of South Africa's prime agricultural soils in the relatively arid western part of the country are wind-blown sand deposits (De Villiers *et al.*, 2002).

Wind erosion may cause off-site effects, such as the covering of the terrain with wind-borne soil particles from distant sources. It is estimated that more than 100 million tonnes of dust per annum are blown westward from the African continent across the Atlantic. The amount of dust arising from the Sahel zone has been reported to be around or above 270 million tonnes per year which corresponds to a loss of 30 mm per m_2 per year or a layer of 20 mm of soil particles over the entire area (WMO, 2005).

9.4.2 | Loss of soil organic matter

The loss of vegetative cover and decline in the level of soil organic matter (SOM) are the root cause of most soil degradation, since all the physical, chemical and biological problems follow a drop in SOM content. Soil organic matter is a key component of any terrestrial ecosystem, and any variation in its abundance and composition has important effects on many of the processes that occur within the system. The amount of organic matter and size of soil carbon stock results from an equilibrium between the inputs into the system, which are mostly from biomass detritus, and outputs from the system, largely decomposition and volatilization. These processes are driven by various parameters of natural or human origins (Schlesinger and Winkler, 2000; Amundson, 2001; Section 2.1). A decrease of organic matter in topsoil can have dramatic negative effects on the water holding capacity of the soil, on the soil structure stability and compactness, on nutrient storage and supply, and on soil biological components such as mycorrhizas and nitrogen-fixing bacteria (Sombroek, Nachtergaele and Hebel, 1993).

Direct causes of SOM decline

Apart from climatic factors that influence carbon changes in the soil, inappropriate land uses and practices are the main cause of decline in SOM. These uses and practices include: monoculture cereal production; intensive tillage; short to no fallow; and reduction or absence of crop rotation systems. The long-term effects of these management actions are now being experienced across SSA.

The SSA experience is not unique. Across the globe, the carbon balance of terrestrial ecosystems is being changed markedly by the direct impact of human activities. Land use change was responsible for 20 percent of global anthropogenic CO_2 emissions during the 1990s (IPCC, 2007). In SSA, land use change is the primary source, much of it through burning of forests.

The impact of land use change varies according to the land use types. The clearing of forests or woodlands and their conversion into farmland in tropical SSA reduces the soil carbon content mainly through reduced production of organic inputs, increased erosion rates and the accelerated decomposition of soil organic matter by oxidation. Various reviews agree that the loss amounts to 20 to 50 percent of the original carbon in the topsoil, with deeper layers less affected, if at all (Sombroek, Nachtergaele and Hebel, 1993; Murty *et al.*, 2002; Guo and Gifford, 2002). However, conversion of forests to pasture does not necessarily change soil carbon (Guo and Gifford, 2002) and may actually increase the soil organic matter content (Sombroek, Nachtergaele and Hebel, 1993). Where shifting cultivation is practiced, soil carbon has been found to reduce to half the level before the land was cleared for use (Detwiler, 1986). Surprisingly, studies suggest that commercial logging and tree harvesting do not result in long-term decreases in soil organic matter (Knoepp and Swank, 1997; Houghton *et al.*, 2001; Yanai *et al.*, 2003). Clearly many factors are at play: changes in the amount of soil organic matter following conversion of natural forests to other land uses depend on several factors such as the type of forest ecosystem undergoing change (Rhoades, Eckert and Coleman, 2000), the post conversion land management practiced, the climate (Pastor and Post, 1986) and the soil type and texture (Schjønning *et al.*, 1999).

Socio-economic causes of SOM decline

In Sub-Saharan Africa, socio-economic pressures to increase production and incomes create incentives for farmers to reduce the length of fallow periods, cultivate continuously, overgraze fields, or remove much of the above-ground biomass for fuel, animal fodder and building materials. These practices can result in the reduction of SOM, water holding capacity and nutrients. They also increase the soil's vulnerability to erosion (Lal, 2004).

Extent of SOM decline in the region

As with negative nutrient balances (see below, 9.4.3), SOM decline threatens soil productivity. In SSA, the concentration of organic carbon in the top soil is reported to average 12 mg kg^{-1} for the humid zone, 7 mg kg^{-1} for the sub-humid zone and 4 mg kg^{-1} or less in the semi-arid zone (Williams *et al.*, 1993). These inherently low concentrations of soil organic carbon are due not only to the low root growth of crops and natural vegetation but also to the rapid turnover rates of organic materials caused by high soil temperature associated with abundant micro-fauna, particularly termites (Bationo *et al.* 2003). There is considerable evidence for rapid decline in SSA of soil organic C levels where cultivation of crops is continuous (Bationo *et al.*, 1995). For sandy soils, average annual losses in soil organic C may be as high as 5 percent, whereas for sandy loam soils reported losses are much lower, averaging 2 percent (Pieri, 1989). Results from long-term soil fertility trials indicate that losses of up to 0.69 tonnes carbon ha^{-1} yr^{-1} in the soil surface layers are common in Africa, even with high levels of organic inputs (Nandwa, 2003; Bationo *et al.*, 2012).

Responses to SOM decline

Appropriate land management could reverse the trend of SOM decline and contribute to soil carbon sequestration. In fact, increasing the SOM content is crucial for future African agriculture and food production (Bationo *et al.*, 2007; Sanchez, 2000). Several studies on SSA have shown that a synergetic effect exists between mineral fertilizers and organic amendments and that this synergy leads to both higher yields and higher SOC content (Palm *et al.*, 2001, Vågen *et al.*, 2005; Bationo *et al.*, 2007).

Barnard *et al.* (2002) emphasized the importance of establishing and maintaining an effective and intimate association between soils and growing plants. Biological measures for stabilizing slopes and decreasing the rate of runoff are essential. It is often necessary to undertake some form of land shaping prior to this, together with chemical amelioration and nutrient augmentation.

There is abundant evidence that soil organic matter plays a major role in stabilizing soil and in preventing its physical, chemical and biological deterioration. This has been demonstrated under South African conditions by several scientists as reported by Barnard *et al.* (2002). For example, Folscher (1984) pointed out that micro-organisms played a vital role in the chemo-biological condition of soils. Under predominantly heterotrophic microbial activity, physical and chemical stability could be expected, while under autotrophic microbial activity, acidification and nutrient decline could be forecast. Much more attention therefore needs to be paid to the dynamic nature of soil and its physical, chemical and biological interactions.

Because nitrogen dynamics are so important in establishing a stable C:N ratio in soil, alternative sources of natural forms of nitrogen such as suitable legumes should be included in rotations. Rhizobial and mycorrhizal associations need to be stimulated and soil organic carbon and nutrient levels need to be systematically monitored and evaluated. Other soil quality indicators relating to specific situations need to be developed and utilized, with emphasis on earthworm populations as an indicator of soil quality. Reduced, minimum and no-till systems also need to be investigated and implemented where possible. These have been introduced worldwide and are being adopted in many parts of South Africa (Van der Merwe *et al.*, 2000).

Land degradation leads to a release of carbon to the atmosphere through oxidation of soil organic matter. With present concerns about climate change and the increase in atmospheric CO_2, it has been suggested that this process could be reversed and that the soil could be used to capture and store carbon. Soil organic matter could be gradually built up again through carbon sequestration. Among the land use changes which could be promoted with this objective in mind are improved agricultural practices, the introduction of agroforestry, and reclamation of degraded land. By such means, the carbon stored in soils could be substantially increased by amounts of the order of 30–50 tonnes ha^{-1}. Thus land use changes which are beneficial to local communities would, in addition, fulfil a global environmental objective.

9.4.3 | Soil nutrient depletion

Nearly 3.3 percent of agricultural gross domestic product (Agricultural GDP) in Sub-Saharan Africa is lost annually due to soil and nutrient losses (Global HarvestChoice, 2011). In Africa, harvesting grains and crop residues from the land removes considerable quantities of soil carbon content. As lost nutrients in SSA are only very partially replaced with fertilizers, these losses contribute to negative nutrient balances (Gray, 2005). As a result, soil fertility decline has been described as the single most important constraint on food production and food security in SSA. Soil fertility decline (also described as soil productivity decline) is a deterioration of chemical, physical and biological soil properties. Besides soil erosion, the main processes contributing to nutrient depletion in SSA are:

- Decline in organic matter and soil biological activity
- Degradation of soil structure and loss of other soil physical qualities
- Reduction in availability of major nutrients (N, P, K) and micro-nutrients
- Increase in toxicity, due to acidification or pollution

In the first assessment of the state of nutrient depletion in SSA, which was carried out in 1990, nutrient balances were calculated for the arable lands of 38 countries across the continent. Four classes of nutrient-loss rates were established (Table 9.2). As discussed above (9.3.3), the average nutrient loss in 1990 was estimated to be 24 kg nutrients ha^{-1} per year (10 kg N; 4 kg P_2O_5, 10 kg K_2O). Countries with the highest depletion rates, such as Kenya and Ethiopia (Table 9.3), also have severe soil erosion.

Table 9.2 | Classes of nutrient loss rate (kg ha^{-1} yr^{-1}). Source: Stoorvogel and Smaling, 1990.

Class	Low	Moderate	High	Very High
N	<10	10-40	21-40	>40
P_2O_5	<4	4-7	8-15	>15
K_2O	<10	10-40	21-40	>40

Direct causes of nutrient decline

Fertility decline is caused by a negative balance between output (harvesting, burning, leaching, and so on) and input of nutrients and organic matter (manure/fertilizers, returned crop residues, mineral deposition through flooding). The estimate of nutrient depletion in SSA cited above is worrying. However, some scientists (Roy *et al.*, 2003) have expressed concern about the approach used, as it is based on approximation and aggregation at country level which could be misleading, masking the 'bright' spots and the 'hot' spots where urgent nutrient replenishment is required. Assessment of fertility decline at micro-watershed or community level would be more appropriate.

Socio-economic causes of nutrient decline

There are various factors that indirectly influence nutrient depletion in SSA and they vary between ecological regions and amongst countries and locations within a given ecological region. The cost of buying mineral fertilizer can put it beyond the reach of many SSA smallholders (World Bank, 1998). Farm-level fertilizer prices in Africa are among the highest in the world (Bationo *et al.* 2012). One metric tonne of urea, for example, costs about US$ 90 in Europe, US$ 500 in Western Kenya and US$ 700 in Malawi. These high prices can be attributed to the removal of subsidies, high transaction costs, poor infrastructure, and poor market development, inadequate access to foreign exchange and credit facilities, transportation costs and lack of training to promote and utilize fertilizers. For example, it costs about US$ 15, US$ 30 and US$ 100 to move 1 tonne of fertilizer 1 000 km in the United States, India and SSA respectively (Bationo *et al.*, 2012).

Many farmers do not follow recommended fertilizer application rates because of cash or labour constraints. In much of Southern and Western Africa there is a dry season lasting 7 to 8 months. In the first weeks of the rainy season many farm operations such as planting, weeding, and fertilizing must take place in rapid succession. Farmers who weed maize twice at critical periods can achieve a higher yield with half the amount of fertilizer used by farmers who only weed once (Kabambe and Kumwenda, 1995), but many farmers do not have sufficient labour to weed more often.

Output price instability is an important factor posing risks for fertilizer users in Western Africa (Byerlee *et al.*, 1994). When markets are sparse, as they are in many rural areas dominated by subsistence production, the variations in market prices of crops tend to be wider than in regions where markets are more fully developed. Overall, the economics of fertilizer use are often not sufficiently positive, especially under rainfed conditions; farmers are cash-poor and so cannot buy expensive inputs; and farmers are highly averse to making cash outlays in unpredictable climatic conditions and with uncertain commercial returns.

Extent of nutrient decline in the region

The results of an FAO study (1983-2000) (Lesschen *et al.*, 2003; Stoorvogel and Smaling, 1990) which assessed N, P and K balances by land use system and by country revealed a generally downward trend in soil fertility in Africa. Overall, the study suggests that all African countries except Mauritius, Reunion and Libya show negative nutrient balances every year. The result for 2000 showed a deteriorating nutrient balance for almost all countries. This was influenced by the FAO estimates for crop production in 2000 and an accompanying expected decrease in fallow areas. For SSA as a whole, the nutrient balances were: -22 kg ha^{-1} in 1983 and -26 kg ha^{-1} in 2000 for N; -2.5 kg ha^{-1} in 1983 and -3.0 in 2000 for P; and -15 kg ha^{-1} in 1983 and -19 kg ha^{-1} in 2000 for K.

Table 9.3 lists nutrient balances for several SSA countries. The study found substantial differences between countries. In 1993-1995 the difference between nutrient inputs and nutrient losses in the continent ranged from -14 kg of NPK per ha per year in South Africa to 136 kg in Rwanda. Burundi and Malawi also experienced rates of nutrient depletion above 100 kg of NPK per ha per year.

Densely populated and hilly countries in the Rift Valley area (Kenya, Ethiopia, Rwanda and Malawi) had the most negative values, owing to high ratios of cultivated land to total arable land, relatively high crop yields, and significant erosion problems. In the semiarid, arid, and Sudano-Sahelian areas that are more densely populated, soils were found to lose 60-100 kg of nitrogen, phosphorus, and potassium (NPK) per ha each year. The soils of these areas are shallow, highly weathered, and subject to intensive cultivation with low-level fertilizer use.

Short growing seasons contribute to additional pressure on the land. In important agricultural areas in the sub-humid and humid regions and in the savannas and forest areas, nutrient losses vary greatly. Rates of nutrient depletion range from moderate (30 - 60 kg of NPK per ha per year) in the humid forests and wetlands in southern Central Africa, to high (> 60 kg NPK per ha per year) in the East African highlands.

More countries fall into the high depletion range than the medium range. Nutrient imbalances are highest where fertilizer use is particularly low and nutrient loss, mainly from soil erosion, is high. The low gains in nutrients, inherently low mineral stocks in these soils, and the harsh climate of the interior plains and plateaus aggravate the consequences of nutrient depletion. The estimated net annual losses of nutrients vary considerably by sub-region: 384 800 metric tonnes for North Africa, 110 900 metric tonnes for South Africa, and 7 629 900 metric tonnes for Sub-Saharan Africa as a whole. This represents a total loss of US$ 1.5 billion per year in terms of the cost of nutrients as fertilizers.

Table 9.3 | Estimated nutrient balance in some SSA countries in 1982-84 and forecasts for 2000.
Source: Stoorvogel and Smaling, 1990; Roy et al., 2003.

Country	N		P		K	
	1982-84	2000	1982-84	2000	1982-84	2000
	(kg ha^{-1} yr^{-1})					
Benin	-14	-16	-1	-2	-9	-11
Botswana	0	-2	1	0	0	-2
Cameroon	-20	-21	-2	-2	-12	-13
Ethiopia	-41	-47	-6	-7	-26	-32
Ghana	-30	-35	-3	-4	-17	-20
Kenya	-42	-46	-3	-1	-29	-36
Malawi	-68	-67	-10	-10	-44	-48
Mali	-8	-11	-1	-2	-7	-10
Nigeria	-34	-37	-4	-4	-24	-31
Rwanda	-54	-60	-9	-11	-47	-61
Senegal	-12	-16	-2	-2	-10	-14
United Republic of Tanzania	-27	-32	-4	-5	-18	-21
Zimbabwe	-31	-27	-2	2	-22	-26

More nitrogen and potassium than phosphorus get depleted from African soils. Nitrogen and potassium losses primarily arise from leaching and soil erosion. These soil problems result mainly from continuous cropping of cereals without rotation with legumes, inappropriate soil conservation practices, and inadequate amounts of fertilizer use. Among West African countries, Guinea Bissau and Nigeria experience the highest annual losses of nitrogen and potassium. Nitrogen loss in East Africa is highest in Burundi, Ethiopia, Malawi, Rwanda, and Uganda, and phosphorus loss is highest in Burundi, Malawi, and Rwanda (IFPRI, 1999).

Responses to nutrient decline

The negative nutrient balances clearly indicate that not enough nutrients are being applied in most areas (Bationo *et al.*, 2012). Annual application of nutrients in SSA averages about 10 kg of NPK per ha. Fertilizer tends to be used mostly on cash and plantation crops because of the higher profitability of fertilizer application in the production of cash crops. Food crops receive less fertilizer because of unfavourable crop/fertilizer price ratios and financial constraints faced by farmers. In addition, food crops are only partly commercialized.

To maintain current average levels of crop production without depleting soil nutrients, Africa as a whole (including North Africa) would require approximately 11.7 million metric tonnes of NPK each year, roughly three times more than the continent currently uses (3.6 million metric tonnes) (Henao and Baanante, 1999). Of this quantity, Sub-Saharan Africa would need by far the largest proportion (76 percent) because the current average level of fertilizer use is so low. Total nutrient requirements per ha per year range from Botswana's 24.5 kg ha^{-1} NPK (a figure 350 percent above current usage) to Reunion's 437.3 kg ha^{-1}NPK (about 20 NPK per ha less than the country consumes). Burkina Faso would have to increase its NPK consumption more than 11 times to maintain crop production levels without depleting nutrients and Swaziland would have to double its consumption. Estimated average use for SSA as a whole would have to increase about 4 times to meet nutrient needs at the current level of production. Generally, more nitrogen is required than potassium, and more potassium than phosphorus.

9.5.1 | Senegal

Introduction

The main objective of the national land resources assessment undertaken in Senegal between 2000 and 2010 was to identify for each land use system the status and trends of land degradation and the major sustainable land management interventions present in the country. This assessment used national and local technical expertise, including that of the land users themselves. The findings have been reported in several documents, maps and web-sites (Ndiaye and Dieng, 2013).

The methodology was based on the premise that land degradation is largely driven by the way people use the environment in which they live. The level of degradation or sustainable use of a given land resource depends to a great extent on the needs and objectives of the land user, which are limited by technical knowledge and level of access to production factors (capital, labour etc.). Choices about land use take place within an integrated production system (Jouve, 1992). Consequently, defining the units in which both degradation and sustainable land management are to be described requires the identification of areas with similar geographic characteristics and then the mapping of the different production or land use systems. In Senegal this mapping was carried out using the 'Framework for characterization and mapping of agricultural land use' (George and Petri, 2006).

In Senegal the following major land use systems were identified and characterized:

1. Aquaculture and fishing, which takes place in areas covered with mangrove and other aquatic vegetation that are regularly flooded.
2. Rainfed subsistence agriculture, which is characterized by the absence of livestock and minimal use of inputs.
3. Agropastoral systems, characterized by a significant presence of rainfed agriculture but with greater levels of livestock activity. These systems are located in areas that receive between 400 and 700 mm of rainfall.
4. Riverbank agriculture, characterized by the use of receding floodwaters to produce crops.
5. Irrigated agriculture, characterized by intensive management and relatively high use of inputs.
6. Forest based systems that exploit trees for timber.
7. Conservation areas that are protected to preserve biodiversity
8. Peri-urban agriculture, characterised by a mix of activities aimed at producing high-value products close to urban markets.
9. Nomadic grazing, which takes place in the driest areas of the country and is characterized by shifting livestock and no permanent agriculture.

The distribution of these land use systems and their extent within the country are given in Figures 9.5 and 9.6 respectively.

The national assessment of land degradation and sustainable management was carried out using available hard data and a questionnaire developed by FAO in collaboration with WOCAT (Liniger *et al.*, 2011). The evaluation describes the actual situation and assesses the trends of land change over the last ten years. The method used local observations and measurements and expert opinion and covered the whole of Senegal. It has achieved the identification and characterization of land change in terms of degradation types, their extent, degree, level, trend, causes and impacts on ecosystem services. Each of these parameters has been mapped and examples are given in Figure 9.7 (extent of dominant degradation type) and Figure 9.8 (rate of change of degradation). All information collected has been captured in a national database and analysed statistically (SOW-VU, 2010).

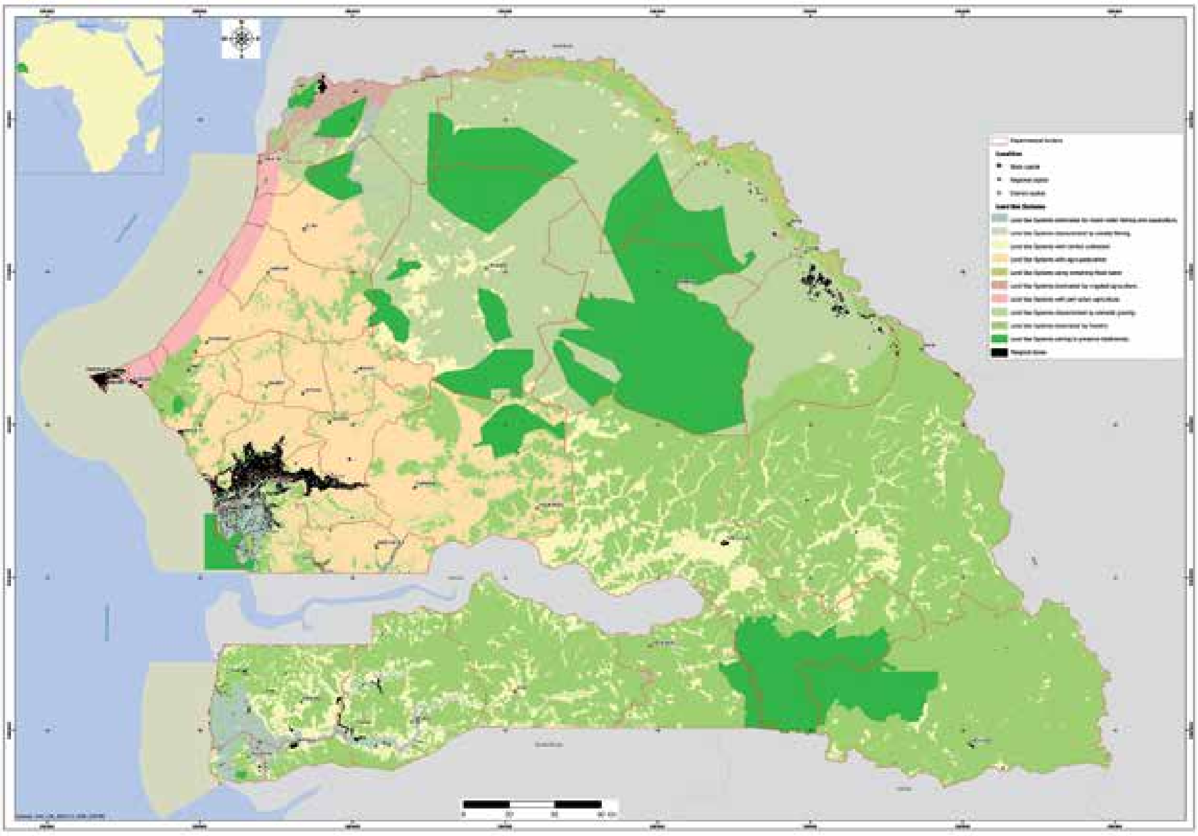


Figure 9.5 | Major land use systems in Senegal.
Source: FAO, 2010.

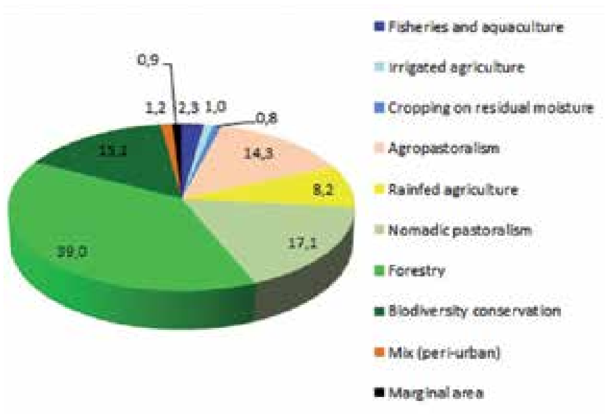


Figure 9.6 | Proportional extent of major land use systems in the Senegal.
Source: Ndiaye and Dieng, 2013.

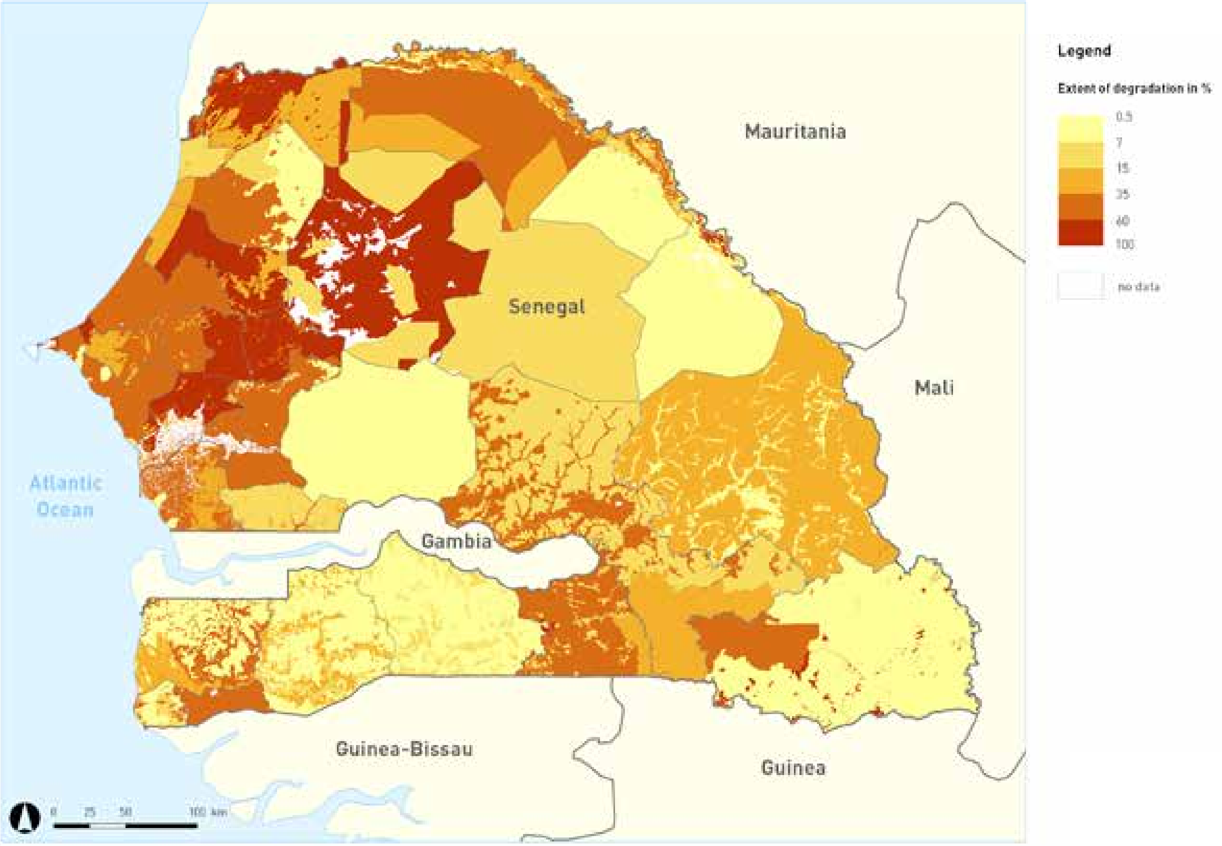


Figure 9.7 | Extent of dominant degradation type in Senegal.
Source: FAO, 2010.

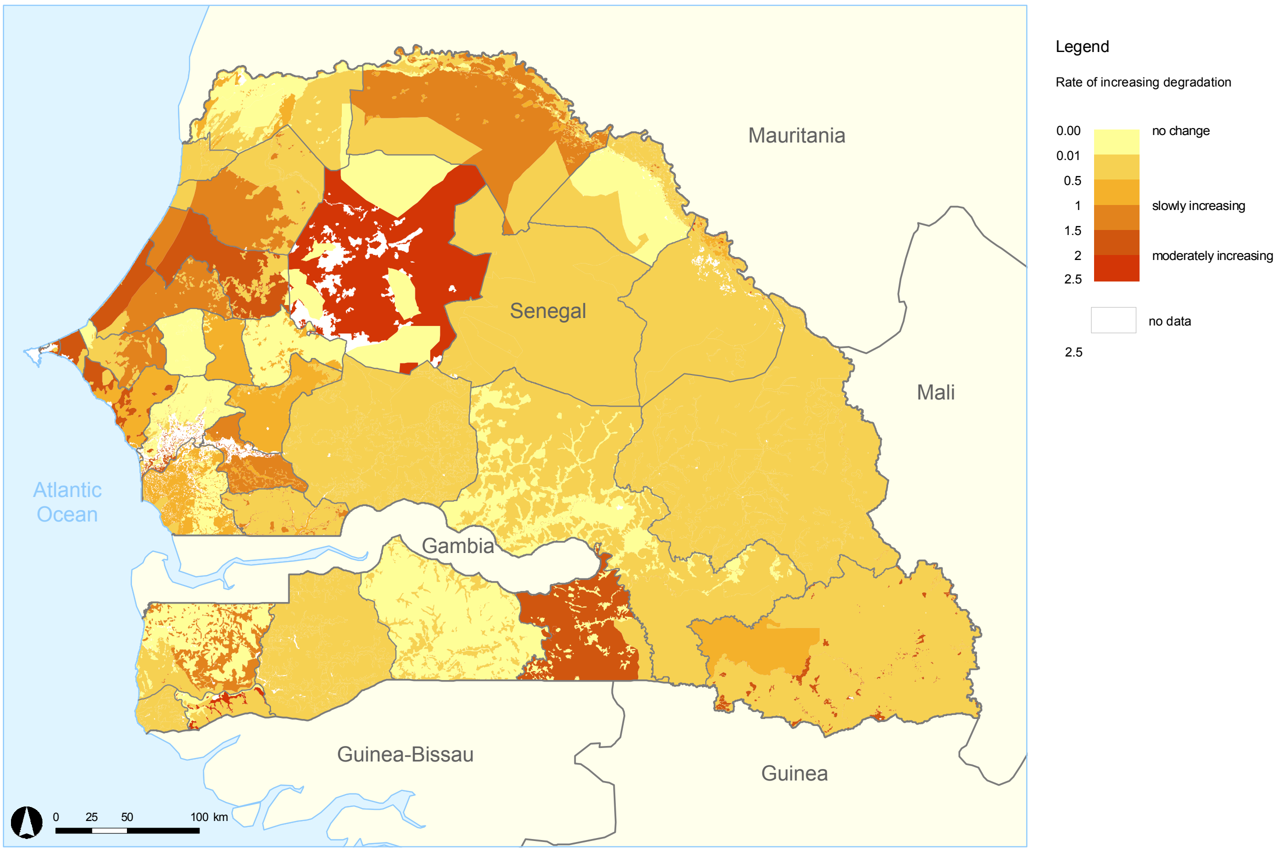


Figure 9.8 | Average rate of degradation in Senegal.
Source: FAO, 2010.

Some examples of analysis of the data show the value of the national assessment:

1. The analysis of the socio-economic drivers of land degradation in the country showed that poverty and population pressure are the main drivers, while governance and education are also significant. Land tenure and conflict situations were reported to be of minor importance as drivers of land degradation in Senegal.
2. It is population pressure and poverty which lead in a majority of cases to deforestation and overgrazing and which, together with lack of access to extension services, lead to unsustainable soil and crop management. Urbanisation and mining are minor pressures in the country.
3. Impacts of land degradation on ecosystem goods and services fell mainly on the productive services (affecting 15 percent of the area), while impacts on ecological services, in particular on biodiversity, were slightly less (13 percent). The influence on the socio-cultural provisioning services was the smallest, affecting only 6 percent of the area.
4.

Further field and socio-economic studies were undertaken at the local level, both in areas that were considered 'hotspots' for degradation and in 'bright spots' where degradation was less prevalent and sustainable management was practiced.

The analysis of results in these local areas is illustrated in Figure 9.9 which gives the impact of degradation on the various ecosystem services. There is a major impact on the net returns of the farmers in all areas, but there are also important differences according to the different situations in each zone.

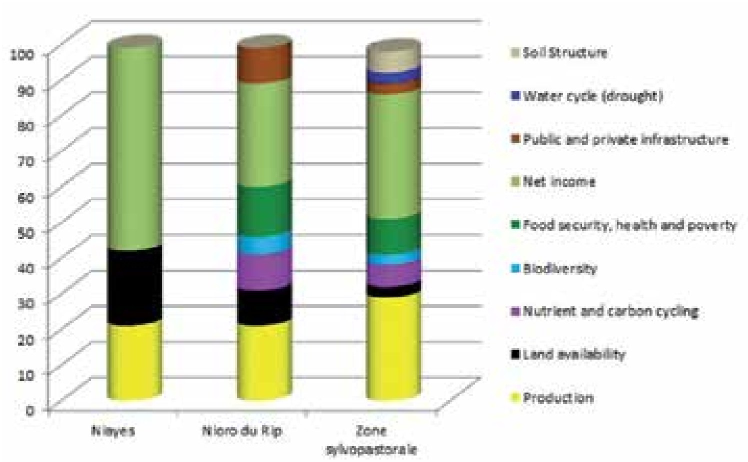


Figure 9.9 | Impact of degradation on ecosystem services in the local study areas in Senegal. Source: Ndiaye and Dieng, 2013.

Responses have included measures implemented by the government, NGOs, the communities themselves and local producers. The principal responses were: assisted natural regeneration, agro-forestry, application of organic amendments, introduction or extension of fallow periods, composting, and using a millet/groundnut rotation. Most of these responses have proved to be efficient, but their adoption by land users has been slow, affected by lack of information and by economic and/or political constraints.

9.5.2 | South Africa

Of South Africa's total area of 123.4 million ha, arable land accounts for only 11-15 percent. Of this arable land, only about a quarter is high potential. This high potential land is thus a critical resource which needs to be protected. South Africa's soils are diverse and complex as a result of varied soil formation and weathering processes. The largest proportion (81 percent) are slightly weathered and calcareous soils. More than 30 percent of soils are sandy (e.g. less than 10 percent clay content) and almost 60 percent of soils have low organic matter content (Scotney, Volschenk and Van Heerden, 1990). The most important soil limitations are shallow depth, extremes of texture, rockiness, severe wetness and high erosion hazard. In terms of soil management, it is important to note that agriculture in South Africa has a dualistic nature, with a well-developed commercial sector on the one hand, and a predominantly subsistence or small-scale agricultural sector in communal areas on the other.

The first, and to date only, nationwide study of soil distribution was done by the Land Type Survey Staff (2003) from 1970 to 2003. The study delineated areas known as 'land types' at 1:250 000 scale - land types were defined as areas displaying a marked degree of uniformity in terms of terrain form, soil pattern and climate. The study included an in-depth analysis of a number of soil profiles, termed modal profiles, selected to represent the range of soils encountered during the survey. Soils from 2 380 profiles across the country were described and analysed for morphological and chemical data and classified according to the binomial classification system developed for South Africa (MacVicar *et al.*, 1977). Each land type includes a collection of soils and their relative distribution in terms of area per landscape position, as well as their characteristics in terms of physical and chemical properties. The resultant national map of broad soil patterns is shown in Figure 9.10.

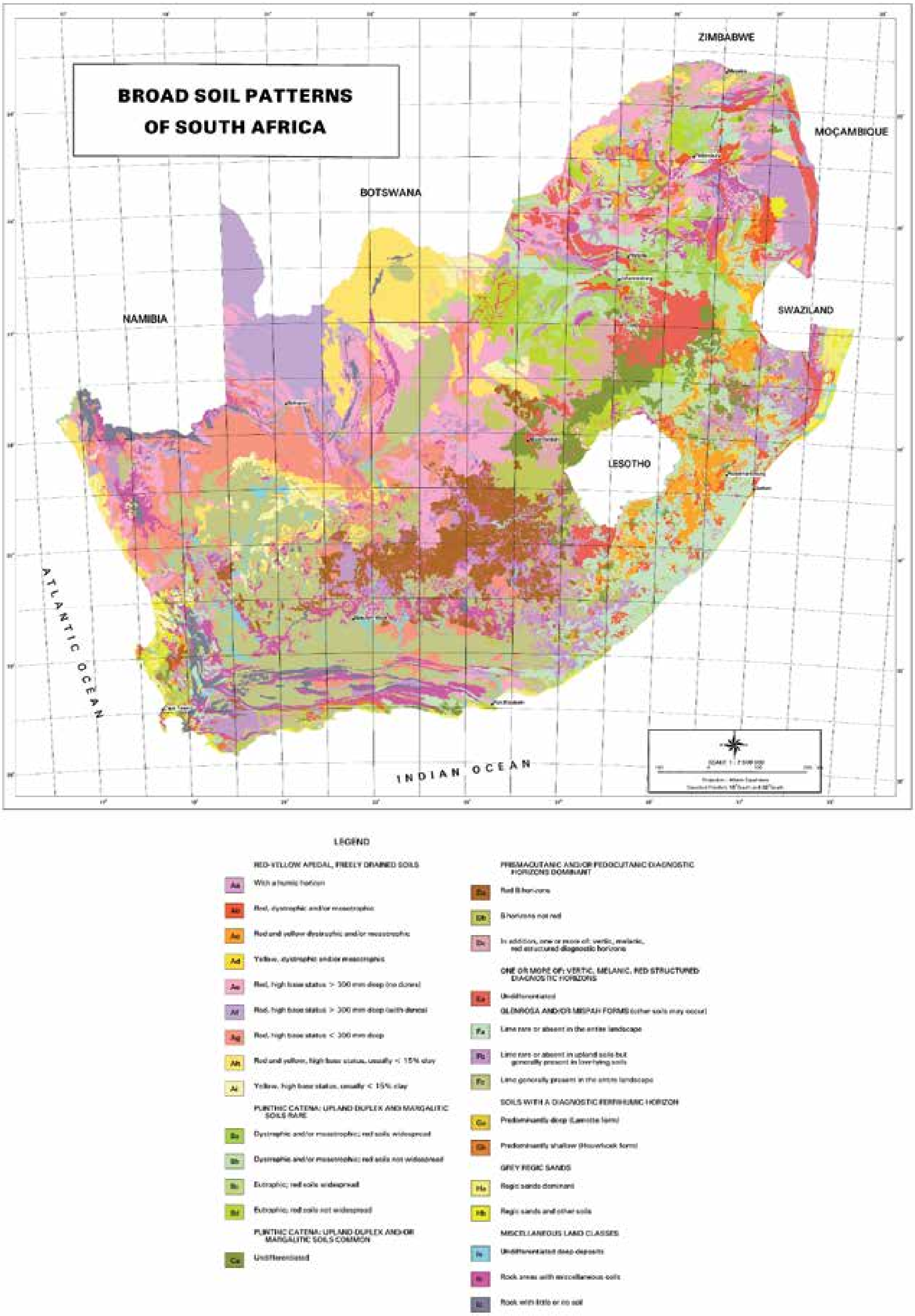


Figure 9.10 | Broad soil patterns of South Africa.
Source: Land Type Survey Staff, 2003.

From 2006, several further national studies were conducted to assess the status of soils, land use trends, land degradation and sustainable land management implementation in the country. Results are summarized in this section. There is much evidence of mismanagement of soil resources which has led to widespread erosion by both wind and water, loss of soil fertility, compaction and acidification.

National land degradation assessment

Since land use is considered the single most important driver of land (and soil) degradation, land degradation assessments were conducted based on land use categories. For this purpose, a national stratification map was developed for South Africa based on amendments to the Land Use System Approach as described by Nachtergaele and Petri (2008) as well as by Pretorius (2009). On this basis, the stratification map adopted the following 18 land use categories:

- Desert
- Azonal vegetation
- Savanna
- Forest
- Grassland
- Nama-Karoo
- Indian Ocean Coastal Belt
- Succulent Karoo
- Fynbos
- Albany Thicket
- Open Water
- Urban
- Cultivated – commercial – rain-fed
- Cultivated – irrigated
- Cultivated – subsistence – rain-fed
- Plantations
- Mines
- Protected areas

By integrating the local municipality boundaries with those of land use, a total of 2 447 unique units were derived for further assessment, as illustrated in Figure 9.11. Land degradation and sustainable land management implementation were then assessed in each of the 2 447 mapping units. The assessment was carried out from 2008 to 2010 and was based on a participatory approach as part of a Land Degradation Assessment in Drylands (LADA) project. The approach relied strongly on the inputs from a range of experienced contributing specialists and land users who were conversant with the areas to be assessed. Data capturing was done through a series of 33 Participatory Expert Assessment (PEA) Workshops throughout the country, involving 728 contributing specialists (Wiese, Lindeque and Villiers, 2011).

In terms of soil, the main forms of degradation at national level were soil erosion by water, biological degradation and chemical soil degradation. For soil erosion, sheet and gully erosion were considered the most serious threats, with river or stream bank erosion and off-site sedimentation considered less critical. Soil acidification and salinization were also highlighted, although their occurrence was more localized and area-specific.

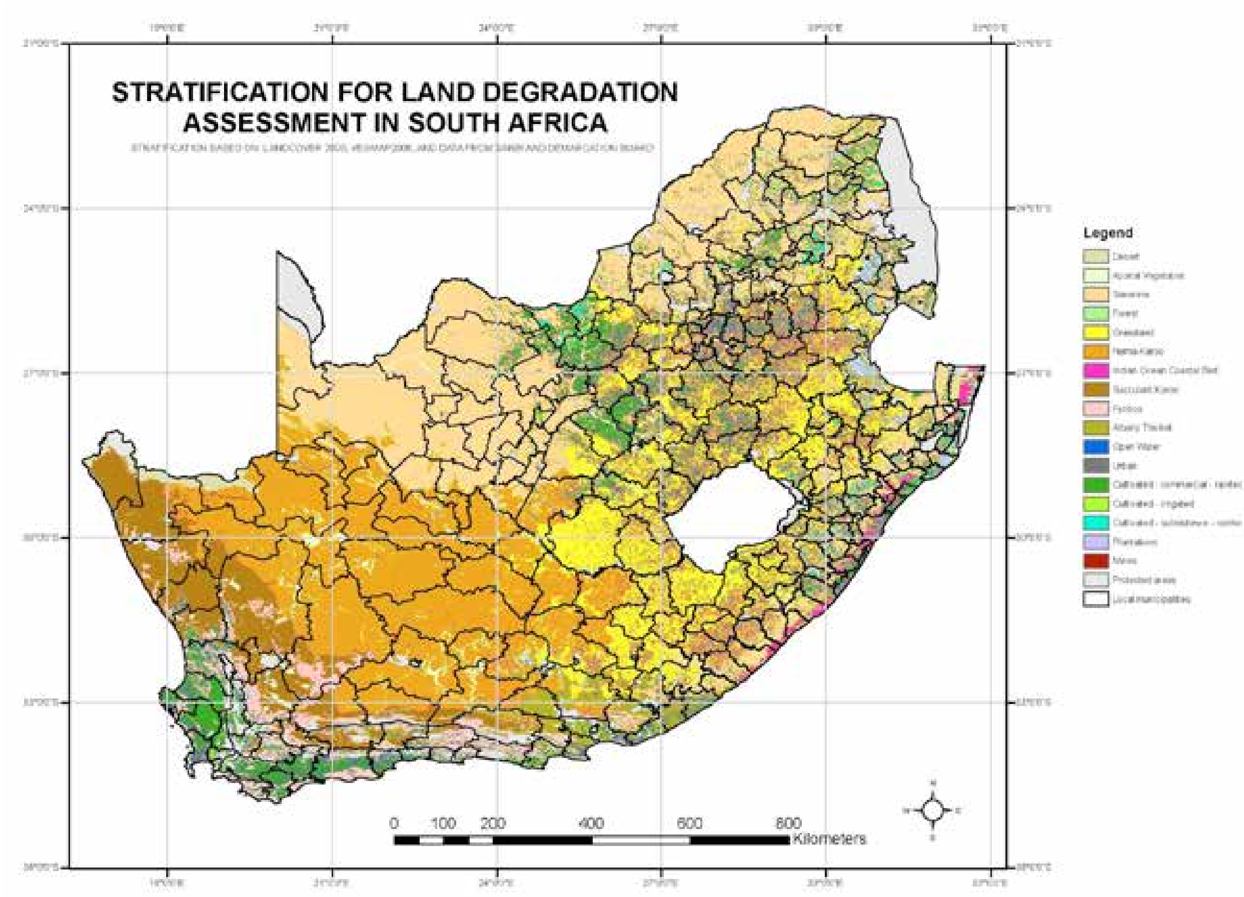


Figure 9.11 | The national stratification used for land degradation assessment in South Africa, incorporating local municipality boundaries with 18 land use classes.
Source: Pretorius, 2009.

Erosion assessment

A separate spatial study was conducted on the extent of gully erosion in South Africa (Le Roux *et al.*, 2008). The study also assessed national erosion potential in terms of soils, climate and topography (Le Roux, 2012). The assessment of water erosion susceptibility indicated that around 20 percent (26 million ha) of the country is classified as having a moderate to severe erosion risk (mainly based on sheet-rill erosion). The affected areas are concentrated in the south-eastern and north-eastern interior, mainly in the Eastern Cape, KwaZulu-Natal, Mpumalanga and Limpopo Provinces. All of these areas are characterized by a combination of high (often intense) rainfall, duplex soils derived from sodium-rich parent materials, and steep slopes (see Figure 9.12). These natural conditions are often exacerbated by poor land use practices, such as incorrect cultivation methods, overgrazing by livestock and high population density. Under such circumstances, potential soil loss can easily be in the 'Very High' class of more than 50 tonnes ha^{-1} yr^{-1} (Le Roux *et al.*, 2006).

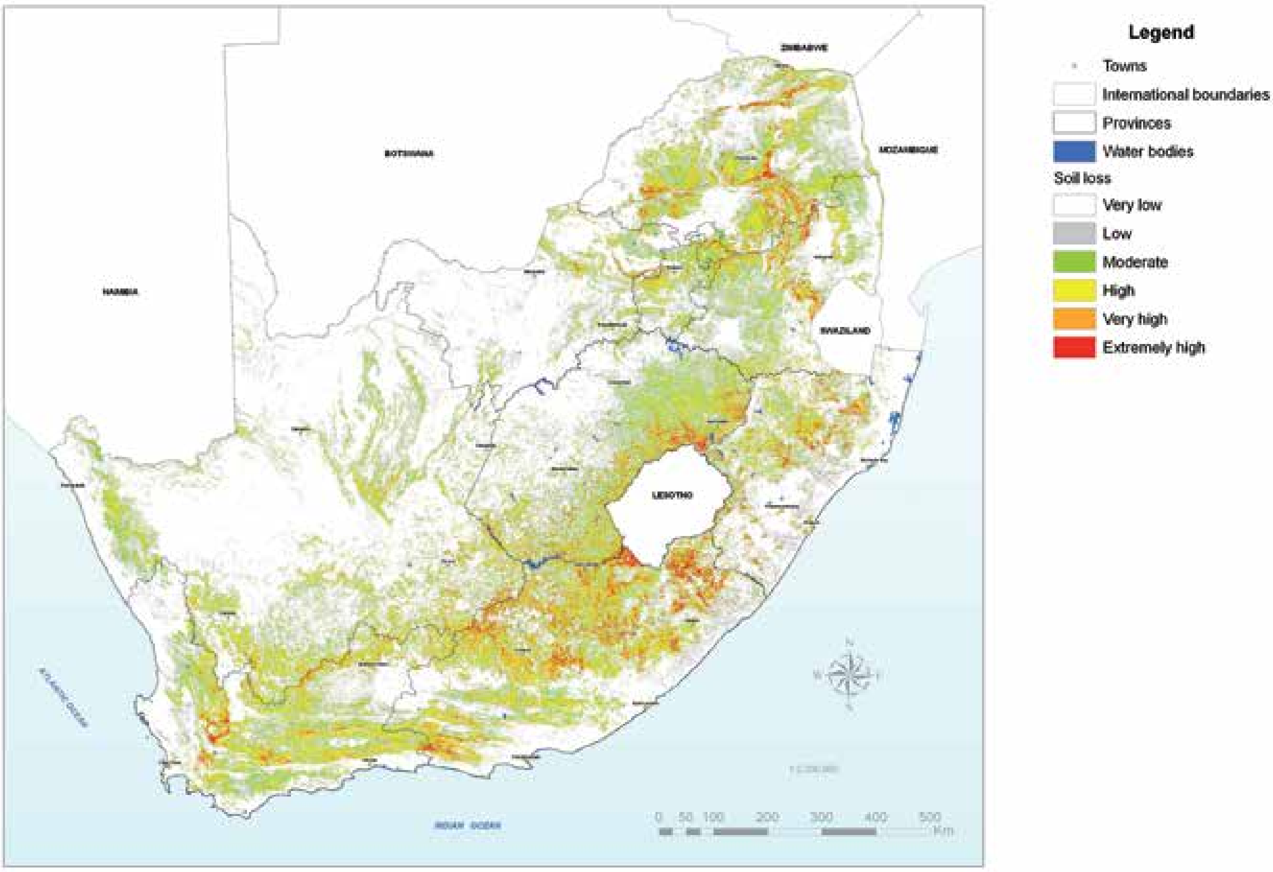


Figure 9.12 | Actual water erosion prediction map of South Africa.
Source: Le Roux et al., 2012.

The erosion process starts when the vegetation cover is disturbed or removed, allowing the rainfall to impact directly on bare soil. If measures to restrict surface run-off are not put in place, the effect is generally two-fold: firstly, the water flowing on the soil surface removes a significant amount of topsoil ('sediment'), especially on steeper slopes; and secondly, the duplex nature of the soils (sandy topsoil abruptly overlying a structured clay subsoil) results in the formation of a surface seal. As a result, very little water is actually able to infiltrate the soil. Research in South Africa (Levy, 1988; Rapp, 1998; Bloem, 1992) indicated that exchangeable sodium percentage (ESP) values play an important role in erosion risk, with problematic values being over 12, although values as low as 5 or 6 (Bloem and Laker, 1994; Laker and D'Huyvetter, 1988) can also cause erosion under poor land use conditions.

Combating soil erosion by water remains a huge challenge in many affected areas of the country due to a combination of lack of resources, poor knowledge or awareness and poor infrastructure, mainly roads. The challenges of treating erosion and the more difficult task of rehabilitating large areas of land, combined with the off-site effects such as silting up of dams, together pose one of the most serious soil management challenges in South Africa today.

Soil nutrient depletion, acidity and organic matter

Although soil erosion by water was confirmed as the main soil degradation type in the country, there are areas in South Africa affected by wind erosion, nutrient depletion, loss of organic matter, soil acidity, salinity and sodicity as well as pollution from mining and industrial sources. Desktop assessments of soil nutrient depletion, acidity and organic matter in South Africa were conducted during 2007-2008 (Beukes, Stronkhorst and Jezile, 2008a,b; Du Preez *et al.*, 2010; Du Preez *et al.*, 2011a,b; Rantoa, Du Preez and Van Huyssteen, 2009).

Nutrient depletion and acidity

A multitude of soil nutrient and acidity studies have been conducted over time in South Africa (Bierman, 2001; Bloem, 2002; Buhmann, Beukes and Turner, 2006; Conradie, 1994; Eweg, 2004; Farina, Manson and Johnston, 1993; Mandiringana *et al.*, 2005; Meyer *et al.*, 1998; Miles and Manson, 2000; Thibaud, 2005). These studies included extensive reviews of international and national documentation, interviews with experts from various national and provincial institutions, and processing of data available from a number of national databases and soil analytical laboratories. Detailed results have been reported for each of the nine provinces in South Africa, but only a national summary is presented here (Beukes, Stronkhorst and Jezile, 2008a,b) with a focus on the agricultural sector.

The impact of the dualistic nature of South African agriculture on soil nutrient depletion was clearly evident, with soils from the **resource-poor/small-scale/upcoming** farmers generally being acidified, severely P depleted, and N, K, Ca and Mg deficient (Manson, 1996; Beukes, Stronkhorst and Jezile, 2008b). Within this group, two sub-groups can be distinguished, as these farmers produce crops at two levels. The first sub-group is the home garden where relatively high fertility levels are evident. This is mainly because these gardens are located next to the homesteads and are therefore easier to manage. The second sub-group consists of crop fields which are larger and further from the homesteads. As a result, these fields are less secure in terms of livestock access and there are transport constraints. In addition, most smallholder farmers are risk-averse due to their limited resources. These fields are therefore generally severely nutrient depleted, especially in terms of P and K deficiencies, while N, Mg and Ca deficiencies are also often noted.

By contrast, **commercial agriculture** operates on a much larger scale and higher levels of management and inputs are maintained on these farms to ensure higher productivity. This is especially the case in the sugar, vine and fruit farming sectors due to higher costs for crop establishment and maintenance. Soils in the commercial sector generally exhibit P deficiency as the main nutrient concern, with K deficiency also occurring in many areas. Commercial pastures may have fewer deficiencies: for example in KwaZulu-Natal, P deficiency is almost negligible and K, Ca and Mg appear well supplied (Beukes, Stronkhorst and Jezile, 2008b).

Naturally occurring acid soils are generally associated with high rainfall areas and certain geological materials which, in South Africa, are located in the western and southern Cape coastal belts, KwaZulu-Natal, Mpumalanga and Limpopo Province (see Figure 9.13). The extent of anthropogenic soil acidity in the country is not easy to estimate, but general trends can be observed. In the winter rainfall region, approximately 560 000 ha is under cultivation, and on 60 percent of this area soils indicate problems with acidity. In Kwa-Zulu-Natal, roughly 35 percent of the more than 660 000 ha of cultivated soils have acid saturation values above 15 percent. In the summer rainfall area west of the Drakensberg, 37 percent of topsoils in the cropped area are acidified (Beukes, 1995).

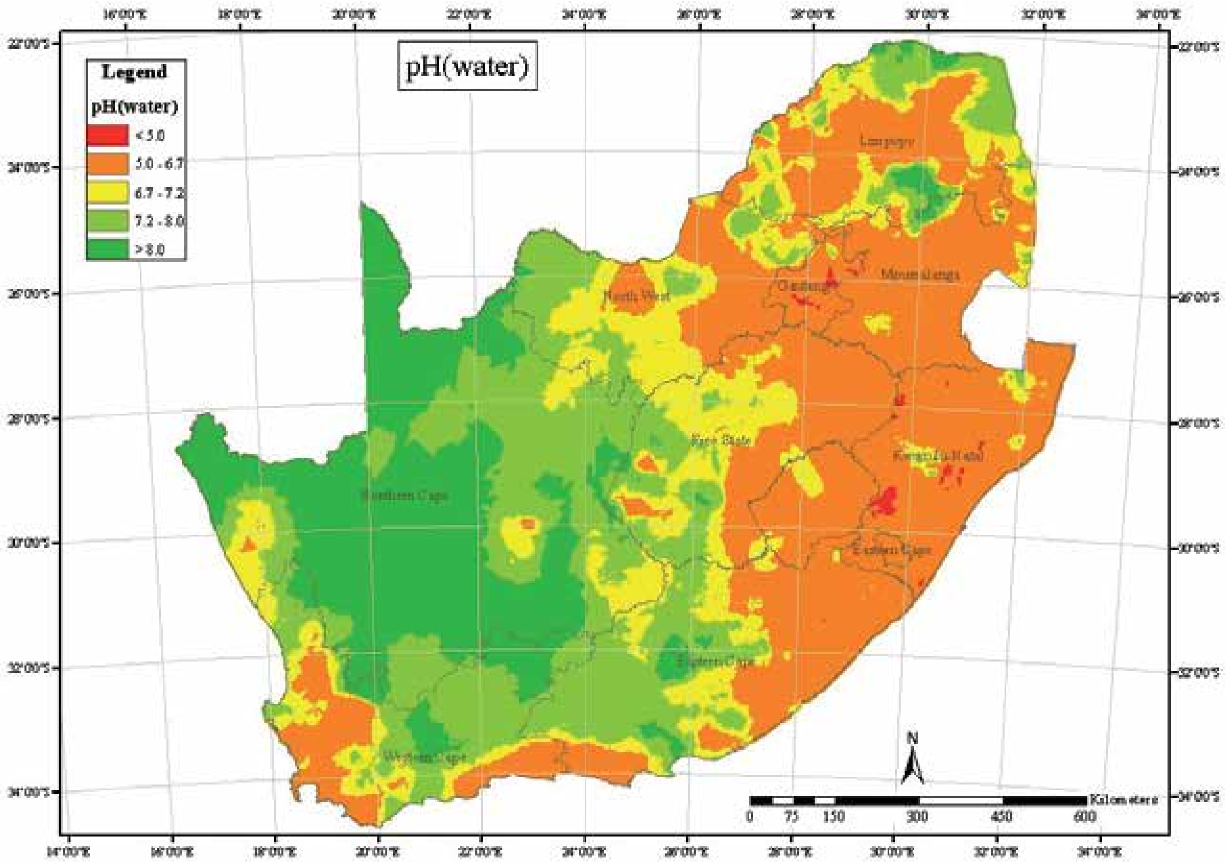


Figure 9.13 | Topsoil pH derived from undisturbed (natural) soils.
Source: Beukes, Stronkhorst and Jezile, 2008a.

Soil organic matter

Although data on soil organic carbon in South Africa are limited, fragmented and uncoordinated, general trends of SOC content can be derived from a range of studies conducted (Barnard *et al.*, 2000; McKean, 1993, Du Toit and Du Preez, 1993, Le Roux *et al.*, 2005; Mills and Fey, 2004; Prinsloo, Willshire and Du Proez, 1990; Van Antwerpen and Meyer, 1996; Birru, 2002; Du Preez, Mnkeni and Van Huyssteen, 2010, 2011a, b). A review of SOC research estimated that approximately 58 percent of South African soils contain < 0.5 percent organic C, 38 percent contain 0.5–2 percent organic C and 4 percent have > 2 percent organic C. These organic C contents vary greatly as a function of soil types, climate, vegetation, topography and soil texture, and are greatly influenced by management practices which result in organic C losses such as overgrazing, high levels of soil disturbance during cultivation, and the use of fire in rangeland management. Soil organic matter losses were generally associated with dryland cropping, but were less prevalent in irrigated agriculture. Increasing SOM is a slow process, but it has been achieved by implementing zero/minimum tillage, by mulching and through reversion of cropland to perennial pastures. Increases have mainly occurred in the upper 300 mm of soil, and in most instances, have been restricted to the upper 50 mm of soil.

Loss of SOM has been found to result in lower nitrogen and sulphur reserves, but not necessarily in lower phosphorus reserves. Loss of SOM also coincided with changes in the composition of amino sugars, amino acids and lignin. It further resulted in a decline of water stable aggregates which are essential in the prevention of soil erosion.

Rantoa, Du Preez and Van Huyssteen, (2009) used data from the approximately 2 200 modal profiles from the land type survey to estimate organic carbon stocks in South African soils with reference to master horizons, diagnostic horizons, soil forms and land cover classes. In summary, the average organic carbon content in the master horizons ranges from 16 percent in the O horizon to 0.3 percent in the C horizons. In the diagnostic horizons, the highest average organic C in topsoils ranged from 21 percent in the O horizons to 1.4 percent in the orthic A horizons. In the diagnostic subsoil horizons, however, values ranged from 1.2 percent in podzol B to 0.2 percent in the dorbank horizons.

Land Cover Change Assessment

Land use trends give an indication of land conversion from one land use to another, which directly affects soil use properties as a function of management. A study of land-cover change was conducted in 2010 based on land-cover data from 1994/1995, 2000 and 2005 employing a cost-effective approach which used Earth Observation data. The study was based on changes in five land-cover classes: urban, mining, forestry, cultivation, and other. These five classes are defined in Table 9.4 (Schoeman *et al.*, 2010).

Table 9.4 | Definitions of the five land-cover classes on which the land-cover change study was based. Source: Schoeman et al., 2010.

Land-cover class	Class definition
Urban	Human settlements, both rural and urban
Mining	Areas covered by mining and related mining activities (also includes mine dumps)
Forestry and plantations	All forestry and plantations including woodlots and clear fell areas (excludes indigenous natural forests)
Cultivation	All areas used for agricultural activities, including old fields and subsistence agriculture
Other	All other areas not covered by those listed above

The land-cover change results (Figure 9.14) indicated that at national level there was a total increase of 1.2 percent in transformed land, specifically associated with Urban, Cultivation, Forestry & Plantation and Mining. This represents an increase from 14.5 percent transformed land in 1994 to 15.7 percent in 2005 across South Africa. On a national basis the areas of Urban, Forestry & Plantation, and Mining have all increased over the 10-year period, whereas Cultivation areas have decreased. Urban has increased from 0.8 percent to 2 percent, Forestry & Plantation from 1.2 percent to 1.6 percent, and Mining from 0.1 percent to 0.2 percent, while Cultivation has decreased from 12.4 percent to 11.9 percent. The spatial patterns do, however, vary geographically across provinces in South Africa.

The increase in urban and mining areas are the biggest concern in terms of soil conservation and future use since urban development involves soil sealing which irreversibly removes soils from other land uses, while mining results in serious chemical and physical soil degradation which can only be restored to a limited extent. For this reason, it is essential that soil suitability and potential for agricultural and environmental purposes be assessed in order to ensure that the high potential and environmentally important soils are reserved and conserved for food production purposes.

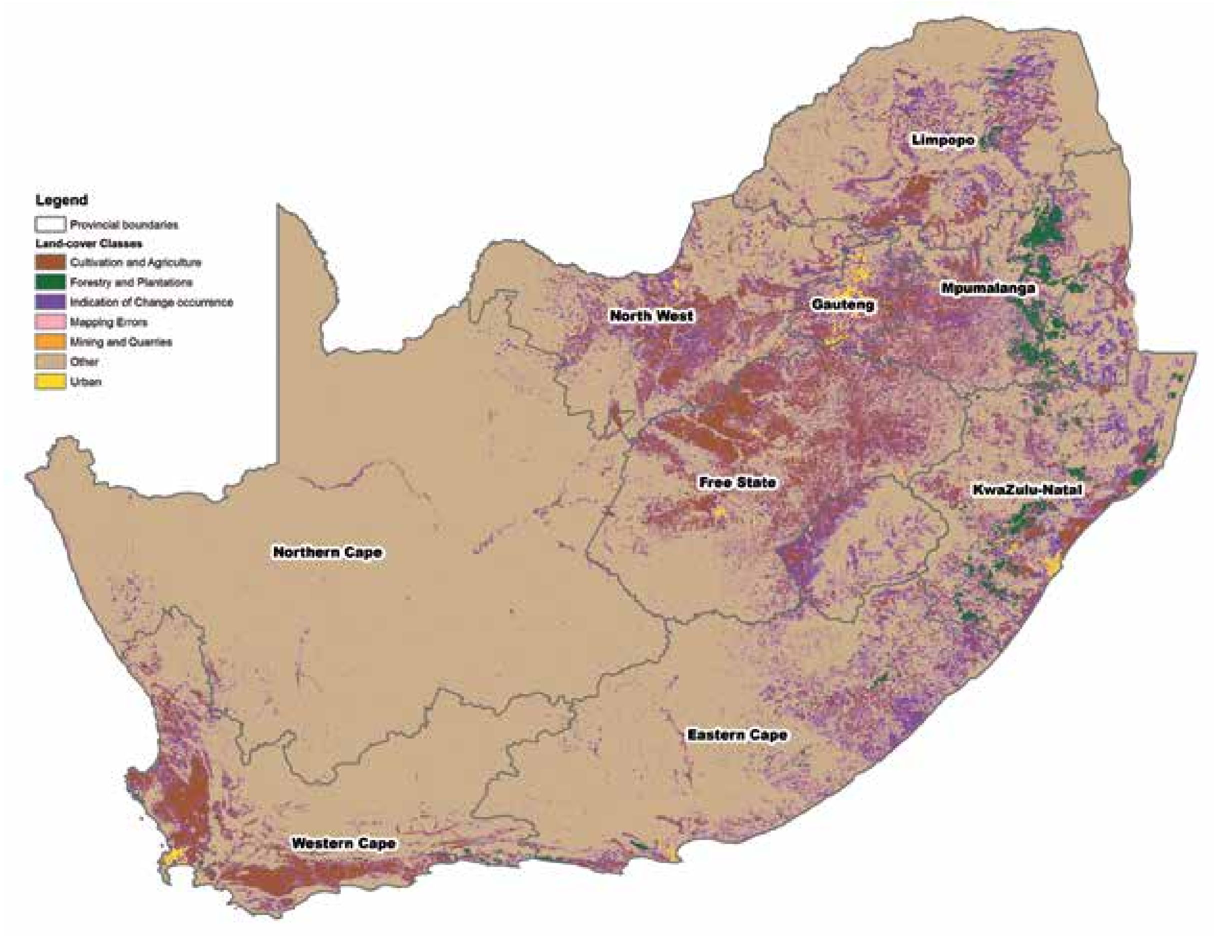


Figure 9.14 | Change in land-cover between 1994 and 2005 as part of the Five Class Land-cover of South Africa after logical corrections. Source: Schoeman et al., 2010.

9.6 | Summary of conclusions and recommendations

Based on the above finding, an assessment is made of the status and trend of the ten soil threats in order of importance for the region. At the same time an indication is given of the reliability of these estimates (Table 9.5).

Soil degradation is considered one of the root causes of stagnating or declining agricultural productivity in SSA. Unless soil degradation can be controlled, many parts of the continent are expected to suffer increasingly from food insecurity. If this decline in the productivity of Africa's soil resources continues, the consequences will be severe, not only for the economies of individual countries, but also for the welfare of the millions of rural households dependent on agriculture for meeting their livelihood needs.

There is an urgent need for proactive interventions to arrest and reverse soil degradation. Rehabilitation of degraded land and conservation of those not yet degraded is the most desirable step for every country in the region, but this can only be achieved if the characteristics of the soil resources are well defined and quantified and soil monitoring systems established in every country.

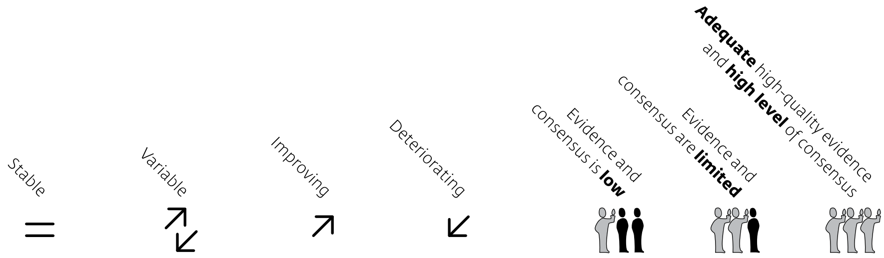


Table 9.5 | Summary of soil threats status, trends and uncertainties in Africa South of the Sahara.

Threat to soil function	Summary	Condition and Trend					Confidence	
		Very poor	Poor	Fair	Good	Very good	In condition	In trend
Soil erosion	Soil erosion constitutes >80% of land degradation in SSA, affecting about 22% of agricultural land and all countries in the region. The majority of causes related to the exposure of the bare soil surface by cultivation, deforestation overgrazing and drought.		↙					
Organic carbon change	The replacement of the natural vegetation reduces nearly always the soil carbon level. Further carbon release from the soil is caused by complete crop removal from farmlands, the high rate of organic mater decomposition by microbial decomposition accentuated by high soil temperature and termite activates in parts of SSA.		↙					
Nutrient imbalance	Nutrient imbalance, which is generally manifested by the deficiency of key essential nutrients is mainly due to the fact that fertilization has not been soil and crop specific, farmers are unable to pay the price for fertilizers and the inability to follow the rates that are recommended. Nearly all countries in the region show a negative nutrient balance.		↙					

Loss of soil biodiversity	SSA suffers the world's highest annual deforestation rate. The areas most affected are the in the moist areas of West Africa and the highland forests of the Horn of Africa. Cultivation, introduction of new species, oil exploration and pollution reduce the population of soil organisms thus reducing faunal and microbial activities.			↙				
Soil acidification	Over 25% of soils in Africa are acidic. Most of these occur in the wetter parts of the continent. In South Africa it poses as a serious chemical problem and the greatest production-limiting factor.		↙					
Waterlogging	Most waterlogging threats are due to rise in water table due to poor infiltration/ drainage or occurrence of impervious layer in the subsoil. Waterlogging generally reduces crop productivity, but in paddy fields is deliberate and beneficial.				=			
Compaction	The major cause of compaction is pressure on the soil from heavy machinery. It is more serious in forested regions where land clearing (and even other cultivation activities) cannot be done without mechanization.				=			
Soil sealing and land take	These constitute problems mainly in peri-urban agriculture and valley sites used for dry season vegetable production.				=			
Soil pollution	Soil contamination by chemicals (fertilizers, petroleum products, pesticides, herbicides, mining) has affected agricultural productivity and other ecosystem services negatively. Nigeria and South Africa are the most affected.				↙			

References

Abdalla, E., Meheissi, B. & Basher, N.E. 2011. *Indicators of impact of climate change on salinity distribution and land-use in Sudan Agricultural Research Corporation, Wad Medani.* Sudan Proceedings of the Global Forum on Salinization and Climate Change (GFSCC 2010), 25–29 October 2010. Valencia.

Adeleke, B.O. & Leong, G.L. 1980. *Certificate Physical and Human Geography,* Oxford University Press Ltd, pp. 42-43.

Adeniran, J.R. 1993. T*he Study of Flood Hazards in Nigeria*. An Unpublished Seminar Paper. University of Ilorin, Department of Geography.

Amundson, R. 2001.The carbon budget in soils. *Annu. Rev. Earth Pl. Sc.*, 29: 535–56.

Ayuke, F.O., Brussaard, L., Vanlauwe, B., Six, J., Lele[i], D.K., Kibunja, C.N. & Pulleman, M.M. 2011. Soil fertility management: impacts on soil macrofauna, soil aggregation and soil organic matter allocation. **Applied Soil Ecology**, 48: 53–62.

Barnard, R.O. 2000. *Carbon sequestration in South African soils*. ARC-ISCW Report No GW/A/2000/48. Pretoria, ARC-Institute for Soil, Climate and Water.

Barnard, R.O., Van der Merwe, A.J., Nell, J.P., De Villiers, M.C., Van der Merwe, G.M.E. & Mulibana, N.E. 2002. *Technical Country report/in-depth study on problem soils including degraded soils in South Africa: Extent, present use, management and rehabilitation (with emphasis on salt-affected soils).* 4th Meeting of FAO Global Network Integrated Soil Management for Sustainable Use of Salt-Affected Soils. Spain, Valencia.

Bationo, A., Buerkert, A., Sedogo, M.P., Christianson, B.C. & Mokwunye, A.U. 1995. A critical review of crop residue use as soil amendment in the West African Semi-arid tropics. In J.M. Powell, S. Fernandez-Rivera, T.O. Williams, C. Renard, eds. *Livestock and sustainable nutrient cycling in mixed farming systems of Sub-Saharan Africa*. Vol. 2. Pp. 305–322. Addis Ababa, International Livestock Centre for Africa (ILCA).

Bationo, A., Kihara, J., Vanlauwe, B., Waswa, B. & Kimetu, J. 2007. Soil organic carbon dynamics, functions and management in West African agroecosystems. *Agricultural Systems*, 94: 13-25.

Bationo, A., Mokwunye, U., Vlek, P.L.G, Koala, S. & Shapiro, B.I. 2003. Soil fertility management for sustainable land use in the West African Sudano-Sahelian zone. In M.P. Gichuru, eds., *Soil Fertility Management in Africa: a regional perspective*. Nairobi, CIAT, Academy of Science Publishers (ASP), Tropical Soil Biology & Fertility.

Bationo, A., Waswa, B., Kihara, J., Adolwa, I., Vanlauwe, B. & Saidou, K. (eds.). 2012. *Lessons learned from long-term soil fertility management experiments in Africa*. Springer Science & Business Media.

Beukes, D.J. 1995. *Benefits from Identifying and Correcting Soil Acidity in Agriculture.* Pretoria, ARC-ISCW. 12pp. ISBN: 1-86849-021-1..

Beukes, D.J., Stronkhorst, L.D. & Jezile, G.G. 2008a. *Development of a soil protection strategy and policy for South Africa: An overview of the soil acidity problem in South Africa.* ARC-ISCW Report Nr. GW/A/2008/08.

Beukes, D.J., Stronkhorst, L.D. & Jezile, G.G. 2008b. *Development of a soil protection strategy and policy for South Africa: An overview of the soil nutrient depletion in South Africa*. ARC-ISCW Report Nr. GW/A/2007/162.

Bielders, C.L., Michels, K. & Rajot, J-L. 1985. On-Farm Evaluation of Ridging and Residue Management Practices to Reduce Wind Erosion in Niger. *Soil Sci. Soc. Am. J.*, 54: 1157–1161.

Bierman, C.R. 2001. *Grondvrugbaarheid en bemesting in die Noordwes Provinsie met verwysing na volhoubare droelandgewasverbouing*. Bloemfontein, University of the Free State. 36 pp. (MSc Thesis)

Birru, T.C. 2002. *Organic matter restoration by conversion of cultivated land to perennial pasture on three agro-ecosystems in the Free State*. Bloemfontein, University of the Free State. (M.Sc. dissertation)

Bloem, A.A. & Laker, M.C. 1994. Criteria for the adaptation and design and management of centre-pivot irrigation systems to the infiltrability of soils. *Water SA*. 20: 127-132.

Bloem, A.A. 1992. *Criteria for the adaptation and design and management of centre-pivot irrigation systems to the infiltration potential of soils (in Afrikaans)*. University of Pretoria. (M.Sc. dissertation)

Bloem, A.A. 2002. *Databasis- en navorsingsresultate dui stikstoflewering van gronde aan*. SA Graan/Grain, Julie 2002. 44 pp.

Bohlen, P.J., Edwards, W.M. & Edwards, C.A. 1995. Earthworm community structure and diversity in experimental agricultural watersheds in Northeastern Ohio. In H.P. Collins, G.P. Robertson & M.J. Klug, eds. *The significance and regulation of soil biodiversity*. pp. 271-278. Kluwer Academic, Netherlands.

Buhmann, C., Beukes, D.J. & Turner, D.P. 2006. Plant nutrient status of soils of the Lusikisiki area, Eastern Cape Province. S Afr J Plant Soil, 23(2): 93-98.

Byerlee, D.A., Anandajayasekeram, P., Diallo, A., Gelaw, G., Heisey, P.W., Lopez–Pereira, M., Mwangi, W., Smale, M., Tripp, R. & Waddington, S. 1994.Maize Research in Sub–Saharan Africa: An Overview of Past Impacts and Future Prospects. CIMMYT Economics working Paper 9403. Mexico, D.F. International Maize and Wheat Improvement Center (CIMMYT).

Charman, P.E.V. & Murphy, B.W. 2005. *Soils - Their Properties and Management*. *Melbourne*, Vic, Oxford University Press in association with NSW Department of Land and Water Conservation,..

CIA World Factbook. 2012. Also available at https://www.cia.gov/library/publications/the-world-factbook/

Conradie, W.J. 1994. *Vineyard Fertilization*. Proceedings of a work session on vineyard fertilization, held at Nietvoorbij on 30 September 1994. Stellenbosch, Nietvoorbij, ARC-Fruit, Vine and Wine Research Institute. , 41 pp.

Crossley, D.A., , Mueller, B.R. & Perdue, J.C. 1992. Biodiversity of Microarthropods in agricultural soils: relations to processes. *Agric. Ecosyst. Environ*., 40: 37-46.

Curry, J.P. 2004. Factors affecting the abundance of earthworms in soils. In C.A. E*dwards, ed. Earthworm Ecology.* CRC Press, Boca Raton.

De Villiers, M.C., Pretorius, D.J., Barnard, R.O., Van Zyl, A.J. & Le Clus, C.F. 2002. L*and degradation assessment in dryland areas: South Africa*. FAO Land Degradation Assessment in Dryland Project. Rome, FAO.

Detwiler, R.P. 1986. Land use change and global carbon cycle: The role of tropical soils. *Biogeochemistry*, 2: 67-93.

Dregne, H. & Chou, N. 1992. Global desertification: dimensions and costs. In H. Dregne, ed. *Degradation and restoration of arid lands*. Lubbock, Texas Tech. University.

Dregne, H. E. 1990. Erosion and soil productivity in Africa. *Journal of Soil &Water Conservation*, 45(4): 431–436.

Du Preez, C.C., Mnkeni, P.N.S. & Van Huyssteen, C.W. 2010. *Knowledge review on land use and soil organic matter in South Africa.* 19th World Congress of Soil Science, Soil Solutions for a Changing World, 1 – 6 August 2010. Australia, Brisbane.

Du Preez, C.C., Mnkeni, P.N.S. & Van Huyssteen, C.W. 2011a. Knowledge review on land use and soil organic matter in South Africa 1. Spatial variability and rangeland stock production. *S. Afr. J. Sci*., 207: 27-34.

Du Preez, C.C., Mnkeni, P.N.S. & Van Huyssteen, C.W. 2011b. Knowledge review on land use and soil organic matter in South Africa 2: Arable crop production. *S. Afr. J. Sci*., 207: 35-42.

Du Toit, M.E. & Du Preez, C.C. 1993. Relationship between organic matter content of certain virgin orthic topsoils, soil properties and climatic data in South Africa. *South African Journal of Plant and Soil*, 10: 168-173.

Eswaran, H., Almaraz, R., van den Berg, E. & Reich, P. 1996. *An Assessment of the Soil Resources of Africa in Relation to Productivity*. Washington, DC, World Soil Resources, Soil Survey Division, USDA Natural Resources Conservation Service.

Eswaran, H., Reich, P. & Beinroth, F. 1997. Global distribution of soils with acidity. In A.C. Moniz, ed. *Plant-Soil Interactions at Low pH*, pp. 159-164. Brazilian Soil Science Society.

Eweg, M.J. 2004. Changing fertiliser practices in the small-scale sector of the South African sugar industry: The role of Extension. *Proc S Afr Sug Technol Ass*, 78: 393-402.

Fakayode, S.O. & Onianwa, P.C. 2002. Heavy metal contamination of soil, and bioaccumulation in Guinea grass (Panicum maximum) around Ikeja Industrial In R. Lal & B. Stewart, eds. *Advances in Soil Science*, pp. 499-510. New York, Springer.

FAO & World Bank. 2001. *Farming Systems and Poverty. Improving Farmers' livelihoods in a Changing World*. Rome & Washington.

FAO. 1993. *The conservation and rehabilitation of African lands*. Rome, FAO. 38 pp.

FAO. 1999. *Integrated Soil Management for Sustainable Agriculture and Food Security in Southern and East Africa*. Agritex.

FAO. 2001. *Soil Fertility Management in Support of Food Security in Sub-Saharan Africa*. Rome, FAO.

FAO. 2005. Global Forest Resource Assessment 2005. FAO Forestry Paper 146. Rome, FAO.

FAO. 2007. *State of the World's Forests 2007*. Rome, FAO.

FAO. 2010. *Land Degradation Assessment for Dry Lands (LADA)*. Rome, FAO.

FAO. 2015. *FAOSTAT*. Rome, FAO. (Also available at http://faostat 3.fao.org/home/E)

Farina, M.P.W., Manson, A.D. & Johnston, M.A. 1993. Maize in Natal. Fertilizer Guidelines. *Natal Maize*, 6: 1-12.

Favis-Mortlock, D. 2005. *Erosion little and large*. Soil Erosion Site. (Also available at www.soilerosion.net/doc/erosion_little_large.html)

Fischer, G., Shah, M., van Velthuizen H. & Nachtergaele, F.O. 2002. *Global Agro-ecological Assessment for Agriculture in the 21st century: Methodology and Results*. FAO/IIASA. Laxenburg, IIASA & Rome, FAO.

Folscher, W.J. 1984. *Stikstofdinamika in grond. In: Proceedings of the nitrogen symposium*. Pretoria, Soil and Irrigation Research Institute, Private BagX 79.

GEF. 2006. *Evaluation Office Monitoring and Evaluation Policy Evaluation Document* No. 1. Global Environment Facility. 42pp.

George, H. & Petri, M. 2006. The rapid characterization and mapping of agricultural land use: A methodological framework approach for the LADA project. LADA Technical Report #1. Rome, FAO.

Global HarvestChoice, 2010. *Agro-ecological Zones of Sub-Saharan Africa*. Washington, DC, International Food Policy Research Institute & St. Paul, MN, University of Minnesota.

Global HarvestChoice, 2011. *Tsunami-wrecked Farmland: Insights from Droppr. Washington*, DC, International Food Policy Research Institute & St. Paul, MN, University of Minnesota.

Gray, K. M. 2005. Changing soil degradation trends in Senegal with carbon sequestration payments. Montana State University. 135 pp. (A M. Sc. Thesis)

Guo, L.B. & Gifford, R.M. 2002. Soil carbon stocks and land use change: a meta analysis. *Global Change Biology*. 8(4): 345 - 360.

Hansen, M.C., Potapov, P.V., Moore, R., Hancher, M., Turubanova, S.A., Tyukavina, A. & Townshend, J.R.G. 2013. High-resolution global maps of 21st-century forest cover change. *Science*, 342(6160): 850-853.

Hein, L. & De Ridder, N. 2006. Desertification in the Sahel: A reinterpretation. *Global Change Biology*, 12: 751–758.

Henao, J. & Baanante, C. 1999. *Nutrient depletion in the agricultural soils of Africa*. 2020 Brief 62. A 2020 Vision For Food, Agriculture and Environment. IFPRI.

Hooper, D.U., Adair, E.C., Cardinale, B.J., Byrnes, J.E.K., Hungate, B.A., Matulich, K.L., Gonzalez, A., Houghton R.A. & Hackler J.L. 2006. Emissions of carbon from land use change in Sub-Saharan Africa. *Journal of Geophysical Research-Biogeosciences*, 111(G 2): G 02003. doi:10.1029/2005JG 000076,2006

Houghton, J.T., Ding, Y., Griggs, D.J., Noguer, M., van der Linden, P.J. & Xiaosu D. (eds). 2001. *Climate Change 2001: The Scientific Basis*. Contributions of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press. 881 pp.

IAASTD. 2009. *International Assessment of Agricultural Knowledge, Science and Technology for Development. Agriculture at a Crossroads*. Synthesis Report. 106 pp.

IFAD. 2009. *The Strategic Investment Program for Sustainable Land Management in Sub-Saharan Africa*. IFAD.

IFPRI. 1999. *Soil Degradation A Threat to Developing-Country Food Security by 2020?* by S.J. Scherr. Food, Agriculture, and the Environment Discussion Paper 27. Washington, DC, International Food Policy Research Institute.

ILCA. 1987. *Assessment of Agricultural Research Priorities: An International Perspective*. ACIAR Monograph, 4th Edition, ILCA's Strategy and Long-Term Plan. Ethiopia, Addis Ababa..

IPCC. 2007. *Fourth Assessment report. Impacts, Adaptation and Vulnerability.* Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press.

ISRIC/UNEP. 1990 *World map of the status of human-induced soil degradation*. The Netherlands, Wageningen, ISRIC.

Jahnke, H.E. 1982. *Livestock production systems and livestock development in tropical Africa*. Germany, Kiel, Kieler Wissenschaftsverlag Vauk.

Jimoh, H.I. 2000. *Individual Rainfall Events and Sediment Generation on Different Surfaces in Ilorin, Nigeria* Ilorin, University of Ilorin, Department of Geography. ,. 220 pp. (Ph. D. Thesis)

Jones, A., Breuning-Madsen, H., Brossard, M., Dampha, A., Deckers, J., Dewitte, O. & Zougmore., R. 2013. *Soil Atlas of Africa*.

Jouve, P. 1992. Le diagnostic en milieu rural, de la région à la parcelle, n°6. Approche systémique des modes d'exploitation agricole du milieu. CNEARC, Juin. 40 pp.

Kabambe, V.H. & Kumwenda, J.D.T. 1995. Weed management and nitrogen rate effects on maize yield in Malawi. In D. Jewell, S.R. Waddington, J.K. Ransom & K.V. Pixley, eds. *Maize Research for Stress Environments Harare. Zimbabwe*, CIMMYT.

Knoepp, J.D. & Swank, W.T. 1997. Forest management effects on surface soil carbon and nitrogen. *Soil Sci. Soc. Am. J.*, 61: 928–935.

Laker, M.C. & D'Huyvetter, J.H.H. 1988. Slope-erodibility relationships for different soils in a semi-arid region of Ciske[i]. *In Proceedings of the Fifth International Soil Conservation Conference*, pp. 641-653. January 18-29, 1988. Bangkok.

Laker, M.C. 2005. Appropriate plant nutrient management for sustainable agriculture in Southern Africa. *Communications in Soil Science & Plant Analysis*, 36(1-3): 89-106.

Lal, R. 1990. Soil erosion and land degradation: The global risks. In R. Lal & B.A. Stewart, ed. *Soil degradation*. Vol. 11. *Advances in Soil Science*, pp. 129-172. Springer

Lal, R. 1995. Erosion-crop productivity relationships for soils of Africa. *Soil Science Society of America Journal*, 59(3): 661-667.

Lal, R. 2004. Soil carbon sequestration to mitigate climate change. *Geoderma*, 123: 1–22.

Lal, R. 2006. Enhancing crop yields in the developing countries through restoration of the soil organic carbon pool in agricultural lands. *Land Degradation and Development*, 17: 197–209.

Land Type Survey Staff. 2003. *Land types of South Africa*. Pretoria, ARC-Institute for Soil, Climate and Water.

Le Roux, J.J. 2012. Water erosion risk assessment in South Africa: towards a methodological framework. PhD Thesis. University of Pretoria, Pretoria

Le Roux, J.J., Morgenthal, T.L., Malherbe, J., Pretorius, D.J. & Sumner, P.D. 2008. Water erosion prediction at national scale for South Africa. *Water* SA, 34: 305-314.

Le Roux, J.J., Morgenthal, T.L., Malherbe, J., Smith, H.J., Weepener, H.L. & Newby, T.S. 2006. Improving spatial soil erosion indicators in South Africa. Report No. GW/A/2006/51. Pretoria, ARC-Institute for Soil, Climate and Water.

Le Roux, P.A.L., Hensley, M., Du Preez, C.C., Kotze, E., Van Huyssteen, C.W., Collins, N.B. & Zere, T.B. 2005. The Weatherley catchment: Soil organic matter and vegetation baseline study. WRC Report No. KV 170/50. Pretoria, Water Research Commission.

Lesschen, J.P., Asiamah, R.D., Gicheru, P., Kanté, S., Stoorvogel, J.J. & Smaling, E.M.A. 2003. *Scaling soil nutrient balances*. Rome, FAO.

Levy, G.J. 1988. *The effects of clay mineralogy and exchangeable cations on some of the hydraulic properties of soil*. University of Pretoria. (Unpublished D.Sc. thesis)

Liniger, H. P., Studer, R. M., Hauert, C. & Gurtner, M. 2011. Sustainable Land Management in Practice– Guidelines and Best Practices for Sub-Saharan Africa. TerrAfrica, World Overview of Conservation Approaches and Technologies (WOCAT) and Food and Agriculture Organization of the United Nations (FAO).

Liu C., Lu M., Cui J., Li B. & Fang C. 2014. Effects of straw carbon input on carbon dynamics in agricultural soils: a meta-analysis. Glob Chang Biol., 20(5): 1366-81.

Lyles, L. 1975. Possible effects of wind erosion on soil productivity. *Journal of Soil Conservation*, 30: 279-283.

Mabogunje, A.L. 1995. The Environmental Challenges in Sub Saharan Africa. Environment. *African Technology Forum*, 8(1): 4-11.

MacVicar, C.N., De Villiers, J.M., Loxton, R.F., Verster, E., Lambrechts, J.J.N., Merryweather, F.R., Le Roux, J., Van Rooyen, T.H. & von Harmse, H.J. M. 1977. *Soil classification - A binomial system for South Africa*. Scientific Bulletin No 390. Pretoria, Department of Agricultural Technical Services.

Mandiringana, O.T., Mkile, Z., Van Averbeke, W., Van Ranst, E. & Verplanke, H. 2005. Mineralogy and Fertility Status of Selected Soils of the Eastern Cape Province, South Africa. *Comm Soil Sci and Plant Anal*, 36: 2431-2446.

Manson, A.D. 1996. *The fertility status of land used for small-scale cropping in KwaZulu-Natal*. Cedara Report No. N/A/96/9. South Africa, Cedara, KwaZulu-Natal Department of Agriculture. 9 pp.

McCann, J.C. 2005. *Maize and grace: Africa's encounter with a new world crop 1500-2000*. Cambridge, Harvard Univ. Press.

McKean, S.G. 1993. *Soil organic matter patterns, processes and effects in two contrasting* savannas.University of Witwatersrand: Johannesburg. (M.Sc. dissertation)

Meadows, M. E. & Hoffman, M. T. 2002. The nature, extent and causes of land degradation in South Africa: legacy of the past, lessons for the future? *Royal Geographical Society*, 34(4): 39-44.

Meyer, J.H., Harding, R., Rampersad, A.L. & Wood, R.A. 1998. Monitoring long term soil fertility trends in the South African sugar industry using the FAS Analytical Database. *Proc S Afr Sug Technol Ass*, 72: 61-68.

Miles, N. & Manson, A.D. 2000. Nutrition of planted pastures. In N Tainton, ed. *Pasture Management in South Africa*, pp. 180-232. South Africa, Pietermaritzburg, University of Natal Press.

Mills, A.J. & Fey, M.V. 2004. Soil carbon and nitrogen in five contrasting biomes of South Africa exposed to different land uses. *South African Journal of Plant and Soil*. 21: 94-102.

Mulugeta, L. 2004. *Effects of Land Use Changes on Soil Quality and Native Flora Degradation and Restoration in the Highlands of Ethiopia: Implications for sustainable land management.* (Doctoral Dissertation) ISSN 14016230, ISBN 91576 6540-0

Murty, D., Krischbaum, M. F., McMurtrie, R. E. & McGilvray, H. 2002. Does conversion of forest to agricultural land change soil carbon and nitrogen? A review of the literature. *Glob. Change Biol*., 8: 105–123.

Nachtergaele, F. & Petri M. 2008. *Mapping Land Use Systems at global and regional scales for Land Degradation Assessment Analysis*. FAO.

Nandwa, S. M. 2003. Perspectives on soil fertility in Africa. In M.P. Gichuru, eds. *Soil fertility management in Africa: a regional perspective.* Nairobi, Academy of Science Publishers (ASP), Tropical Soil Biology & Fertility of CIAT.

Ndiaye, D.S. & Dieng, N. 2013. *La gestion durable des terres au Sénégal :outils et politiques.* Dakar, CSE.

NEST. 1991. *Nigeria's Threatened Environment: a National Profile*. Nigeria, NEST.

Odai, S.N., Mensah, E., Sipitey, D., Ryo, S. & Awuah, E. 2008. Heavy metals uptake by vegetables cultivated on urban waste dumpsites: Case study of Kumasi, Ghana. Res. J. Environ. *Toxicol.*, 2: 92-99.

Ofomata, G.E.K. 1985. *Soil erosion in Nigeria: the views of a Geomorphologist.* University of Nigeria Inaugural Lecture Series No.7. Nigeria.

Okigbo, D. 1977. Farming Systems and Soil Erosion in West Africa. In J. Greenland & R. Lal,eds. Soil and Water Conservation and Management in the Humid Tropics. U.K, John Wiley & Sons.

Okwakol, M.J.N. 2000. Changes in termite (Isoptera) communities due to the clearance and cultivation of tropical forest in Uganda. Afr. J. *Ecology*, 38: 1-7.

Oldeman, L. R. 1991. G*lobal extent of soil degradation*. Biannual report. The Netherlands, Wageningen, International Soil Reference and Information Centre.

Oldeman, L.R., Hakkeling, R.T.A. & Sombroek, W.G. 1991. *World Map of the status of human- induced soil degradation: An explanatory note*. Wageningen, The Netherlands, IFPRI & Kenya, Nairobi, United Nations Environment Programme.

Oldeman, R. 2002. *Assessment of Methodologies for Drylands Land Degradation Assessment.* First Meeting of Technical Advisory Group and Steering Committee of the Land Degradation Assessment in Drylands. FAO, Rome, January 22 25. The Netherlands, Wageningen.

Otte, M.J. & Chilonda, P., 2002. Cattle and small ruminant production in sub-Saharan Africa: A systematic review. Rome, FAO.

Palm, C.A., Gachengo, C.N., Delve, R.J., Cadisch, G. & Giller, K.E. 2001. Organic inputs for soil fertility management in tropical agroecosystems: application of an organic resource database. Agriculture. *Ecosystem and Environment*, 83: 27-42.

Paoletti, M.G., Foissner, W. & Coleman, D.C. 1993. *Soil Biota, Nutrient Cycling and farming Systems*. USA, Boca Raton, Fl, Lewis.

Pastor, J. & Post, W.M. 1986. Influence of climate, soil moisture and succession on forest carbon and nitrogen cycles. *Biogeochemistry*, 2: 3-27.

Perrings, C. & Lovett, J. 1999. *Policies for Biodiversity Conservation: The Case of Sub-Saharan Africa. International Affairs*. Wiley. (Also available at http://onlinelibrary.wiley.com/doi/10.1111/inta.1999.75.issue-2/issuetoc)

Pieri, C. 1989. *Fertilité des terres de savane*. Bilan de trente ans de recherche et de développement agricoles au Sud du Sahara. Paris, Ministère de la Coopération, CIRAD. 444 pp.

Pretorius, D. 2009. *Mapping Land Use Systems at a National Scale for Land Degradation Assessment Analysis in South Africa*. South Africa, Department of Agriculture..

Prinsloo, M.A., Willshire, G.H. & Du Proez, C.C. 1990. Loss of Nitrogen fertility and its restoration in some Orange Free State soils. *S. Afr. J Plant Soil*, 7: 55-61.

Rantoa, N.R., Du Preez, C.C. & Van Huyssteen, C.W. 2009. *Estimating organic carbon stocks in South African Soils*. South Africa, Bloemfontein, University of the Free State. (MSc. Thesis)

Rapp, I. 1998. *Effects of soil properties and experimental conditions on the rill erodibility of selected soils.* University of Pretoria. (Unpublished Ph.D. thesis)

Reich, P.F., Numbem, S.T., Almaraz, R.A. & Eswaran, H. 2001. Land resource stresses and desertification in Africa. In E.M. Bridges, I.D. Hannam, L.R. Oldeman, F.W.T. Pening de Vries, S.J. Scherr, and S. Sompatpanit, eds. *Responses to Land Degradation.* Proc. 2nd. International Conference on Land Degradation and Desertification, Khon Kaen, Thailand. India, New Delhi, Oxford Press.

Rhoades, C.C., Eckert, G.E. & Coleman, D.C. 2000. Soil carbon differences among forest, agriculture, and secondary vegetation in lower montane Ecuador. *Ecological Applications*, 10, 497–505.

Ringius, L. 2002. Soil carbon sequestration and the CDM: Opportunities and challenges for Africa. *Climate Change*, 54: 471–495.

Roy, R.N., Misra, R.V., Lesschen, J.P. & Smaling, E.M. 2003. Assessment of soil nutrient balance. *Approached and methodologies.* FAO Fertilizer and Plant Nutrition Bulletin 14. Rome, FAO. 15 pp.

Sanchez, P.A. 2000. Linking climate change research with food security and poverty reduction in the tropics. *Agr. Ecosyst. Environ.*, 82: 371–383.

Schjønning, P., Thomsen, I.K., Møberg, J.P., de Jonge, H., Kristensen, K. & Christensen, B.T. 1999. Turnover of organic matter in differently textured soils: I. Physical characteristics of structurally disturbed and intact soils. *Geoderma*, 89: 177-198.

Schlesinger, P. & Winkler, J.P. 2000. Soils and the global carbon cycle. In T.M.L. Wigley & D.S. Schimel, eds. The Carbon Cycle, pp. 93-101. UK, Cambridge, Cambridge University Press.

Schneider, A., Friedl, M.A., Potere, D.A. 2010. Mapping global urban areas using MODIS 500-m data: New methods and datasets based on 'urban ecoregions'. *Remote Sensing of Environment*. 114(8): 1733–1746.

Schoeman, F., Newby, T.S., Thompson, M.W. & Ven den Berg, E.C. 2010. *South African National Land-Cover Change Map*. ARC-ISCW Report Number: GW/A/2010/47.

Scotney, D.M. & Dijkhuis, F.J. 1990. Changes in the fertility status of South African soils. *S. Afr. J. Science*, 86: 395-402.

Scotney, D.M., Volschenk, J.E. & Van Heerden, P.S. 1990. *The potential and utilization of natural agricultural resources of South Africa.* Pretoria, Department of Agricultural Development. 11pp.

Shepherd, K.D. & Soule, M.J. 1998. Soil fertility management in West Kenya. *Agriculture, Eco systems and Environment*, 71(1-3): 133- 147.

Smaling, E. 1993. *An agro- ecological frame work for integrated nutrient management, with special reference to Kenya*. The Netherlands, Wageningen, Agricultural University. (Doctoral thesis)

Smaling, E., Nandwa, S. & Janssen, B. 1997. Soil fertility in Africa is at stake. In R.J. Buresh & P.A. Sanchez, eds. *Replenishing soil fertility in Africa*. SSSA Special Publication. USA, Madison, Wisconsin, Soil Science Society of America and American Society of Agronomy.

Smith P. 2008. Soil organic carbon dynamics. In A.K. Braimoh & P.L.G. Vick, eds. *Land Use and Soil Resources*. Springer Science & Business Media B.V.

Sombroek, W., Nachtergaele, F. O. & Hebel, A.1993. Amounts, dynamics and sequestering of carbon in tropical and subtropical soils. *Ambio*, 22: 417–426.

SOW-VU. 2010 National land degradation assessment Senegal and Review of global socio-economic parameters in the LADA database. Project Report. Amsterdam, SOW-VU.

Stoorvogel, J.J. and Smaling, E.M.A. 1990. *Assessment of soil nutrition depletion in Sub-Saharan Africa*: 1983-2000. Report No. 28. Vol. 1. The Netherlands, Wageningen, Winand Staring Centre.

Stoorvogel, J.J., Smaling, E.M.A. & Janssen, B.H. 1993. Calculating soil nutrient balances in Africa at different scales. 1. Supra-national scale. *Fert Res*, 35: 227–235.

Swift, M.J. & Shepherd, K.D. 2007. *Africa's Soils: Science and Technology for improved soil management in Africa*. Nairobi, World Agroforestry Centre.

Thibaud, G.R. 2005. NPK fertilization and liming in no-till. In A. Venter, T. Matchett, W. Mackenzie, D. MacNicol, B. Berry, B. Russell & V. Brummer, eds. *A Guide to No-till Crop Production in KwaZulu-Natal*. Howick, No-Till Club.

Tieszen, L.L., Tappan, G.G. & Toure A. 2004. Sequestration of carbon in soil organic matter in Senegal: an Overview. *Journal of Arid Environments*, 59: 409–425.

Todaro, M. 2000. *Economic Development*. Essex, Pearson Education Limited.

Tschakert, P., Khouma, M. & Sene M. 2004. Biophysical potential for soil carbon sequestration in agricultural systems of the Old Peanut Basin of Senegal. *Journal of Arid Environments*, 3: 511–533.

Udo, R.K. 1970. *Geographical regions of Nigeria*. London, Heinemann & Nigeria, Ibadan.

UNDP. 2005. *Population Division*. World Population Prospects: the 2004 Revision and World Urbanization Prospects. Department of Economic and Social Affairs of the United Nations Secretariat.

UNEP. 1982. World's soil policy. Kenya, Nairobi, United Nations Environment Programme.

UNEP. 2007. *Global Environment Outlook 4 (GEO-4)*. Nairobi. 572 pp.

UNEP. 2013. *Second Global Conference on Land – Oceans Connection (GLOC-2)*. October 2-4, 2013. Jamaica, Montego Bay.

UNSO/SEED/BDP. 1999. *Promoting Farmer Innovation: Harnessing local environmental knowledge in East Africa Regional Land Management (RELMA)*. Unit Workshop Report No. 2, Urban Food Production: A Survival Strategy of Urban Households. Report of a Workshop on East and Southern Africa. May 3-5, 1998. Kenya, Nairobi.

Vågen, T.G., Lal, R. & Singh, B.R. 2005. Soil carbon sequestration in sub-Saharan Africa: A review. *Land Degrad*. Dev., 16: 53–71.

Van Antwerpen, R. & Meyer, J.H. 1996. Soil degradation under sugarcane cultivation in Northern KwaZulu-Natal. *Proceedings of the South African Sugar Technology Association*, 70: 29-33.

Van der Merwe, A.J. 1995. *Wise land use: The basis for sustainable growth and development in South Africa'*. Proceedings of the Wise Land Use Symposium. Pretoria: ISCW. pp. 2-8.

Van der Merwe, A.J., De Villiers, M.C., Barnard, R.O., Beukes, D.J., Laker, M.C. & Berry, W.A.J. 2000. *Technical report on guidelines on the management and rehabilitation of acid and fertility declined soils in South Africa*. Proc. 2nd MADS-SEA-Network Meeting, September 18-22. Pretoria.

Wiese, L., Lindeque, L. & De Villiers, M. 2011. *Land Degradation Assessment in Drylands*. Project Policy Report. ARC-ISCW Project Nr. GW/A/2011/52.

Williams, C.A., Hanan, N.P., Neff, J.C., Scholes, R.J., Berry, J.A., Denning, A.S., Baker, Windmeijer P.N. & Andriesse W. 1993. *Inland valleys in West Africa: an agro-ecological characterization of rice-growing environments.* Publication 52. The Netherlands, Wageningen, ILRI, International Institute for Land Reclamation and Improvement.

WMO. 2005. *World Meteorological Organization*. ISBN 92-63-10989-3. 34 pp.

Woomer, P.L., Toure, A. & Sall, M. 2004. Carbon stocks in Sahel transition zone. J. *Arid Environ.*, 32: 134–147.

World Bank. 1992. *Trade and the Environment: A Survey of the Literature*. Background paper for World Development Report 1992. WB.

World Bank. 1997. *World Development Report 1997: The State in a Changing World*. New York, Oxford University Press.

World Bank. 1998. *Soil Fertility Management in Sub-Saharan Africa*. World Bank Technical Paper No. 408.

World Bank. 2002. World Development Report 2002: *Building Institutions for Markets*. Oxford, Oxford University Press.

Yanai, R.D., Stehman, S.V., Arthur, M.A., Prescott, C.E., Friedland, A.J., Siccama, T.G. & Binkley, D. 2003. Detecting change in forest floor carbon. *Soil Sci. Soc. Am. J.*, 67: 1583– 1593.

Zivin J.G. & Lipper, L. 2008. Poverty, Risk, and the Adoption of Soil Carbon Sequestration. *Environment and Development Economics*, 13: 353-373.

10 | Regional Assessment of Soil Change in Asia

Regional Coordinator/Author: Kazuyuki Yagi (ITPS/Japan)

Contributing Authors: Fahmuddin Agus (Indonesia), Tomohito Arao (Japan), Milkha S. Aulakh (ITPS/India), Zhaohai Bai (China), Rodel Carating (The Philippines), Kangho Jung (South Korea), Atsunobu Kadono (Japan), Masayuki Kawahigashi (Japan), Seung Heon Lee (South Korea), Lin Ma (China), G.P. Obi Reddy (India), G. S. Sidhu (India), Yusuke Takata (Japan), Tran Minh Tien (Vietnam), Renkou Xu (China), Xiaoyuan Yan (China), Kazunari Yokoyama (Japan), Fusuo Zhang (China), Dongmei Zhou (China).

10.1 | Introduction

This chapter describes the status of soil resources in the member countries of the Asian Soil Partnership (ASP), which includes East Asia (five countries: China, Democratic People's Republic of Korea, Japan, Mongolia, and Republic of Korea), Southeast Asia (11 countries: Brune[i] Darussalam, Cambodia, Indonesia, Lao People's Democratic Republic, Malaysia, Myanmar, Philippines, Singapore, Thailand, Timor Leste, and Viet Nam), and South Asia (eight countries: Afghanistan, Bangladesh, Bhutan, India, Maldives, Nepal, Pakistan, and Sri Lanka).

Asia, the Earth's largest and most populous continent, is located largely in the eastern and northern hemispheres. The region, which consists of the above-mentioned ASP countries, covers 4.1 percent of the Earth's total surface area and comprises 16 percent of its land area. With approximately 3.9 billion people, Asia hosts 55 percent of the world's population. Population density is high – averaging 1.87 persons ha^{-1} – compared to the world average of 0.54 person ha^{-1}. Like most areas of the world, Asia has experienced rapid population growth rate in the modern era. In the twentieth century, Asia's population nearly quadrupled, as did the world population.

In general, Asia enjoys a warm and seasonally humid climate and is well-endowed with natural resources for agriculture. The unique combination of the monsoon climate and the exceptionally large lowland area has made Asia the rice basket of the world (Kyuma, 2004). Sustained high levels of staple food production have enabled Asian countries to support a large population within a limited area of arable land. However, recently, Asian countries have faced rapid changes in both socio-economic and natural factors and these have had major impacts on agro-environments in the region. In particular, rapid economic development and urbanization are changing land management systems in many countries, and climate change has emerged as a significant source of risks. These changes are having major impacts on the status of soil resources in the region.

10.2. Stratification of the region

10.2. 1 | Climate and agro-ecology

The map of Asia shows many vast rivers with large alluvial plains and deltas. Major rivers include the Yellow, Yangtze, Mekong, Chao Phraya, Irrawaddy, Ganges, Brahmaputra, and Indus. The Himalayan mountain range runs for more than 2 400 km, separating the Indian subcontinent from the rest of Asia. Many of Asia's major rivers have their source in the Himalayas and adjacent plateaus. These rivers have created vast areas of fertile land suitable for farming. On the east and southeast shores of the continent lie a number of islands chains or island arcs, many characterized by mountainous landscape with volcanic activities.

Most areas of the Asian region are strongly influenced by the monsoon, a seasonal reversing wind accompanied by corresponding changes in precipitation. For this reason, the region is often called 'Monsoon Asia'. The Asian monsoon is a highly significant component of the global climate system. It has a huge influence on how people live and on their livelihoods, and it provides water resources throughout the region (Salinger *et al.*, 2014). The East Asian monsoon carries moist air from the Pacific Ocean and the Indian Ocean to East and Southeast Asia in summer. In winter, it reverses and carries cold, dry air from the Eurasian Continent. In south Asia, winds blow from June to September from a south-westerly direction from the Indian Ocean onto the Indian landmass, bringing rain to most parts of the subcontinent. Subsequently, from around October, the winds reverse direction and start blowing from a north-easterly direction, from the subcontinent onto the Indian Ocean. These winds carry less moisture and bring rain to only limited parts of India. Dry areas predominate in parts of the north-western interior where the influence of oceanic winds is less. This wide diversity of climate is a major factor in the stratification of the region into different agro-ecological zones (Figure 10.1)

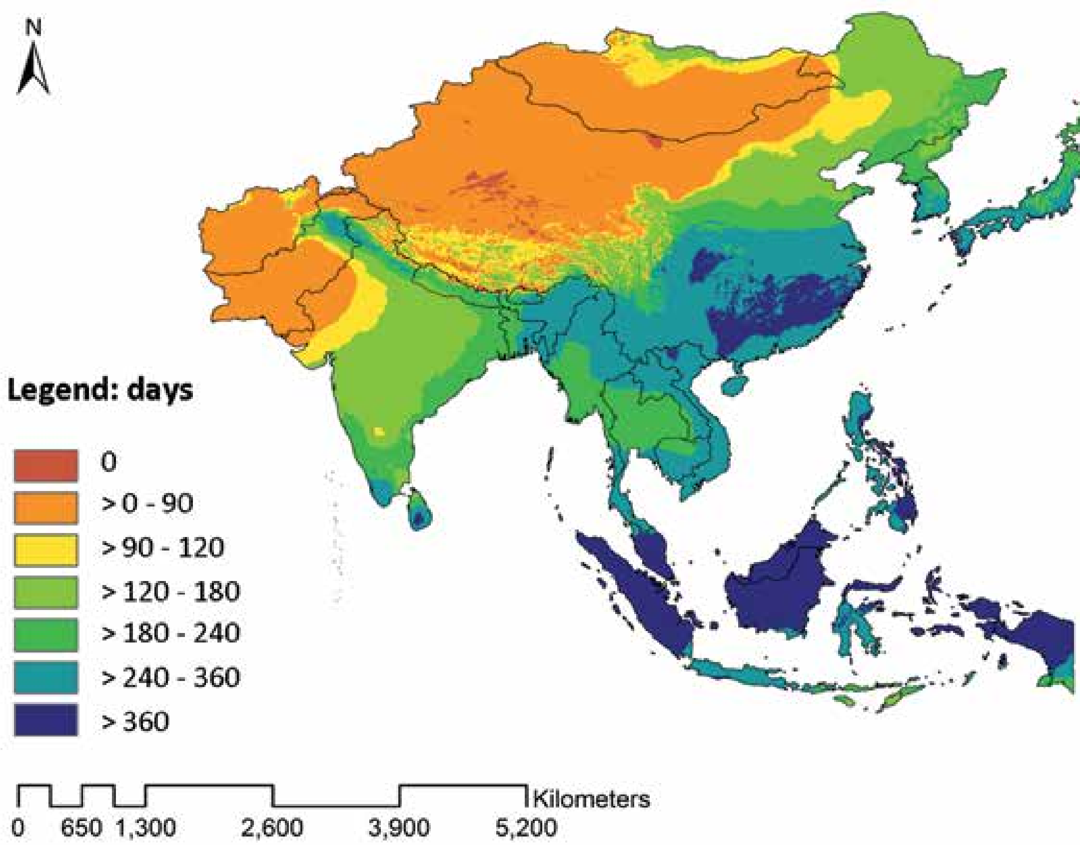


Figure 10.1 | Length of the available growing period in Asia (in days yr⁻¹).
Source: Fischer et al., 2012.

A distinctive feature of monsoon Asia is its very high share of the world's area and production of rice (Kyuma, 2004). Some 90 percent of the global acreage and output of rice are concentrated in the region, earning it the title of 'rice granary' of the world. This dominance of rice cultivation is due to several factors, notably high precipitation and temperatures and the existence of extensive lowlands suitabe for paddy prduction. The vast expanse of lowlands is a unique feature of the region, resulting from a combination of geological instability and the high precipitation. Rice cultivation originally emerged as an adaptation to extensively inundated lowlands, but with time it was expanded to areas that could support rice only with irrigation. High productivity and high sustainability are the outstanding advantages of rice cultivation. By contrast, upland cultivation of dryland crops in Monsoon Asia is handicapped by low soil fertility and high susceptibility to soil erosion.

10.2.2 | Previous regional soil assessments

Based on the report by Oldeman (1991), the GLASOD project estimated that human-induced soil degradation in Asia region (including non-ASP west Asian countries) accounted for 31 percent of the inhabited land area, the highest share of any of the global regions. Soils in Asia were found to have been degraded by several factors: water erosion (59 percent), wind erosion (30 percent), chemical degradation (10 percent) and physical degradation (2 percent). Deforestation was identified as the most dominant causative factor for soil degradation, followed by agricultural activities and overgrazing. In most Asian countries the mining of soil nutrients is causing decline of average crop yields. Fertile soil is washed away by the erosive forces of water, or blown away by wind. This so-called first generation of environmental problems is leading not only to negative nutrient balances but also to habitat destruction and loss of biodiversity (Oldeman, 2000).

Following GLASOD, the need for more detailed and more country-specific degradation assessments became apparent. In 1993, the members of the Asian Network on Problem Soils recommended the preparation of a qualitative assessment for South and Southeast Asia at a scale of 1:5 million. This recommendation was acknowledged by FAO and UNEP. FAO assigned ISRIC to prepare a new physiographic map and database at 1:5 million scale. UNEP prepared and implemented the Assessment of the Status of Human-Induced Soil Degradation in South and Southeast Asia (ASSOD). Sixteen national institutions for natural resources in the region collaborated on the project under the coordination of ISRIC. The ASSOD project published revised sub-regional Guidelines for General Assessment of the Status of Human-Induced Soil Degradation and produced regional maps on the status of human-induced soil degradation at a scale of 1:5M together with digitized version of the map (van Lynden and Oldeman, 1997).

The different soil degradation types inventoried by ASSOD are described below and shown in Figure 10.2:

- Water erosion: Water erosion covers 21 percent of the total land area in the region (or 46 percent of the total degraded area). It is predominant in large parts of China (>180 million ha) except for the northern parts, on the Indian subcontinent (>90 million ha) and in the sloping parts of Indochina (40 million ha), the Philippines (10 million ha) and Indonesia (22.5 million ha). In relative terms, as a percentage of the total country area, moderate to extreme water erosion is particularly important in India (10 percent), the Philippines (38 percent), Pakistan (12.5 percent), Thailand (15 percent) and Vietnam (10 percent).
- Wind erosion: Wind erosion (9 percent of the total area, 20 percent of all degradation) is concentrated mainly in the most western and northern arid and semi-arid desert regions of Pakistan (>9 million ha on-site and >2 million ha off-site), India (20 million ha on-site, 3.6 million ha offsite) and China (>70 million ha on-site, >8.5 million ha off-site). Although large parts of these regions may be considered deserts, some human-induced wind erosion was also reported.
- Chemical deterioration: Chemical deterioration is distributed in patches, probably also partly due to different perceptions of this type of degradation. About 11 percent of the total area (or 24 percent of the degraded area) is affected by some kind of chemical deterioration. High relative extents of chemical deterioration (>30 percent of total country area) can be observed in Bangladesh, Cambodia, Sri Lanka, Malaysia, Pakistan and Thailand, generally with negligible to light impact.
- Physical deterioration: Occurrence of physical deterioration (affecting about 4 percent of the total area or 9 percent of the total degraded area) is even more dispersed and infrequent than chemical deterioration. Waterlogging and aridification are the main subtypes, in particular in Bangladesh, China, India and Pakistan. Compaction or crusting/sealing is relatively unimportant except in Thailand and the Philippines, although they occur in most countries. Waterlogging and compaction as a result of paddy cultivation are not considered as degradation. Loss of productive function as a result of urbanisation, industrialisation and infrastructure has been indicated for only a few countries although this phenomenon is on the rise.

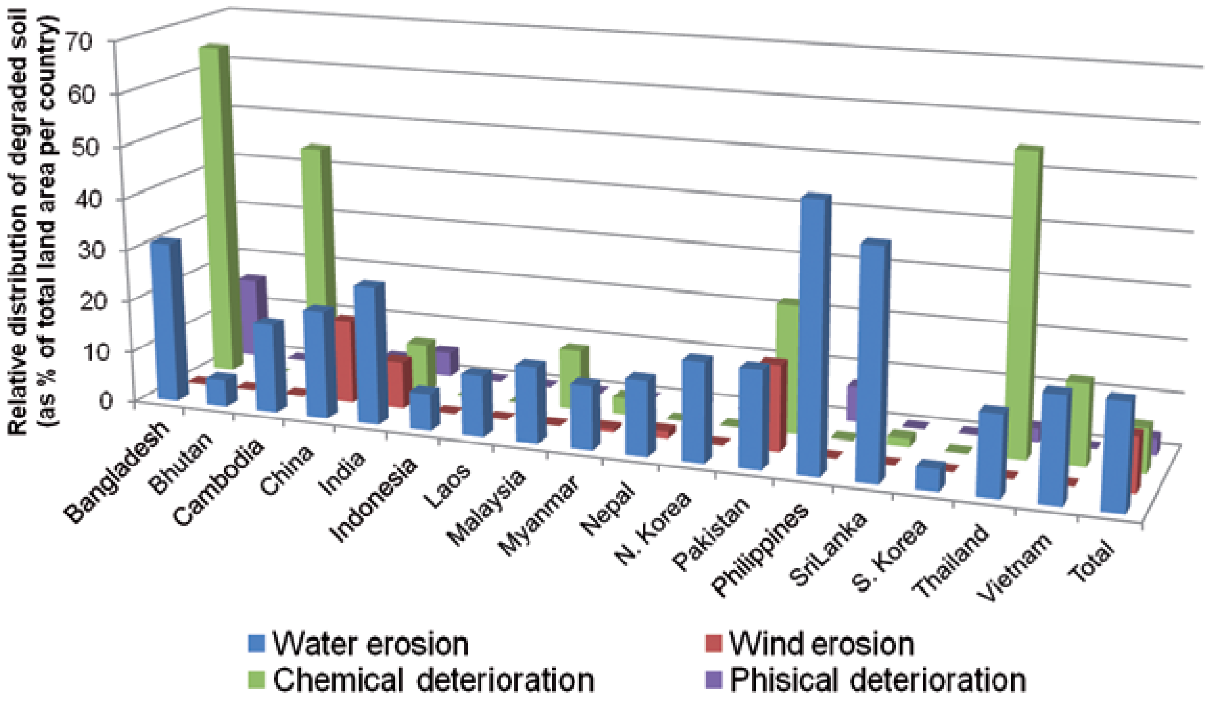


Figure 10.2 | Threats to soils in the Asia region by country.

10.3 | General threats to soils in the region

10.3.1 | Erosion by wind and water

Soil erosion is the most important threat to soil in Asia. Water erosion by rainfall and surface water flow is dominant in humid regions with torrential rains, as in South and East Asia. In drier or desert areas, wind is the driving force inducing soil erosion. This threat is discussed below in Section 10.4.1 in detail.

10.3.2 | Soil organic carbon change

Data for evaluating soil organic carbon (SOC) change in Asian countries are limited because countries do not generally monitor SOC stock and changes. However, data from available literature show that where there are increases in crop yield in croplands of East and Southeast Asia, SOC is retained. SOC has also been shown to accumulate in forest areas. However, in South Asia SOC is decreasing. This is because crop residues are widely used as fuel and fodder and are not returned to the soil. In Indonesia, three anthropogenic activities – deforestation, poor land management, and intensive cropping – contribute to SOC change in mineral and peat soils (Section 10.5.2). Throughout the region, the degradation of grassland has generally caused great losses of SOC stock. This threat is discussed in detail below in Section 10.4.2.

10.3.3 | Soil contamination

Sources of contamination of arable land in most Asian countries include (i) parent material, (ii) mining, (iii) smelting, (iv) agrochemicals and sewage sludge applications, and (v) livestock manure uses (Luo *et al.*, 2009). There is an urgent need to reduce hazardous chemical concentrations of Cd, As, Pb, Cu and Zn, especially in paddy soil and rice grains. In many regions of Southeast Asia (Bangladesh, India, China, Vietnam, Taiwan - Province of China, Thailand and Nepal), arsenic is naturally present in groundwater and represents a threat to sustainable agriculture (Smedley, 2003; Brammer and Ravenscroft, 2009). This enrichment is magnified considerably when mining takes place, as it is the case of the Ron Phibun district, in southern Thailand (Williams *et al.*, 1996). Arsenic contamination particularly affects rice, as this crop not only requires large amounts of water but is also grown under the anaerobic conditions favoured in rice fields where As is mainly present in its trivalent form which is readily available to plants (Brammer and Ravenscroft, 2009). The agricultural rice soils of the Guandu Plain in Taiwan, Province of China are seriously contaminated with As and Pb (Zhuang *et al.*, 2009; Chang *et al.*, 1999, 2007). Cadmium pollution of paddy fields has been found downstream of a Zn mineralized area in Thailand (Simmons *et al.*, 2005). China contributes around 28 percent of global Hg emissions, with India, Japan and the Korea Democratic Republic also among the ten countries that contribute the most to global Hg emissions (Li *et al.*, 2009).

According to the Agricultural Land Soil Pollution Prevention Law in Japan, the maximum allowable limit of Cd in paddy fields is set in terms of the Cd concentration in rice grains produced in the field, not the soil Cd concentration of the field. This is because the amount of bio-available Cd in soil is affected dramatically by water management practices used for rice cultivation (Asami, 1981).

With rapid industrialization, urbanization and intensive use of farmland, China is now facing serious soil pollution (Ministry of Environmental Protection, the People's Republic of China, 2014). About 19.4 percent of farmland has high levels of Cd, Ni and As pollution. Soil contamination has been estimated to cause a reduction of more than 107 tonnes of food supply annually (Wei and Chen, 2001). In 2006-2010, China's Ministry of Environmental Protection and Ministry of Land and Resources jointly launched a nationwide investigation of soil pollution status, covering an area of 6.3 million square kilometers. In 2014, a Communiqué on Nationwide Soil Contamination was issued by the two ministries indicating that the overall situation of the soil environment in China is not encouraging. Some regions are heavily polluted. There is concern over the quality of farmland soils and over other soil environmental issues also caused by anthropogenic activities, e.g. those related to mining and industrial activities which cause atmospheric deposition, and to the use of livestock manures (Luo *et al.*, 2009). Trace elements are pollutants of major concern, especially in the southern area of China. Currently, the evaluation of soil pollution in China is primarily based on the Environmental Quality Standard for Soils which was promulgated and implemented in 1995. The Standard needs further improvement because of its stringent limitations. At present, China's Ministry of Agriculture is working on an Implementation Plan on Prevention and Control of Heavy Metal Pollution in Agricultural Products.

Historic and current rates of intensive pesticide and fertilizer use in agricultural land and also industrial development have caused the accumulation of organic pollutants and heavy metals in soils of Indonesia. Earlier research showed high concentration of organo-chlorines in vegetables (Dibiyantoro, 1998). However, residue levels in foodstuffs have gradually reduced to within acceptable daily intake levels as established by WHO (Shoiful *et al.*, 2012; Rahmawati *et al.*, 2013). The tapioca industry in Java is now recognized as a contributor to cyanide levels which have risen above background levels in river water (Indrayatie *et al.*, 2013). Mining plays an important role in the Indonesian economy but this mining, particularly artisanal and small-scale mining (ASGM), can have a major impact on the environment (Limbong *et al.*, 2003; Prasetyo *et al.*, 2010). During ASGM, Hg is used to recover Au from the ore during grinding. The process is inefficient and releases a significant amount of Hg to soil, water and the atmosphere (Limbong *et al.*, 2003; Edinger *et al.*, 2008). Tailings from Hg amalgamation are then leached with cyanide. Ultimately, the final waste, contaminated with metals and cyanide, is released into the environment (Veiga *et al.*, 2009). Many ASGM operations also release As and Sb to the environment, although this depends on the composition of the host ore (Edinger *et al.*, 2008). These operations are unlicenced and illegal. Indonesia has now signed the Minamata Convention on Hg and has decentralized control of ASGM operations to provincial governments. Environmental protection has become a primary objective for government regulation.

10.3.4 | Soil acidification

There is large area of acid soils distributed across the tropical and subtropical regions of Asia, mainly in Southeast Asia, parts of East Asia and parts of South Asia. Acid sulphate soils are widely distributed in the coastal plains of Southeast Asia and southern China. The total area of acid sulphate soils in Southeast Asia is 7.5 million ha and there are about 112 thousand ha of these soils in China (Shamshuddin *et al.*, 2014). The soils in these regions are also sensitive to external acid such as acid deposition (Hicks *et al.*, 2008).

In Vietnam, ferralitic, basaltic, and grey degraded soils, which cover about one third of the country, have strong potential for acidification and degradation because of their nature (NISF, 2012). A leading contributor to soil acidification in Vietnam is the unbalanced and unsuitable application of chemical fertilizers. Data from the International Fertilizers Association (IFA) show that from 1961 to 2012, total NPK chemical fertilizer use in Vietnam increased 31 times (IFA, 2012). The increase in fertilizer application and unbalanced use lead to inadequate presentations of acidic factors in soil solutions. Some types of fertilizers (including organic fertilizers) can make the soil more acidic (Nguyen, 2014). Another reason for the acidification process in Vietnam is the presence of sulphate soil. The area of sulphate soil in the Red River delta has increased by 7 000 ha and the content of S in this soil has also strongly increased. In the Mekong river delta, the area has reduced by 261 000 ha and the content of total exchangeable cations and total dissolved salts has also reduced. Increase in temperature, change of precipitation, and sea level rise due to climate change may also be having negative effects on this process in Vietnam.

The total area of Indonesian acid upland soils (pH <5 and <50 percent base saturation) is about 102.8 million ha, of which more than half (55.8 million ha) are suitable for various types of agriculture and plantation (Puslitbangtanak, 2000). Relatively low cost technologies are widely available for rectifying soil acidity problems. These include liming, organic matter application, balanced fertilization and selection of acid tolerant crops. However, farming practices, such as application of acidifying fertilizers like ammonium sulphate, aggravate the problem (Kamprath, 1984).

In China, acid soils are mainly distributed in tropical and subtropical regions south of the Yangtze River. The total area of acid soils is about 204 million ha, 22.7 percent of the total land area in the country. In the past three decades, soil acidification has accelerated in southern China due to serious acid deposition and the heavy application of ammonium-based fertilizers. In subtropical regions of the country, soil pH decreased on average by 0.23 units for cereal cropland and 0.30 units for cash cropland between the 1980s and 2000s (Guo *et al.*, 2010). Soil acidification in cropland in China was accelerated mainly by ammonium-based fertilizers, while acid deposition was mainly responsible for acidification in forest soils. Although the emission of SO_2 was reduced by 20 percent in China in the past ten years, acid deposition is still a serious problem and its impact on soil acidification is unlikely be eliminated in the near future (Zhao *et al.*, 2009).

Tea is cultivated extensively in subtropical regions of China, Japan, Southeast Asia and parts of South Asia. There is serious acidification of soils in tea gardens (Wang *et al.*, 2010). Excess application of ammonium-based fertilizers and lack of leaching were responsible for acidification and salinization of soils in vegetable greenhouses (Guo *et al.*, 2010).

10.3.5 | Soil salinization and sodification

The threat of salinization/sodification in the region takes varying forms. In the semiarid and arid zones of central and west Asia, salt-affected soils are widely distributed. On the other hand, salt-affected soils are also developing in certain coastal areas in monsoon zones, caused mainly by salt water intrusion in South and Southeast Asia and by coastal tideland reclamation. Although the coastal area affected is relatively small, this could become a serious problem for lowland rice production. This threat is discussed in detail below in Section 10.4.3.

10.3.6 | Loss of soil biodiversity

Potentially the greatest contributor to soil biodiversity loss in Asia is land use change. In China, for example, an assessment of land-use change across the country indicated that the largest changes were associated with the conversion of productive cultivated land to urban areas, thereby removing fertile land from agricultural production. Conversion of cultivated areas, forest and grasslands between 1996 and 2008 has been estimated to be 1 475 × 104 ha, 269 ×104 ha, and 536 ×104 ha, respectively (Wang *et al.*, 2012). There have been attempts at land restoration to maintain soil biodiversity, such as in the coastal lands in the Jiangsu Province of China. Generally, there is a higher diversity of soil macrofauna in uncultivated land and forests compared to less diverse wheat farms and bulrush land (Baoming *et al.*, 2014). Soil faunal diversity is significantly impacted by land use with a strong relationship between vegetation and macrofauna distribution and composition (Baoming *et al.*, 2014). In Thailand, the forested land area has decreased by more than 50 percent in the past 40 years (Fisher and Hirsch, 2008). Although reforestation measures can be used to restore degraded lands, difficulties with establishment of native trees in these areas may lead to the use of substitute trees, resulting in long lasting effects on soil biodiversity. For example, a study of soil microbial communities of an Acacai tree plantation established on degraded land in Thailand found reduced microbial activities compared to natural evergreen forest, suggesting that key microbes had been lost (Doi and Ranamukhaarachchi, 2013).

Studies in India have also indicated an effect of land use change on soil organisms. The Nilgiri biosphere reserve in the Western Ghats of India, a global hotspot for aboveground biodiversity, has been under pressure because of high population density in the surrounding region (Mujeeb Rahman, Mujeeb, Varma and Sileshi, 2012). In this area, land management had a significant effect on soil macrofauna, with larger densities and diversity found in the forest sites and a clear response of macro-invertebrates to land use (Rossi and Blanchart, 2005). Interestingly, there was a high similarity in macrofauna between primary forest and disturbed forest plots, which indicated the potential of land restoration. A separate morphological analysis of soil invertebrate density at 15 different land use sites (from intensively managed agricultural systems to pristine forests) identified a wide range of soil faunal groups including earthworms, termites, ants, grasshoppers, crickets, mole crickets, bugs, coccids, cicadas, woodlice, centipedes, millipedes, and spiders (Mujeeb Rahman, Mujeeb, Varma and Sileshi, 2012). The natural forests had significantly higher taxonomic richness at the family level than soils from the annual cropping system and a significantly higher total number of individuals than annual crops, agroforestry and plantations. The highest richness (identified to family level) was found in the sites with the least anthropogenic disturbance, and the greatest diversity of earthworms, ants and termites (determined to species level) was found in the more complex forest ecosystems. An earlier study in the region recorded almost double the number of earthworm species in soils from forest compared to pasture (Blanchart and Julka, 1997). These results indicate a decrease in earthworm biodiversity associated with forest loss and a lower diversity of soil macrofauna with more intensive land use.

In addition to anthropogenic pressures on soil biodiversity, natural disturbances such as tsunamis can affect soil biodiversity. Tsunami-affected areas in the Phang Nga province of Thailand had a higher proportion of prokaryotes (archaea and bacteria; 83.25 percent) compared to non-affected areas (72.5 percent), whereas the non-affected areas were more hospitable to eukaryotes (animals, plants, fungi and protists) (Somboonna *et al.*, 2014). Increased occurrences of tsunamis and other natural disasters may result in losses to above and belowground biodiversity.

There has not been a comprehensive analysis of threats to soil biodiversity in Asia completed to date. Scientific evaluation of soil biodiversity over large regions has been extremely difficult due to: (1) size of organisms, (2) large abundance and diversity of organisms, and (3) lack of research and gaps in the information collected. In Japan, the fusion of complex systems theory and computer technology has made it possible to employ the tools of statistical physics to develop a totally new technology for assessment of farmland soils (Yokoyama, 1993). The technology is used primarily for a health check on farmland soil, to measure the environmental affinity of production. Commercial analysis services have been started to add value to the products (Sakuramot, Yokoyama and Iekushi, 2010; Yokoyama and Taguchi, 2013).

In paddy rice cultivation and upland farming with sustainable agriculture, crop rotation and organic amendments have generally maintained good soil microbial diversity. However, loss of significant microbial diversity is notable in large-scale and intensive vegetable production areas due to: (i) continuous monoculture cropping, (ii) over-cultivation, (iii) intensive use of chemical fertilizers, (iv) chemical fumigation of soils to prevent soil-borne diseases, (v) soil contamination and non-target effects of chemical pesticides such as fungicides, nematicides, herbicides and insecticides, and (vi) over-application of herbicides for weed control. The loss of diversity in the soil is conducive to further diseases and other cropping issues from processes (ii) to (vi).

Recently, the effect of poor management practices on soil biodiversity has led to concerns. As a result, environmental and safety-oriented consumers have tried to promote organic farming and to accelerate the branding of sustainably produced agricultural products based on scientific evidence. The idea is to create incentives for farmers to consider soil biodiversity in their farming practices. The private sector has undertaken evaluation of microbial diversity, demonstrating high soil microbial diversity in organically managed paddy fields in Taiwan - Province of China and in the Philippines, as well as in upland crops in southern China. However, in soils of the semi-arid areas of northern China, the loss of microbial diversity is severe. Here the processes mentioned –in points (i) to (vi) above may lead to soil drying, increases in salt concentrations and a loss of stable soil surface. These changes could potentially result in loss of soil organisms, although research exploring the loss of ecosystem functions and services from these soils has yet to be conducted.

10.3.7 | Waterlogging

Two types of waterlogging may occur – permanent waterlogging in natural swampland, and occasional waterlogging in flood prone areas along the flat coastal regions and flood plains of main rivers. Constructed wetlands such as paddy fields are intentionally flooded as part of the management system. Waterlogging can have other anthropogenic causes such as poor drainage systems in settlements, industrial and urban development, or deforestation in upstream areas, all of which may increase the threat of water logging in flood prone areas.

In the GLASOD estimate, waterlogging affects 4.6 million ha in Asia, largely in the irrigated areas of India and Pakistan. Waterlogging is closely linked with salinization. Since the start of large scale irrigation schemes in the 1930s, the progressive rise in the water table beneath irrigation areas on the Indo-Gangetic plains has been monitored (e.g. Ahmad and Kutcher, 1992). For India, monitoring results suggest a waterlogged area more than twice the GLASOD estimate. For Pakistan, four sources give total areas affected by waterlogging of between 1.6 and 3.7 million ha, compared with the GLASOD value of 0.96 million ha. Since the Pakistan country data come from at least two independent surveys, show good agreement and are believed to result from detailed field surveys, these country estimates are likely to be more accurate than the much lower GLASOD estimates.

10.3.8 | Nutrient imbalance

Harvested crops remove nitrogen, phosphorus, and other nutrients from agricultural soils. Hence, sustaining agricultural production requires replacement of those nutrients, whether through biological processes like nitrogen fixation or through the addition of animal wastes or mineral fertilizer to fields (Vitousek *et al.*, 2009). Balanced nutrient supply is essential for achieving high crop yields, but excessive and/or imbalanced nutrient input may pose risks to the environment, human health and ecosystems. Nutrient inputs in Asia vary considerably amongst countries, areas and farming systems.

In some countries or regions in Asia, removal of nutrients from the soil in crop harvest appears substantially to exceed inputs through natural replacement or fertilizer application. For example, negative soil nutrient balances have been reported for each of the 15 agro-climatic regions of India (Biswas and Tewatia, 1991; Tandon, 1992).

Problems are also caused by imbalances in fertilizer application. Fertilizer use in the region is often dominated by nitrogen (N), relative to phosphorous (P) and potassium (K). This trend originated in the early years of the 'green revolution'. When fertilizers are first applied to a soil, a high response is frequently obtained from nitrogen. The improved crop growth depletes the soil of other nutrients; "In such systems, nitrogen is simply used as a shovel to mine the soil of other nutrients" (Tandon, 1992). Long-term experiments in India show depletion of soil P and K is higher for plots with N fertilizer, and depletion of K is higher still with N+P fertilizer (Tandon, 1992). The use of N fertilizer in Indonesia has been continually increasing since the late 1960s, reaching 2.4 million Mg yr^{-1} in 2012. However, the consumption of P and K has not followed the same trend (IFADATA, 2007). This is partly due to the fact that N fertilizers are cheaper and more accessible, and also to the rapid crop response to N fertilization. A study on nutrient balances in rice fields under intensive cultivation found a positive balance for N, P, Ca, Mg and Na, but a negative balance for K and Si (Husnain, Masunaga and Wakatsuki, 2010).

For secondary nutrients and micronutrients, an increasing incidence of sulphur and zinc deficiency is occurring in the region. Sulphur deficiency has been reported for India, Pakistan and Sri Lanka, and zinc deficiency for India and Pakistan (FAO/RAPA, 1992). For Bangladesh, 3.9 million ha are reported to be deficient in sulphur and 1.75 million ha in zinc, including areas of continuous swamp rice cultivation (Bangladesh, 1992; Shaheed, 1992). Because of its generally alkaline soils, Pakistan is particularly liable to micronutrient deficiencies (Twyford, 1994).

On the other hand, nutrient additions to many fields in some Asian countries far exceed those in the United States and Northern Europe, and much of the excess fertilizer is lost to the environment, degrading both air and water quality. This important threat to soil is described more in detail in Section 10.4.4.

10.3.9 | Compaction

Slightly compacted soil conditions are conducive to soil productivity as they reduce soil erosion and maintain soil structure. Highly compacted soil is, however, a physically deteriorated condition affecting plant productivity under various land uses. Decrease in soil porosity that affects water content, hydraulic conductivity and gas permeability is a major disadvantage brought about by soil compaction. Loading of heavy equipment strongly compresses surface soil and/or subsoil in cropland, grassland and timber forests.

Mechanization of cultivation and harvest in Asian countries has increased, resulting in soil erosion and soil compaction due to tractor loading (Zhang *et al.*, 2006). Some studies of soils in rice/wheat cropping areas of India showed increases in compactness of subsurface soils as indicated by increased bulk density as high as 1.80 g cm^{-3}. This was due to the use of heavy machinery in conjunction with puddling activities (Sidhu *et al.*, 2014; Singh, Jalota and Sharma, 2009; Aggarwal *et al.*, 1995; Kukal and Aggarwal, 2003). Heavy machines for harvesting and skidding logs also compress soils considerably, and may be accompanied by rutting on the ground and by removal of organic matter (Kozlowski, 1999; Hattori *et al.*, 2013). The increase in the area of plantation forests in Asia, which has been especially rapid in China in recent decades (FAO, 2006), has led to compaction of soils through the use of heavy equipment for management. Livestock trampling is also a major cause of surface soil compaction in grassland and hilly grazing areas (Drewry and Paton, 2000). Soil compaction due to heavy grazing has led to severe land degradation in the extensive pastoral steppe regions of Mongolia and Inner Mongolia of China (Kruemmelbein, Peth and Horn, 2008). Where soils have been compacted in agricultural and forested areas, water surface runoff followed by soil erosion is a major threat. Soils in urban green areas are commonly compressed by human traffic and by vehicles for park management. This results in damage to plant roots and reduces productivity (Jim, 1998a) because in urban park vegetation over 50 percent of root density is concentrated in the top 50cm of the soil depth (Millward, Paudel and Briggs, 2011). Surface runoff on compacted soil generates flooding and delivers contaminants such as heavy metals and persistent organic pollutants into receiving water environments.

10.3.10 | Sealing and capping

Sealing and capping on the soil surface is mainly required in urban areas to construct roads and buildings. Although the impervious surface area (ISA) covers only 0.43 percent of the global land area at present (Schneider, Friedl and Potere, 2009), ISA is constantly on the increase, as shown by satellite image data and by measures of the constant increase in urban population (Elvidge *et al.*, 2007). More than half of the global population is now concentrated into cities. The Asia region has the largest ISA ratio in the world (Schneider, Friedl and Potere, 2009). China has the largest ISA in Asia followed by India, Indonesia, Japan and Bangladesh (Elvidge *et al.*, 2007). Increase in the ISA causes environmental issues such as the formation of urban heat islands (Changnon, 1992), increases in surface water runoff (Booth, 1991), and reduction of carbon sequestration due to reduction of the forested area (Milesi *et al.*, 2003).

Inter-regional differences in properties of sealed soils are not well documented. However, in general, construction processes of sealing affect soils physically and chemically due to disturbance of surface and subsoil by excavation and filling, and by addition of construction materials. In Japan, very large volumes of soil are excavated every year for land levelling, and much of this soil is carried away to other sites. Soils sealed by construction are compressed to enhance their physical strength, making them hugely compacted structures. Additives such as lime, which is used to enhance the sub-base strength of a road, make soils alkaline (Jim, 1998b). Pervious asphalt paving and inter-locking covers for light traffic roads are now recommended to drain rainwater by infiltration to the sub-soil. However, this infiltration treatment can make soil solution and drainage water alkaline. Cracks on a paved road surface due to heavy traffic may allow rainwater to infiltrate into the subsoil, reducing the strength of the roadbed and making the sub-soil alkaline.

10.4 | Major threats to soils in the region

10.4.1 | Erosion

Soil erosion is one of the major threats to soil quality in Asia. Soil erosion is the action of exogenic processes such as water flow and wind to move soil from its location. The processes inducing soil erosion vary with climate. Asia can be divided into several climate zones: tropical and subtropical in South Asia, humid subtropical and temperate in East Asia, semiarid in China, and arid in Mongolia and East Asia. Most regions of Asia are affected by the Asian-Australian monsoon which causes dry and wet seasons. Water erosion is the major type of erosion in the regions of South and East Asia with alternating dry and wet seasons. On the other hand, wind is the crucial driving force inducing soil erosion in the drier and desert areas.

Soil erosion by rainfall and surface water flow is generally affected by five factors: rainfall erosivity, soil erodibility, topography, surface coverage, and support practices. In humid regions, soil erosion is of little concern in well-established forests and in paddy fields. However, bare lands such as logged forests, construction areas and upland crop fields on slopes are exposed to a high risk of soil erosion. Annual soil loss in paddy fields has been generally reported in case studies to be lower than 1 tonnes ha^{-1} (Chen, Liu and Chen, 2012; Choi *et al.*, 2012; Kim *et al.*, 2013). By contrast, soil loss from upland crop areas on slopes is much greater – for example, 38 million tonnes ha^{-1} from fields in South Korea where no conservation practices were applied (Jung *et al.*, 2005). In semiarid regions, soil erosion is also of concern especially for slope areas with scant vegetation. In these areas, several hundred mm of rainfall in the rainy season can result in massive gully erosion. For these reasons, soil erosion is regarded as the most important threat to soil in Asia, especially for poorly-covered lands and bare soil.

Major threats of soil erosion by water are found in the hilly and mountainous landscapes of Indonesia. Natural conditions, anthropogenic influences on land cover and intensive land use make the steep and densely populated islands of Java, Sumatra and Sulawesi the most threatened areas. Approaches to coping with erosion problems range from engineering measures such as terracing, sediment pit construction and waterways improvement to vegetative measures including agroforestry approaches, contour strips and cover crops (Agus and Widianto, 2004).

In the Philippines, a monsoon country with high rainfall, erosion by water is one of the major causes of land degradation. In the 2013 Land Degradation Assessment final report by the Bureau of Soils and Water Management, the estimated annual soil loss for agricultural land for the whole country based on the 2003 Land Use System Map is about 61.8 tonnes ha^{-1} yr^{-1}. Paningbatan (1987) estimates that soil loss of about 10 tonnes ha^{-1} can be considered tolerable for Philippines conditions. In the 1950s, USDA (quoted by Schertz, 1983) established soil loss tolerance values for the Philippines at about 5 to 12 tonnes ha^{-1} yr^{-1} at soil bulk density of 1 200 kg m^{-3}. An analysis made by the Department of Environment and Natural Resources (DENR) on the state of the Philippine environment showed that, overall, 75 percent of total croplands are vulnerable to erosion of various degrees. To counter high rates of erosion, sustainable land management practices are being promoted. These include the application of various soil conservation and management strategies in highland agriculture as well as other technologies like agroforestry and multiple-storey cropping. The Philippines is a member of the UN Convention to Combat Desertification (UNCCD), and the Department of Agriculture and other government agencies, academia and non-government organizations aggressively pursue various programs. These approaches seek to engage farmers as partners in development rather than treat them as the cause of the problem.

In South Korea, deviation of annual precipitation was 251 mm yr^{-1} between 1981 and 2010 and rainfall erosivity ranged from 2 264 MJ mm ha^{-1} yr^{-1} hr^{-1} to 6 856 MJ mm ha^{-1} yr^{-1} hr^{-1} in the same period (Park et al, 2011). The national average rainfall erosivity was 4 276 mm ha^{-1} yr^{-1} with EI 30 data between 1973 and 1996 (Jung *et al.*, 2004) and was 4 147 MJ mm ha^{-1} yr^{-1} hr^{-1} with EI 60 data between 1981 and 2010 (Park *et al.*, 2011). The variation of rainfall is expected to increase with climate change based on the RCP scenario (CCIC, 2014), which implies that the probability of extreme rainfall erosivity would also increase in future. Soil erodibility in South Korea was 0.027 million tonnes hr MJ^{-1} mm^{-1} and ranged from 0.001 million tonnes hr MJ^{-1} mm^{-1} to 0.102 million tonnes hr MJ^{-1} mm^{-1} with soil series. The soil erodibility of paddy fields is the greatest at 0.036 million tonnes hr MJ^{-1} mm^{-1} followed by upland crop fields (0.026 million tonnes hr MJ^{-1} mm^{-1}) and forests (0.020 million tonnes hr MJ^{-1} mm^{-1}) (Jung *et al.*, 2004).

Differences in soil erodibility between land use types are affected by geological characteristics. Forests are located in mountains and upland crop fields are generally placed on lower slopes below the mountains. This means that forests and upland crop fields have been chronically exposed to past overland flow and soil erosion, which has resulted in their current status with less easily-eroded particles such as silt and very fine sand. By contrast, paddy fields in the bottomlands are formed from sediments with greater soil erodibility. However, actual annual soil loss in paddy fields has been generally reported to be lower than 1 million tonnes ha^{-1} in case studies because paddy is protected from rainfall by the water already ponded in the field and by the ridges which store water inside paddy fields (Chen, Liu and Chen, 2012; Choi *et al.*; 2012, Kim *et al.*, 2013). Levels of soil erosion are tolerable in well-managed forests and grasslands (Kitahara *et al.*, 2000; Lee, 1994). Where grass was sown in spring, soil loss on grasslands was found to reach only half of that from bare soil in the first year. Losses decreased abruptly after the second year, to only 3 percent or less of those on bare soil (Jung, 1998; Jung and Oh, 1993).

Soil loss from upland crop fields is much greater than from paddy fields and forests. The national average of soil loss in upland crop fields was 38 million tonnes ha^{-1} in South Korea (Jung *et al.*, 2005). A variety of conservation practices have been applied to reduce soil erosion including agronomic and engineering practices. On low slopes, agronomic practices such as mulching and grass strips can reduce soil loss by 80-90 percent. Engineering practices such as terraces, channels and drop spillways can reduce soil loss on steeper slopes (Jo *et al.*, 2009). Based on the Law of Water Quality Conservation and the Law of Soil Environment Conservation, soil erosion in South Korea has been monitored in severely eroded areas and best management practices have been proposed to regional governments and farmers.

10.4.2 | Soil organic carbon change

ASSOD describes soil organic carbon change as "a net decrease of available nutrients and organic matter in the soil". Not all Asian countries monitor the soil organic carbon (SOC) stock and its change. Even where soil properties are monitored, some data may not be appropriate for aggregation at the country scale as they lack certain parameters such as bulk density, or the number of observations may be insufficient. However, data compiled from the literature are adequate to draw a rough sketch of SOC change in the region. Piao *et al.* (2012) compared three methods to estimate SOC change 1990-2009 in five East Asian countries. Using an inventory-remote sensing model approach, they concluded that the sub-region was a net ecosystem carbon sink (+0.293 Pg C yr^{-1}). They found an estimated net SOC increase in the forest (+0.014 Pg C yr^{-1}), shrub land (+0.022) and cropland (+0.022) and SOC decrease in the grassland (-0.003).

China occupies the largest land area of the region (46 percent of the total). China reported SOC changes in the range -0.143 Pg C yr^{-1} to +0.094 Pg C yr^{-1} during 1980-2000. India occupies the second largest area (14 percent of the total). In India, it is estimated that forest accumulated SOC at the rate of +0.041 Pg C yr^{-1} over the one hundred year period 1880-1981. The relevant data are listed in Table 10.1. Overall, there is a tendency towards SOC accumulation in forested areas and towards a decrease in grassland areas. To confirm these findings and allow analysis to guide future decisions, more detailed and comparable datasets for both SOC stock and change should be compiled in the region.

Table 10.1 | Soil organic carbon change in selected countries in Asia

Sub-region/Country		Area M ha	SOC stock Pg C	SOC stock Mg C ha^{-1}	SOC change Pg C yr^{-1}	Reference
East Asia sub-region	Total 1990 - 2010	1156			0.055	Piao *et al.* (2012)
China	Total 1980 - 2000	871	89.6	102.9		Xie *et al.* (2007)
			86.8		-0.143	
	Total 1981 - 2000	760	85.8	112.9		Tian *et al.* (2011)
			87.6		0.094	
	Total 2000		101.9		0.029	Houghton and Hackler (2003)
	Total				0.075	Piao *et al.* (2009)
India	Forest 1880 - 1981	64.2	6.8	106.1		Chhabra *et al.* (2003)
			10.9		0.041	
	Total		47.5			Velayuthum *et al.* (2000)
	Total	329	24.0	73.1		Bhattacharyya *et al.* (2008)
Indonesia	Total	183	20.8	113.4		Shofiyati *et al.* (2010)
Japan	Arable land 1979 - 1998	5.4	0.44	81.8		Takata (2010)
		5.0	0.45	89.6	0.0001	
	Forest 1950s - 1990s	24.3	2.18	90.0		Morisada *et al.* (2004)
	Total	29.3	2.63	89.9		Takata (2010) and Morisada *et al.* (2004)

China is a vast country – 6 percent of the global land area and 46 percent of the area of the region. Estimates in China of the totoal SOC storage in the 0-100 cm soil profile show high variability, ranging from 50 to 183 Pg (Xie *et al.*, 2007). The reasons for this variability include uncertainty about the area of croplands, the quantity of soil profiles and the methods applied for scaling up from soil profiles to a national level. Using data from the Second National Soil Survey carried out in the 1980s, Xie *et al.* (2014) estimated areas and SOC stocks as follows: paddy lands 29.87 million ha and 2.91 Pg C; uplands 125.89 million ha and 10.07 Pg C; forest 249.32 milion ha and 34.23 Pg C; and grassland soils 278.51 million ha and 37.71 Pg C.

Farmland and forests were found to have SOC sequestration rates of 23.62 and 11.72 Tg C yr −1, respectively, resulting in 0.472 and 0.234 Pg SOC respectively being accumulated in these soils during the period 1980 to 2000. However, degradation of grassland depleted 3.56 Pg C over the same period. Thus during twenty years, a net amount of 2.86 Pg C was lost, approximately 3.4 percent of total SOC storage in China.

With an artificial neural network model to link SOC change to six parameters - latitude, longitude, elevation, soil type, land use type, and original SOC in early 1980s - Yu *et al.* (2009) estimated an increase of 260 Tg C occurred in Chinese cropland in the period 1980 to 2000 in the topsoils (0-20 cm). The increase of SOC content is mainly attributed to the large increase in crop yields and the increased residues retained in fields. By contrast, SOC storage in grassland is dwindling, especially in the northwest and southwest parts of China, mainly due to the degradation of grassland. Nationally, an area of 5.29 million ha grassland had degraded 1986-1999 (Han and Gao, 2005). Unlike the grassland ecosystem, SOC storage in forestland has increased (see also above). On the forested area of 249.32 million ha, estimates of carbon sequestration range from 234 to 304 Tg SOC in the period of 1980 to 2000, mainly attributed to forest expansion and regrowth (Zhou *et al.*, 2006; Xie *et al.*, 2007).

Other studies show conflicting results, with some showing recent Chinese soils as a net carbon sink. Tian *et al.* (2011) estimated SOC change using two process-based terrestrial ecosystem models with considering factors including climate change and land use change. They concluded that biomass and soils in China accumulated organic carbon at a rate of 0.121 and 0.094 Pg C yr^{-1} respectively from 1981 to 2000. Piao *et al.* (2009) adopted a bottom-up approach and showed consistent results (0.105 and 0.075 Pg C yr^{-1}, respectively) in the same period, while Houghton and Hackler (2003) reported net C loss from Chinese ecosystems (-0.008 PgC yr^{-1}) in 1990s, including loss in biomass (-0.028 Pg C yr^{-1}) and net C sink in soils (0.029 Pg C yr^{-1}).

Velayutham and Bhattacharyya (2000) estimated soil organic C stock in different soil orders and different agro-ecological regions in India (Sehgal and Abrol, 1992). For the top 1 m depth, they estimated soil organic C stock as 47.5 PgC, which is double the previous estimates of Dadhwal and Nayak (1993) and Gupta and Rao (1994). The trend may, however, be negative as crop residues are widely used as fuel and fodder and not returned to the soil, which would result in a decrease in soil organic content. In Bangladesh, the average organic content is said to have declined by half, from 2 percent to 1 percent, over the past 20 years (Bangladesh, 1992). For the Indian State of Haryana, soil test reports over 15 years show a decrease in soil carbon (Chaudhary and Aneja, 1991). Decreased organic content leads to: (i) degradation of soil physical properties, including water holding capacity, as has developed in India (Indian Council of Agricultural Research, personal communication); (ii) reduced nutrient retention capacity; and (iii) longer release of nutrients, including micronutrients, from mineralization of organic master. As a consequence of all these effects, there may be longer responses to fertilizer.

The threat to soil organic carbon (SOC) change in Indonesia's mineral and peat soils is mainly caused by deforestation, poor land management or intensive cropping, or by a combination of these factors (Hartanto *et al.*, 2003; Lal, 2004). In lowland rice grown on mineral soils, SOC tends to be maintained or increased because of anoxic condition (Kyuma, 2004). Land use types greatly influence soil loss. Annual crop based systems where soil conservation measures are not practiced are associated with high erosion

(Valentin *et al.*, 2008). For peatland, the threat to SOC occurs when peat forest is cleared and drained (Page, Rieley and Banks, 2011). The change from saturated to unsaturated conditions leads to the enhancement of aerobic microbial activities. This is the main factor in the SOC loss which occurs through the resulting accelerated microbial decomposition (Hooijer *et al.*, 2014; Agus and Subiksa, 2008). Peat fire is another cause of SOC loss. Maintaining a high water table is the key to reducing SOC losses through both these pathways. More details on the loss of soil carbon in Indonesia are included in the country case study section (10.5.2).

Permanent meadows and pastures area occupy 73 percent of the land area of Mongolia, while forest and arable land area account for 7 percent and 0.4 percent, respectively (FAO, 2012). During the 1990s and 2000s, the forest area decreased (loss of 8.19×102 km^2 yr^{-1}), and as a result forest biomass in Mongolia has decreased at a rate of 0.004 PgC yr^{-1} (Piao *et al.*, 2009). Li *et al.* (2005) reported net ecosystem exchange in a Mongolian steppe under grazing using the eddy covariance technique and suggested that the steppe was almost carbon neutral.

Forest occupies 69 percent of the land area of Japan, while arable land and permanent meadows and pastures area account for 12 percent and 2 percent, respectively (FAO, 2012). Total SOC stock in forest area was estimated separately by Morisada, Ono and Kanomata, (2004) and Ugawa *et al.* (2012) using a bottom-up approach but with different data sets. Morisada *et al.* (2004) used 3 391 soil profile data sampled from the 1950s to the 1970s and calculated the weighted average SOC stock as 90 Mg C ha^{-1} (0-30cm) and 188 Mg C ha^{-1} (0-100cm). Ugawa *et al.* (2012) compiled 4 km mesh (around 3 000 profiles) data sampled from 1999 to 2003 and estimated average SOC stock as 69.4 Mg C ha^{-1} (0-30cm). Since the Japanese forest area has changed little during the last 40 years, and has even increased slightly, the difference of SOC values between the two studies could be attributed to the difference in the methodology of sampling. Assuming the average SOC stock represented the total forest area (25.0×106 ha), total SOC stock in forest area would be in the range of 1.7 to 2.2 Pg C. Takata (2010) compiled soil survey data (1979-1998) to calculate SOC stock in Japanese arable soils, and reported that 0.44 Pg C in 1979 increased very slightly to 0.45 Pg C, mainly due to an increase of both area and stock per unit area in grassland.

10.4.3 | Soil salinization and sodification

As outlined above section (10.3.5), the threat of salinization/sodification in the region takes varying forms. In the semiarid and arid zones of central and west Asia, salt-affected soils are widely distributed. On the other hand, salt-affected soils are also developing in certain coastal areas in monsoon zones, caused mainly by salt water intrusion in South and Southeast Asia and by coastal tideland reclamation. Although the coastal area affected is relatively small, this could become a serious problem for lowland rice production.

In the GLASOD study, the region is estimated to have 42 million ha affected by salinization, nearly all of which is located in the dry zone. Of this salinized area in the drylands, there are estimated to be approximately 4 million ha in both India and Pakistan. Salinization is also a major problem on irrigated land: GLASOD estimates that 10 percent of irrigated lands in India are affected, 23 percent in Pakistan and 9 percent in Sri Lanka, although these percentages are probably overstated since some of the salinization results from saline intrusion into unirrigated land.

GLASOD provides estimates of areas subject to strong salinization. These numbers are important, as by definition they refer to land abandoned and taken out of cultivation. However, there is sometimes a wide difference between GLASOD figures for strongly saline soils and country estimates, although it should be noted that some of these include naturally occurring saline soils. For India, country estimates range between 7 and 26 million ha, all higher than the GLASOD value of 4 million ha. For Pakistan, there is better agreement; leaving aside three estimates of 9-16 million ha, the GLASOD and six country estimates lie in the range 4-8 million ha. Two apparently independent surveys, by the Soil Survey of Pakistan and the Water and Power Development Authority, show relative agreement at 5.3 and 4.2 million ha, respectively.

In Bangladesh, an extension inland of coastal soil salinity has been noted in recent years. Lower river flows, reduced by upstream abstraction for irrigation, have proved insufficient to dilute and displace sea water. In Sri Lanka, small areas of light salinization have appeared on irrigated lands of the Mahaweli scheme; the problem has not yet reached serious proportions, but needs to be monitored.

Estimates of the extent of saline soils need to be associated with the dates of survey. Through successful reclamation, the extent of saline soils has been reduced in some areas, particularly as a consequence of the series of Salinity Control and Reclamation Projects (SCARP) in Pakistan. For example in the Pakistan Punjab the area of waterlogged and saline soils, which had risen from 61 000 ha in 1960 to 68 000 in 1966, had been reduced to 23 000 ha by 1985 (Chopra, 1989).

Tideland reclamation projects have been carried out for centuries on the western coast of Korea. Records show that tideland reclamation in Korea began at the Ganghwa island in Gyeonggi-Do in 1235 (the 22nd year of King Gojong, Goryeo Dynasty). Reclamation in the early years was on a small scale, but has expanded over the years. Large scale modern reclamation projects started in 1960s as part of the national development program. Some of these projects have been as substitutes for the more than 20 000 ha of farmland which have been converted each year into industrial estates or other urban purposes. Since 1945, 75 738 ha of tideland have been reclaimed for paddy fields in 185 project areas by the Korean government and private companies (Park, 2001).

The main constraints to crop production on reclaimed tideland are soil salinity, a high water table with poor drainage, and an unfavorable soil chemical composition. Resalinization of the surface soil is caused by evapotranspiration during the dry season and by capillary rise of saline water from groundwater resources. High soil salinity in reclaimed tidelands needs to be managed by controlling the amount and quality of irrigation water (Jung, Joo and Yoon, 2002). Soil characteristics on reclaimed land change continuously as desalinization progresses. However, in general, newly reclaimed saline soils have poor chemical properties and weak physical soil characteristics of soil thickness, soil structure and water logging, so that it is difficult to grow crops economically (Park, 1991).

10.4.4 | Nitrogen imbalance

Balanced nutrient supply is essential for achieving high crop yields, but excessive or unbalanced nutrient inputs may pose risks to the environment, human health and ecosystems. Nutrient losses may occur via emission to the air or discharge to the water through runoff, leaching and erosion.

Nutrient inputs in Asia vary considerably amongst countries, districts and farming systems. Nutrient inputs tend to be higher than in other regions, and the trend may continue. FAO has predicted that for the coming four decades 60 percent of the world population increase will be in Asia (FAO, 2014). Feeding this rapidly growing population while minimizing harm to the environment is of great importance for both Asia and the world. Innovations in policy, science and farming practice are urgently needed to achieve this goal.

Nutrient cycles link agricultural systems to their societies and surroundings and create the need for decisions on tradeoffs. Inputs of nitrogen and other nutrients are essential for high crop yields, but downstream and downwind losses of these same nutrients diminish environmental quality and human well-being (Vitousek *et al.*, 2009). In this section, the issue of nitrogen imbalances in agriculture in Asia is highlighted as an important threat to soil.

One study found that nutrient balances differ among Asian countries, varying generally with the level of economic development (Vitousek *et al.*, 2009). The study examined six countries representing developing (China and India), developed (Japan and South Korea) and least developed countries (Laos and North Korea). China had the highest input of nitrogen (505 kg N ha^{-1} of arable land), which is nearly ten times levels applied in Laos (59 kg N ha^{-1}). On the other hand, the mean N output was 108 kg N ha^{-1} in China, while North Korea achieved half that level (44 kg N ha^{-1}). The N input was dominated by fertilizer application, especially in the booming economies of China and India, where the fertilizer N inputs accounted for 76 percent and 62 percent respectively of the total N inputs. In the least developed countries, such as Laos and North Korea, the N input mainly consists of animal manure, crop straw and biological N fixation. For developed countries, such as Japan and South Korea, the N inputs were more evenly contributed by fertilizer and manure. However, because of relatively high N input, higher N losses were observed in China, Japan and South Korea, compared with the least developed countries. The nitrogen use efficiency (NUE) was relatively high in Laos and North Korea, at the expense of depleting the soil N. The average soil N depletion rate ranged between 10–30 kg N ha^{-1} yr^{-1} in Laos and North Korea, while in China soil N accumulated by 26 kg N ha^{-1} yr^{-1}.

In the least developed countries, manures were considered as precious nutrient resource for crop production. This was also the case in China before the 1980s, when artificial fertilizer was not subsidized. However, from the 1980s on, fertilizer was more widely used than manure in China. The amount of fertilizer applied in China far exceeds rates in other countries in Asia, and even exceeds rates in the United States and EU (Ju *et al.*, 2009; Vitousek *et al.*, 2009). In China in 2005 lack of regulation led to more than half of manure being discharged untreated to water bodies (Ma *et al.*, 2010). By contrast, the environment is highly protected in Japan. More than 80 percent of manure is treated before being applied to crop land (Mishima, 2012). The N inputs and outputs also showed large variations between different crops. The highest nutrient inputs and accumulation in cash crop production were observed in China (Yan *et al.*, 2013). As both livestock and cash crop production are expected to increase considerably in Asia in the future as demand rises from a fast growing and increasingly urbanized population (FAO, 2014), N use is likely to increase but there may be a more intelligent and sustainable approach to N inputs and outputs.

Nitrogen use efficiency (NUE) is defined by the N output as crop production divided by the N inputs that are added through fertilizer, biological N fixation and N deposition (Ma *et al.*, 2010). In a study across Asia and the Middle East, national average NUE ranged from 131 percent in Laos to 19 percent in the United Arab Emirates.

The NUE was more than 100 percent in Laos, Nepal and Myanmar, because these countries relied to a great extent on recycled N inputs such as manure and crop straw. The NUE showed a reverse trend of N surplus among countries, decreasing from high N depletion countries such as Nepal and Laos to the high N surplus countries such as the United Arab Emirates and China. None of the crop production systems were sustainable in high N depletion or surplus countries. Although lower N losses and higher NUE were observed in the least developed countries, the continuous N depletion will limit crop yields and food production. In N surplus countries, the high levels of N accumulation may lead to higher N losses and consequently to serious environmental problems (Guo *et al.*, 2010; Liu *et al.*, 2013).

The average N losses varied among countries, from 23 kg N ha^{-1} yr^{-1} in Afghanistan to 327 kg N ha^{-1} yr^{-1} in China (Figure 10.3). This was due to the differences in N input rate, the crop production structure and the area of arable land. For a variety of reasons, large areas of arable land are not presently cultivated in Afghanistan, Iraq, Mongolia and Syria. The contribution of the different N loss pathways varied among countries, for example, ammonia emission contributed to 50 percent of the N losses in Philippines, but to only 22 percent of the N losses in Laos.

In conclusion, N inputs and outputs vary considerably amongst countries in Asia, with variations mainly attributable to levels of development and to policies. The nutrient imbalance in Asia could have a large impact on crop production and on the environment. These impacts may increase in the future with the further rapid development of livestock and cash crop production. Both the N imbalance and the N losses can be improved greatly without any sacrifice of crop yield. For example, balanced N fertilizer application and maximum manure N application standards have already proved effective in the EU (Velthof *et al.*, 2009). Recent studies show that crop yields in China can be increased by 20-30 percent with no increase in input of fertilizer simply by using ISSM technologies (Chen *et al.*, 2014). However, in the least developed countries increased use of fertilizer would greatly boost crop yields, as has happened in China during the past 30 years. For the developed countries, promoting mixed crop and livestock production systems could be the best choice to mitigate nutrient losses

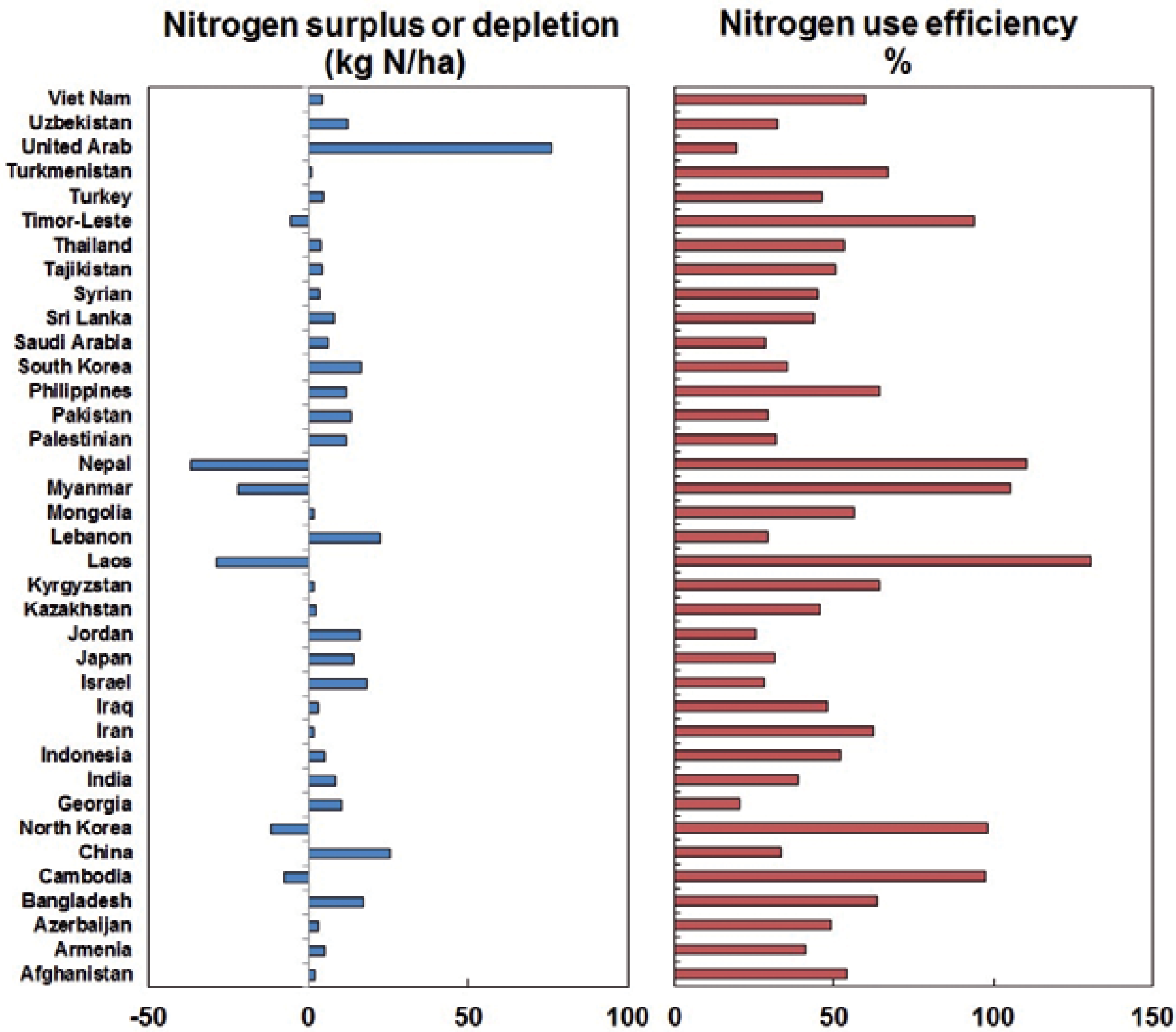


Figure 10.3 | Nitrogen surplus or depletion, and nutrient use efficiency in crop production in Asia and the Middle East in 2010.

10.5 | Case studies

10.5.1 | Case study for India

The area of India is 328.2 million ha, of which 141 million ha is under cultivation. The country is bordered by the Arabian Sea in the west, the Bay of Bengal in the east, and the Himalayas in the north. Physiographically, India is divided into four broad divisions: (i) Himalayan Range (Northern Mountains); (ii) Hill regions, Indian Peninsula and Eastern Plateau; (iii) the great Indo-Gangetic Plains and Coastal Plains; and (iv) the Islands (Singh, 1971). India is endowed with diverse climates and there are three distinct main seasons (rainy, winter and summer). The country is influenced by monsoonal type climate and rainfall. Annual rainfall varies from less than 100 mm in the cold desert area of Jammu & Kashmir, Lahaul Spiti and the Thar Desert of Rajasthan to over 11 000 mm in Cherrapunji in Meghalaya. Mean annual temperatures in the country vary from 8 to 28°C. Variation in the mean summer and mean winter temperature in the northern region is <10°C and in the south <5°C. In all, 20 Agro-Ecological Regions (AERs) have been identified, subdivided into 60 Agro-Ecological

Sub-Regions. Inceptisols are the dominant soils covering 39.75 percent of the total area, followed by Entisols (28.08 percent), Alfisols (13.55 percent), Vertisols (8.52 percent), Aridisols (4.28 percent), Ultisols (2.51 percent), Mollisols (0.4 percent) and others (2.92 percent) (Bhattacharyya *et al.*, 2013).

Degraded and wastelands of India

Velayutham and Bhattacharyya (2000) reported that the total area subject to soil degradation in India is 45.9 percent. Of this, 37.0 percent is affected by water erosion, followed by wind erosion (4.0 percent), salinization (2.2 percent), loss of nutrients (1.1 percent) and waterlogging (1.6 percent). Land not fit for agriculture (ice-caps, salt-flats, arid mountain and rock outcrops) constitutes 5.5 percent of the total area. About 27.5 percent of soils have no degradation problem and the remaining 9.8 percent of the area is classed as 'stable terrain', e.g. natural conditions. Recently the National Academy of Agricultural Sciences (NAAS) inventoried the soil degradation status of the country based on reconciliation of databases gathered from different organizations (Figure 10.4 and Table 10.2). This confirmed that soil erosion by water is the most serious threat, affecting 82.57 percent of the total area, followed by wind erosion (12.40 percent), acidic soils (17.94 percent), salt-affected soils (6.74 percent), waterlogged soils (0.88 percent) and mining and industrial waste (0.19 percent).

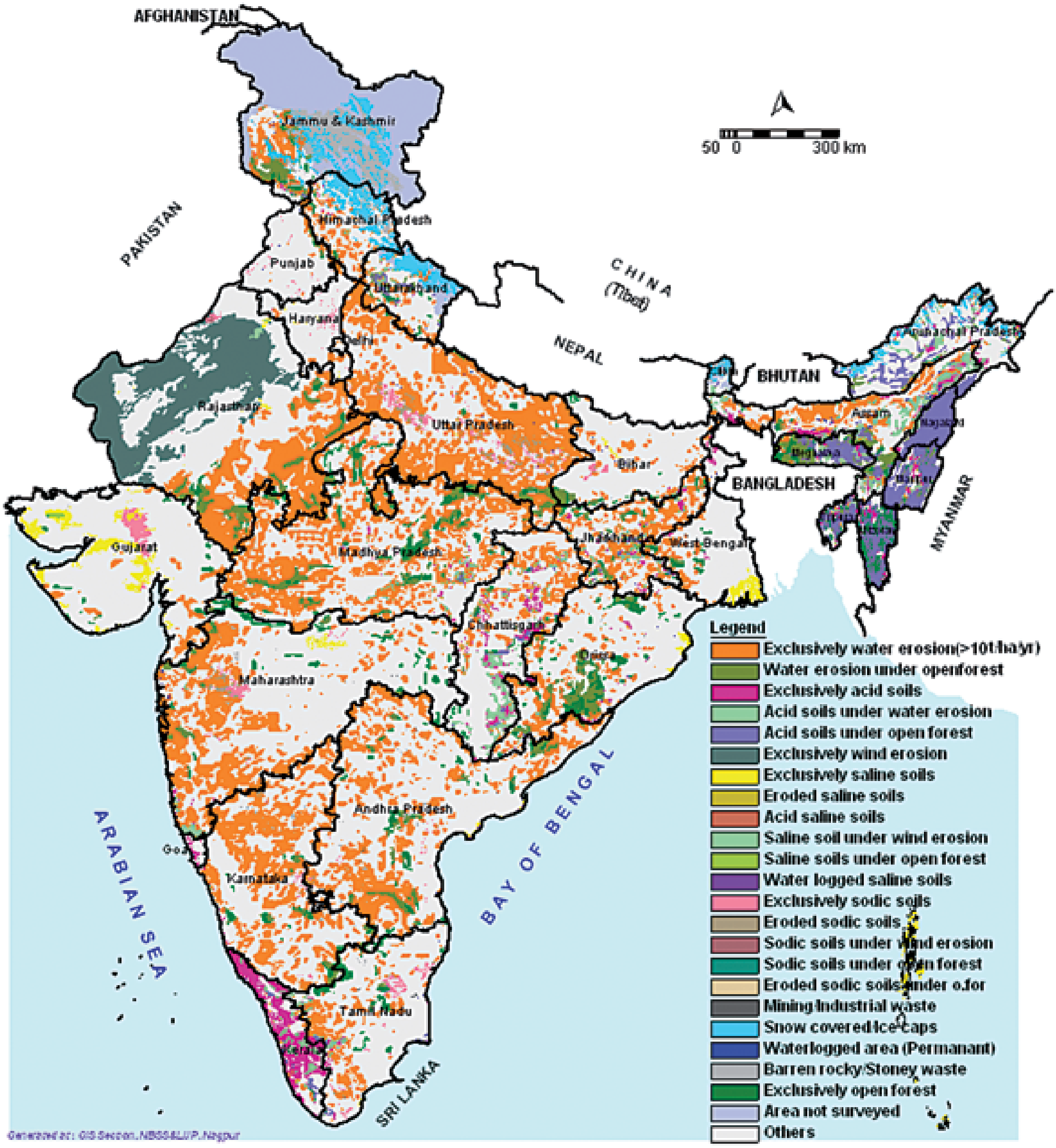


Figure 10.4 | Degradation and wastelands map of India.
Source: ICAR and NAAS, 2010.

Table 10.2 | Harmonized area statistics of degraded and wastelands of India.
Source: ICAR and NAAS, 2010.

Notes: FSI (1999) was used to exclude degraded land under dense forest; Unculturable Wastelands: Barren rocky/stony waste: 6 M ha, are the source for runoff water and building material; Snow covered/Ice-caps: 6 M ha, are best source of water and are not treated as wastelands
#For acid soils, areas under paddy growing and plantation crops were also included in the total acid soils
$Sub-surface waterlogging not considered.

Degradation type	Arable land (million ha)	Open forest (<40 percent canopy) (million ha)	Data source
Erosion			
Water erosion (>10 tonnes ha^{-1} yr^{-1})	73.27	9.30	Soil Loss Map of India–CSWCR&TI
Wind erosion (Aeolian)	12.40	–	Wind Erosion Map of India–CAZRI
Sub-total	**85.67**	**9.30**	
Chemical degradation			
Exclusively salt- affected soils #	5.44		Salt-Affected Soils Map of India, CSSRI, NBSS&LUP, NRSA and others
Salt-affected and water eroded soils	1.20	0.10	
Exclusively acidic soils (pH< 5.5) #	5.09	-	Acid Soil Map of India NBSS&LUP Acidic (pH < 5.5) and water
Acidic soils (pH< 5.5) and eroded soils	5.72	7.13	
Sub-total	**17.45**	**7.23**	
Physical degradation			
Mining and industrial waste	0.19		Wasteland Map of India–NRSA
Waterlogging (permanent surface inundation) $	0.88		
Sub total	**1.07**		
Total	**104.19**	**16.53**	
Grand total (arable land and open forest)	**120.72**		

10.5.2 | Case study for Indonesia

Topography, climate, soil parent materials and anthropogenic factors determine the various types and degree of soil threats in Indonesia. These threats include soil carbon depletion both for mineral and organic soils, erosion by water, soil contamination, soil acidification and nutrient imbalance. Soil organic carbon depletion in peat is the most significant threat.

Soil erosion threats are found in almost all hilly and mountainous landscapes of the Indonesian archipelago due to the very high (>2000 mm) annual rainfall over most of the area (83 percent). Most other sloping areas with lower rainfall are also affected by erosion due to the high intensity of the monsoonal rainfall during the rainy season (Agus, Amien and Sutono, 2002). Besides causing erosion, the high rainfall also leads to leaching of basic cations and hence to soil acidification. The main problems associated with managing acid soil are low pH, P-fixation, low basic cation concentration, low cation exchange capacity (CEC) and toxicity of soluble Al and Fe.

The total area of Indonesian acid upland soils (pH <5 and <50 percent base saturation) is about 102.8 million ha (Puslitbangtanak, 2000). Relatively low cost technologies are widely available for rectifying soil acidity problems. These include liming, organic matter application, balanced fertilization, and selection of acid tolerant crops (Kamprath, 1984; von Uexküll and Mutert, 1995). Nutrient imbalance is related to insufficient or unbalanced fertilizer use (Tan, Lal and Wiebe, 2005). The use of nitrogen fertilizer in various farming systems in Indonesia has been continually increasing since the late 1960s, reaching 2.4 Mg yr^{-1} in 2012. However, the application of P and K has not followed the same trend (IFADATA, 2007). This is in part because N fertilizers have been easier to obtain, they are cheaper, and they give a more rapid crop response to the farmer. For intensive cultivation of vegetable crops, over-fertilization has led to accumulation of N, P, Ca, Mg and Na, but to negative balances of K and Si (Husnain, Masunaga and Wakatsuki, 2010; Widowati *et al.*, 2011). K deficiency was also reported in newly-developed rice fields (Sukristiyonubowo, Nugroho and Ritung, 2012). For low input systems, nutrient removal through harvest is seldom sufficiently replenished by proper fertilization.

Soil organic carbon depletion

Depletion of soil organic carbon (SOC) is considered as one of the major threats. In extreme cases, it contributes to the physical, chemical and biological degradation of the soil. The rate of SOC depletion in mineral soils is relatively small. In peat soils, organic C is the main soil constituent and the rate of depletion is high for drained organic soils (IPCC, 2014; Agus, Hairiah and Mulyani, 2011).

The loss of carbon from mineral soil

Research findings differ on the trend of soil organic matter depletion in Indonesia (Agus *et al.*, 2013). One study (Murty *et al.*, 2002) suggested that SOC stock decreased by about 30 percent when forest is converted to continuous agricultural production; forest transformed to degraded land would lose 50 percent of its C stock, while forest converted to plantation would lose about 30 percent of its C stock. Another study found that SOC stocks in the 0–15 cm soil profile of Dipterocarp forest in Sumatra decreased by 48.1 Mg ha^{-1} when the forest degraded to Imperata grassland. However, Imperata grasslands are not necessarily associated with low SOC. In some cases, this grassland may have similar C content to forest (Santoso *et al.*, 1997). For Sumatra, van Noordwijk *et al.* (1997) found some decrease in SOC when forest is converted to cropland. Tanaka *et al.* (2009), on the other hand, did not observe significant differences in SOC between secondary forests, 9 and 19 year old oil palm plantations and a 30 year old rubber plantation.

Management systems can restore SOC. Under intensive management, it was found that in the first and second cycles of oil palm, organic matter increased 32 percent and SOC 15 percent in the 0-45 cm soil layer, relative to secondary forest as the initial land use (Mathews, Tan and Chong, 2010). Regular application of palm fronds increased SOC, especially in the 20 percent of the plantation area receiving an equivalent of 4.8 Mg C ha^{-1} yr^{-1} from palm fronds (Haron *et al.*, 1998). If degraded land is converted to plantation, it was found to gain about 30 percent SOC stock (Murty *et al.*, 2002; Germer and Sauerborn, 2008). In East Kalimantan, natural regeneration from Imperata grassland to secondary forest increased soil carbon content by 14 percent, from 14.5 g kg^{-1} to 16.5 g kg^{-1} (van der Kamp, Yassir and Buurman, 2009). In Java, agricultural top soil with continuous rice cropping accumulated more than 1.7 Tg C per year over the period of 1990-2000 (Minasny *et al.*, 2012). Kyuma (2004) also suggested that SOC in newly established paddy rice soils tends to increase with time. Strategies to increase the soil carbon pool include soil restoration and woodland regeneration, no-till farming, cover crops, nutrient management, manuring and sludge application, improved grazing, water conservation and harvesting, and efficient irrigation (Lal, 2003).

The loss of carbon from organic soils

Indonesian peatland is estimated to cover about 14.9 million ha (Ritung *et al.*, 2011). It is a massive store of carbon, storing around 27 Pg C (Agus *et al.*, 2013). This huge C store is formed under saturated condition in concave areas with the annual rate of peat formation in the range of 0-3 mm thickness (Noor, 2001) or equivalent to zero to about 1.5 Mg C ha yr^{-1} carbon accumulation (Agus and Subiksa, 2008; Parish *et al.*, 2007). This relatively slow process has led to the formation of peat domes that began between 6 800 and 4 200 years ago (Andriesse, 1994). Some formations may be as old as 26 000 years (Page *et al.*, 2002).

Carbon stock in peat ranges from 420 to 820 Mg C ha^{-1}. Peat depths range from 0.5m to over 10 m (Agus, Hairiah and Mulyani, 2011) and the peat C stock is strongly determined by peat thickness (Warren *et al.*, 2012). Under saturated natural conditions, peat C is slowly emitted in the forms of CO_2 and CH_4 due to anaerobic microbial activities. As peat forest is cleared and drained, the peats become unsaturated and CO_2 emission escalates at a pace far exceeding the rate of sequestration. Currently, from the 14.9 million ha classified as peatland, forested areas amount to just over half (52 percent), shrub cover is about 21.7 percent, and the rest is under agriculture or settlements. Peat soil C loss runs in parallel to the level of local development. Peatland areas of Sumatra and Kalimantan have been drained on a wide scale, and hence are fast losing carbon. On the other hand, Papua peatlands are mostly conserved (Figure 10.5).

The major processes of carbon loss from drained peatland are (i) peat decomposition under aerobic condition and (ii) peat fire. Natural phenomena such as lengthy droughts aggravate peat fire risks (Page *et al.*, 2002; IPCC, 2014; Parish *et al.*, 2007; Agus and Subiksa, 2008; Wosten *et al.*, 2008).

The literature varies in its estimates of peat carbon loss through decomposition. One source (IPCC, 2014), based on research data from Malaysia and Indonesia, presented peat CO_2-C emission factors based on land cover classes. In this study, degraded forest is expected to emit as much as 5.3 Mg CO_2-C ha^{-1} yr^{-1}, while various agricultural and forestry uses emit a wide range of CO_2-C as shown in Table 10.3. The 95 percent confidence interval was very wide for each land cover class, which suggests the need for site specific emission factors. A case study in Riau Province (Husnain *et al.*, 2014) showed insignificant differences in peat emissions under an oil palm plantation, an Acacia plantation, a secondary forest and a rubber plantation. The emission rates were 18.0 ± 6.8; 16.1 ± 5.3; 16.6 ± 6.8; 14.2 ± 4.6 Mg CO_2-C ha^{-1} yr^{-1}, respectively. For bare land sites, the rates measured lay between 15.2 ± 8.2 and 18.2 ± 6.5 Mg CO_2-C ha^{-1} yr^{-1}. The findings of other studies (Marwanto and Agus, 2014; Dariah, Marwanto and Agus, 2014) were similar to those of the IPCC (2014), with emissions from oil palm plantations on peat soils of about 10 to 11 Mg CO_2-C ha^{-1} yr^{-1}.

A study in Central Kalimantan Province revealed a linear relationship between average water table depth and peat CO_2 emission. Degraded drained forest was found to emit about 2.7 Mg CO_2-C ha^{-1} yr^{-1} in areas with zero mean water table depth, and about 15 Mg CO_2-C ha^{-1} yr^{-1} for areas where average water table depth was 1 m (Hooijer *et al.*, 2014). Regardless of the variations found amongst the studies, the values recorded demonstrate rapid carbon depletion in drained peatland.

Peat fire is another important process that may cause a huge amount of carbon loss in a short period of time, and the literature is in agreement on the importance of peat fire as one of the main causes of peat carbon loss (Page *et al.*, 2002; Hooijer *et al.*, 2014). However, determination of peat C loss through peat fire is a future research challenge, especially with regards to gathering firm data, for example on the volume (area and depth) of burnt scars (IPCC, 2014).

Table 10.3 | Emission factors of drained tropical peatland under different land uses and the 95 percent confidential interval. Source: IPCC, 2014.

1) Emission of primary peat forest is assumed to be zero.
2) Some confidence intervals contain negative values because calculation was based on error propagation of uncertainties. However, all underlying CO_2 fluxes were positive.

Land cover type[1]	Emission ($Mg\ CO_2\text{-}C\ ha^{-1}\ yr^{-1}$)	95 percent Confidence level[2] ($Mg\ CO_2\text{-}C\ ha^{-1}\ yr^{-1}$)	
Drained forest land and cleared forest land (shrubland)	5.3	-0.7	9.5
Plantations, drained, unknown or long rotations	15	10	21
Plantations, drained, short rotations, e.g. acacia	20	16	24
Plantations, drained, oil palm	11	5.6	17
Plantations, shallow-drained (typically less than 0.3 m), typically used for agriculture, e.g. sago palm	1.5	-2.3	5.4
Cropland and fallow, drained	14	6.6	26
Cropland, drained, paddy rice	9.4	-0.2	20
Grassland, drained	9.6	4.5	17

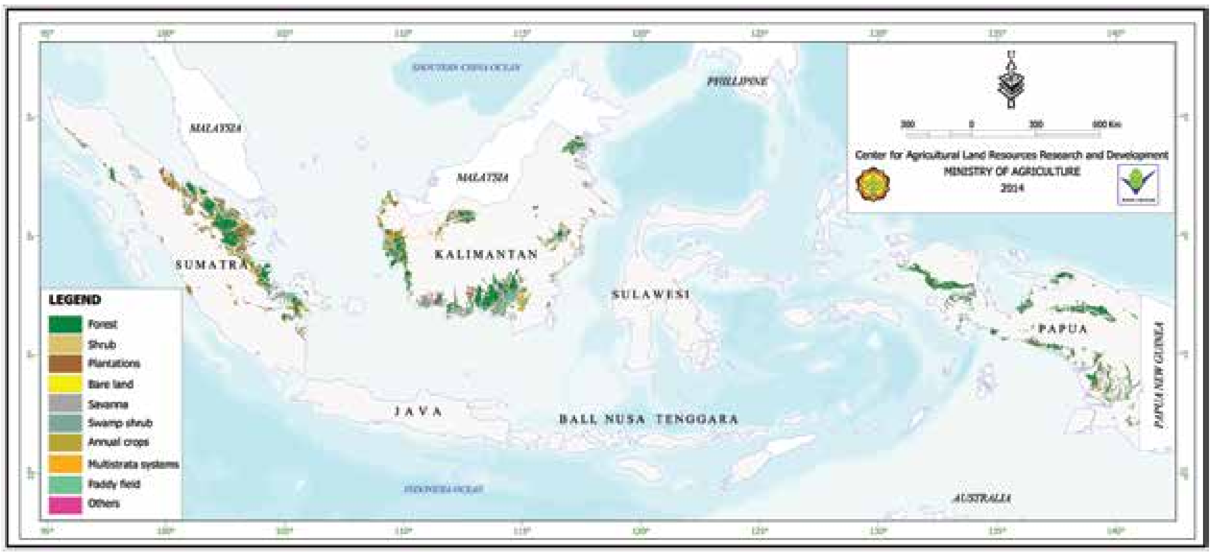


Figure 10.5 | Indonesian peatland map overlaid with land cover map as of 2011.
Source: Wahyunto et al., 2014.

10.5.3 | Case study for Japan

The islands which make up Japan are located in the one of the most active parts of the Circum-Pacific Ring of Fire. The major islands are Hokkaido, Honshu, Shikoku and Kyushu. There are 110 active volcanoes in Japan (16 volcanoes in Hokkaido, 38 volcanoes in Honshu and 11 volcanoes in Kyushu). Large quantities of tephras from volcanoes have been deposited on the Pleistocene terraces, constituting a main parent material of Japanese soil (Andosols). Japan's total land area is about 378 000 km^2. About 72 percent of Japan's land area is mountainous, and rivers are characterized by their steep gradients and relatively short lengths. About two-thirds of the total land area consists of forest. The plains cover only about 28 percent of the total land area. Most plains are located along the seacoast. Arable lands account for 12.1 percent of the total land surface, mainly distributed in the plains. The climate of Japan is influenced by a monsoonal flow that carries moist air from the Indian Ocean and Pacific Ocean. In general, Japan has four distinct seasons: spring (March to May), summer (June to August), autumn (September to November) and winter (December to February).

Japan's arable land covers 4.5 million ha, with paddy fields accounting for just over half the area (54 percent). Grey lowland soils (Fluvic Hydragric Anthrosols or Gleyic Fluvisols (FAO/IUSS/ISRIC, 2006) comprise the largest cultivated soil area; followed by Gley soils (Gleyic Fluvisols), Andosols (Aluandic or Silandic Andosols), Brown forest soils (Haplic Cambisols), Brown lowland soils (Haplic Fluvisols), and Wet Andosols (Gleyic Andosols). Urban sprawl and other changes in land use (including abandonment of cultivation) led to a shrinking of the agricultural land area by about 1 million ha between 1973 and 2001. Urbanization and consequent soil sealing advanced into flat lowland areas, largely into paddy fields on Grey Lowlands soils and Gley soils. In addition, loss of Andosols to expanding urbanization was widely observed over the flat upland fields in the middle part of Hunshu islands. By contrast, upland fields on steep slopes in the western part of Japan, largely on distributed Brown Forest Soils, were simply abandoned (Takata *et al.*, 2011b).

Soil organic carbon change

Spatio-temporal variations in soil organic carbon (SOC) content in arable land were evaluated by both model-based (Yagasaki and Shirato, 2014) and monitoring-based (Takata, 2010) approaches. In the model-based approach, SOC stock change was simulated using the original Rothamsted Carbon model (Coleman and Jenkinson, 1996) and two modified Rothamsted Carbon models (Shirato, Yagasaki and Nishida, 2011; Takata *et al.*, 2011a). The rate of change in the total SOC stock in Japanese agricultural lands evaluated with 10 year intervals was estimated to be -0.95 Tg C yr^{-1} between 1980 and 1990. A greater loss of SOC, equal to -1.06 Tg C yr^{-1}, was found subsequently for the period from 1990 to 2000.

An agricultural soil monitoring project named 'Basic Soil-Environmental Monitoring in Japan' has been conducted since 1979. This soil monitoring project has taken readings at repeated five year intervals at about 20 000 fixed points. The rate of change in the total SOC stock evaluated by this monitoring was 2.3 Tg C yr^{-1} from 1979 to 1989. The project found that SOC increased in arable land. However, the monitoring detected a greater loss of SOC for a later period (1989-1998), equal to -2.3 Tg C yr^{-1}. During this period, the agricultural land area decreased from 5.4 to 5.0 million ha. At the same time, soil carbon content in arable land gradually rose from 88 to 90 tonnes C ha^{-1}. These results indicated that the decline of SOC stock in Japanese arable land 1989-1998 was mainly influenced by the fluctuation in the arable land area.

Spatial variation of SOC content in forest soils has been monitored since 2005. The mean SOC content of forest soil at 0-30cm was 69.4 tonnes C ha^{-1} (Ugawa *et al.*, 2012), lower than the value of 90 tonnes C ha^{-1} in arable land.

Heavy metal contamination

Rapid industrialization in Japan during the 1960s polluted arable soil with heavy metals such as cadmium (Cd), Copper (Cu), and Arsenic (As). There were four main pollution sources: mining activity, factories and incinerators, fertilizer, and precipitation and irrigation water (Makino, 2010). In 1970, the Japanese government enacted the Agricultural Land Soil Pollution Prevention Law to regulate heavy metal pollution. The law designated Cd, As, and Cu as hazardous substances to be regulated. The allowable limitation of Cd was set in terms of the Cd concentration in rice grains (1 mg kg^{-1}). The allowable limitations on As and Cu were set to 15 (1M HCl soluble) and 125 (0.1M HCl soluble) mg kg^{-1} soil, respectively. The amount of bioavailable Cd in soil is affected by many factors, so setting an allowable concentration in terms of the soil Cd content is impractical (Asami, 1981). The area of polluted arable land was assessed as 7 592 ha (Cd, 7 050 ha; Cu, 1 405 ha; As, 391 ha). About 92 percent of the total polluted area has been remediated by uncontaminated soil dressing (MOE, 2014).

Radioactive Cs contamination

As a result of the accident at the Fukushima Dai-ichi Nuclear Power Station (FDNPS) operated by the Tokyo Electric Power Company, radioactive cesium (Cs) was released into the surrounding environment. To determine the extent of decontamination required in arable land and to consider management options, the Ministry of Agriculture, Forestry and Fishery (MAFF) surveyed and measured soil Cs concentrations in 3 461 agricultural fields, and used these data to construct a distribution map of radioactive Cs concentration in agricultural soil in eastern Japan (Takata *et al.*, 2014). The distribution map of radioactive Cs concentration in agricultural soil is shown in Figure 10.6.

Contamination level 4 (>25 000 Bq kg^{-1}) was observed only in a 20 km evacuation area (EA) surrounding FDNPS (EA-20km) and the Deliberate Evacuation Area (DEA); 77 percent of the level 4 areas were observed in the EA-20km. Farmers of level 4 contaminated fields were advised to solidify their topsoil with a fixation

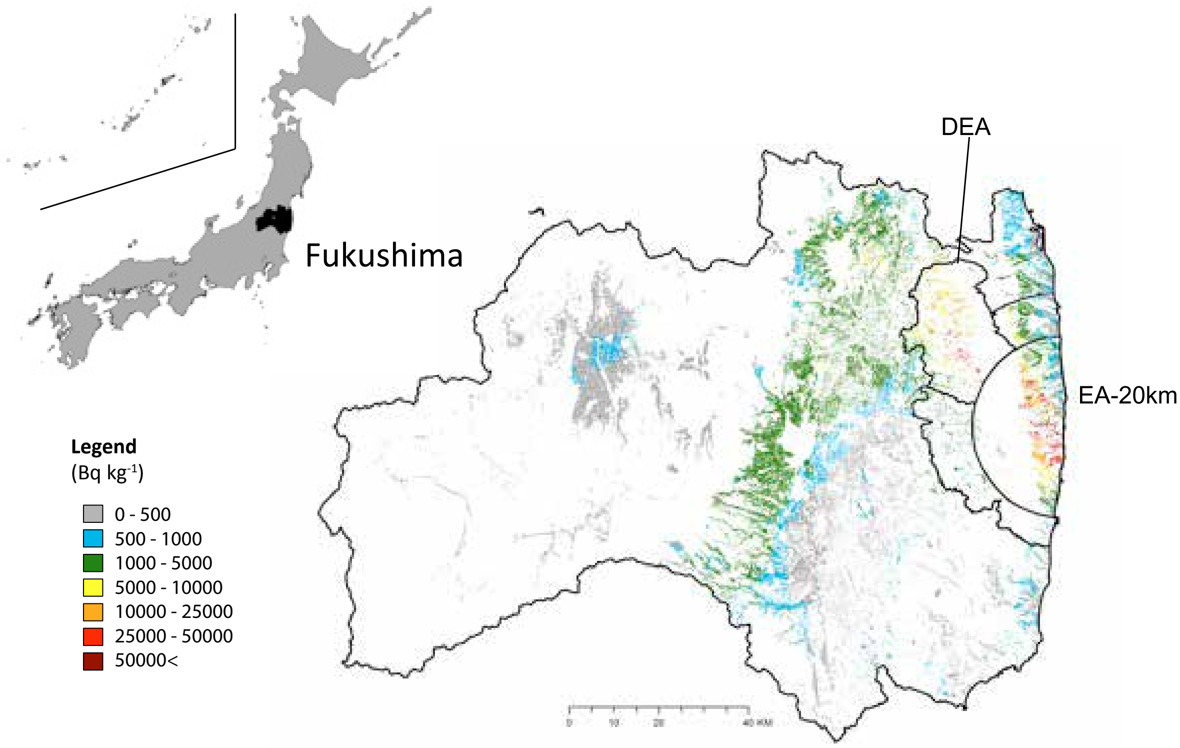


Figure 10.6 | Distribution map of radioactive Cs concentration in soil in Fukushima prefecture (reference date of 5 November, 2011). Source: Takata et al., 2014.

agent to prevent scattering of contaminated soil during a topsoil removal operation. Contamination level 3 (10 000–25 000 Bq kg^{-1}), which indicated a need to remove topsoil, was distributed in only the EA-20km and DEA, with more than half (55 percent) in the EA-20km. Eighty percent of fields with contamination level 2 (5 000–10 000 Bq kg^{-1}) were distributed in the evacuation zone, with the remaining 20 percent distributed in the non-evacuation zone in Fukushima Prefecture. Paddy fields with contamination level 2 (2 100 ha) have three options for decontamination: topsoil removal, fine-textured topsoil removal using water, and topsoil burying. Upland fields, orchards, and meadows that are at contamination level 2 (1 200 ha) have two options for decontamination: topsoil removal and topsoil burying.

Nutrient imbalance

The soil surface nitrogen (N) and phosphate (P) balance in Japanese arable land has been improving (Mishima, Endo and Kohyama, 2010a; 2010b). These values serve as the index of the impact of arable land on the environment and of the farm-gate balance of nutrients. The soil surface N (and P) balance is defined as the total N (P) input (N kg ha^{-1}) minus the total N (P) output (N kg ha^{-1}). Chemical fertilizer application in Japan declined continuously during the period from 1985 to 2005. The application rate of livestock manure also peaked in 1990 and declined thereafter. Crop production, however, remained constant during this period. Between 1985 and 2005, the surplus N and P (positive value of soil surface N balance) declined from 89.9 to 49.3 kg N ha^{-1} and from 153 to 105 kg P ha^{-1} respectively (Mishima, Endo and Kohyama, 2010a, 2010b). However, this trend was not consistent at the regional level because organic amendment applications were largely related to the availability and movement of livestock excreta (Mishima, Endo and Kohyama, 2010a) and to soil type (Leon *et al.*, 2012). High surplus P and low crop P uptake compared with N, P input for crop production could be reduced. This limited negative environmental effects such as eutrophication of soil and water and conserved limited P resources.

The Basic Soil-Environmental Monitoring Project found excess soil Ca in paddy fields, upland fields and orchards during 1979-1998 (MAFF, 2008), but also a gradual increase in soil Mg deficit over the same period. Thus the balance of Ca and Mg has been deteriorating in Japanese arable land. In addition, the soil pH of paddy fields gradually decreased from 5.8 to 5.7 during 1979-1998 (MAFF, 2008).

Soil erosion

Spatial estimation of soil loss from arable land at national scale was carried out using the Universal Soil Loss Equation (USLE) and environmental inventories (Kohyama *et al.*, 2012). Hourly rainfall and runoff factors (R: resolution 1 km) were calculated by the amount of precipitation analysed by radar-AMeDAS. Topographic factors (LS) are shown in Figure 10.7, calculated using a digital elevation model (resolution 50 m) and ALOS satellite imagery. The soil erodibility factor (K) of arable land was calculated using the physico-chemical soil properties (soil texture, soil organic matter content, etc.) as measured in the Basic Soil-Environmental Monitoring Project (Taniyama, 2003). The K factor of arable land was determined by soil series groups. The K factor was relatively higher in clayey lowland soil group than in humic Andosol groups. The cover and management factor (C) of each crop was determined by Taniyama (2003), and was delineated using the agro-environmental census data map (Kohyama *et al.*, 2003).

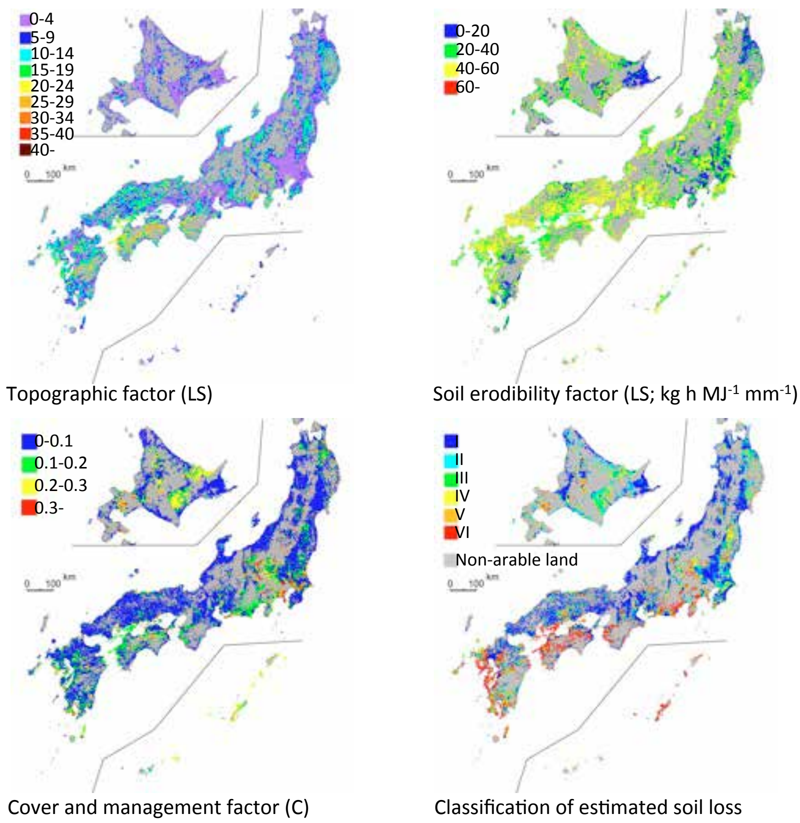


Figure 10.7 | Distribution map of the parameters of USLE and classification of estimated soil loss. Class I: less than 1 tonnes ha^{-1} yr^{-1}; Class II: 1-5 tonnes ha^{-1} yr^{-1}; Class III: 5-10 tonnes ha^{-1} yr^{-1}; Class IV: 10-30 tonnes ha^{-1} yr^{-1}; Class V: 30-50 tonnes ha^{-1} yr^{-1}; Class VI: more than 50 tonnes ha^{-1} yr^{-1}.
Source: Kohyama et al., 2012.

Loss of soil in Japanese arable land was categorized into six classes: Class I; less than 1 tonnes ha^{-1} yr^{-1}, Class II; 1-5 tonnes ha^{-1} yr^{-1}, Class III; 5-10 tonnes ha^{-1} yr^{-1}, Class IV; 10-30 tonnes ha^{-1} yr^{-1}, Class V; 30-50 tonnes ha^{-1} yr^{-1}, Class VI; more than 50 tonnes ha^{-1} yr^{-1}. The proportion of soils in these classes was: 43 percent in Class I; 18 percent in Class II; 9 percent in Class III; 12 percent in Class IV; 5 percent in Class V; and 14 percent in Class VI. Highly erodible zones were mainly distributed in areas of western Japan which are characterized by complex topography and heavy precipitation.

10.5.4 | Case study of greenhouse gas emissions from paddy fields

Rice is the staple crop for the majority of the world's population. In Asia, rice cultivation areas roughly account for 89 percent of the global total (Yan, Akimoto and Ohara, 2003). While rice production is thus vital for feeding the world's population, it is also an important source of greenhouse gas emissions, notably methane (CH_4) and nitrous oxide (N_2O). CH_4 is converted from substrate by methanogenic bacteria in strictly anaerobic environments, while N_2O is an intermediate production of nitrification and denitrification. Both of these two gases possess considerably greater infrared absorbing capability than carbon dioxide (CO_2) on a mass basis: 25 times for CH_4 and 298 times for N_2O.

Using the tier 1 method described in the 2006 Intergovernmental Panel on Climate Change (IPCC) Guidelines for National Greenhouse Gas Inventories (IPCC, 2006), and country-specific estimates of rice harvest area and data on agricultural activities, Yan and colleagues estimated that global CH_4 emission for 2000 was 25.6 Tg CH_4 yr^{-1}, with a 95 percent uncertainty range of 14.8 to 41.7 Tg CH_4 yr^{-1}, considerably lower than earlier estimates (Yan *et al.*, 2009). Rice paddies in monsoon Asia countries contributed far and away the largest share of these emissions, estimated at 23.7 Tg CH_4 yr^{-1}. China, with an amount of 7.41 Tg CH_4 yr^{-1}, was estimated to be the largest CH_4 emission country, followed by India, Bangladesh, Indonesia, Vietnam, Myanmar and Thailand. The areas with the greatest emission intensity were the delta regions of large rivers in Bangladesh, Myanmar and Vietnam, the island of Java in Indonesia, central Thailand, southern China and the southwestern portion of the Korean peninsula (Figure 10.8).

CH_4 emission from rice fields is the net result of three processes: production, oxidation and transport. Three main factors affect one or more of these processes: organic amendment, the water regime during the rice-growing season, and water status in pre-season. Appropriate agricultural management of organic amendments and the water regime should therefore be promoted to mitigate CH_4 emissions. Techniques could include off-season straw incorporation and midseason drainage (Yan *et al.*, 2009).

Unlike CH_4, N_2O emission from rice paddies has been found to be much lower than from upland crops, because under a flooded environment, nitrification is weak and denitrification proceeds to the end step with N_2 as the dominant product. The majority of N_2O emissions from rice paddies usually occur shortly after the mid-season drainage and nitrogen fertilizer additions (Yan *et al.*, 2000). The availability of substrates and soil moisture condition are critical controlling factors for N_2O emission since they affect the activity of nitrifiers and denitrifiers. The fertilizer-induced N_2O emission factor for rice fields averages approximately 0.3 percent, probably fluctuating dependent on the water regime (Akiyama, Yagi and Yan, 2005). Mitigation of N_2O emissions from paddy fields should also not be overlooked because of the excessive nitrogen fertilizer consumption by rice production. Since a tradeoff relationship between CH_4 and N_2O emissions from rice paddies is frequently observed, any mitigation options pursued should take their comprehensive global warming potential fully into account.

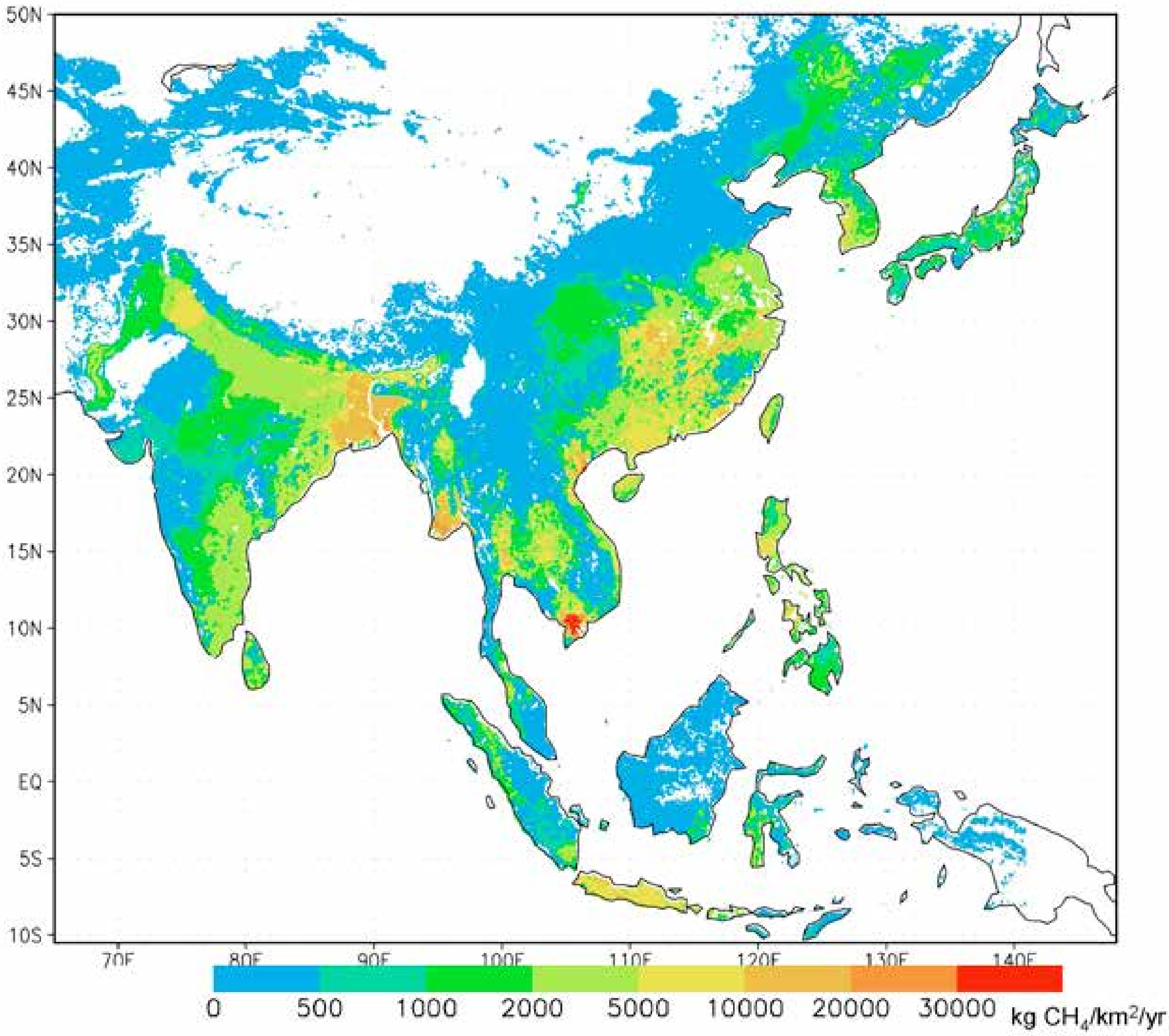


Figure 10.8 | Estimate CH_4 emission from rice paddy in Asia.
Source: Yan et al., 2009.

10.6 | Conclusion

In Asia, management of land and water resources has been identified as one of the priority ways to achieve sustainable food security by raising land productivity, reversing land degradation and water loss, and increasing biodiversity and the quality of the environment. Asian countries have also committed themselves to strengthening regional cooperation and national capacities to develop a more integrated approach to the management of natural resources. An integrated approach is needed to improve the ability of countries to plan and monitor the better use and management of their land resources to increase agricultural productivity while maintaining land and environmental quality.

However, since the GLASOD and ASSOD projects of the 1980s and 1990s, no extended assessment of the status of soil resources has been carried out in the region. There have been extensive scientific communications amongst experts in the region, including the activities of the East and Southeast Asia Federation of Soil Science Societies (ESAFS). Based on the above finding, a provisional assessment is made of the status and trend of the 10 soil threats in order of importance for the region. At the same time an indication is given of the reliability of these estimates (Table 10.4).

However, a number of the country reports that contributed to this chapter emphasized that rapid socio-economic change and resulting changes in land use and its management, as well as climate change, have had great impacts on the soil resources in Asian countries. Therefore, there is a need to conduct a new and extensive assessment of changes in soil resources in the region.

Responding to this need, a regional conference on soil information was held in Nanjing, China, on February 2012 to share the latest soil information and knowledge about advanced science and technologies on soil resources in Asian region. The conference recognized the benefits to be gained from further sharing of information and data on soil surveys, soil mapping and capacity development. The conference saw the establishment of the initial Asia Soil Partnership (ASP) and the signing of the Nanjing Communiqué (GSP, 2012), which put the following goals as the priorities:

1. sharing and transferring soil knowledge and new technology within and beyond the region
2. providing soil information to all those with an interest in the sustainable use of soil and land resources
3. building consistent and updated Asian soil information systems and starting to contribute to the global soil information system through initiatives such as GlobalSoilMap.net
4. training new generations of experts in soil science and land management

After the endorsement of the global Plans of Action in GSP, the next step in the Asian Region is the development of the regional implementation plan for sustainable soil management, which can translate the plans in the Nanjing Communiqué into practice.

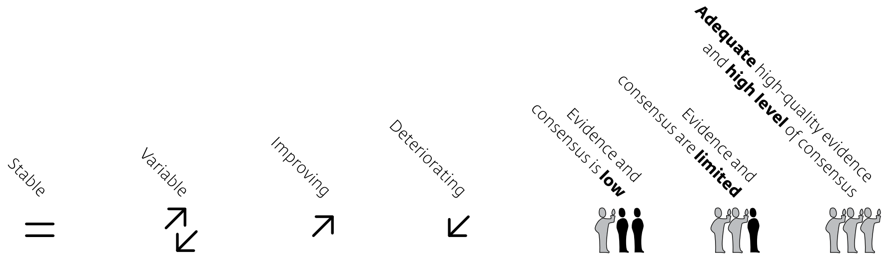


Table 10.4 | Summary of Soil Threats Status, trends and uncertainties in Asia

Threat to soil function	Summary	Condition and Trend					Confidence	
		Very poor	Poor	Fair	Good	Very good	In condition	In trend
Soil erosion	Serious water erosion occurs in regions with dry and wet seasons covering South Asia to East Asia, particularly in the hilly and mountainous landscapes. However, it is of little concern for well-established forests and paddy fields. Wind erosion is concentrated mainly in the most western and northern arid and semi-arid regions of Afghanistan, Pakistan, India, and China.		↙					
Organic carbon change	Increase in crop yield retains soil organic carbon (SOC) in croplands of East and Southeast Asia. Whereas, SOC is decreasing in South Asia, because crop residues are widely used as fuel and fodder, and not returned to the soil. The degradation of grassland has caused great losses of SOC stock.		↗↙					
Salinisation and sodification	The threat of salinisation/ sodification in the Asia region is widespread but variable. In semiarid and arid zones of central Asia, salt-affected soils are widely distributed. On the other hand, salt- affected soils are developed in certain coastal areas in monsoon zones, mainly by salt water intrusion in South and Southeast Asia.		↗↙					
Nutrient imbalance	Negative soil nutrients balances have been reported for N, P, K and micronutrients in many…" South Asian countries. Whereas, large excess of nutrients, in particular N, causes serious environmental problems in other countries.		↙					

Contamination	Rapid urbanization, industrialization, and intensive farming causes contamination of heavy metals (Cd, Ni, As, Pb, Zn, etc.) and pesticides in various parts of Asia, which, in turn, poses a serious risk to human health.		↙					
Soil sealing and land take	Rapid urbanization and development of mega-cites significantly increased the rate of impervious surface area (ISA). Asia region has the largest ISA within the global regions.		↙					
Soil acidification	There is substantial area of acid soils distributed in tropical and subtropical regions of Asia, mainly in Southeast Asia, parts of East and South Asia. This is mainly caused by unbalanced and unsuitable application of chemical fertilizers. Distribution of acid sulphate soils in tropical Asia also limits crop production.		↙					
Compaction	Mechanization of land management has increased compaction of surface soil and/or subsoil in cropland, grassland and timber forests. Increase in livestock trampling is also a major cause of surface soil compaction in grassland and hilly region.		↙					
Waterlogging	Anthropogenic activates such as poor drainage system and deforestation in the upstream areas increase the threat to waterlogging in the flood prone areas.			↙				
Loss of soil biodiversity	Limited information is available for soil biodiversity in Asia. Some reports show high microbial biodiversity in the soils of organic farming lands.			↗ ↙				

References

Aggarwal, G.C., Sidhu, A.S., Du, N.K., Sandhu, K.S. & Sur, H.S. 1995. Puddling and N management effects on crop response in a rice-wheat cropping system. Soil Till. Res., 36: 129-139.

Agus, F. & Subiksa, I.G.M. 2008. *Lahan Gambut: Potensi untuk Pertanian dan Aspek Lingkungan (Peatland: The potential for Agriculture and environmental aspects)*. Booklet. Indonesia, Bogor, Balai Penelitian Tanah (Indonesian Soil Research Institute) & World Agroforestry Centre (ICRAF) SE Asia,

Agus, F. & Widianto. 2004. *Petunjuk praktis konservasi tanah lahan kering (Practical guidelines for upland soil conservation)*. Bogor, World Agroforestry Centre (ICRAF) Southeast Asia. 102 pp.

Agus, F., Amien, L.I. & Sutono, S. 2002. *Farming systems and best practices for drought-prone areas in Indonesia*. Proceedings, Expert Group Consultations on Farming Systems and Best Practices for Drought Prone Areas, January 21-25, 2002. India, Hyderabad, FAO&CRIDA.

Agus, F., Hairiah, K. & Mulyani, A. 2011. *Measuring carbon stock in peat soils: practical guidelines*. World Agroforestry Centre (ICRAF) Southeast Asia Regional Program and Indonesian Centre for Agricultural Land Resources Research and Development. Indonesia. Bogor. 60 pp.

Agus, F., Henson, I.E., Sahardjo, B.H., Harris, N., van Noordwijk, M. & Killeen, T.J. 2013. Review of emission factors for assessment of CO_2 emission from land use change to oil palm in Southeast Asia. In T.J. Killeen & J. Goon, eds. *Roundtable for Sustainable Palm Oil (RSPO)*, pp. 7-27. Kuala Lumpur.

Ahmad, M. & Kutcher, G.P. 1992. *Irrigation planning with environmental considerations: a case study of Pakistan's Indus basin*. World Bank Technical Paper 166. Washington, DC, World Bank. 196 pp.

Akiyama, H., Yagi, K. & Yan, X.Y. 2005. Direct N_2O emissions from rice paddy fields: Summary of available data. *Global Biogeochem. Cycles*, 19: GB 1005.

Andriesse, J.P. 1994. Constrainsts and opportunities for alternative use options of tropical peat land. In B.Y. Aminuddin, ed. *Tropical Peat*. Proceedings of International Symposium on Tropical Peatland, May 6-10, 1991. Malaysia, Sarawak, Kuching.

Asami, T. 1981. Maximum allowable limits of heavy metals in rice and soil. In: K. Kitagawa & I. Yamane, eds. *Heavy Metal Pollution in Soils of Japan*, pp. 257–274. Tokyo, Japan Scientific Societies Press.

Bangladesh. 1992. *Land Degradation*. Paper presented to FAO 21st Regional Conference for Asia and the Pacific. New Delhi.

Baoming, G.E., Zhang, D., Tang, B. & Zhou, C. 2014. Effect of land cover on biodiversity and composition of a soil macrofauna community in a reclaimed coastal area at Yancheng, China. T*urkish Journal of Zoology*, 38: 229-233.

Bhattacharyya, T., Pal, D.K., Mandal, C., Chandran, P., Ray, S.K., Dipak Sarkar, Velmourougane, K., Srivastava, A., Sidhu, G.S., Singh, R.S., Sahoo, A.K., Dutta, D., Nair, K.M., Srivastava, R., Tiwary, P., Nagar, A.P. & Nimkhedkar, S.S. 2013. Soils of India: historical perspective, classification and recent advances. C*urrent Science*, 104: 1308- 1323.

Biswas, A. & Tewatia, R.K. 1991. Nutrient balance in agro-climatic regions of India - an overview. *Fert. News*, 36(6): 13- 18.

Blanchart, E. & Julka, J.M. 1997. Influence of forest disturbance on earthworm (Oligochaeta) communities in the Western Ghats (South India). *Soil Biol. Biochem.*, 29: 303–306.

Booth, D.B. 1991. Urbanization and the natural drainage system-impacts, solutions and prognoses. *Northwest Environ*. J., 7: 93-118.

Brammer, H. & Ravenscroft, P. 2009. Arsenic in groundwater: A threat to sustainable agriculture in South and South-east Asia. *Environ. Int.*, 35 : 647-654.

CCIC. 2014. *Official web site.* Climate Change Information Center. (Also available at http://ccs.climate.go.kr/index.html)

Chang ,T.K., Shyu, G.S., Chang, W.L., Huang, W.D., Huang, J.H., Lin, J.S. & Lin, S.C. 2007. Monitoring and investigation of heavy metal in soil of Taipe[i] City. DEP-95-056. [In Chinese, with an English summary]

Chang, T.K., Shyu, G.S., Lin, Y.P., & Chang, N.C. 1999. Geostatistical analysis of soil arsenic content in Taiwan. J. *Environ. Sci. health*, 34: 1485-1501.

Changnon, S.A. 1992. Inadvertent weather modification in urban areas: Lessons for global climate change. *Bull. Am. Meteorol. Soc.*, 73: 619-627.

Chaudhary, M.K. & Aneja, D.R. 1991. Impact of green revolution on long-term sustainability of land and water resources in Harayana. *Indian Journal of Agricultural Economics*, 45: 428-432.

Chen, S.K., Liu, C.W. & Chen, Y.R. 2012. Assessing soil erosion in a terraced paddy field using experimental measurements and universal soil loss equation. *Catena*, 95: 131-141.

Chen, X., Cui, Z., Fan, M., Vitousek, P., Zhao, M., Ma, W., Wang, Z., Zhang, W., Yan, X., Yang, J., Deng, X., Gao, Q., Zhang, Q., Guo, S., Ren, J., Li, S., Ye, Y., Wang, Z., Huang, J., Tang, Q., Sun, Y., Peng, X., Zhang, J., He, M., Zhu, Y., Xue, J., Wang, G., Wu, L., An, N., Wu, L., Ma, L., Zhang, W. & Zhang, F. 2014. Producing more grain with lower environmental costs. *Nature*, 514: 486-489.

Choi, D.H., Jung, J.W., Yun, K.S., Lee, K.S., Choi, W.J., Lim, S.S., Park, H.N., Yim, B.J. & Hwang, T.H. 2012. Estimation of TOC concentration using BOO, COD in runoff from paddy fields. J. *Korean Soc. Water Environ.*, 28: 813-818.

Chopra, H. 1989. Land degradation: dimensions and casualties. *Indian Journal of Agricultural Economics*, 44: 45-54.

Coleman, K. & Jenkinson, D.S. 1996. RothC-26.3 – A model for the turnover of carbon in soil. In D.S. Powlson, P. Smith& J.U. Smith, eds. *Evaluation of Soil Organic Matter Models, Using Existing Long-Term Datasets*, pp. 237-246. Berlin, Springer

Dadhwal, V.K. & Nayak, S.R. 1993. A preliminary estimate of biogeochemical cycle of carbon for India. *Science & Culture*. 59: 9-13.

Dariah, A., Marwanto, S. & Agus, F. 2014. Root- and peat-based CO_2 emissions from oil palm plantations. *Mitig. Adapt. Strateg. Glob. Change*, 19: 831–843.

Dibiyantoro, A.L.H. 1998. Pesticide residues on some vegetables and reductions possible by integrated pest management.. In I.R. Kennedy, J.H. Skerritt, G.I. Johnson & E. Highley, eds. *Seeking Agricultural Produce Free of Pesticide Residue*s, pp. 374-379. Proceedings of an International Workshop held in Yogyakarta, Indonesia, February 17–19, 1998.

Doi, R. & Ranamukhaarachchi, S.L. 2013. Slow restoration of soil microbial functions in an Acacia plantation established on degraded land in Thailand. *Int. J. Environ. Sci. Technol.*, 10: 623-634.

Drewry, J.J. & Paton, R.J. 2000. Effects of cattle treading and natural amelioration on soil physical properties and pasture under dairy farming in Southland, New Zealand. *New Zealand Agr. Res.*, 43: 377-386.

Edinger, E.N., Azmy, K., Diegor, W. & Siregar, P.R. 2008. Heavy metal contamination from gold mining recorded in Porites lobata skeletons, Buyat-Ratototok district, North Sulawesi, Indonesia. *Marine Pollution Bulletin*, 56: 1553-1569.

Elvidge C.D., Tuttle B.T., Sutton P.C., Baugh K.E., Howard A.T., Milesi C., Bhaduri B.L. & Nemani R. 2007. Global distribution and density of constructed impervious surfaces. *Sensors*, 7: 1962-1979.

FAO. 2006. *Global planted forests thematic study: results and analysis*, by A. Del Lungo, J. Ball & J. Carle. Planted Forests and Trees Working Paper No. 38. Rome. (Also available at www.fao.org/forestry/site/10368/en).

FAO. 2012. *FAOSTAT*. Rome, FAO. (Also available at http://faostat 3.fao.org/browse/R/RL/E)

FAO. 2014. *FAOSTAT*. Rome, FAO. (Also available at http://faostat.fao.org/site/291/default.aspx)

FAO/IUSS/ISRIC. 2006. *World Reference Base for Soil Resources*. World Soil Resources Reports 103. Rome, FAO.

FAO/RAPA. 1992. *Environmental Issues in Land and Water Development*. Bangkok, FAO/RAPA.. 488 pp. (Includes country papers on Bangladesh, India, Nepal, Pakistan and Sri Lanka)

Fischer, G., Nachtergaele, F.O., Prieler, S., Teixeira, E., Toth, G.,Velthuizen, H.v., Verelst, L. & Wiberg, D. 2012. *Global Agro-Ecological Zones* (GAEZ v 3.0) - Model Documentation. Laxenburg, IIASA & Rome, FAO.

Fisher, R. & Hirsch, P. 2008. Poverty and agrarian-forest interactions in Thailand. *Geogr. Res.*, 46: 74-84.

FSI. 1999. *State of Forest Report*. Dehra Dun, Forest Survey of India.

Germer, J. & Sauerborn, J. 2008. Estimation of the impact of oil palm plantation establishment on greenhouse gas balance. *Environ. Develop. Sustain.*, 10: 697-716.

GSP. 2012. *Official web site*. Nanjing Communiqué on Asian Soil Partnership. (Also available at http://www.fao.org/globalsoilpartnership/regional-partnerships/asia/en/)

Guo, J.H., Liu, X.J., Zhang, Y., Shen, J.L., Han, W.X., Zhang, W.F., Christie, P., Goulding, K.W.T., Vitousek, P.M. & Zhang, F.S. 2010. Significant Acidification in major Chinese croplands. *Science*, 327: 1008-1010.

Gupta, R.K. & Rao, D.L.N. 1994. Potential of wastelands for sequestering carbon by reforestation. *Curr. Sci.*, 66: 378-380.

Han, Y.W. & Gao, J.X. 2005. Analysis of main ecological problems of grasslands and relevant countermeasures in China. *Res. Environ. Sci.*, 18: 60–62. [in Chinese]

Haron, K., Brookes, P.C., Anderson, J.M. & Zakaria, Z.Z. 1998. Microbial biomass and soil organic matter dynamics in oil palm (Elaeis guineensis Jacq.) plantations, West Malaysia. *Soil Biol. Biochem.*, 30: 547–552.

Hartanto, H., Prabhu, R., Widayat, A.S.E. & Asdak, C. 2003. Factors affecting runoff and soil erosion: plot-level soil loss monitoring for assessing sustainability of forest management. *For. Ecol. Manage.*, 180: 361–374.

Hattori, D., Tanaka, K., Okamura, I.K., Kendawang, J.J., Ninomiya, I. & Sakurai, K. 2013. Effects of soil compaction on the growth and motality of planted dipterocarp seedlings in a logged-over tropical rainforest in Sarawak, Malaysia. *For. Ecol. Manag.*, 310: 770-776.

Hicks, W.K., Kuylenstierna, J.C.I., Owen, A., Dentener, F., Seip, H.M. & Rodhe, H. 2008. Soil sensitivity to acidification in Asia: status and prospects. *AMBIO*, 37: 295-303.

Hooijer, A. S. Page, P. Navratil, R. Vernimmen, M. Van der Vat, K. Tansey, K. Konecny, F. Siegert, U. Ballhorn & N. Mawdsley. 2014. *Carbon emissions from drained and degraded peatland in Indonesia and emission factors for measurement, reporting and verification (MRV) of peatland greenhouse gas emissions A summary of KFCP research results for practitioners.* Indonesia, Jakarta, IAFCP.

Houghton, R.A. & Hackler, J.L. 2003. Sources and sinks of carbon from land-use change in China. *Global Biogeochem. Cycles*, 17: 1034.

Husnain, H., Wigena, I.G.P., Dariah, A., Marwanto, S., Setyanto, P. & Agus, F. 2014. CO_2 emissions from tropical drained peat in Sumatra, Indonesia. *Mitig. Adapt. Strateg. Glob. Change*, 19: 845-862.

Husnain, Masunaga, T., Wakatsuki, T. 2010. Field assessment of nutrient balance under intensive rice-farming systems, and its effects on the sustainability of rice production in Java Island, Indonesia. *Journal of agricultural, food and environmental science*, 4(1).

ICAR & NAAS. 2010. *Degraded and Wastelands of India - Status and Spatial Distribution*, by S.M. Virmani, R. Prasad & P.S. Pathak, ed. Indian Council of Agricultural Research and National Academy of Agricultural Sciences, and Indian Council of Agricultural Research, New Delhi. 158 pp.

IFA. 2012. *Global supply and demand outlook for fertilizer and raw materials.* IFA. (Also available at www.fertilizer.org)

IFADATA. 2007. *Production and International Trade Statistics*. International Fertilizer Industry Association. (Also available at http://www.fertilizer.org/ifa/statistics.asp, retrieved 12 October 2014)

Indrayatie, E.R., Utomo, W.H., Handayanto, E. & Anderson, C.W.N. 2013. The use of vetiver (Vetiveria zizanioides L.) for the remediation of wastewater discharged from tapioca factories. Int. J. of Environ. Waste Manage, 12: 1-16.

IPCC. 2006. *Guidelines for National Greenhouse Gas Inventories*, by H.S. Eggleston, L. Buendia, K. Miwa., T. Ngara & K. Tanabe, eds. National Greenhouse Gas Inventories Programme. Japan, IGES.

IPCC. 2014. *Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands*, by T. Hiraishi, T. Krug, K. Tanabe, N. Srivastava, J. Baasansuren, M. Fukuda & T.G. Troxler, eds. Switzerland, IPCC.

Jim, C.Y. 1998a. Soil characteristics and management in an urban park in Hong Kong. *Env. Manage*, 22: 683-695.

Jim, C.Y. 1998b. Physical and chemical properties of a Hong Kong roadside soil in relation to urban tree growth. *Urban Ecosyst.*, 2: 171-181.

Jo, I.S., Jung, P.K., Kim, L.Y., Ha, S.K. & Chung, D.Y. 2009. Soil Physical properties and soil conservation. Korean. *J. Soil. Sci. Fert.*, 42(s): 28-52.

Ju, X.T., Xing, G.X., Chen, X.P., Zhang, S.L., Zhang, L.J., Liu, X.J., Cui, Z.L., Yin, B., Christie, P., Zhu, Z.L. & Zhang, F.S. 2009. Reducing environmental risk by improving N management in intensive Chinese agricultural systems. *Proc. Natl. Acad. Sci. USA*, 106: 3041-3046.

Jung, K.H., Kim, W.T., Hur, S.O., Ha, S.K. Jung, P.K. & Jung, Y.S. 2004. USLE/RUSLE factors for national scale soil loss estimation based on the digital detailed soil map. Korean. *J. Soil. Sci. Fert.*, 37(4): 199-206.

Jung, K.H., Sonn, Y.K., Hong, S.Y., Hur, S.O. & Ha, S.K. 2005. Assessment of national soil loss and potential erosion area using the digital detailed soil map. Korean. *J. Soil. Sci. Fert.*, 38(2): 59-65.

Jung, P.K. & Oh, S.J. 1993. Soil loss with grass species in grasslands conversed from mountainous forests. *In Research report* 1992, pp 149-153.. Republic of Korea, Suwon, National Institute of Agricultural Technology.

Jung, P.K. 1998. Soil and water conservation (Review). *Korean. J. Soil. Sci. Fert.*, 31(S 1): 27-35.

Jung, Y.S., Joo, J.H. & Yoon, S.Y. 2002. *A Management guideline for soil and irrigation water in the reclaimed saline land.* Korea, IAS, Kangwon Nat. Univ.

Kamprath, E. 1984. Crop response to lime on soils in the tropics. In F. Adam, ed. *Soil acidity and liming, Second edition, Agronomy* 12, pp. 349-368. USA, Madison, WI, Soil Science Society of America.

Kim, M.K., Kwon, S.I., Jung, K.B., Hong, S.C., Chae, M.J., Yun, S.K. & So, K.H. 2013. Small-scale pond effects on reducing pollutants load from a paddy field. *Korean J. Environ. Agri.*, 32: 355-358.

Kitahara, H., Okura, Y., Sammori, T. & Kawanami, A. 2000. Application of universal soil loss equation (USLE) to mountainous forests in Japan. *J. For. Res.*, 5: 231-236.

Kohyama, K., Hojito, M., Sasaki, H. & Miyaji, H. 2003. Generation of agricultural statistics mesh data using digital national land information. *Jap. J. Soil Sci. Plant Nutr.*, 74: 415-424. [in Japanese with English summary]

Kohyama, K., Taniyama, I., Ohkura, T. & Nakai, M. 2012. USLE parameter data of 1 km grid to estimate soil erosion in farmland. *Inventory*, 10: 3-9. [in Japanese]

Kozlowski, T. 1999. Soil compaction and growth of woody plants. Scand. J. For. Res., 14, 596-619.

Kruemmelbein, J, Peth, S. & Horn, R. 2008. Determination of pre-compression stress of a variously grazed steppe soil under static and cyclic loading. *Soil Till.*, 99: 139-148.

Kukal, S.S. & Aggarwal, G.C. 2003. Puddling depth and intensity effects in rice–wheat system on a sandy loam soil, I: Development of subsurface compaction. *Soil Till. Res.*, 72: 1–8.

Kyuma, K. 2004. Paddy soil science. Kyoto University Press & Trans Pacific Press.

Lal, R. 2003. Soil erosion and the global carbon budget. Environ. *Internat.*, 29: 437-450.

Lal, R. 2004. Soil carbon sequestration impacts on global climate change and food security. *Science*, 304: 1623-1627.

Lee, K.S. 1994. Application of GIS to the Universal Soil Loss Equation for Quantifying Rainfall Erosion in Forest Watersheds. *J. Korean For. Soc.*, 83(3): 322-330.

Leon, A., Kohyama, K., Mishima, S., Ohkura, T., Shirato, Y., Takata, Y., Taniyama, I. & Obara, H. 2012. Factors controlling organic amendment application rate and long-term change in application rate in Japanese paddy field using longitudinal questionnaire survey dataset (the Basic Soil Environment Monitoring Project, Stationary Monitoring, 1979-1998). *Soil Sci. Plant Nutr.*, 58: 104-120.

Li, P., Feng, X., Qiu, G., Shang, L., Wang, S. & Meng, B. 2009. Atmospheric mercury emission from artisanal mercury mining in Guizhou Province, Southwestern China. *Atmos. Environ.*, 43: 2247-2251.

Li, S.-G., Asanuma, J., Eugster, W., Kotani, A., Liu, J.-J., Urano, T., Oikawa, T., Davaa, G., Oyunbaatar, D. & Sugita, M. 2005. Net ecosystem carbon dioxide exchange over grazed steppe in central Mongolia. *Global Change Biol.*, 11: 1941-1955.

Limbong, D., Kumampung, J., Rimper, J. Arai, T. & Miyazaki, N. 2003. Emissions and environmental implications of mercury from artisanal gold mining in North Sulawesi, Indonesia. *Sci. Total Environ.*, 302(1-3): 227-236.

Liu, X., Zhang, Y, Han, W., Tang, A., Shen, J., Cui, Z., Vitousek, P., Erisman, J.W., Goulding, K., Christie, P., Fangmeier, A. & Zhang, F. 2013. Enhanced nitrogen deposition over China. *Nature*, 494: 459-462.

Luo, Y., Wu, L., Liu, L., Han, C. & Li, Z. 2009. *Heavy Metal Contamination and Remediation in Asian Agricultural Land*. MARCO Symposium 2009: Challenges for Agro-Environmental Research in Monsoon Asia.

Ma, L., Ma, W.Q., Velthof, G.L., Wang, F.H., Qin, W., Zhang, F.S. & Oenema, O. 2010. Modelling nutrient flows in the food chain of China. J. *Environ. Qual.*, 39: 1279-1289.

MAFF. 2008. *Trend and Issue of cultivated soil in Japan.* Ministry of Agriculture, Forestry and Fishery. (Also available at http://www.maff.go.jp/j/study/dozyo_kanri/01/pdf/ref_data 1.pdf)

Makino, T. 2010. Heavy metal pollution of soil and risk alleviation methods based on soil chemistry. Pedologist, 53; 38-49.

Marwanto, S. & Agus, F. 2014. Is CO_2 flux from oil palm plantations on peatland controlled by soil moisture and/or soil and air temperatures? *Mitig. Adapt. Strateg. Glob. Change*, 19: 809-819.

Mathews. J., Tan, T.H. & Chong, K.M. 2010. Indication of soil organic carbon augmentation in oil palm cultivated inland mineral soils of Peninsular Malaysia. *The Planter, Kuala Lumpur*, 86: 293-313.

Milesi, C., Elvidge, C.D., Nemani, R.R. & Running, S.W. 2003. Assessing the impact of urban land development on net primary productivity in the Southeastern United States. *Remote Sens. Environ.*, 86: 401-410.

Millward, A.A., Paudel, K. & Briggs, S.E. 2011. Naturalization as a strategy for improving soil physical characteristics in a forested urban park. *Urban Ecosyst.*, 14: 261-278.

Minasny, B., McBratney, A.B., Hong, S.Y., Sulaeman, Y., Kim, M.S., Zhang, Y.S., Kim, H.Y. & Han, K.H. 2012. Continous rice cropping has been sequestering carbon in soils in Java and South Korea for the past 30 years. *Global Biogeochem. Cycle*, 26: 1-8. GB 3207.

Ministry of Environmental Protection, the People's Republic of China. 2014. *China alerted by serious soil pollution*. China. (available at http://english.mep.gov.cn/News_service/media_news/201404/t20140421_270801.htm)

Mishima, S., Endo, A. & Kohyama, K. 2010a. Recent trend in residual nitrogen on national and regional scales in Japan and its relation with groundwater quality. *Nutr. Cycling Agroecosys.*, 83: 1-11.

Mishima, S., Endo, A. & Kohyama, K. 2010b. Recent trends in phosphate balance nationally and by region in Japan. *Nutr. Cycling Agroecosys.*, *86*: 69-77.

Mishima, S., Kimura, S.D., Eguchi, S. & Shirato, Y. 2012. Estimation of the amounts of livestock manure, rice straw, and rice straw compost applied to crops in Japan: a bottom-up analysis based on national survey data and comparison with the results from a top-down approach. *Soil Sci. Plant Nutr.*, 58: 83-90.

MOE. 2014. *Enforcement status of Agricultural Land–Soil Pollution Preventation Law in 2014 fiscal year*. Japan, Ministry of the Environment. (Also available at: http://www.env.go.jp/press/files/25644.pdf) (in Japanese)

Morisada, K., Ono, K. & Kanomata, H. 2004. Organic carbon stock in forest soils in Japan. *Geoderma*, 119: 21-32.

Mujeeb Rahman, P., Varma, R.V. & Sileshi, G.W. 2012. Abundance and diversity of soil invertebrates in annual crops, agroforestry and forest ecosystems in the Nilgiri biosphere reserve of Western Ghats, India. *Agroforest. Syst.*, 85: 165-177.

Murty, D., Kirschbaum, M.U.F., McMurtrie, R.E. & McGilvray, A. 2002. Does conversion of forest to agricultural land change soil carbon and nitrogen? A review of the literature. *Global Change Biol.*, 8: 105–123.

Nguyen, N.D. 2014. Adequate use of Sulphur. *Vietnam Agriculture Newspaper,* February 10, 2014.

NISF. 2012. *The basic information of main soils unit of Vietnam*. National Institute for Soils and Fertilizers. Hanoi, Thegioi publishers.

Noor, M. 2001. *Pertanian lahan gambut: potensi dan kendala (Peatland: potential and constraints).* Jakarta, Penerbit Kanisius.

Oldeman, L.R. 1991. Global extent of soil degradation. *In ISRIC Bi-Annual Report* 1991-1992, pp. 19-36. ISRIC.

Oldeman, L.R. 2000. Impact of soil degradation: A Global Scenario. *In Proceedings of International Conference on Managing Natural Resources for Sustainable Agricultural Production in the 21st Century*, pp 79-86. India, New Delhi.

Page, S. E., Rieley, J.O. & Banks, C.J. 2011. Global and regional importance of the tropical peatland carbon pool. *Glob. Change Biol.*, 17: 798–818.

Page, S.E., Siegert, F., Rieley, J.O., Boehm, H.V., Jaya, A. & Limin, S. 2002. The amount of carbon released from peat and forest fires in Indonesia during 1997. *Nature*, 420: 61-65.

Paningbatan, E.P. 1987. Soil erosion, problem and control in the Philippines. *In Paper presented during the Training Workshop on Soil Conservation Measures and Agroforestry,* pp. 7-23. PCARRD. Laguna, Los Baños.

Parish, F., Sirin, A., Charman, D., Joosten, H., Minayeva, T., Silvius, M. & Stringer, L. (eds.). 2007. *Assessment on Peatlands, Biodiversity and Climate Change: Main Report.* Kuala Lumpur, Global Environment Centre & Wageningen, Wetlands International.

Park, C.W., Sonn, Y.K., Hyun, B.K., Song, K.C., Chun, H.C., Moon, Y.H. & Yun, S.G. 2011. The redetermination of USLE rainfall erosion factor for estimation of soil loss at Korea. Korean. *J. Soil. Sci. Fert.*, 44: 977-982.

Park, K.R. 1991. Suitability Assessment and Production management of upland field crops in the reclaimed saline land. In A.M. Mashali, Choi, W.Y. & D.W. West, eds. *Training manual agricultural use of reclaimed coastal land*, pp. 174-192. FAO & Korea, Kwangju, Chonnam National University.

Park, S.H. 2001. Tideland reclamation. In S.K. Kwun, ed. *Rice Culture in Asia*, pp. 116-118. Korea, Ansan, KCID

Piao, S., Fang, J., Ciais, P., Peylin, P., Huang, Y., Sitch, S. & Wang, T. 2009. The carbon balance of terrestrial ecosystems in China. *Nature*, 458: 1009-1013.

Piao, S.L., Ito, A., Li, S.G., Huang, Y., Ciais, P., Wang, X.H., Peng, S.S., Nan, H.J., Zhao, C., Ahlström, A., Andres, R.J., Chevallier, F., Fang, J.Y., Hartmann, J., Huntingford, C., Jeong, S., Levis, S., Levy, P.E., Li, J.S., Lomas, M.R., Mao, J.F., Mayorga, E., Mohammat, A., Muraoka, H., Peng, C.H., Peylin, P., Poulter, B., Shen, Z.H., Shi, X., Sitch, S., Tao, S., Tian, H.Q., Wu, X.P., Xu, M., Yu, G.R., Viovy, N., Zaehle, S., Zeng, N. & Zhu, B. 2012. The carbon budget of terrestrial ecosystems in East Asia over the last two decades. *Biogeosci.*, 9: 3571-3586.

Prasetyo, B., Krisnayani, B.D., Utomo, W.H. & Anderson, C.W.N. 2010. Rehabilitation of artisanal gold mining land in West Lombok, Indonesia. Vol 2. Arbuscular mycorrhiza status of tailings and surrounding soils. *J. Agric. Sci.*, 2: 202-209.

Puslitbangtanak. 2000. *Atlas Sumber Daya Tanah Eksplorasi Indonesia. Skala 1:1.000.000 (Exploration soil resource atlas, at 1:1 000 000 scale).* Pusat Penelitian dan Pengembangan Tanah dan Agroklimat. Bogor, Puslitbangtanak.

Rahmawati, S., Margana, G., Yoneda, M. & Oginawati, K. 2013. Organochlorine pesticide residue in Catfish (*Clarias sp.*) collected from local fish cultivation at Citarum watershed, West Java Province, Indonesia. Procedia Environ. Sci., 17: 3-10.

Ritung, S., Wahyunto, Nugroho, K., Sukarman, Hikmatullah, Suparto & Tafakresnanto, C. 2011. *Peta Lahan Gambut Indonesia Skala 1:250.000 (Indonesian peatland map at the scale 1:250 000).* Indonesia, Bogor, Indonesian Center for Agricultural Land Resources Research and Development.

Rossi, J.P. & Blanchart, E. 2005. Seasonal and land use induced variations of soil macrofauna composition in the Western Ghats, southern India. *Soil Biol. Biochem.*, 37: 1093–1104.

Sakuramot, N., Yokoyama, K. & Iekushi, T. 2010. Advocacy of positive environmental assessment using soil microbial diversity and its vitality. *In Proceeding of AFITA 2010 International Conference*, pp. 137-140. AFITA.

Salinger, M.J., Shrestha, M.L., Ailikun, Dong, W., McGregor, J.L. & Wang, S. 2014. Climate in Asia and the Pacific: Climate variability and change. In M. Manton & L.A. Stevenson, eds. *Climate in Asia and the Pacific*. Asia-Pacific Network for Global Change Research. Dordrecht, Springer Science & Business Media.

Santoso, D., Adiningsih, S., Mutert, E., Fairhurst, T. & van Noordwijk, M. 1997. Site improvement and soil fertility management for reclamation of Imperata grasslands by smallholder agroforestry. *Agrofor. Syst.*, 36: 181-202.

Schertz, D.L. 1983. The basis for soil loss tolerance. J. *Soil Water Conserv.*, 38: 10-14.

Schneider, A., Friedl, M.A. & Potere, D. 2009. A new map of global urban extent from MODIS satellite data. *Environ. Res. Lett.*, 4(044003): 1-11.

Sehgal, J. & Abrol, I.P. 1992. Land degradation statue: India. *Desertification Bulletin*, 21: 24-31.

Shaheed, S.M. 1992. Environmental issues in land development in Bangladesh. *In Environmental Issues in Land and Water Development*, pp. 105-127. FAO & Bangkok, RAPA.

Shamshuddin, J., Azura, A.E., Shazana, M.A.R.S., Fauziah, C.I., Panhwar, Q.A. & Naher, U.A. 2014. Properties and measurement of acid sulfate soils in Southeast Asia for sustainable cultivation of rice, oil palm, and cocoa. *Adv. Agron.*, 124: 91-142.

Shirato, Y., Yagasaki, Y. & Nishida, M. 2011. Using different versions of the Rothamsted Carbon model to Simulate soil carbon in long-term experimental plots subjected to paddy-upland rotation in Japan. *Soil Sci. Plant Nutr.*, 57: 597-606.

Shofiyati, R., Las, I. & Agus, F. 2010. Indonesian soil data base and predicted stock of soil carbon. *In Proceedings of an international workshop on evaluation and sustainable management of soil carbon sequestration in Asian countries*, pp.73-83. Indonesian Soil Research Institute (ISRI), Food & Fertilizer Technology Center for the Asian and Pacific Region (FFTC) and National Institute for Agro-environmental Sciences (NIAES).

Shoiful, A., Fujita, H., Watanabe, I. & Honda, K. 2012. Concentrations of organochlorine pesticides (OCPs) residues in foodstuffs collected from traditional markets in Indonesia. *Chemosphere*, 90: 1742-1750.

Sidhu, G.S., Bhattacharyya, T., Sarkar, D., Ray, S.K., Chandran, P., Pal, D.K., Mandal, D.K., & Prasad, J. 2014. Impact of soil management levels and land use changes on soil properties in rice-wheat cropping system of the Indo-Gangetic Plains (IGP). *Current Science*, 107(9): 1487-1501.

Simmons, R.W., Pongsakul, P., Saiyasitpanich, D. & Klinphoklap, S. 2005. Elevated levels of cadmium and zinc in paddy soils and elevated levels of cadmium in rice grain downstream of a zinc mineralized area in Thailand: implications for public health. *Environ. Geochem. Health*, 27: 501-511.

Singh, K. B., Jalota, S. K. & Sharma, B. D. 2009. Effect of continuous rice-wheat rotation on soil properties from four agro-ecosystems of Indian Punjab. *Commun. Soil Sci. Plant Anal.*, 40: 2945-2958.

Singh, R.L. 1971. *Regional Geography*, by R. L. Singh, ed. Silver Jubilee Publication. National Geographical Society of India, Varanasi-5.

Smedley, P.L. 2003. Arsenic in groundwater -south and east Asia. In A.H. Welch & K.G. Stollenwerk, eds. *Arsenic in Ground Water*, pp. 179-209. Boston, Kluwer Academic Publishers.

Somboonna, N., Wilantho, A., Jankaew, K., Assawamakin, A., Sangsrakru, D., Tangphatsornruang, S. & Tongsima, S. 2014. Microbial ecology of Thailand tsunami and non-tsunami affected terrestrials. *PLoS ONE*, 9.

Sukristiyonubowo, Nugroho, K. & Ritung, S. 2012. Rice growth and water productivity of newly opened wetlands in Indonesia. Internat. Res. *J. Agric. Sci. Soil Sci.*, 2: 328-332.

Takata, T., Kohyama, K., Obara, H., Maejima, Y., Ishitsuka, N., Saito, T. & Taniyama, I. 2014. Spatial prediction of radioactive Cs concentration in agricultural soil in East Japan. *Soil Sci. Plant Nutr.*, 60: 393-403.

Takata, Y. 2010. Estimation of soil carbon stock changes in Japanese agricultural soils using national resource inventory. *In Proceedings of an international workshop on evaluation and sustainable management of soil carbon sequestration in Asian countries*, pp 15-28. Indonesian Soil Research Institute (ISRI), Food & Fertilizer Technology Center for the Asian and Pacific Region (FFTC) and National Institute for Agro-environmental Sciences (NIAES).

Takata, Y., Ito, T., Ohkura, T., Obara, H., Kohyama, K. & Shirato, Y. 2011a. Phosphate adsorption coefficient can improve the validity of RothC model for Andosols. *Soil Sci. Plant Nutr.*, 57: 421-428

Takata, Y., Obara, H., Nakai, S. & Kohyama, K. 2011b. Process of the decline in the cultivated soil area with land use changes in Japan. *Jpa. J. Soil Sci. Plant Nutr.*, 82: 15-24. (in Japanese with English summary)

Tan, Z.X., Lal, R. & Wiebe, K.D. 2005. Global soil nutrient depletion and yield reduction. *J. Sustainable Agric.*, 26: 123-146.

Tanaka, S., Tachibe, S., Wasli, M.E.B., Lat, J., Seman, L., Kendawang, J.J., Iwasaki, K. K. & Sakurai, K. 2009. Soil characteristics under cash crop farming in upland areas of Sarawak, Malaysia. *Agric. Ecosys. Environ.*, 129: 293–301.

Tandon, H.L.S. 1992. *Assessment of Soil Nutrient Depletion*. Paper presented to FADINAP seminar, Fertilization and the environment. Thailand, Chiang Mai.

Taniyama, I. 2003. Study of soil factor related to production of surface runoff from cropland and development of MI. In Agriculture, Forestry and Fisheries Research Council, eds. *Comprehensive evaluation of implication of agriculture, forestry, fisheries and trade of products on natural resources and the environment*, p. 149-152. Tokyo, Agriculture, Forestry and Fisheries Research Council(in Japanese)

Tian, H., Melillo, J., Lu, C., Kicklighter, D., Liu, M., Ren, W., Xu, X., Chen, G., Zhang, C., Pan, S., Liu, J. & Running, S. 2011. China's terrestrial carbon balance: Contributions from multiple global change factors. *Global Biogeochem. Cycles*, 25: GB 1007.

Twyford, I. 1994. *Fertilizer Use and Crop Yields*. Paper presented to 4th National Congress of the Soil Science Society of Pakistan, Islamabad.

Ugawa, S., Takahashi, M., Morisada, K., Takeuchi, M., Matsuura, Y., Yoshinaga, S., Araki, M., Tanaka, N., Ikeda, S., Miura, S., Ishizuka, S., Kobayashi, M., Inagaki, M., Imaya, A., Nanko, K., Hashimoto, S., Aizawa, S., Hirai, K., Okamoto, T., Mizoguchi, T., Torii, A., Sakai, S., Ohnuki, Y. & Kaneko, S. 2012. Carbon stocks of dead wood, litter, and soil in the forest sector of Japan: general description of the National Forest Soil Carbon Inventory. *Bulletin of FFPRI*, 11: 207-221.

Valentin, C., Agus, F., Alamban, R., Boosaner, A., Bricquet, J.P., Chaplot, V., de Guzman, T., de Rouw, A., Janeau, J.L., Orange, D., Phachomphonh, K., Do Duy Phai, Podwojewski, P., Ribolzi, O., Silvera, N., Subagyono, K., Thiébaux, J.P., Tran Duc Toan & Vadari, T. 2008. Runoff and sediment losses from 27 upland catchments in Southeast Asia: Impact of rapid land use changes and conservation practices. *Agric. Ecosys. Environ.*, 128: 225-238.

van der Kamp, J., Yassir, I. & Buurman, P. 2009. Soil carbon changes upon secondary succesion in Imperata grasslands (East Kalimantan, Indonesia). *Geoderma*, 149: 76-83.

van Lynden, G.W.J. & Oldeman, L.R. 1997. *The assessment of the status of human-induced soil degradation in South and Southeast Asia*. The Netherlands, Wageningen, International Soil Reference and Information Centre.

van Noordwijk, M., Cerri, P., Woomer, P.L., Nugroho, K. & Bernoux, M. 1997. Soil carbon dynamic in the humid tropical forest zone. *Geoderma*, 79: 187-225. doi: 10.1016/S 0016-7061(97)00042-6.

Veiga, M.M., Nunes, D., Klein, B., Shandro, J.A., Velasquez, P.C. & Sousa, R.N. 2009. Mill leaching: a viable substitute for mercury amalgamation in the artisanal gold mining sector? *J. Cleaner Prod.*, 17: 1373-1381.

Velayutham, M. & Bhattacharyya, T. 2000. *Soil Resource Management. In Natural Resource Management for Agricultural Production in India*, byJ.S.P. Yadav and G.B. Singh, eds. Published by Secretary General International Conference on Managing Natural Resources for Sustainable Agricultural Production in the 21st Century.

Velthof, G.L., Oudendag, D., Witzke, H.P., Asman, W.A.H., Klimont, Z. & Oenema, O. 2009. Integrated assessment of nitrogen losses from agriculture in EU-27 using MITERRA-EUROPE. *J. Environ. Qual.*, 38: 402-417.

Vitousek, P.M., Naylor, R., Crews, T., David, M.B., Drinkwater, L.E., Holland, E., Johnes, P.J., Katzenberger, J., Martinelli, L.A., Matson, P.A., Nziguheba, G., Ojima, D., Palm, C.A., Robertson, G.P., Sanchez, P.A., Townsend, A.R. & Zhang, F.S. 2009. Nutrient imbalances in agricultural development. *Science*, 324: 1519-1520.

von Uexküll, H.R. & Mutert, E. 1995. Global extent, development and economic impact of acid soils. *Plant Soil*, 171: 1-15.

Wahyunto, W., Nugroho, K., Ritung, S. & Agus, F. 2014. Indonesian peatland map: Method, Certainty, and uses. pp. 81-96. *In* Prosiding Seminar Nasional Pengelolaan Berkelanjutan Lahan Gambut Terdegradasi untuk Mitigasi Emisi GRK dan Peningkatan Nilai Ekonomi. Indonesian Agency or Agricultural Research and Development, Jakarta, Indonesia.

Wang, H., Xu, R.K., Wang, N. & Li, X.H. 2010. Soil acidification of Alfisols as influenced by tea plantation in eastern China. *Pedosphere*, 20: 799-806.

Wang, J., Chen, Y., Shao, X., Zhang, Y. & Cao, Y. 2012. Land-use changes and policy dimension driving forces in China: Present, trend and future. *Land Use Policy*, 29: 737-749.

Warren, M.W., Kauffman, J.B., Murdiyarso, D., Anshari, G., Hergoualc'h, K., Kurnianto, S., Purbopuspito, J., Gusmayanti, E., Afifudin, M., Rahajoe, J., Alhamd, L., Limin, S. & Iswandi, A. 2012. A cost-efficient method to assess carbon stocks in tropical peat soil. *Biogeosciences*, 9: 4477–4485.

Wei, C.Y. & Chen, T.B. 2001. Hyperaccumulators and phytoremediation of heavy metal contaminated soil: a review of studies in China and abroad. *Acta Ecologica Sinica*, 21: 1196-1203.

Widowati, W., De Neve, L.R., Sukristiyonubowo, S., Setyorini, D., Kasno, A., Sihaputar, I.A. & Sukristiyohastomo. 2011. Nitrogen balances and nitrogen uses efficiency of intensive vegetable rotations in South East Asian tropical Andisols. *Nutr. Cycling Agroecosys*., 91: 131-143.

Williams, M., Fordyce, F., Paijitprapapong, P. & Charoenchaisri, P. 1996. Arsenic contamination in surface drainage and groundwater in part of Southeast Asia tin belt, Nakhon Si Thammarat province, Southern Thailand. *Environ. Geol.*, 27: 16-33.

Wosten, J.H.M., Clymans, E., Page, S.E., Rieley, J.O. & Limin, S.H. 2008. Peat-water interrelationships in a tropical peatland ecosystem in Southeast Asia. *Catena*, 73: 212-224.

Xie, Z., Liu, G., Be[i], Q., Chen, C., Cadisch, G., Liu, Q., Lin, Z., Hasegawa, T. and Zhu, J. 2014. Soil organic carbon stocks, changes and CO_2 mitigation potential by alteration of residue amendment pattern in China. In A. Hartemink & K. McSweeney, eds. *Soil Carbon*,, pp.457-466. Springer International Publishing.

Xie, Z., Zhu, J., Liu, G., Cadisch, G., Hasegawa, T., Chen, C., Sun, H., Tang, H. & Zeng, Q. 2007. Soil organic stocks in China and changes from 1980s to 2000s. *Global Change Biol.*, 13:1989-2007.

Yagasaki, Y. & Shirato, Y. 2014. Assessment on the rates and potentials of soil organic carbon sequestration in agricultural lands in Japan using a process-based model and spatially explicit land-use change inventories – Part 1: Historical trend and validation based on nation-wide soil monitoring. *Biogeosciences*, 11: 4429-4442.

Yan, X., Du, L., Shi, S. & Xing, G. 2000. Nitrous oxide emission from wetland rice soil as affected by the application of controlled-availability fertilizers and mid-season aeration. *Biol. Fertil. Soils*, 32: 60-66.

Yan, X.Y., Akimoto, H. & Ohara, T. 2003. Estimation of nitrous oxide, nitric oxide and ammonia emissions from croplands in East, Southeast and South Asia. *Global Change Biol.*, 9,:1080-1096.

Yan, X.Y., Akiyama, H., Yagi, K. & Akimoto, H. 2009. Global estimations of the inventory and mitigation potential of methane emissions from rice cultivation conducted using the 2006 Intergovernmental Panel on Climate Change Guidelines. *Global Biogeochem. Cycles*, 23: GB 2002. doi:10.1029/2008GB 003299.

Yan, Z., Liu, P., Li, Y., Ma, L., Alva, A., Dou, Z., Chen, Q. & Zhang, F. 2013. Phosphorus in China's intensive vegetable production systems: Overfertilization, soil enrichment, and environmental implications. *J. Environ. Qual.*, 42: 982-989.

Yokoyama, K. & Taguchi, Y-H. 2013. Microbiology and biodiversity-based modelling of suppression of cottony leak of scarlet runner bean in soils with diverse and uniform ecology. *J. Agric. Sci. Appl.*, 2: 35-44.

Yokoyama, K. 1993. Evaluation of biodiversity of soil microbial community. *In Symbiosphere: Ecological Complexity for Promoting Biodiversity*, p. 74-78. Biology International special issue 29. International Union of Biological Sciences.

Yu, Y.Y., Guo, Z.T., Wu, H.B., Kahmann, J.A. & Oldfield, F. 2009. Spatial changes in soil organic carbon density and storage of cultivated soils in China from 1980 to 2000. *Global Biogeochem. Cycles*, 23: GB 2021, doi: 2010.1029/2008GB 003428.

Zhang, X.Y., Cruse, R.M., Sui, Y.Y. & Jhao, Z. 2006. Soil compaction induced by small tractor traffic in northeast China. *Soil Sci. Soc. Am. J.*, 70: 613-619.

Zhao, Y., Duan, L., Xing, J., Larssen, T., Nielsen, C.P. & Hao, J.M. 2009. Soil acidification in China: Is controlling SO_2 emissions enough? *Environ. Sci. Tech.*, 43: 8021-8026.

Zhou, G.Y., Liu, S.G., Li, Z.A., Zhang, D.Q., Tang, X.L., Zhou, C.Y., Yan, J.H. & Mo, J.M. 2006. Old-growth forests can accumulate carbon in Soils. *Science*, 314: 1417.

Zhuang, P., Zou, B., Li, N.Y. & Li, Z.A. 2009. Heavy metal contamination in soils and food crops around Dabaoshan mine in Guangdong, China: implication for human health. *Environ. Geochem. Health*, 31: 707-715.

11 | Regional assessment of soil changes in Europe and Eurasia

Regional Coordinator/Lead Author: Pavel Krasilnikov (ITPS/Russia)

Contributing Authors: Irina Alyabina (Russia), Dominique Arrouays (ITPS/France), Svyatoslav Balyuk (Ukraine), Marta Camps Arbestain (ITPS/New Zealand), Laziza Gafurova (Uzbekistan), Hakki Emrah Erdogan (Turkey), Elena Havlicek (Switzerland), Maria Konyushkova (Russia), Ramazan Kuziev (Uzbekistan), Marc van Liedekerke (EC), Vitaliy Medvedev (Ukraine), Luca Montanarella (ITPS/EC), Panos Panagos (EC), Manuela Ravina da Silva (EC), Bülent Sönmez (Turkey).

11.1 | Introduction

The majority of reports on the global state of soil degradation regard the European region as less disturbed compared with the situation in other regions. According to an ISRIC estimation (Oldeman, 1998), the average cumulative loss of productivity during the post-Second World War period due to human-induced soil degradation was estimated as 7.9 percent while in Africa it was 25 percent, and in Central America it was as high as 36.8 percent. However, the extent of soil degradation in Europe appears to be underestimated, because soil degradation on the territory of the European region has many facets, not all considered in previous estimates.

The processes of human-induced soil degradation started in many parts of the region in ancient times, because many centres of agrarian civilization emerged in Europe and Eurasia several millennia ago: Greece, Anatolia and the Amu Darya delta are just the most remarkable examples. Since that time the pressure on the land has increased because of growing populations and the intensive migration of people due to a decline in natural resources and climatic fluctuations. The western part of the European region in comparison to other regions of the world has a history of over 200 years of industrialization which have placed additional pressures on the soil.

Soil changes can occur naturally but are now under increasing threat from a wide range of anthropogenic pressures. Today these pressures represent the main reason for soil degradation in many parts of Europe. Soil resources are being over-exploited, degraded and irreversibly lost due to poor management practices, industrial activities and land-use changes. These issues in the region threaten soil's key role as the basis for provision of food, feed, fibre and energy as well as for ecosystem services and mitigation of climate change.

Knowledge on the state of soil resources in the region is good because of the generally high development of soil science and soil monitoring in the countries of the region. Nonetheless, an overview of the state of soil resources and of the development of land degradation for the whole region remains difficult because of the lack of harmonization of data, which were often obtained at different times using different methodologies (Jones and Montanarella, 2003; Morvan *et al.*, 2008).

In this chapter, we focus on anthropogenic degradation, e.g. alteration of soil properties induced by human activities that leads to declines in soil productivity and ecosystem services. The human activities in question include improper agricultural use, and soil disturbance and contamination due to urbanization, industrial and mining activities.

11.2 | Stratification of the region

The European region as considered in this report includes Europe sensu stricto plus Turkey and Eurasia. This larger definition extending beyond Europe proper entails consideration of a wider diversity of bioclimatic and soil resources and consequently of land use. The importance of agriculture varies among the countries of the region (Table 11.1). In terms of percentage area under agricultural use, the five leaders are Kazakhstan, Moldova, United Kingdom, Ukraine and Turkmenistan and the five least agricultural countries are Greenland, Norway, Finland, Sweden, and the Russian Federation. For the countries with the largest agricultural area, it should be noted that the figures require interpretation. For example, the high percentage of agricultural lands in Kazakhstan does not mean that the country has the highest pressure on natural ecosystems, because almost 90 percent of its agricultural area is in fact occupied by rangelands. The countries with the least percentage of agricultural lands are the coldest countries of the region, where bioclimatic condition do not allow the extension of agricultural activities.

Table 11.1 | The percentage of agricultural land area of total land area in the countries of the European region
Source: FAO, 2015.

Area name	Agricultural area of total land area, percent
Kazakhstan	77.00
Republic of Moldova	76.05
United Kingdom	71.35
Ukraine	71.32
Turkmenistan	69.29
Hungary	64.41
Uzbekistan	63.04
Denmark	62.82
Ireland	62.79
Romania	61.67
Serbia	57.81
Azerbaijan	57.54
Spain	57.44
Netherlands	57.12
Armenia	56.21
Kyrgyzstan	55.96
Greece	55.38
Czech Republic	55.09
Serbia and Montenegro	54.81
France	53.81
Poland	53.17
Turkey	52.27
Luxembourg	49.95
Italy	49.76
Germany	48.59
Bulgaria	47.95
The former Yugoslav Republic of Macedonia	46.78
Belgium	45.57
Belarus	44.20
Lithuania	43.69
Liechtenstein	42.44
Slovakia	42.17
Bosnia and Herzegovina	42.07
Andorra	41.81
Albania	41.69
Channel Islands	41.34
Portugal	41.17
Austria	39.59
Georgia	38.34
Montenegro	38.19
Switzerland	38.11
Tajikistan	33.47
Malta	30.34
Latvia	27.80
Slovenia	24.62
Croatia	21.98
Estonia	20.45
Iceland	18.43
San Marino	16.67
Cyprus	15.38
Russian Federation	13.17
Sweden	7.66
Finland	7.47
Norway	3.39
Greenland	0.57

The internal stratification within the region and within countries, including the extent of agricultural lands, depends mainly on bioclimatic conditions which determine agro-ecological zones. According to standard agro-ecological zoning (Fischer *et al.*, 2002), the European region lies in the following agro-ecological zones: Arctic (only Greenland and Russia); Boreal continental (Eastern Russia); Boreal sub-continental (Russia, Scandinavian countries, Greenland); Boreal oceanic (Iceland and Greenland); Temperate continental (Russian and Kazakhstan); Temperate sub-continental (Eastern and Central Europe, Turkey, South Caucasian countries, Russia, Central Asian Countries); Temperate oceanic (Western, Central and Northern Europe); Sub-tropical with winter rainfall (Southern Europe, Turkey, South Caucasus); and Sub-tropical with summer rainfall (only small areas in Spain and southern France).

The other approach for characterizing the internal stratification of the region is its division according to the major biomes. The digital map of terrestrial eco-regions presented below (Figure 11.1) delineates the major biomes found in the European region, based on the World Wildlife Fund's eco-regions (The Nature Conservancy, 2009).

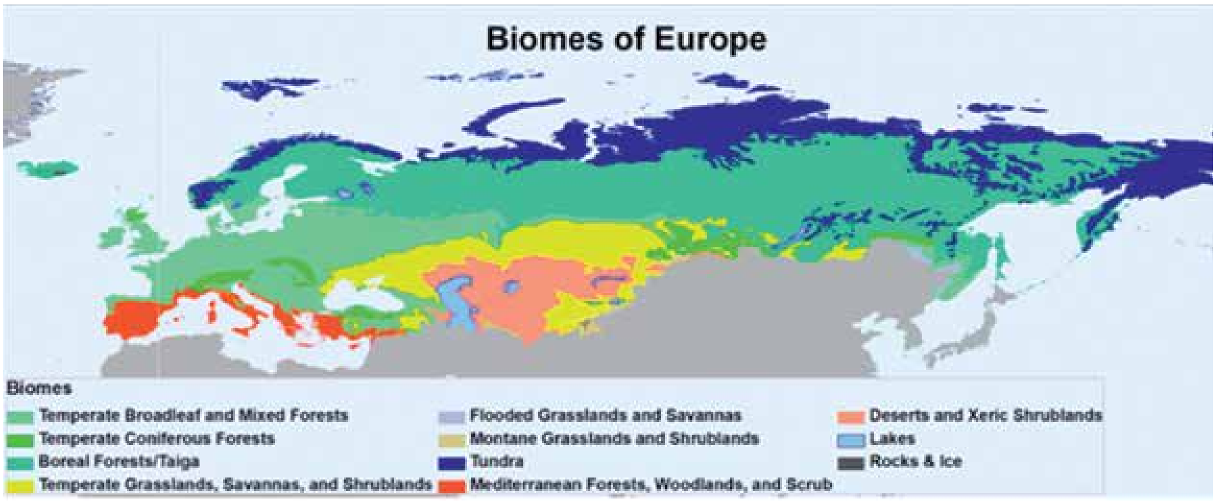


Figure 11.1 | Terrestrial eco-regions of the European region.
Source: Olson et al., 2001.

In brief, the relation of these major zones to agricultural development and soil degradation processes is the following:

Polar and tundra, and taiga zone

This zone represents a treeless polar ecosystem located in high latitudes in the European region in Russia and Scandinavia. The climate is characterized by long winters with months of total darkness and extremely frigid temperatures. Vegetation is mainly scattered, although sometimes it can be patchy, reflecting changes in soil and moisture gradients. Most precipitation falls in the form of snow during winter time. Soils tend to be acidic and saturated with water where not frozen. The region is sparsely populated, with agriculture in the tundra limited to reindeer grazing. Thus agricultural pressure on soils is not very strong; however huge areas are affected by mining and petroleum extraction. Some land degradation processes are triggered by humans indirectly. For example, in Siberia the processes of permafrost melting due to climatic change is resulting in the alteration of topography and causing severe damage to roads and buildings. Waterlogging is also a challenge in many areas.

Boreal forest/taiga

These ecosystems cover extensive areas in northerly latitudes and with low annual temperatures. There are large expanses in central and eastern Russia, with medium precipitation of 40-100 cm yr^{-1}, partly in the form of snow. Predominant tree species are coniferous including Abies, Picea, Larix and Pinus as well as deciduous such as Betula spp. And Populus spp. The ground cover is mainly dominated by mosses and lichens. These biomes are known for slow regeneration of mature forests, due to the challenging climate and soil conditions. Forests are sensitive to acid rain and other forms of pollutant. Agriculture in the taiga is restricted to relatively small areas used for livestock and production of such crops as rye, flax, millet and vegetables. Some two decades ago, soil acidification induced by industrial contamination of the atmosphere, so called 'acid rain', was an important menace to soil quality in these areas. However, today the pressure of technogenic acid precipitation is considerably reduced (Jones *et al.*, 2011).

Broad-leaf and mixed forest zone

This zone stretches across the European region from the British Isles to Western Siberia, and most of this territory is actually under cultivation. In the temperate climate, forests experience a wide variability in temperature and precipitation. Species such as oak (Quercus spp.), beech (Fagus spp.), birch (Betula spp.) and maple (Acer spp.) typify the composition of this biome. Structurally, these forests are characterized by four layers: a canopy composed of mature full-sized dominant species; a slightly lower layer of mature trees; a shrub layer; and an understory layer of grasses and other herbaceous plants. In contrast to tropical rain forests, most biodiversity is concentrated much closer to the forest floor. The zone has a favorable humid climate and soils with a relatively high natural productivity. Anthropogenic pressure is, however, strong due both to the intensive practice of agriculture and to high population density. The main threats to soils in this zone are water erosion favoured by intensive deforestation, and soil sealing and capping due to the high urbanization rate and dense infrastructure. In addition, the high degree of industrialization in this biome results in extensive contamination of the soils.

Temperate coniferous forest

This ecosystem is also known as 'temperate evergreen forest'. It sustains the highest levels of biomass of any terrestrial ecosystem after tropical rainforest. The area has warm summers and cool winters, resulting in a high variation of the vegetation, e.g. needle leaf trees, broadleaf evergreen trees or a mix of both types. Temperate evergreen forests are common in the coastal areas of regions with mild winters and heavy rainfall, or inland in drier climates or hilly areas. Predominant tree species include pine, cedar, fir and redwood. This biome is mostly located in mountainous regions and the use of these areas is not very intensive.

Temperate grassland zone

This zone possesses the soils with the highest natural productivity such as Chernozems and Kastanozems. This high potential results in an intensive use of the land for agriculture which in places occupies up to 90-95 percent of the total land area. The main threats to soils in this zone are water and wind erosion. These processes are the main reasons for the loss of organic carbon in soils; however, the loss of carbon by mineralization from arable lands is also a common process. Since the population density and the development of industry are high in this zone, soil sealing and capping and contamination are also threats. In places, especially in endorheic valleys, soil salinization may be observed.

Mediterranean zone

This zone by definition is typical of the regions around the Mediterranean Sea. The ecosystems are characterized by hot and dry summers with cool and moist winters, with precipitation mostly during the winter months. Plant species are uniquely adapted to the stresses caused long, hot and dry summers. Most of the vegetation is adapted to fire and in fact depends on this disturbance for its sustainability. Some wildlife species undertake seasonal migration according to resource availability. The natural communities in this biome are highly sensitive to habitat fragmentation, grazing, and alteration of fire regimes (over-burning or fire suppression). Native species are also at risk from exotic species that easily establish and spread. To maintain the natural communities fire regimes need to be managed and exotic species controlled (WWF, 2014). The ecosystems and soils are vulnerable due to the dryness of climate and to the abundance of shallow limestone soils. In many countries, transformation of pastures represents a serious problem. Erosion, organic carbon loss and decline in biodiversity are the main challenges for areas with Mediterranean climate. In places soil salinity may also limit the agricultural use of soils.

Sub-arid and arid zone

This zone includes deserts and semi- deserts, mostly located in Central Asia but also in some areas of Anatolia (Camci Çetin *et al.*, 2007) and in the Iberian Peninsula. The vegetation in this zone is sparse, and all the ecosystems are subject to desertification. The main threats to soils in the zone are salinization, sodification, and wind erosion. Salinization is a natural process in many areas, also favored by initially saline parent material of marine origin. However, the most menacing process is irrigation-induced soil salinization, which leads to a drastic decline in soil fertility.

11.3 | General threats to soils in the region

Soil threats in the European region are complex, and although they are unevenly spread in the region, their dimension is continental and they are frequently inter-linked. If not managed, soil threats will lead to soil degradation, and the capacity of the soil to carry out its vital ecosystem functions will be lost. When many threats occur simultaneously, the combined effect tends to aggravate soil degradation (Jones *et al.*, 2005). Climate change is likely to affect soil quality and cause land degradation through changes in soil moisture content (Calanca *et al.*, 2006; Wong *et al.*, 2011; García-Ruiz *et al.* 2011). For example, throughout central and northern Europe, evapotranspiration has increased by about 0.3 mm day^{-1} and this has the potential to deplete the normally adequate soil water store and limit plant growth. More frequent and severe droughts may cause plant cover to reduce leading to the onset of erosion and desertification (Jones *et al.*, 2011). However, the precise impacts of climate change on soil degradation in Europe are still uncertain (Kovats *et al.*, 2014). General threats to soils in the European region include the following.

(1) Erosion by wind and water

A recent report (Jones *et al.*, 2011) estimated that in the 1990s 105 million ha, or 16 percent of Europe's total land area (excluding Russia), were affected by water erosion, and that 42 million ha were affected by wind erosion. A new model of soil erosion by water constructed by the JRC has estimated the surface area affected in EU-27 at 1.3 million km², with almost 20 percent subject to soil loss in excess of 10 tonnes ha^{-1} yr^{-1}. In Russia the area affected by medium and strong water erosion is 51 million ha, 26 percent of the agricultural land area and about 3.5 percent of the total land area (Ministry of Natural Resources, 2006). In Ukraine the area affected by water and wind erosion is about one third of all agricultural land, or 14.4 million ha. In Moldova the area affected by water erosion is about 840 thousand ha, one third (33.6 percent) of the total area of arable lands in the republic (Leah, 2012). In Belorussia the area affected by water erosion is 467 thousand ha, and by wind erosion 89 thousand ha; totally eroded lands cover about 10 percent of the territory of the country. In Turkey, where 80 percent of soils are located on slopes steeper than 15 percent, the area affected by moderate, severe and very severe erosion is 61.3 million ha, or 78.7 percent of the total area of the country; wind erosion is active on about 500 thousand ha (Senol and Bayramin, 2013). The area of eroded soils in South Caucasus varies between 35 and 43 percent of total agricultural lands, aggravated by the mountainous topography of the region. In Central Asia as a whole, the total area affected by water erosion is over 30 million ha, and by wind erosion about 67 million ha: in Uzbekistan up to 80 percent of agricultural land is affected by water erosion, and in Tajikistan the area of agricultural lands affected by water erosion is estimated by different sources at between 60 and 97 percent (CACILM, 2006). Long-term observations in Russia show that soil erosion on average decreases the yield of leguminous crops by 15 percent, of wheat by 32 percent, of potatoes by 45 percent, and of perennial grasses by 25 percent (State Committee of Russian Federation on Land Resources and Land Planning, 1999).

(2) Soil organic carbon change

Organic matter is a key component of soil, controlling many vital functions (Jones *et al.*, 2011). The loss of organic matter in soils is due both to erosion and to the increased rate of mineralization of organic carbon in arable soils. Methodologically, it is difficult to separate erosion and mineralization-driven loss of humus in soils. However, soils with negligible erosion loss commonly also lose organic carbon under cultivation. The rate of soil organic matter loss differs between mineral and peat soils. In the latter group, soil degradation after drainage may be fairly quick, producing an intensive flux of carbon dioxide to the atmosphere and in places leading to complete mineralization of the organic layer and exposure of infertile underlying sediments. This threat is discussed in detail in Section 11.4.3 below.

(3) Soil contamination

Due to more than 200 years of industrialization, soil contamination is a widespread problem in Europe. The most frequent contaminants are heavy metals and mineral oil. The number of sites where potentially polluting activities have taken place now stands at approximately three million. The issue is discussed in detail below in Section 11.4.1.

(4) Soil acidification

Acidification involves the loss of base cations (e.g. calcium, magnesium, potassium, sodium) through leaching and their replacement by acidic compounds, mainly soluble aluminum and iron complexes. Acidification is always accompanied by a decrease in a soil's capacity to neutralize acid, a process which is irreversible in nature except over very long periods. Regulatory controls initiated in recent decades to mitigate global warming have had a significant impact on the emissions of pollutants that cause acidification, mainly by decreasing SO_2 emissions. By 2020, it is expected that the risk of ecosystem acidification will only be an issue in some hot spots, in particular in the border area between the Netherlands and Germany (EEA, 2010a). Recovery from acid deposition is characterised by decreased concentrations of sulphate, nitrate and aluminium in soils. An increase in pH and acid-neutralising capacity (ANC) coupled with higher concentrations of base cations would, in turn, improve the potential for biological recovery. However, given the delay in the response of soil to decreases in acid deposition, many decades are likely to be required for affected sites to recover fully. Additional information on trends in acidification is presented in the SOER 2010 Air Pollution Assessment (EEA, 2010a).

(5) Salinization and sodification

In Europe, salinization is generally the result of the accumulation of salts from irrigation water and fertilizers. High levels of salt eventually make soils unsuitable for plant growth. Improper irrigation and the use of highly mineralized irrigation water lead to rapid accumulation of soluble salts in soil profiles. This form of salinization affects approximately 3.8 million ha in Europe. Though a number of practices of saline soil reclamation exist, most of them are expensive and not very effective, and all of them are site-specific. This makes salinity control a challenging task. Geographically this threat is localized in the drier parts of the region, mostly in central Asia, in southern Russia, in Turkey, Azerbaijan, Greece, Hungary, and Spain. This threat is discussed in more detail in Section 11.4.4 below.

(6) Waterlogging

Waterlogging occurs in many soils affected either by a high groundwater table or by rainwater stagnation due to poor permeability. Most waterlogged soils have excessive moisture due to natural causes, but waterlogging may also be caused by improper irrigation practices or through disturbance of landscape hydrology by construction, mining and traffic activities. The Russian Federation is the country that has the most extensive area of waterlogged soils in the world. Excessively moist soils, both mineral and organic, occupy 360 million ha (21 percent of the total area of the country). Among the agricultural lands of Russia 23.9 million ha or 10.1 percent are waterlogged. Unlike Russia, where the distribution of waterlogging is mostly due to natural reasons, in Central Asia excessive moisture is caused by irrigation. In Uzbekistan, the groundwater table is less than 2 meters below the surface in about one-third of irrigated lands. The area of waterlogged land varies from 40 percent in the Fergana Valley to 80 percent in downstream Amu Darya. In dry areas, waterlogging is commonly associated with salinization.

(7) Nutrient imbalance

The situation with the balance of nutrients in soils is much better in the European region than in most other parts of the world. However, there is considerable heterogeneity in the distribution of nutrients in soils. In Western Europe, the concentration of nutrients in soils is high due to application of high doses of fertilizer (Grizzetti, Bouraoui and Aloe, 2007). The absolute leader in the use of nitrogen fertilizers is the Netherlands, where in places the dose exceeds 170 kg of N per ha. Germany, France and the United Kingdom also apply nitrogen fertilizers intensively. The highest doses of phosphorus fertilizers – in doses higher than 21 kg P per ha – are used in some regions of Italy, Spain, France and Greece. These regions are running a risk of contaminating the ecosystem with excessive fertilizers (e.g. for France: Lemercier *et al.*, 2008, Follain *et al.*, 2009). In Eurasia, by contrast, the use of fertilizers is much lower, although Russia and Belorussia are major exporters of mineral fertilizers. The restricted use of fertilizers in these countries is in part due to the high natural productivity of their soils, but is also due in part to economic reasons. Underutilization of fertilizers in Central Asia is caused by the fact that small farmers cannot afford them or make money out of them.

(8) Compaction

Compaction can be induced by the use of heavy machinery in agriculture. Compaction reduces the capacity of soil to store and conduct water, makes it less permeable for plant roots and increases the risk of soil loss by water erosion. Estimates of areas of Europe at risk of soil compaction vary. Some authors estimate that 36 percent of European subsoil has a high or very high susceptibility to compaction (Jones *et al.*, 2011). Other sources report 32 percent of soils as being highly susceptible and 18 percent as being moderately affected (Jones *et al.*, 2011). In Russia and Central Asia, soil compaction is also a challenge, especially where soils are naturally susceptible to compression, for example, in soils with vertic or natric properties, or in heavy textured soils. These kinds of soils are located mainly in the southern part of the Russian Federation, and in many places in Kazakhstan, Uzbekistan, Kyrgyzstan and Tajikistan.

(9) Sealing and capping

Sealing occurs when agricultural or non-developed land is lost to urban sprawl, industrial development or transport infrastructure. It normally includes the removal of topsoil layers and leads to the loss of important soil functions, such as food production, water storage or temperature regulation. The population of the Europe is approximately 11 percent of the world population (740 million) and it has grown at a rate of 0.15 percent a year during the last decade. The urbanization rate is high, particularly in the small and highly developed countries of Western Europe. It is the rapid and intensive expansion of urban and industrial development onto soils that makes soil sealing an important challenge for Europe. The issue is discussed in more detail in Section 11.4.2 below.

11.4. Major threats to soils in Europe and Eurasia

11.4.1 | Soil contamination

Local contamination of soils is generally associated with intensive industrial activities, inadequate waste disposal, mining, military activities or accidents. Management of these contaminated sites is a tiered process starting with a preliminary survey (searching for sites that are likely to be contaminated), followed by site investigations where the actual extent of contamination and its environmental impacts are defined, and finally remedial and after-care measures.

As discussed above (11.3.3), the number of sites where potentially polluting activities have taken place in Europe now stands at approximately three million. Due to improvements in data collection, the number of recorded sites is expected to grow as investigations continue. If current trends continue and no changes in legislation are made, the numbers reported above are expected to increase by 50 percent by 2025 (Jones *et al.*, 2011; EEA, 2014). There is some evidence of progress in remediation of contaminated sites, although the rate is slow. In recent years, around 17 000 sites have already been treated while many industrial plants have attempted to change their production processes to generate less waste. In addition, most countries now have legislation to control industrial wastes and prevent accidents. In theory, this should limit the introduction of pollutants into the environment. However, recent events – such as the flooding of industrial sites in Germany during extreme weather events which led to the dispersal of organic pollutants, and the collapse of a dam at an aluminium plant in Hungary in October 2010 – show that soil contamination can still occur from potentially polluting sites. Trends in the deposition of heavy metals from industrial emissions are discussed in the SOER 2010 Assessment on Air Pollution (EEA, 2010b).

Diffuse soil contamination is one of the specific threats to soils in European region. Even though this form of pollution may be only barely apparent or may not even be directly apparent at all, it can cover very large areas. The contaminants include inorganic compounds such as metallic trace-elements and radionuclide, and organic compounds like natural and xenobiotic molecules. Radionuclides originate from nuclear accidents and nuclear tests, but there are also other sources such as fertilizer application (P). The most common xenobiotic chemicals include PAH, PCB and many pesticides still in use or inherited from the past (e.g. DDT and its metabolites).

The economic feasibility of soil surveys is hindered by the diversity of contaminants – particularly of persistent organic pollutants (POPs), which are in constant evolution due to agrochemical developments – and by the transformation of organic compounds in soils by biological activity into diverse metabolites. The most widespread metallic trace elements in European soils comprise notably As, Cd, Cr, Cu, Hg, Ni, Pb, Zn, which are mobilized in one place but may then be transported to distant areas.

The real extent of diffuse soil contamination by metallic trace elements is not clearly known. Even though some EU member states and other countries have already implemented long-term soil surveys, they lack a harmonized soil monitoring system.

The case study of Austria (Section 11.5.1) includes the description of: (i) local and diffuse pollution affecting Austrian soils; (ii) the effect of regulation on contamination trends; and (iii) the remediation activities being carried out.

11.4.2 | Sealing and capping

Soil sealing is especially intensive in Western Europe. On average, built-up and other manmade areas account for around 4 percent of the total area in EEA countries (data exclude Greece, Switzerland and the United Kingdom), but not all of this is actually sealed (EEA, 2009). EU member States with high sealing rates exceeding 5 percent of the national territory are Malta, the Netherlands, Belgium, Germany and Luxembourg (EC, 2011).The EEA has produced a high resolution soil sealing layer map for the whole of Europe for the year 2006 based on the analysis of satellite images. Much more detail can be found in the SOER Assessments on the Urban Environment (EEA, 2010c) and Land Use (EEA, 2010d), as well as in EC (2011).

Productive soil continues to be lost to urban sprawl and transport infrastructure. Between 1990 and 2000, the sealed area in the EU-15 increased by 6 percent and at least 275 ha of soil were lost per day in the EU, amounting to 1 000 km² per year. Between 2000 and 2006, the EU average loss increased by 3 percent, but by 14 percent in Ireland and Cyprus, and by 15 percent in Spain (EC, 2011). A study by Huber *et al*. (2008) provides an interesting insight into the development of baselines and thresholds to monitor soil sealing. See also the SOER 2010 Assessment on Land Use (EEA, 2010b) for additional details on urbanisation.

Soil sealing causes adverse effects on soil functions, or even their complete loss, and it prevents soil from fulfilling important ecological functions. Fluxes of gas, water and energy are reduced which affects, for example, soil biodiversity. The water retention capacity and groundwater recharge function of soil are reduced, resulting in several negative impacts such as a higher risk of floods. The reduction in the ability of soil to absorb rainfall leads to rapid flow of water from sealed surfaces to river channels, resulting in damaging flood peaks. Above-ground biodiversity is affected through fragmentation of habitats and the disruption of ecological corridors. These indirect impacts affect areas much larger than the sealed areas themselves. Built-up land is lost for other uses such as agriculture and forestry, as the soils which are sealed are often fertile and high value soils. Soil sealing appears to be almost irreversible and may result in an unnecessary loss of good quality soil. Soil sealing can lead to the contamination of soil and groundwater sources because of higher volumes of unfiltered runoff water from housing, roads and industrial sites. This is exacerbated during major flood events, as was demonstrated by the 2002 floods on the Elbe which deposited levels of dioxins, PCBs and mercury from industrial storage areas onto the soils of floodplains, well in excess of national health thresholds (Umlauf *et al.*, 2005).

In the Russian Federation the density of population is low and the area of settlements constitutes only 0.3 percent of the national territory. However, the vicinities of megacities suffer intensive urban sprawl which takes land out of agriculture.

11.4.3 | Soil organic matter decline

Soil organic matter is essentially derived from residual plant and animal material, transformed (humified) by microbes and decomposed under the influence of temperature, moisture and ambient soil conditions. The stable fraction of organic substances in soil is known as complexed organic matter (previously referred as humus). Soil organic matter (SOM) plays a major role in maintaining soil functions because of its influence on soil structure and stability, water retention and soil biodiversity, and because it is a source of plant nutrients.

The primary constituent of SOM is soil organic carbon. Some 45 percent of soils in Europe have low or very low organic matter content (0–2 percent organic carbon). This is particularly evident in the soils of many southern European countries, but is also the case in parts of France, the United Kingdom, Germany, Norway and Belgium. A key driver is the conversion of woodland and grassland to arable crops.

The soils of EU-27 Member States are estimated to store between 73 and 79 billion tonnes of carbon, which is equivalent to almost 50 times annual greenhouse gas emissions from the EU. However, intensive and continuous arable production may lead to a decline of soil organic matter. In 2009, European cropland emitted an average of 0.45 tonnes of CO_2 per hectare, much of which resulted from land conversion (EEA, 2011).

The conversion of peatlands and their use is particularly worrying. For instance, although only 8 percent of the farmland in Germany is on peatland, it is responsible for about 30 percent of the total greenhouse gas emissions of its whole farming sector (EC DG Environment and JRC, 2010). However, with appropriate management practices, soil organic matter can be maintained and even increased. Apart from peatlands, particular attention should be paid to the preservation of permanent pastures and the management of forests soils, as carbon age in the latter can be as high as 400-1 000 years (EC DG Environment and JRC, 2010). Maintaining carbon stocks is essential for the fulfilment of the present and future emission reduction commitments of the EU.

In Russia more than 56 million ha of mineral soils on agricultural land are characterized by the loss of organic matter (Shoba *et al.*, 2010), in Ukraine, the area subject to loss of SOM is 18.4 million ha (Laktionova *et al.*, 2010), and in Moldova more than 1 million ha (Leah, 2012). Turkey was reported to be losing SOM from about 70 percent of its agricultural soils (Senol and Bayramin, 2013), but the rate and extent of dehumification are unknown yet. In the South Caucasus republics, the loss of soil organic matter is not well documented, largely because in mountainous countries it is hard to separate out erosion and mineralization-driven loss of humus. A similar situation exists in the data reported for Central Asia: the cultivation of virgin lands in Kazakhstan resulted in the loss of approximately 570 million tonnes of carbon from soils, but a significant part has been transported by the wind.

Several factors are responsible for a decline in SOM and many of them relate to human activity: conversion of grassland, forests and natural vegetation to arable land; deep ploughing of arable soils; drainage; fertiliser use; tillage of peat soils; crop rotations with reduced proportion of grasses; soil erosion; and wild fires (Kibblewhite *et al.*, 2005). High soil temperatures and moist conditions accelerate soil respiration and thus increase CO_2 emissions (Brito *et al.*, 2005). Excess nitrogen in the soil from high fertiliser application rates and/or low plant uptake can cause an increase in mineralisation of organic carbon which, in turn, leads to an increased loss of carbon from soils. Maximum nitrogen values are reached in areas with high livestock populations, intensive fruit and vegetable cropping, or cereal production with imbalanced fertilisation practices. While in extreme situations, the surplus soil nitrogen can be as high as 300 kg N ha^{-1} (EC, 2002), estimates show that 15 percent of land in the EU-27 exhibits a surplus in excess of 40 kg N ha^{-1}. For reference, while rates vary from crop to crop, the IRENA Mineral Fertiliser Consumption Indicator (EEA, 2005a) estimates that average application rates of nitrogen fertiliser for EU-15 in 2000 ranged from 8 to 179 kg N ha^{-1}.

There is growing realisation of the role of soil, in particular peat, as a store of carbon and recognition of soil's role in managing terrestrial fluxes of atmospheric carbon dioxide (CO_2). Other than in tropical ecosystems, soil contains about twice as much organic carbon as above-ground vegetation. Soil organic carbon stocks in the EU-27 are estimated to be between 73 to 79 billion tonnes, of which about 50 percent is to be found in the peatlands and forest soils of Sweden, Finland and the United Kingdom (Schils *et al.*, 2008). Peat soils contain the highest concentration of organic matter in all soils. Peatlands are currently under threat from unsustainable practices such as drainage, clearance for agriculture, fires, climate change and extraction. The current area of peatland in the EU is estimated at more than 318 000 km^2, mainly in the northern latitudes.

While there is no harmonised exhaustive inventory of peat stocks in Europe, the CLIMSOIL report (Schils *et al.*, 2008) estimated that more than 20 percent (65 000 km^2) of all peatlands have been drained for agriculture, 28 percent (almost 90 000 km^2) for forestry and 0.7 percent (2 273 km^2) for peat extraction. The degradation of organic soils is especially pronounced in Belorussia, where about 190 thousand ha of peat soils are strongly degraded: the peat layer was completely mineralized, and infertile sands were exposed on an area of 18.2 thousand ha. In the Russian Federation, the mineralization of peat has occurred on extensive drained areas with Histosols located mainly in the north of the European part of the country. The total area of drained peatland in Russia was estimated as 3.86 million ha (Inisheva, 2005), but the area of deeply degraded drained peat soils is unknown. Extensive areas of previously drained peatland have been abandoned as agriculture has shifted to more climatically favorable regions closer to markets. These dry peatlands are subject to fires in dry summer periods; in 2010 peat fires in Central European Russia caused an ecological catastrophe, driving millions of people from their homes and disrupting air transportation because of dense smoke over thousands of square kilometers. Another common practice in Russia is forest drainage, designed to improve forest productivity in waterlogged areas. Currently about 3 million ha of forested organic soils are drained in Russia, mainly in the European part. Though productivity has increased, the loss of organic carbon from soils is also evident. In Ireland, peatlands are widely used for energy production. Annual production of peat in the country peaked in 1995 at 8.0 million tonnes (Devlin and Talbot, 2014).

11.4.4 | Salinization and sodification

While naturally saline soils exist in certain parts of Europe, the main concern is the increase in salt content in soils resulting from human interventions such as inappropriate irrigation practices, use of salt-rich irrigation water and/or poor drainage conditions. Locally, the use of salt for de-icing can also be a contributing factor. The primary method of controlling soil salinity is to use excess water to flush the salts from the soil. In most cases where salinization is a problem, this must inevitably be done with high quality irrigation water.

Thresholds to define saline soils are highly specific and depend on the type of salt and land use practices (Huber *et al.*, 2008). Excess levels of salts are believed to affect around 3.8 million ha in Europe (EEA, 1995). While naturally saline soils occur in Spain, Hungary, Greece and Bulgaria, artificially induced salinization is affecting significant parts of Sicily and the Ebro Valley in Spain and more locally in other parts of Italy, Hungary, Greece, Portugal, France and Slovakia (Figure 11.2).

In the post-Soviet countries, saline soils cover the most extensive areas in Kazakhstan, Russia, Uzbekistan, Turkmenistan, Ukraine, and Azerbaijan (Figure 11.3). In other countries saline soils are present only locally, either on marine marsh deposits or in areas of salt mining. In Belorus, salt-affected soils occupy less than 1 thousand hectares around the potassium fertilizer mines in Solikamsk region. The area occupied by saline soils in different countries is shown in Table 11.2.

Table 11.2 | The areas of saline soils in the countries with major extent of soil salinization in the European region

Country	Area of salt-affected soil (million ha)	Data source
Kazakhstan	111.5	Borovskii, 1982
Russia	54.0	Shishov and Pankova, 2006
Uzbekistan	20.8	Kuziev and Sektimenko, 2009
Turkmenistan	14.1	Pankova, 1992
Ukraine	4.0	Novikova, 2009
Turkey	1.5	Senol and Bayramin, 2013
Spain	0.63	Tóth *et al.*, 2008
Hungary	0.56	Tóth and Szendrei, 2006
Azerbaijan	0.51	Ismayilov, 2013

Kazakhstan

The area of saline soils in Kazakhstan, including Solonetz, alkaline soils, and complexes of Solonetz with other soils, is 111.55 million ha, or 41 percent of the national territory (Borovskii, 1982). Saline soils are present everywhere in the country except in mountainous areas. They are common in the steppe zone, where they cover about 30 percent of the area. In dry steppe, semi-desert and desert zones these soils occupy up to 50 percent of the area. Salt-affected soils are represented mainly by Solonetz and alkaline soils. Solonchaks cover only 1–3 percent of the area of salt-affected soils in the steppe zone, and 7–13 percent of the area of salt-affected soils in the semi-desert and desert zones.

Russian Federation

The area of salt-affected soils in Russia, including Solonetz, alkaline soils and combinations of salt-affected soils with other soil groups, is 54 million ha or 3.3 percent of the total area of the country. The category of salt-affected soils includes all the soils with a salic horizon starting within the first 1 meter. Salt-affected soils in Russia are located in the European part of the country and in Eastern Siberia within the steppe, dry steppe and semi-desert zones. The major part of the area is occupied by Solonetz and alkaline soils. In Eastern Siberia and in the Far East, salt-affected soils are localized in closed inter-mountain basins with dry steppe landscapes and semiarid to arid climate. In Yakutia (Sakha), salt-affected soils form over permafrost in thermo-karst depressions (so-called alases) in taiga larch forest. Sporadically saline soils are present along the northern and eastern shores of the country, but their area is insignificant (Shishov and Pankova, 2006).

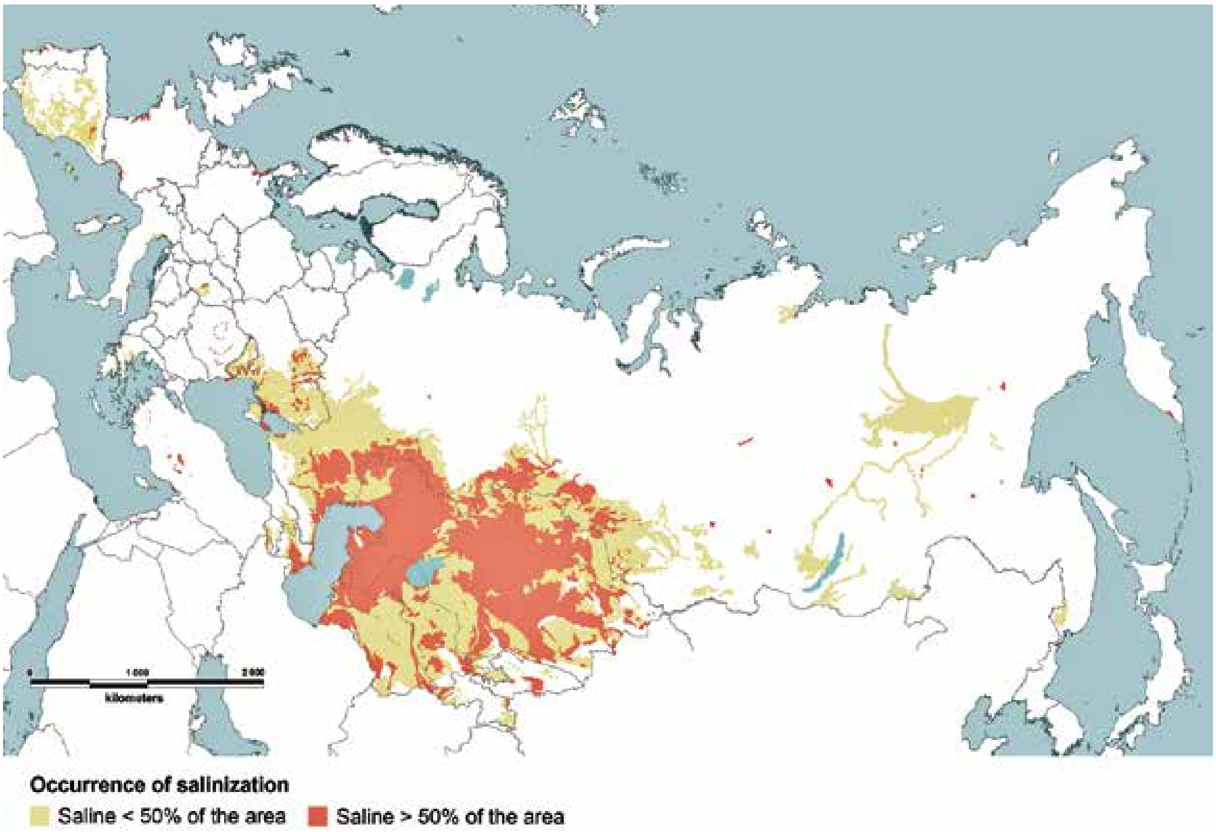


Figure 11.2 | Soil salinization on the territory of the European region.
Source: Afonin et al., 2008; Toth et al., 2008; GDRS, 1987.

Uzbekistan

The area of salt-affected soils in Uzbekistan is 20.8 million ha, or 46.5 percent of the country's territory. Saline soils are everywhere in the country except in mountainous regions and well drained flood valleys. Salt-affected soils are represented by Solonchaks in closed depressions and the bottom of the largely dried-up Aral Sea, by Salic Calcisols in uplifted areas such as the Ustyurt plateau and the Kyzyl-Kum desert, by Takyric Salic Fluvisols in the alluvial and deltaic plains, and by extensive irrigated soils of various grades of human-induced salinization.

Turkmenistan

The area of salt-affected soils in Turkmenistan is 14.1 million ha or 28.7 percent of the total area of the country. These soils are distributed all over the country, but the major concentration is in the western part of the country close to the Caspian Sea. Most of these soils are not classified as Solonchaks and belong to the groups of Salic Calcisols and Takyric Salic Calcisols.

Ukraine

The area covered by salt-affected soils in Ukraine is about 4 million ha or 6.6 percent of the national territory (Novikova, 2009). Most of these lands are not used in agriculture; they are located largely in the southern and eastern parts of the country within the steppe and forested steppe zones. Most of these soils are classified as Solonetz and other soils with various grades of alkalinity.

Turkey

The major areas with salt-affected soils in Turkey are: Konya-Eregli, the Aksaray and Malya plains of Central Anatolia, and the alluvial plains of lower Seyhan, Igdır, Menemen, Bafra, Söke, Acıpayam and Salihli. The distribution of the salt-affected arable lands is: 60 percent slightly saline, 19.6 percent saline, 0.4 percent alkali and 8 percent saline-alkali. Although sodium salts are the main components of the salt-affected soils, there are also magnesium soils in Denizli-Acıpayam, potassium-nitrate-alkali soils in Nigde- Bor, Kayseri, and gypsiferous soils in Central Anatolia.

Spain

In Spain 3 percent of the 3.5 million ha of irrigated land is severely affected, reducing markedly its agricultural potential, while another 15 percent is at serious risk. Soil salinization is a frequent problem in arid and semiarid regions like Southeast Spain (Hernández Bastida, Vela de Oro and Ortiz Silla, 2004). In these areas, demand for water for agriculture and increasing frequency of drought events have led farmers to irrigate with poor quality water. This has caused processes of soil degradation and salinization, limiting crop growth and impairing productive capacity (Pérez-Sirvent *et al.*, 2003; Acosta *et al.*, 2011).

Hungary

In Hungary Solonchak soils, which are by definition saline soils, occupy a total area of only 4.7 thousand ha. They are mainly located in low-lying areas, typically along the shorelines of saline/sodic lakes in the region between the Danube and Tisza Rivers, but they also occur in patches east of the Tisza River. These soils are not cropped, but sustain native halophyte vegetation which is grazed. Solonchak-Solonetz soils with a total area of 65.9 thousand ha are also found largely between the Danube and Tisza Rivers, but above deeper groundwater levels of around one metre. These soils also sustain a native halophyte vegetation which is grazed. Mollic Gleyic Solonetz soils occupy a total area of 274.9 thousand ha and are characterized by large exchangeable sodium percent and a not very high salt content. Mollic Solonetz soils, with a total area 212.2 thousand ha, have only minor limitations for cultivation of crops and are typically under arable farming.

Azerbaijan

The area of salt-affected soils in Azerbaijan is estimated to be 510 thousand ha or 5.9 percent of the territory of the country. The saline soils are located mainly on the coastal plain of the Caspian Sea, in the Kura-Araksinskaya depression and in the Salyan, Mugansk, and Milsk plains. These soils are represented mainly by Solonchaks, Salic Gleysols and, in rare cases, by Salic Calcisols.

While several studies show that salinization levels in soils in countries such as Spain, Greece and Hungary are increasing (De Paz *et al.*, 2004), systematic data on trends across Europe are not available.

11.5 | Case studies

11.5.1 | Case study: Austria

Austria is a relatively small country that is land-locked in central Europe and shares borders with eight countries (Gentile *et al.*, 2009). Austria's location in the middle of Europe gives rise to specific environmental issues such as the pressures from intensive freight transit traffic (e.g. air emissions, habitat disruption) and the trans-boundary exchange of acidifying air pollutants and tropospheric ozone precursors (e.g. damage to forests and soil). In addition, only 37 percent of the national territory is suitable for permanent settlements.

This is due to the country's geo-morphological conditions with more than 60 percent of the territory occupied by mountains. As a consequence, urban sprawl and land consumption occurs in restricted areas, with resulting high pressures on the environment.

Overall, the Austrian soils are in a good condition, but their ecological functionality is at risk from diffuse and local accumulation of pollutants, intensive use of land, sealing and erosion. The more affected areas are located in the Alpine region, where forest soils are threatened by air deposition, and in urban areas, where sealing and contamination put urban soils under a growing pressure. Sealing is also present in rural areas due to the increasing urban sprawl and new road construction. In arable land, According environmental measures within Cross Compliance in arable land a treatment of the soil is prohibited, if the soil is water-satisfied. Soil erosion is higher on steeper slopes and land under permanent crops (vineyards, orchards), as well as on land cultivated with maize, sugar beet, potatoes and vegetables.

The use of heavy machinery, especially in case of wet soil conditions, often results in the compaction of the topsoil, which can reach in some cases the subsoil layer. Soil compaction mainly occurs in areas with intensive agriculture and, locally, in other areas due to forest management activities. According environmental measures within Cross Compliance in arable land a treatment of the soil is prohibited, if the soil is water-satisfied.

Floods happen occasionally in the floodplains of eastern Austria after extraordinary weather conditions (heavy rainfalls) whereas landslides occur quite often in the alpine regions with steep slopes. Adequate management measures for the protection of forests and afforestation, as well as technical engineering measures, are being implemented to prevent the consequences of such events and reduce the risks.

Salinization is found in the areas around the lake Neusiedl, which is located in the north-eastern part, at the Hungarian border. Salinization is causing problems only in small areas with intensively managed and irrigated agricultural soils. Information on decline of organic matter is scarce. Some areas of arable land show low organic matter content. The evaluation of the Austrian environment programme for agriculture (ÖPUL) concerning organic matter contents in soil showed positive trends.

Contamination from heavy metals is mainly due to long-range trans-boundary air pollution, especially in forest soils due to the high filter capacity of vegetation cover and the barrier effects of the Alps to air mass circulation. Contamination by heavy metals and persistent organic pollutants can also be found in restricted areas originating from different sources, e.g. local industrial sources, traffic especially near bigger population centres or agricultural sources.

The restructuring of the industrial sector and in particular the decline of the heavy industry in the 1990s did not have observable effects on environmental pressures. Despite the decrease of the overall production, the contribution of this sector to the overall emissions is still considerable. Adverse effects of soil degradation are still to be expected, despite the continuing improvement in the implementation of regulations and the reduction of pollutant emissions, since many pollutants (e.g. heavy metals) are accumulating in the soil. The major indirect impacts are on biodiversity and the quality of groundwater resources.

The threats to soil in Austria

Contamination

Diffuse contamination

Soil surveys targeted at the four most relevant heavy metals (mercury, lead, cadmium and copper) showed increased lead and cadmium concentrations in topsoil with respect to background values in the regions of the Northern and Southern Limestone Alps. This may be attributed both to local sources of pollution and to long-range trans-boundary air pollution. Lead enrichment is particularly high in grassland and forest soils – the latter due to the high filtering effect of the vegetation cover. The guidance values for lead established by the Austrian Standard (Önorm, 1975, 2004[1]) were exceeded in more than 5 percent of monitored grassland sites, and in more than 3 percent of forest sites. Cadmium concentrations exceeded the guidance value in 5 percent of the monitoring sites in forests and in 6 percent of the sites in grassland areas.

On the other hand, soil pollution with mercury and copper only occurs in restricted areas. In particular, copper contamination can be found mostly in the surroundings of industrial sites processing copper ore and in areas with intensive animal husbandry. The latter is due to the application of high amounts of pig manure with high copper content, of which the source is copper-enriched ready-made food. Other sources of copper inputs in soil are sewage sludge and compost as well as the application of pesticides containing copper. About 2 percent of the forest and grassland monitored sites exceeded the guidance value for copper.

Contamination from Persistent Organic Pollutants (POPs) was found in a limited number of sites, some of which required to be cleaned-up. POP pollution mainly occurs in urban areas and around industries. It can also derive from long-range trans-boundary air pollution. Emissions of POPs have been substantially reduced in the past years. This should have resulted in lower concentrations in the soil. However, a systematic survey targeted at organic pollutants in soil has not been carried out. For this and other reasons, such as the low mobility of these pollutants and the appearance on the market of new chemical products, the importance of POP pollution may be expected to increase in the future (Umweltbundesamt, 2004, 2007b).

Contamination from local sources

In Austria, soil contamination requiring clean up may be present at 2 500 sites. Potentially polluting activities are estimated to have occurred at 80 000 sites (including the 2 500 sites already mentioned) and investigation is needed to establish whether remediation is required. Approximately 70 heavily contaminated sites have been cleaned up in the past two decades. Industrial production and commercial services, municipal and industrial waste treatment, and oil storage are reported to be the most important sources of heavy contamination. National reports indicate that heavy metals, polycyclic aromatic hydrocarbons, cyanides and mineral oil are the most frequent soil contaminants at investigated sites. Nearly two-thirds of the remediation expenditure come from public budgets. Although considerable efforts have been made already, it will take decades to clean up the legacy of contamination. New contamination is not expected due to the implementation of prevention measures in place.

Salinization

In Austria, salinization is of little relevance as compared to other soil threats. According to an agricultural soil mapping survey carried out in the period 1958-1970, the areas where soil is affected by salts amounted to only 2 500 ha. Conditions for potential salinization do occur in small areas in Eastern Austria. These are areas with a negative water balance, salt-sensitive soils, a low groundwater table and salty groundwater.

1 An ÖNORM standard is a national standard published by the Austrian Standards Institute. ÖNORMs are voluntary standards drawn up in committees of the Austrian Standards Institute

Future changes in climate and land management practices could lead to the salinization of soils also in these areas. In addition, soda-containing soils are estimated to cover only 2 000 ha. In these areas, strict rules for agricultural production apply.

Erosion

Soil erosion in Austria is mainly increased by unsustainable agricultural practices, construction of buildings and roads, and the use of leisure infrastructures. National estimates report that about 13 percent of the agricultural land or more than 5 percent of the total territory is potentially under a high risk for water erosion. The spatial distribution of potential erosion risk is very heterogeneous. The most affected areas include the productive areas of the southeast and northeast plains and hills, the Alpine foreland and the Carinthian basin (Strauss and Klaghofer, 2006).

Except for the results of some scientific studies, Information on wind erosion is scarce. Loss of soil by wind has been observed in the lowlands of Eastern Austria. Areas at risk are sandy soils and, in the dry season, some areas covered with black soils (chernozems). In the past, some measures, such as reforestation of lowlands, were carried out to protect soil against wind. New windbreaks are planted annually, thus increasing the protected areas by several thousand hectares per year. The presence of wind erosion risk in sandy areas has been acknowledged since the 18th century. This early recognition of the problem and the measures adopted have resulted in the stabilisation of erosion in these areas (Strauss and Klaghofer, 2006).

Soil erosion is not expected to increase in the future due to the implementation of prevention and reduction measures such as the measures included in the national Agri-Environmental Programme. In 2008 these measures reduced the soil erosion rate by 3 to 18% depending on the region, in average by 10 percent (AGES, 2011). However, major pressures may come from future climate and land use changes (conversion of grassland into arable land) or significant changes in crop rotation, although these are not very likely to occur.

Decline in soil organic matter

According to the results of a recent monitoring programme that measured the content of organic carbon in topsoil, more than half of all grassland and forest sites in Austria have a content of organic matter in topsoil of over 8 percent, which is comparatively high by global standards. On the other hand, in arable land, most of the monitored sites show an organic matter content ranging from 2 percent to 4 percent (medium by global standards), while a low or very low content (< 2 percent) is found in a quarter of the sites. In the areas with low organic matter content, the natural soil functions will be at risk in the long run (Umweltbundesamt, 2004). Measurements of organic matter content in topsoil also provide an indication of the content in soil organic carbon.

Overall, the organic carbon stock in Austrian soils is not expected to decline in the future, due to the implementation of soil organic matter preservation measures in agriculture. Such measures of the national Agri-Environmental Programme have increased the humus content of arable soils from 1991-1995 until 2006-2009 by 0.1 up to 0.4 percent depending on region (AGES, 2011).

Sealing

Following the general trend in Europe, the sealing of the soil due to the increase of built-up areas and transport infrastructure has shown a growing trend in Austria in the past decades, although population has increased only slowly.

In total, about 7 percent (or around 5 500 km²) of the country area is occupied by buildings or transport infrastructures, and about 56 percent of this area is sealed. About half of the new residential buildings in 2001 were single family dwellings or semi-detached houses which, in comparison to multi-family residences or other high-density structures, occupy a considerably larger surface area (Umweltbundesamt, 2006). In the period 2012-2014, the average increase in built-up areas was about 19 ha day^{-1}. This resulted in a daily increment of soil actually sealed of 10 ha in 2012 and 2014. This figure still exceeds by a factor eight the relevant policy target (Umweltbundesamt, 2007a). These high rates may lead to the saturation of the available space in some regions. In Vorarlberg, for example, 29 percent of the permanent settlement area is already built-up.

These increases are due to changes in the standard of living and in lifestyles and to the development of associated settlement and transport activities, rather than to population growth. This is particularly evident in rural regions where the built-up area continues to grow despite a net decrease in population.

Hydro-geological risks

Erosion and erosion control have been a major issue for a long time in Austria, due to the country's specific geo-morphological configuration. More than 60 percent of the territory is occupied by mountains. The focus of past and current activities is on the control of torrents and avalanches, as these are major threats to human life in alpine environments (Strauss and Klaghofer, 2006).

According to BMLFUW (Austrian Federal Ministry of Agriculture, Forestry, Environment and Water Managemen[2]), about 67 percent of the territory may be classified as either part of a torrent watershed, avalanche watershed or a general risk area. The regional coverage ranges from 16 percent in Burgenland to 91 percent in Tyrol. The amount of budget available for measures against these risks increased from 70 million EUR in 2001 to 148 million EUR in 2009.

In case of extraordinary weather conditions (heavy rainfalls), floods happen occasionally in the floodplains of eastern Austria. The flood events in August 2002 affected large parts of the national territory. Particularly Upper Austria and Lower Austria suffered heavy damage, as floods reached areas that were previously considered as safe. More details on this event can be found in the special chapter on floods of the 7th national State of Environment report (Umweltbundesamt, 2004).

Cross-cutting issues

Brownfields

In general, the remediation of contaminated sites based on fit-for-use remediation goals should be seen not only as bringing an improvement to the status of the environment through the restoration of soil functions but also as bringing benefits to economy and society. In Austria, there is a potential for brownfield redevelopment. The average consumption of green-field areas for housing and traffic was 7 ha day^{-1} in 2014. On the other hand, only 37 percent of the national territory is suitable for permanent settlements.

According to an unofficial definition, brownfields are sites of formerly industrial or commercial land, now derelict or underused, or sites that have been affected by former uses of the site or surrounding land. The latter may require intervention before they can be returned to beneficial use, particularly where there are contamination problems. The number of brownfield sites in Austria is in the range of 3 000-6 000, covering an area between 8 000 and 13 000 ha. According to estimates based on their previous use, about 85 percent of the industrial brownfield sites may present no or little contamination problem and could be revitalised and reused without public funding for remediation.

2 http://www.bmlfuw.gv.at/en.html

Considering an increase of industrial brownfield sites of about 3 ha per day, about a quarter of the annual land requirement for housing and economic activities could be saved by reconverting brownfields to a productive use. Some measures have been proposed to this end. These include policy measures, sustainable and innovative land management, and mechanisms for the involvement of stakeholder groups. However, redevelopment activities are yet to be started. For more details see (Umweltbundesamt, 2007b).

Soil services

The main soil services in Austria include:

- protection of groundwater and spring water in mountain areas, resulting in the availability of water in sufficient quality and quantities (about 99 percent of drinking water supplies originate from groundwater and spring water)
- high diversity and mosaic distribution of geology and soils, which enables a high level of diversity of landscapes and biodiversity
- availability of highly productive soils for agricultural and forestry production
- availability of soils for building purposes, although this function is limited due to the relatively small permanent settlement area available (37 percent of the total area of the country)
- The main impacts of soil degradation in Austria are:
- biodiversity decline from soil sealing and contamination
- impairment of groundwater quality from diffuse pollution and local contamination - there may be 2 500 sites in Austria needing remediation, of which less 3 percent have been cleaned up since 1989
- destruction of natural landscapes from soil sealing and unsustainable agricultural practices

Hot spots

The Alpine region is an environmentally sensitive area with a high level of diversity of landscapes, soils, flora and fauna. This region is under threat of acidification and contamination from deposition of air-borne pollutants on the one hand, and from erosion and landslides because of its steep slopes on the other hand.

Some industrial areas are seriously contaminated from diffuse sources. These include, in particular, the city of Linz, the Inn valley in the Tyrol, and Arnoldstein in Carinthia. In addition, a high concentration of sites which are potentially contaminated can be found in the most urbanised and industrialised regions, in particular the cities of Vienna, Linz and Salzburg, the Inn valley in the Tyrol and the Mur and Mürz valley in Styria.

Outlook

Soil resources in Austria are on average in a good condition; however soil functions are still being threatened by the deposition of airborne pollutants, by a legacy of contamination in industrial and urban areas, and by the continuing increase in the built-up area.

There are some uncertainties on future trends of soil contamination due to the lack of data on the presence of organic pollutants in soil and the appearance on the market of new chemical products whose effects on the environment are not fully understood (Umweltbundesamt, 2004 and 2007b). However, the inputs of pollutants (in particular lead, cadmium and POPs) in the soil are expected to decrease, since emissions and thus depositions are diminishing due to the implementation of regulations and preventive measures in place. On the other hand, acidifying substances, in particular NO_x from traffic sources, are expected to increase. Moreover, soil contamination and its adverse effects are still to be expected in the long run since many pollutants (e.g. heavy metals and POPs) have low mobility and high persistence and accumulate in the soil. In addition, the increase of the emissions of acidifying substances may result in an increase of the pressures on forests and forest soils. The major indirect impacts will be in terms of the loss of biodiversity and the quality of groundwater resources.

In urban and industrial areas, no new contamination is expected due to prevention measures in place. Nevertheless, the clean-up of historical contamination will continue to pose a challenge. In fact, although considerable efforts have been made already, in particular in the investigation and monitoring of contamination, a slow progress is made in the implementation of remedial measures. According to the national vision for contaminated sites the identification of all historically contaminated sites and the remediation measures at heavily contaminated sites shall be completed until 2050 (BMLFUW, 2008).

The amount of soil actually sealed through the construction of buildings and infrastructures is currently increasing at a rate of 10 ha day^{-1}. This figure exceeds ten times the national 2010 sustainability target. As in Austria only 37 percent of the territory is suitable for permanent settlements, high increases of built-up areas may also lead to the saturation of the available space. This is more likely to occur in some regions currently registering the highest rates, especially in rural areas. On the other hand, the redevelopment of brownfields and the clean-up of historical contamination are expected to provide opportunities for the reduction of the consumption of green-land, as well as opportunities for economic and technological development, and job creation. Brownfields in Austria could cover about one quarter of the current needs for land.

Soil degradation in agricultural areas, especially erosion, compaction and decline in organic matter content is not expected to increase in the future due to the implementation of prevention and reduction measures such as notably those measures included in the national Agri-Environmental Programme. For the same reasons, the organic carbon stock in Austrian soils is expected to remain stable on average.

Climate change and the development of the tourist sector may result in increased hydro-geological risks.

Investigations and monitoring of historical contamination are quite advanced. More is known on contaminated sites but remediation activities are progressing slowly. If current trends are maintained, it will take decades to clean-up a legacy of contamination. In Austria, remediation is aimed at removing the source of contamination and restoring the soil functions to a certain extent. The objective is to make the soil again fit for specific uses, in particular for the protection of groundwater resources (ultimately, the source for about 50 percent of drinking water supplies). Further progress with the clean-up of historical contamination and brownfield redevelopment will also provide opportunities for the reduction of land consumption, economic and technological development, and job creation.

11.5.2 | Case study: Ukraine

About 60 percent of the soils of Ukraine are Chernozems – soils known for their unique structure, chemical properties and inherent fertility. These soils are distinguished by a very deep (more than 1 m) humus-enriched layer, perfectly expressed granular structure, almost optimal bulk density, and a good and satisfactory stock of nutrients. However, these favourable soil properties are only present in soils under virgin ecosystems. Chernozems are the best soils in the world (the 'tsar of soil') according to Vasiliy Dokuchaev, the founder of modern soil science. However, are very sensitive to anthropogenic intervention. In particular, when intensively tilled, they rapidly degrade. As a result, in recent years these soils have been characterized in Ukraine by a significant decline of their potential productivity, equal to 80-90 million tonnes of grain annually, or up to 2 tonnes per head of the population. It has proved almost impossible to maintain good ecological conditions for these soils.

The fragility of the soils of Ukraine is well known and has been the subject of many studies. Nevertheless, this has not deterred intensive development which has led to their severe degradation. About a third of arable land is eroded, 30 percent of organic matter has been lost, approximately 40 percent of soils have a compacted layer, and stocks of nutrients have noticeably decreased. Where soils have been improved, numerous problems have also been observed (Balyuk and Medvedev, 2012).

Comparison of cultivated soils with virgin analogues shows that for the last 40-50 years the most typical processes were (Bulygin, Breus and Seminozenko, 1998; Grekov *et al.*, 2011; Medvedev, 2012):

- Loss of humus in arable soils: at the end of the 1980s, 0.5-1.5 tonnes per ha were being lost annually. Between 2005 and 2009 the rate of organic matter loss was 0.42-0.51 tonnes per ha annually.
- Increasing deficiency of labile nutrients, especially nitrogen (declining from -41.4 kg per ha in 2001 to – 56.4 kg per ha in 2009) and potassium (declining from -32.9 to – 64.2 kg per ha between 2001 and 2009).
- Acidification of Chernozems, especially in the Cherkassy and Sumy regions, where the drop in pH was 0.3-0.5 units.
- Compaction, particularly in the western forested steppe but widespread on 40 percent of arable land nationwide. Compaction is characterized by a destruction of structure, lumpiness and crusting.
- Reduction of the depth of upper layer of the soil due to erosion, reaching several centimeters in Chernozems according to modelled data, and also in the drained soils of Polissiya.
- Secondary alkalinization and salinization of irrigated soils, accompanied by a reduction of peat depth.

Among other negative processes of local importance are: contamination with radionuclides and heavy metals; waterlogging; flooding; iron, calcium carbonate and aluminum accumulation; desertification; and alkalinization and soda formation.

The types and extent of degradation of arable soils in Ukraine are shown in Table 11.3 and Figure 11.3. The estimation of soil degradation has been carried out using the technique proposed by van Lynden (1997). The sources of data have been: (i) the results of the agrichemical certification of fields conducted every five years since 1965; and (ii) the database of the National Scientific Center (the 'O.N. Sokolovsky Institute for Soil Science and Agrochemistry Research'). The database has provided information on the morphological, physical, physicochemical and chemical properties of more than 2 500 soil profiles, and also information from long field experience on tillage and application of fertilizers (Laktionova *et al.*, 2010; Grekov *et al.*, 2011).

Soil degradation in Ukraine is mainly the result of the use of inappropriate farming technology. Chernozems are vulnerable to mechanical deformation due to their low bulk density before tillage in the spring, and also to the influence of moisture owing to the low stability of the swelling smectite minerals which predominate in their mineralogical composition.

The problem has been aggravated because state and regional programmes of soil protection slowed down significantly after 1991. These programmes had obtained important results in protecting and restoring soils up to the end of the 1980s but during the last two decades measures aimed at improving soil fertility have been significantly reduced.

Soil degradation is a major problem in Ukraine. There is little realization of the threat it represents for the present and especially future generations. Issues include the absence of effective mechanisms to enforce laws on soil protection, and unbalanced and insecure land tenure. Combating soil degradation requires raising awareness at all levels of society, wide educational activity, active dissemination of knowledge, and gradual formation of a new attitude to soil resources.

Table 11.3 | Types and extent of soil degradation in Ukraine

Type of degradation	Extent, percent of arable land
Fertility decline and reduced humus content	43.2
Compaction	38.2
Sealing and crusting	37.5
Water erosion, surface wash	16.8
Soil acidification	14.1
Waterlogging	12.9
Soil pollution by radionuclides	10.9
Wind erosion: loss of topsoil	10.5
Soil contamination with pesticides and other organic contaminants	9.2
Soil contamination with heavy metals	8.0
Salinization / alkalinization	4.3
Water erosion: terrain deformation by gullying	2.6
Off-site effects of water erosion	2.5
Lowering of the soil surface	0.4
Wind erosion: terrain deformation	0.4
Desertification	0.2

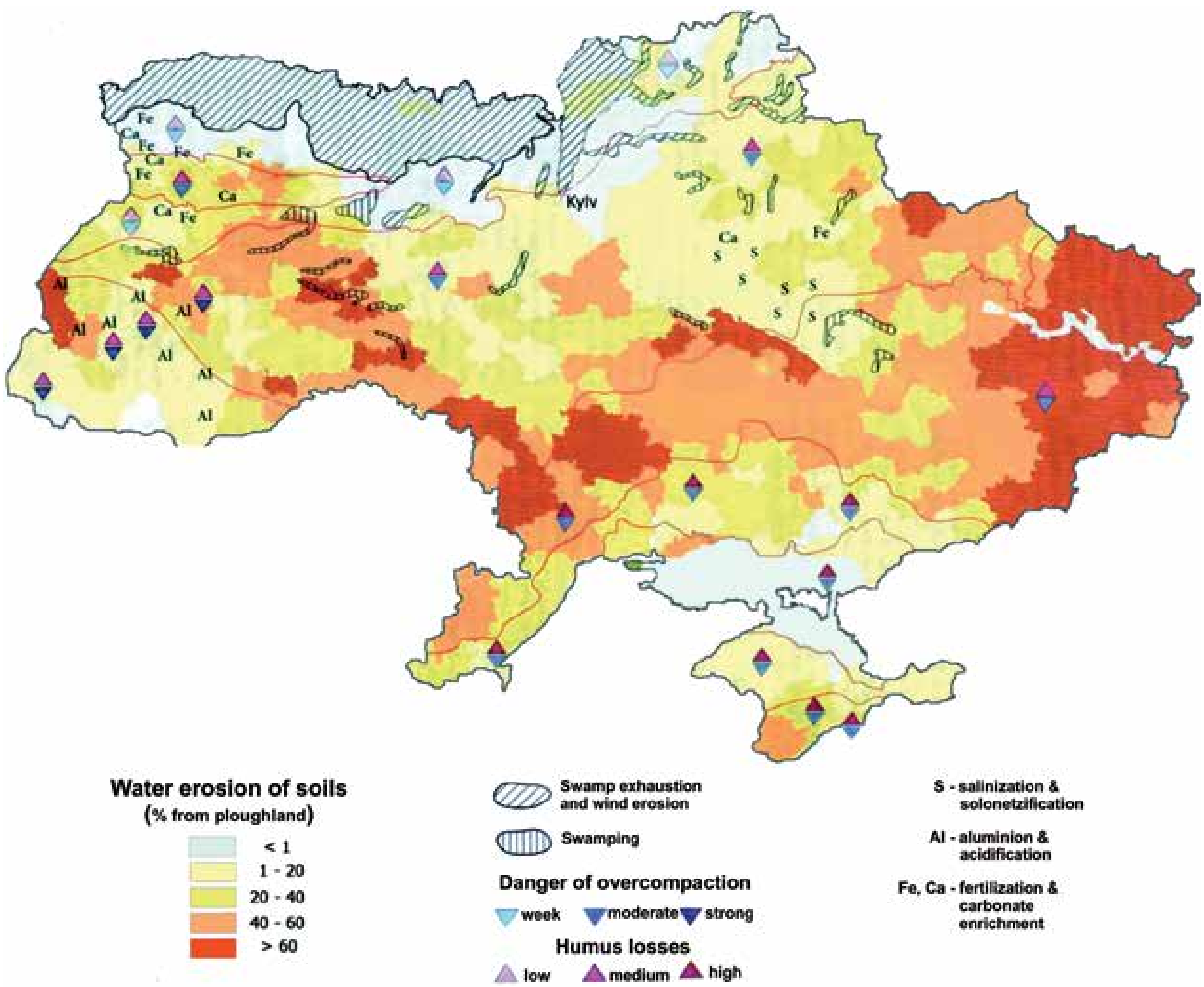


Figure 11.3 | Some types and extent of soil degradation in Ukraine.
Source: Medvedev, 2012.

11.5.3 | Case study: Uzbekistan

Uzbekistan is one of the flattest countries of Central Asia. About 80 percent of the national territory consists of plains, with mountains located in the extreme east of the republic. The climate of Uzbekistan is 'continental dry', with significant daily and seasonal fluctuations in air temperature. The summer is long and hot, the autumn is relatively wet, and the weather in winter is variable. The arid climate favors desertification and maintains relict accumulations of soluble salts in soils and sediments.

The entire area of the country is 44.9 million ha, and agricultural lands constitute 46.1 percent of the national territory. The distribution of soils in Uzbekistan reflects a complex system of pedo-geographical regularities (Figure 11.4). The westernmost part of the republic is occupied by a desert zone that can be subdivided into a sub-boreal desert (Central Kazakhstan) and a subtropical desert (Turan) desert. The boundary between the two types of desert corresponds to the northern limit of possible cotton cultivation. In the lower belt of the piedmont subtropical semi-desert there occur mainly Calcisols which are replaced as elevation increases by mountainous Cambisols under steppe and forest vegetation. Anthropogenic factors strongly modify the morphology and pedogenesis of the soils. Irrigated soils are mainly transformed into Anthrosols - some of the soils in the Amu Darya delta have been cultivated and irrigated for more than three thousand years. The country is vulnerable to negative environmental impacts due to its natural climatic conditions. Irrigated agriculture is localized in the plain and piedmont parts of the republic and is characterized by varying levels of technology and intensity and by the varying quality of the land.

The natural drivers of land degradation and desertification are the following:

- Climatic characteristics, such as aridity, continentality, wind action etc., which cause such phenomena as drought, hot winds, deflation, and atmospheric transportation of sand, salts and dust;
- Topography, with slopes favouring the development of water erosion and landslides, and flat areas with depressions creating conditions for waterlogging and salt accumulation. Topography also favours the formation of specific intensive winds which play an important role in wind erosion;
- Parent material, whose peculiarities are reflected in the soil profile (texture, gypsum and salts content) and which also determine the susceptibility of soils to wind erosion, karst phenomena, and the soil buffering capacity; and
- Extreme natural phenomena, such as forest and grassland fires, floods etc. which affect the development of soils.

Anthropogenic factors affect land resources and trigger degradation processes in the following ways:

- Irrigation without a proper drainage system and inappropriate regulation of the collector-drainage water lead to salinization and waterlogging;
- Inappropriate use of pastures leads to overgrazing, formation of exposed soil surface and destruction of the soil structure and, as a result, to the development of deflation under the effect of wind and high temperature; and
- Forestry strategy allows excessive logging which causes soil erosion on the slopes in mountainous areas, soil deflation and the expansion of sands on the fertile lands on the plains (Arabov, 2014).

Other economic activities such as industry, municipal and domestic wastes, transport emissions, and unreclaimed mining spoil also contribute to land degradation. The natural processes of land degradation and desertification are slow; their effect becomes evident only after decades or centuries. However, human activities accelerate these natural processes, and the results of anthropogenic degradation processes appear in a short period of time.

Box 1 | The catastrophe of the Aral Sea

An illustrative example of the menacing scale of ecological and socioeconomic disasters caused by inappropriate use of natural resources is the catastrophe of the Aral Sea. Its volume has been reduced more than 13 times, and its area by more than seven times. The primary cause being the diversion of inflowing rivers for irrigation projects. The shoreline has moved hundreds of kilometers. Salt concentrations have reached 120 g per liter in the western part and 280 g per liter in the eastern part of the sea (Arabov, 2014). The sea has split and is now on the verge of extinction: only two small components separated by a dam are left, the deeper western part, and the 'Small Aral' in Kazakhstan, Most of the shore is surrounded by a rind of salt cover over marshy clays and sands.

The processes of environmental change in the region have combined with global climatic changes and resulted in the intensification of seasonal droughts. The Aral catastrophe has aggravated the continentality of climate, increasing dryness and temperature in summer and prolonging cold and severe winters. In the Aral region, the number of days with the temperature over 40°C doubled, while in the rest of Uzbekistan it has gone up by about 1.5 times. According to expert evaluation, by 2035-2050 the air temperature in the region may increase by a further 1.5-3.0°C. On the dried surface of the sea bottom, there are extensive white salt crusts, covered in places by wind-blown sand. This territory forms a new desert called 'Aral Kum' that covers 5 million ha. This Aral Kum desert, which is still growing, has already absorbed 2 million ha of arable lands and led to degradation of pastures, riparian forest and other vegetation. Satellite images illustrate the penetration of plumes of salts and dust from the Aral Kum for 8 001 000 km, deep into densely populated zones.

The plains in the basins of Amu Darya and Syr Darya are lowlands with no natural drainage. Due to the dry climate, the low precipitation and the strong evaporation, these plains act as accumulators of easily soluble salts in the topsoil. For this reason, the development of irrigated agriculture, starting from the piedmont areas, requires careful attention to current or historic salt accumulation in the sediments. Farmers also have to be aware of the danger of secondary salinization. Salt-affected soils currently constitute more than 46 percent of the total irrigated area in Uzbekistan, including moderately saline soils (25 percent of the total area), and strongly saline soils (over 6 percent). The worst affected areas are the regions of Karakalpakstan, Bukhara, Khorezm, Dzhizak, Syrdarya, Andijan, Kashkadarya, Navoi, Samarkand, Surkhandarya. Some districts in Tashkent and Fergana regions are also affected. In the Samarkand region, the prevailing type of salinity is magnesium-carbonate salts accumulation. Salinization of some newly irrigated lands is followed by the formation of gypsum-enriched soils that are difficult to reclaim. Gypsum layers and horizons impede water infiltration and decrease the efficiency of leaching doses designed to wash salts from the soil profile. The total area of gypsum-containing soils is about 350 000 ha.

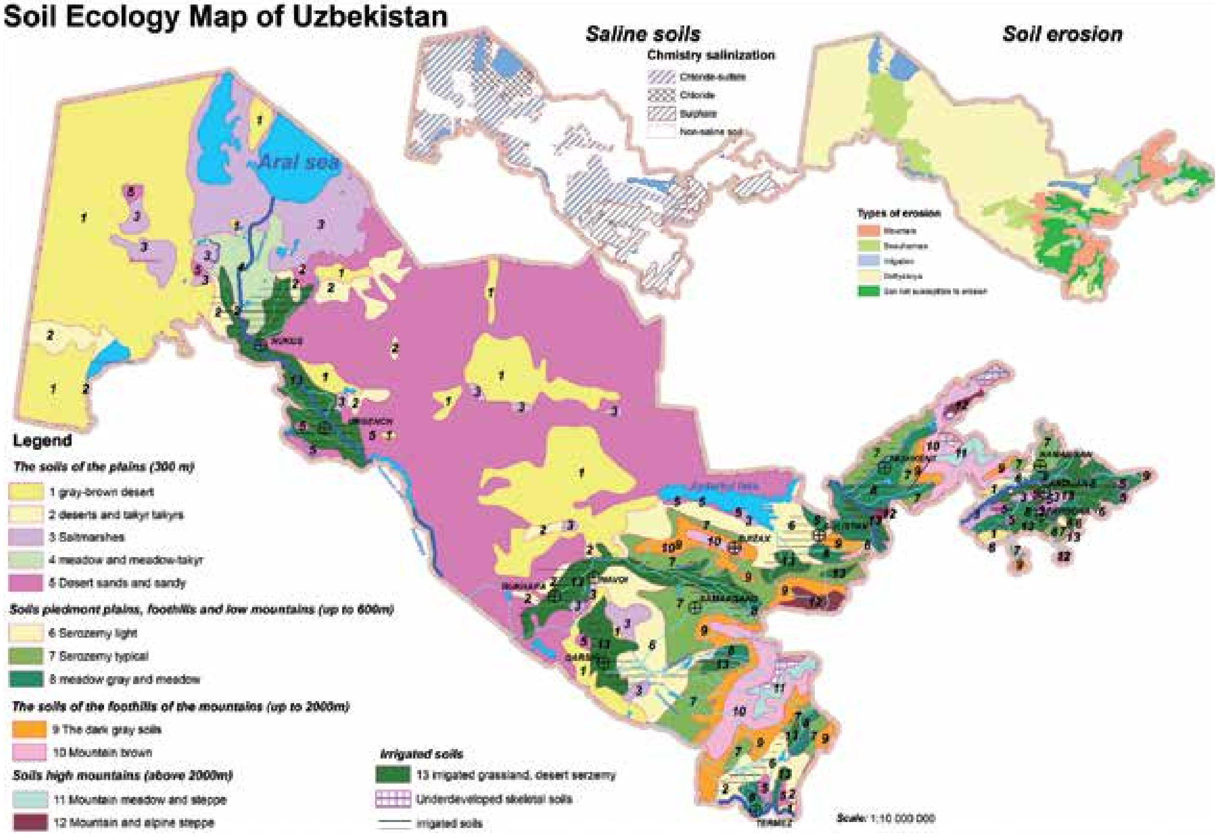


Figure 11.4 | Soil map and soil degradation extent in Uzbekistan.
Source: Arabov, 2010.

Across Uzbekistan, all types of erosion can be found: surface runoff and irrigation erosion, destructive mudflows, wind erosion, and direct negative effects of wind on plants. Wind erosion and negative wind effect on plants affect 21.4 million ha or 80 percent of all agricultural lands (Figure 11.5). Of the 3.7 million ha of irrigated lands, three quarters – 2.8 million ha – are eroded to various extents. Agricultural lands also suffer from water and irrigation erosion. Moderately and strongly eroded soils constitute 12 percent of the agricultural land pool and about 5 percent of the irrigated land pool (Kurbanov, 2001).

The major part of the country is occupied by pastures that cover an area of 20.6 million ha and serve as the main source of fodder for livestock (Arabov, 2014). Rational use and protection of the soils of pastures are issues in natural resources conservation and use. The state of pastures is currently endangered. During the last 70-80 years, the soils of pastures suffered a drastic decrease in humus and nutrient content and they have been affected by salinization and by water and wind erosion. Other negative processes include soil compaction, alkalinization, and decline in biological activity and resulting loss of soil

Thus, the key challenges in soil degradation in Uzbekistan are: (i) secondary salinization of irrigated lands; (ii) waterlogging of irrigated agricultural lands; (iii) depletion of soils, including the loss of humus and nutrients; (iv) compaction; (v) surface runoff and irrigation erosion in mountainous and piedmont areas; (vi) deflation and pasture degradation in desert regions where transhumance is practiced; (vii) deforestation and loss of biological diversity; (viii) soil contamination with agrochemicals and industrial waste; (ix) desertification in the regions bordering the bottom of the dried-up Aral Sea; (x) inappropriate methods of land management; (xi) poor crop management (lacking or wrong crop rotation, in places insufficient or excessive use of fertilizers); (xii) insufficient irrigation; and (xiii) breakdown of the rules governing sustainable management of pastures (Gafurova *et al.*, 2012).

Clearly the protection of land and soils and their sustainable economic use are huge challenges for Uzbekistan. Article 55 of the Constitution reads: "The earth, minerals, water, flora and fauna, and other natural resources are national wealth, requiring their rational use and protection of the State". In order to provide comprehensive rehabilitation, conservation, protection and improvement of soils and their fertility, and to improve overall environmental conditions, the following steps are required:

- develop agricultural techniques aimed at restoring and enhancing soil fertility, including approaches to land reclamation and the promotion of farming practices which improve the physical, chemical and ecological status of the soil
- develop rapid methods of large scale soil mapping and automate the process of compiling maps and soil assessments, using remote sensing and GIS technologies
- develop techniques for preventing the processes of surface, gully and wind erosion, for predicting soil erodibility, and for recovering and improving the fertility of eroded soils
- develop approaches to stopping processes of soil salinization and introduce more effective and innovative ways of desalinization and reclamation of saline soils
- conduct targeted research to establish the levels and pathways of soil contamination by various toxicants, including fluorine, heavy metals, pesticides and others, and develop measures to prevent soil pollution by these substances
- develop integrated science-based recommendations for the assessment of soil fertility of both arable lands and pastures to promote sustainable economic land use in the republic

11.6 | Conclusion

The inherent complexity and spatial variability of soil makes the evaluation of the impact of any change difficult. Transformations of features such as texture and mineralogical composition will only occur over geological time spans while properties such as pH, organic matter content or microbial activity will show a more rapid reaction. In addition, the response of a particular soil type may be both positive and negative depending on the function in question. For example, rising temperatures and precipitation may support increased agricultural productivity on soils previously deemed marginal, but such a transformation can lead to a deterioration of soil biological diversity and an increased risk of erosion. Quantitative assessments of future trends in soil characteristics and properties are limited. As a consequence, this chapter provides an outlook only for a selected number of issues. Considerably more effort is required to model changes in the state of soil conditions in relation to drivers such as changes in land use and climate.

Based on the above finding, an assessment is made of the status and trend of the ten soil threats in order of importance for the region. At the same time an indication is given of the reliability of these estimates (Table 11.4).

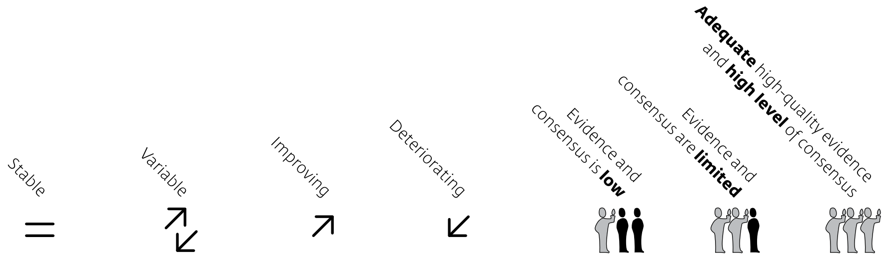


Table 11.4 | Summary of soil threats status, trends and uncertainties in Europe and Eurasia

Threat to soil function	Summary	Condition and Trend					Confidence	
		Very poor	Poor	Fair	Good	Very good	In condition	In trend
Soil sealing and land take	In densely populated Western Europe soil sealing is one of the most threatening phenomena.		↙					
Salinization and sodification	Salinization is a widespread threat in Central Asia, and it is challenging in some areas in Spain, Hungary, Turkey, and Russia.		↙					
Contamination	Soil contamination is a widespread problem in Europe. The most frequent contaminants are heavy metals and mineral oil. The situation is improving in most regions.		↗					
Organic carbon change	The loss of organic carbon is evident in most agricultural soils. Peatland drainage in northern countries also leads to rapid organic carbon loss. In Russia, extensive areas of agricultural lands were abandoned that resulted in quick organic mater accumulation; however, some of these areas are now again used for agriculture.		↗↙					
Nutrient imbalance	In the western part of the region the loss of nutrients is compensated by application of high doses of fertilizers. In the eastern part the use of fertilizers is insufficient, and in most soils nutrient mining results in intensive mineral weathering.		↗↙					
Soil erosion	Water erosion is active in all the cultivated mountainous and rolling areas; the worst situation is observed in Turkey, Tajikistan and Kyrgyzstan. Due to the attention paid to this threat it is controlled in most areas, especially in the EU.			↗				

Loss of soil biodiversity	Loss of biodiversity is expected in the most urbanized and contaminated areas of the region. However, there are almost no qualitative estimations of the biodiversity loss in soils.			↙				
Soil acidification	Acidification due to acid rain was a challenge in Northern and Western Europe. The situation is now improving, though several decades will be needed for complete soil recovery.			↗				
Waterlogging	Waterlogging is mostly associated with irrigation in Central Asian countries. Most cultivated irrigated soils there are waterlogged. This phenomena in Central Asia is commonly associated with salinization.			↗↙				
Compaction	The use of heavy machinery and overgrazing are threatening in almost all the agricultural areas.			↗↙				

References

Acosta, J.A., Faz, A., Jansen, B., Kalbitz, K. & Martínez-Martínez, S. 2011. Assessment of salinity status in intensively cultivated soils under semiarid climate, Murcia, SE Spain. *Journal of Arid Environments*, 75(11): 1056-1066.

Afonin, A.N., Greene, S.L., Dzyubenko, N.I. & Frolov, A.N. (eds.) 2008. *Interactive Agricultural ecological atlas of Russia and neighboring countries. Economic Plants and their Diseases, Pests and Weeds.* AgroAtlas.

AGES. 2011. *Bodenschutz durch umweltgerechte Landwirtschaft*. Wien

Arabov, S.A. (ed.) 2010. *Atlas of soil resources of the Republic of Uzbekistan*. Tashkent, Goskomzemgeodezkadastr of the Republic of Uzbekistan, State Scientific Production Enterprise Kartografiya. [in Russian]

Arabov, S.A. (ed.) 2014. *National report on the state of land resources of the Republic of Uzbekistan*. Tashkent, Goskomzemgeodezkadastr of the Republic of Uzbekistan, State Scientific Production Enterprise Kartografiya. [in Russian]

Balyuk S.A. & Medvedev, V.V. (eds). 2012. *Strategy of balanced use, reproduction and management of soil resources of Ukraine*. Kiev, Agrarian science. 240 pp. [in Ukranian]

BMLFUW. 2008. (*Bau)Land in Sicht (translation: Land for Building in Sight – Good reasons to recycle industrial and commercial brownfields*). Federal Ministry of Agriculture, Forestry, Environment and Water Management.

Borovskii, V.M. 1982. Formation of saline soils and haologeochemical regions of Kazakhstan. Alma-Ata, Nauka publ. 256 pp. [in Russian]

Brito, L.F., La Scala Jr, N., Merques Jr, J. & Pereira, G.T. 2005. Temporal variability of CO_2 soil emissions and its relation with soil temperature in different positions of the landscape passing to areas farmed with sugarcane. *In Simpósio sobre Plantio direto e Meio ambiente; Seqüestro de carbono e qualidade da agua, pp. 210-212. Anais. Foz do Iguaçu, 18-20 de Maio 2005*. [in Portuguese]

Bulygin, S.J., Breus, N.M. & Seminozenko, 1998. To a technique of definition of a degree of soil erodity on slopes. *Pochvovedenie* J., 6: 714-718. [in Russain]

CACILM. 2006. *CACILM Multicountry Partnership Framework Project Document*. Central Asian Countries Initiative for Land Management, Asian Development Bank. 70 pp.

Calanca, P., Roesch, A., Jasper, K. & Wild, M. 2006. Global warming and the summertime evapotranspiration regime of the Alpine region. *Climatic Change*, 79: 65–78.

Camci Çetin, S., Karaca, A., Haktanir, K. & Yildiz, H. 2007. Global Attention to Turkey Due to Desertification. *Environ Monit Assess*, 128: 489–493.

de Paz, J.M., Visconti, F., Zapata, R. & Sanchez, J. 2004. Integration of two simple models in a geographical information system to evaluate salinization risk in irrigated land of the Valencian Community, Spain. *Soil Use and Management*, 20: 333-342.

Devlin, G. & Talbot, B. 2014. Deriving cooperative biomass resource transport supply strategies in meeting co-firing energy regulations: A case for peat and wood fibre in Ireland. *Applied Energy*, 113: 1700-1709.

EC DG Environment & JRC. 2012. The State of Soil in Europe - A contribution of the JRC to the European Environment Agency's Environment State and Outlook Report – SOER 2010. No. EUR 25186 EN.

EC. 2002. Implementation of Council Directive 91/676EEC concerning the protection of waters against the pollution caused by nitrates from agricultural sources. Synthesis from year 2000. European Commission, Brussels.

EC. 2011. *Overview of best practices for limiting soil sealing or mitigating its effects in EU-27*, by G. Prokop, H. Jobstmann & A. Schönbauer. Final report of a study contract for the European Commission, DG Environment. Brussels, European Communities.

EEA. 1995. In D. Stanners & P. Boureau, eds. *Europe's Environment: The Dobris Assessment*. Office for Official Publications of the European Communities, Luxemburg

EEA. 2005a Agriculture and environment in EU – 15 – The IRENA indicator report. EEA report no 6/2005. European Environment Agency.

EEA. 2011. European Pollutant Release and Transfer Register. (Also available at http://prtr.ec.europa.eu/pgAbout.aspx)

EEA. 2009. *Degree of soil sealing 100 m — EEA Fast Track Service Precursor on Land Monitoring*. ETC/LUSI. European Environment Agency.

EEA. 2010b. *The European environment — state and outlook 2010: air pollution*. Copenhagen, European Environment Agency.

EEA. 2010c. *Exposure of ecosystems to acidification, eutrophication and ozone* (*CSI 005*). European Environment Agency.

EEA. 2010d. *The European environment — state and outlook 2010: land use*. Copenhagen, European Environment Agency.

EEA. 2014. *Progress in Management of Contaminated Sites*. Copenhagen, European Environment Agency.

EEA. 2010a. *The European Environment State and Outlook 2010: Freshwater Quality*. Denmark, Copenhagen.

Fischer, G., Van Velthuyzen, H.T., Shah, M.M. & Nachtergaele, F.O. 2002. Global agro-ecological assessment for agriculture in the 21st century: methodology and results. IIASA Research Report RR-02-002. Laxenburg.

FAO. 2015. *FAOSTAT*. Rome, FAO. (Also available at http://faostat 3.fao.org/home/E)

Follain, S., Schvartz, C., Denoroy, P., Villette, C., Arrouays, D., Walter, C., Lemercier, B. & Saby, N.P.A. 2009. From quantitative to agronomic assessment of soil available phosphorus content of French arable topsoils. *Agronomy for Sustainable Development*, 29: 371-380.

Gafurova, L., Abdrakhmanov, T., Zhabborov, Z. & Saidova, M. 2012. Soil degradation. *Tashkent*. pp. 18-204. [In Uzbek]

García-Ruiz, J.M., López-Moreno, J.I., Vicente-Serrano, S.M., Lasanta-Martínez, T. & Baguería, S. 2011. Mediterranean water resources in a global change scenario. *Earth-Science Reviews*, 105(3-4): 121-139.

GDRS. 1987. *General Management Planning of Turkey, Soil Conservation Main Plan*. General Directorate of Rural Services. Ministry of Agriculture, Forestry and Villages. Ankara.

Gentile, A.R., Barceló-Cordón, S. & Van Liedekerke, M. 2009. *Soil Country Analyses – Austria*. Luxembourg, Office for Official Publications of the European Communities. 56 pp.

Grekov, Datsko, L.V., Zhilkin, V.A., Maistrenko, M.I. & Datsko, M.O. 2011. *Methodical instructions for soil protection*. Kyiv, The State Center of Soil Fertility Protection. 108 pp. [in Ukranian]

Grizzetti, B., Bouraoui, F. & Aloe, A. 2007. *Spatialised European Nutrient Balance*. Institute for Environment and Sustainability. EUR 22692 EN

Hernández Bastida, J.A., Vela de Oro, N. & Ortiz Silla, R. 2004. Electrolytic Conductivity of Semiarid Soils (Southeastern Spain) in Relation to Ion Composition. *Arid Land Research and Management*, 18(3): 265-281.

Huber, S.,Prokop, G., Arrouays, D., Banko, G., Bispo, A., Jones, R.J.A., Kibblewhite, M.G., Lexer, W., Möller, A., Rickson, R.J.;,Shishkov, T., Stephens, M., Toth, G., Van den Akker, J.J.H., Varallyay, G., Verheijen, F.G.A. & Jones, A.R. (eds.) 2008. Environmental Assessment of Soil for Monitoring: Volume I Indicators and Criteria. EUR 23490 EN/1. Office for the Official Publication of the European Communities, Luxembourg, 339 pp.

Inisheva, L.I. (ed.). 2005. *Concept of protection and rational use of peatlands of Russia*. Tomsk, Central Peat Research Institute Publ. 99 pp. [In Russian]

Ismayilov, A. 2013. Soil Resources of Azerbaijan. In Y. Yigini, P. Panagos & L. Montanarella, eds. *Soil Resources of Mediterranean and Caucasus Countries*, pp. 16-36. Luxembourg, Office for Official Publications of the European Communities.

Jones, J.A. & Montanarella, L. (eds). 2003. Land Degradation. European Soils Bureau Research Report # 10. EUR 20559 EN. 324pp.

Jones, R.J.A., Hiederer, B., Rusco, F. & Montanarella, L. 2005. Estimating organic carbon in the soils of Europe for policy support. European Journal of Soil Science 56: 655 -671.

Jones, A., Panagos, P., Barcelo, S., Bouraoui, F., Bosco, C., Dewitte, O., Gardi, C., Erhard, M., Hervás, J., Hiederer, R., Jeffery, S., Lükewille, A., Marmo, L., Montanarella, L., Olazábal, C., Petersen, J.-E., Penizek, V., Strassburger, T., Tóth, G., Van Den Eeckhaut, M., Van Liedekerke, M., Verheijen, F., Viestova, E. & Yigini, Y. 2011. *The State of Soil in Europe*. Luxembourg, Publications Office of the European Union. 71 pp.

Kibblewhite, M. Jones, R.J.A., Baritz, R. Huber, S., Arrouays, D., Michéli, E. & Dufour, M.J.D. 2005. Environmental Assessment of Soil for Monitoring. European Commission Desertification meeting. Brussels, 12-13 Oct 2005.

Kovats, R.S., Valentini, R., Bouwer, L.M., Georgopoulou, E., Jacob, D., Martin, E., Rounsevell, M. & Soussana, J.-F. 2014. *Europe. In* V.R. Barros, C.B. Field, D.J. Dokken, M.D. Mastrandrea, K.J. Mach, T.E. Bilir, M. Chatterjee, K.L. Ebi, Y.O. Estrada, R.C. Genova, B. Girma, E.S. Kissel, A.N. Levy, S. MacCracken, P.R. Mastrandrea & L.L. White, eds. *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects*, pp. 1267-1326. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. UK, Cambridge, Cambridge University Press & USA, New York, NY.

Kurbanov, E.K. (ed.). 2001. *Atlas: Land Resources of the Republic of Uzbekistan*. Tashkent, Goskomzemgeodezkadastr of the Republic of Uzbekistan, Publ. Patent-press. [In Russian]

Kuziev, R.K. & Sektimenko, V.E. 2009. *The Soils of Uzbekistan*. Tashkent, Extremum Press. 351 pp. [in Russian]

Laktionova, T.M., Medvedev, V.V., Savchenko, K.V., Bihun, Shejko, S.M. & Nakisko, S.G. 2010. *Structure and the order of data base using Soils Properties of Ukraine*. (Instruction). Kharkiv, Apostrophe. 96 pp. [in Ukranian]

Leah, T. 2012. Land resources management and soil degradation factors in the Republic of Moldova. In *The 3rd International Symposium, Agrarian Economy and Rural Development - realities and perspectives for Romania*, 11-13 October 2012, pp. 194-200. Romania, Bucharest.

Lemercier, B., Gaudin, L., Walter, C., Aurousseau, P., Arrouays, D., Schvartz, C., Saby, N.P.A., Follain, S. & Abrassart, J. 2008. Soil phosphorus monitoring at the regional level by means of a soil test database. *Soil Use and Management*, 24: 131-138.

Medvedev, V.V. 2012. *Soil monitoring of the Ukraine. The concept. Results. Tasks. (2-nd reconsidered and added edition).* Kharkiv, The City printing house. 536 pp. [in Russian]

Ministry of Natural Resources. 2006. *State Report On the State and Protection of Environment in Russian Federation in 2005. Land Resources of Rsuuian Federation for the 1st of January 2006*. Moscow. 45 pp. [in Russian]

Morvan, X.P.P., Saby, N.P.A., Arrouays, D., Le Bas, C., Jones, R.J.A., Verheijen, F.G.A., Bellamy, P.H., Stephens, M. & Kibblewhite, M.G. 2008. Soil monitoring in Europe: a review of existing systems and requirements for harmonisation. *Sci. Tot. Env.*, 391: 1-12.

Novikova, A.V. 2009. *The study of saline and solonetz soils: their genesis, melioration, and ecology. Kharkiv, Dkukarnya*. 720 pp. [in Russian]

Oldeman, L.R. 1998. *Soil Degradation: A Threat to Food Security?* Report 98/01. The Netherlands, Wageningen, International Soil Reference and Information Centre. 14 pp.

Olson, D.M., Dinerstein, E., Wikramanayake, E.D., Burgess, N.D., Powell, G.V.N., Underwood, E.C., D'Amico, J.A., Itoua, I., Strand, H.E., Morrison, J.C., Loucks, C.J., Allnutt, T.F., Ricketts, T.H., Kura, Y., Lamoreux, J.F., Wettengel, W.W., Hedao, P. & Kassem, K.R. 2001. Terrestrial ecoregions of the world: a new map of life on Earth. *Bioscience*, 51(11): 933-938.

Pankova, E.I. 1992. *Genesis of salinization in the soils of deserts*. Moscow, Dokuchaev Soil Science Institute Publ. 136 pp. (In Russian)

Pérez-Sirvent, C., Martínez-Sánchez, M.J. & Sánchez, J.V. 2003. The role of low-quality irrigation water in the desertification of semi-arid zones in Murcia, SE Spain. *Geoderma*, 113: 109-125.

Schils, R., Kuikman, P., Liski, J., Van Oijen, M., Smith, P., Webb, J., Alm, J., Somogyi, Z., Van den Akker, J., Billett, M., Emmett, B., Evans, C., Lindner, M., Palosuo, T., Bellamy, P., Jandl, R. & Hiederer, R. 2008. Review of existing information on the interrelations between soil and climate change (ClimSoil). Final report. Brussels, European Commission

Senol, S. & Bayramin, I. 2013. Soil Resources of Turkey. In Y. Yigini, P. Panagos & L. Montanarella, eds. **Soil Resources of Mediterranean and Caucasus Countries**, pp. 225-237. Luxembourg, Office for Official Publications of the European Communities.

Shishov, L.L. & Pankova, E.I. (eds.) 2006. *Salt-affected Soils of Russia*. Moscow, Publ. Center Akademkniga. 857 pp. [in Russian]

Shoba, S.A., Alyabina, I.O., Kolesnikova, V.M., Molchanov, E.N., Rojkov, V.A., Stolbovoi, V.S., Urusevskaya, I.S., Sheremet, B.V. & Konyushkov D.E. 2010. *Soil Resources of Russia*. *Soil-Geographic Database. Moscow*, GEOS. [in Russian]

State Committee of Russian Federation on Land Resources and Land Planning. 2000. *Report 1999*. Pskov. 68 pp. (In Russian)

Strauss, P. & Klaghofer, E. 2006. Status of soil erosion in Austria. *In* J. Boardman & J. Poesen, eds. *Soil Erosion in Europe*, pp. 205–212. London & New York, John Wiley.

The Nature Conservancy. 2009. Conservation connections. Annual Report 2009. Worldwide Office. Virginia, USA.

Tóth, G., Adhikari. K., Várallyay, Gy., Tóth, T., Bódis, K. & Stolbovoy, V. 2008. Updated Map of Salt Affected Soils in the European Union. In G. Tóth, L. Montanarella, & E. Rusco, eds. 2008. *Threats to Soil Quality in Europe EUR 23438 EN*, pp. 65-77. Luxembourg, Office for Official Publications of the European Communities. 151 pp.

Tóth, T. & Szendre[i], G. 2006. Types and distribution of salt affected soils in Hungary and the characterisation of the processes of salt accumulation. *In Topographia Mineralogica Hungariae*, pp. 7-20. Budapest, Akadémiai Kiadó. 103 pp. (In Hungarian)

Umlauf, G., Bidoglio, G., Christoph, E., Kampheus, J., Krűger, F. Landmann, D., Schulz, A.J., Schwartz, R., Severin, K., Stachel, B. & Dorit, S. 2005. The situation of PCDD/F's and Dioxine-like PCB's after the flooding of river Elbe and Mulde in 2002. *Acta Hydrochim. Hydrobiol*, 33(5): 543 – 554.

Umweltbundesamt. 2004. **The environmental situation in Austria**. 7th Report on the State of the Environment inn Austria, report of the Federal Minister of Environment to the National Assembly of the Austrian Parliament. Federal environment agency.

Umweltbundesamt. 2006. *Website on spatial planning - land consumption*. Federal environment agency.

Umweltbundesamt. 2007a. *8th Report on the State of the Environment in Austria*. Vienna.

Umweltbundesamt. 2007b. Website on brownfields in Austria.

van Lynden, G.W.J. 1997. *Guidelines for the Assessment of Soil Degradation in Central and Eastern Europe*. SOVEUR Project. Revised edition. The Netherlands, Wageningen, ISRIC. 22 pp.

Wong, W.K., B. Stein, E. Torill, Ingjerd H. & Hege, H. 2011. Climate change effects on spatiotemporal patterns of hydroclimatological summer droughts in Norway. *Journal of Hydrometeorology*, 12(6): 1205-1220.

WWF. 2014. *Terrestrial Ecoregions*. World Wildlife Fund. (Also available at http://www.worldwildlife.org/biome-categories/terrestrial-ecoregions)

12 | Regional assessment of soil changes in Latin America and the Caribbean

Regional Coordinator: Maria de Lourdes Mendonça-Santos (ITPS/Brazil)

Regional Lead Author: Juan Comerma (Venezuela)

Contributing Authors: Julio Alegre (ITPS/Peru), Ildefonso Pla Sentis (Spain), Carlos Cruz Gaistardo (Mexico), Rodrigo Vargas (Mexico), Diego Tassinari (Brazil), Moacir de Souza Dias Junior (Brazil), Sebastián Santayana Vela (Peru), Maria Laura Corso (Argentina), Vanina Pietragalla (Argentina), María Nery Urquiza Rodríguez (Cuba), Candelario Alemán García (Cuba), Segundo Sacramento Urquiaga Caballero (Peru/Brazil), Maria de Lourdes Mendonça-Santos (ITPS/Brazil), Miguel Taboada (ITPS/Argentina), Olegario Muniz (Cuba), Carlos Henriquez (ITPS/Costa Rica) and David Espinosa (ITPS/Mexico).

12.1 | Introduction

This chapter discusses the status of soil resources in Latin America and the Caribbean (LAC). The LAC region, as defined by FAO, includes the following countries: Antigua and Barbuda, Argentina, Bahamas, Barbados, Belize, Bolivia, Brazil, Chile, Colombia, Costa Rica, Cuba, Dominica, Dominican Republic, Ecuador, El Salvador, Grenada, Guatemala, Guyana, Haiti, Honduras, Jamaica, Mexico, Nicaragua, Panama, Paraguay, Peru, Saint Lucia, Saint Kitts and Nevis, Saint Vincent and the Grenadines, Suriname, Trinidad and Tobago, Uruguay and Venezuela. The emphasis will be on human-induced 'anthropogenic' changes, and not on natural causes, although it is not always easy to separate them.

The LAC region extends from Mexico at latitude 32 degrees north to Tierra del Fuego at 55 degrees south. It also includes the Caribbean islands. The presence of 12 out of 14 terrestrial biomes (Olson *et al.*, 2001), the rugged relief in Central America and along the western side of South America, and the large lowlands in central South America that also include interior wetlands, all combine to make LAC the most bio-diverse region in the world. In fact, eight of the 17 mega-diverse countries of the world are located in Latin America: Bolivia, Brazil, Colombia, Costa Rica, Ecuador, Mexico, Peru, and Venezuela.

In terms of natural resources, Latin America is one of the richest regions of the world. With only 8 percent of the population, it has 23 percent of the world's potential cropland, 12 percent of the actually cultivated land, 46 percent of the globe's tropical forest and 31 percent of the planet's fresh water (Garret, 1997). The region could provide a further 800 million ha of land for agriculture (Laegreid, Bockman and Kaarstad, 1999). However, most of these potential areas are under tropical rainforest and clearing would cause severe environmental changes with dramatic effects on many ecosystem functions. Agricultural conversion of natural ecosystems (grass-shrub-savannas and forest) as a percentage in LAC is of the order of 30 percent, representing slightly over 600 million ha of agro ecosystems, a figure similar to Africa but smaller than Europe and much smaller than Asia. Compared to other regions of the world, these converted agro-ecosystems had a medium to low intensity use of fertilizers and irrigation for much of the 20th century (UNDP, 2000). However, use of fertilizer and irrigation increased dramatically in recent decades as agriculture expanded onto the temperate and subtropical plains (Grau and Aide, 2008; Viglizzo and Jobbagy, 2010).

Another important characteristic of this region is that agriculture started in the mountains, mainly because of the presence of serious diseases in the lowlands, notably malaria. Consequently, the sloping lands of the Andes and the Central America Mountains have been cultivated the longest, starting with the Incas and Mayas. Mountainous areas in LAC still have high populations that practice both intensive agriculture like horticulture and more extensive land uses like pasture and coffee growing. Exploitation of the flatter lowlands such as the Pampas and Cerrados, with tropical, subtropical and temperate climates, started only more recently. This is where most of the present intensive farming takes place, including the use of fertilizers and machinery in the production of cereals, legumes and other crops (Viglizzo and Jobbagy, 2010).

The main soil threats are related to natural features of physiography and to the type of vegetation cover. Anthropic and cultural features also play an important role, especially inappropriate agricultural practices which are a consequence of inequitable and insecure land tenure, insufficient research and lack of extension services.

Water erosion and landslides are prominent threats in the sloping lands of the mountains, especially when the slopes have been burned and overgrazed. Loss of soil carbon mostly occurs after deforestation and change of land use to permanent grassland. In semiarid and arid areas where irrigation is applied, salinity and sodicity are important threats. In areas with more intense land use and the employment of heavy farm machinery, compaction occurs. Also in these areas, the use of amendments, fertilizers and other agrochemicals bring the threats of nutrient imbalance, acidification, contamination and loss of biodiversity. In addition to induced flooding on rice fields, waterlogging and flooding are on the rise in the region as a consequence of the combination of higher rainfall promoted by the phenomenon of ENSO (the El Niño Southern Oscillation), and by land use changes which decrease ground cover by perennial vegetation. Acidification also occurs, mostly in deltas where the drainage of marine clays produces acid sulphate soils.

Overall, the most important ecosystems services affected in LAC are: (1) climate regulation, through the carbon and nitrogen cycles, especially due to the immense deforestation rate up to 2004 of the humid tropical forests, mostly in the Amazon Basin; (2) water regulation, through changes in quantity and quality of water production in the mountains, which is also due to deforestation of sloping lands accompanied by strong water erosion and landslides; and (3) loss of biodiversity, another ecosystem service threatened by deforestation and change in land use/land cover (Viglizzo and Frank, 2006; Gardi *et al.*, 2014).

12.2 | Biomes, ecoregions and general soil threats in the region.

In order to give information on soil, land use and ecosystem services affected by soil threats, this chapter uses the 'Biomes' (environmental zones with similar climate, fauna and flora) outlined in the Soil Atlas of Latin America and the Caribbean (Gardi *et al.*, 2014, Figure 12.1). The boundaries of the Biomes correlate with major soil-landscapes. We will describe and discuss the Biomes in sequence starting from the most extensive. However, this sequence does not necessarily follow the significance and gravity of soil threats.

Figure 12.1 | Biomes in Latin America and the Caribbean.
Source: Olson et al., 2001.

Tropical and Subtropical Moist Broadleaf Forest is by far the most extensive Biome in LAC. Essentially it is a permanent very humid, high temperature forest. The landscape goes from flat plains to rolling slopes. The soils are dominated by medium to highly weathered Acrisols, Ferralsols and Plinthosols, which are in general acid and unfertile. This Biome has the greatest biodiversity of plant and animals and also shows little resilience to human intervention. It spreads from the eastern coasts of Mexico and Central America, through part of the Caribbean islands, the Pacific Coast of Colombia and the Atlantic coast of Guyana and Venezuela, down to most of the Amazon basin in Ecuador, Peru and Brazil, and finally to the western and southern coast of Brazil.

Because this Biome has had the largest deforestation rate in LAC (FAO, 2005, 2010) and because this practice removes a large and continuous addition of organic matter to the soil, the loss of land cover and soil organic carbon is clearly the most important threat to ecosystem function. Another threat in deforested areas is water erosion, which occurs when heavy rains fall on bare soil, resulting in major erosion problems on the widespread gentle and moderate slopes. As most of the nutrients in these soils are contained in the organic matter of the topsoil, its removal and rapid decomposition affects soil biodiversity, and in the long run nutrient imbalances may appear. Other ecosystem services affected include: the carbon and nitrogen cycles and their contribution to climate regulation; water quality and quantity; and landscape stability.

Tropical and Subtropical Grasslands, Savannahs and Shrub lands are the second largest Biome in LAC. The climate is characterized by alternating wet and dry periods and by high temperatures. The vegetation consists predominantly of grasses with different densities of trees. In general, the topography is flat to gently undulating. For the most part, and especially in Brazil (Cerrados), eastern Venezuela and Colombia, the dominant soils are acid lowfertility Acrisols, Ferralsols and Arenosols, while in the younger surfaces, the dominant soils are more fertile Luvisols, Phaeozems and Vertisols.

In these areas, the principal crops are annuals such as corn, soybean, sorghum, beans, cassava and cotton, grasses like Brachiarias, and legumes like Stylosanthes. In recent years, sugarcane and other biofuel crops have also been planted (Miyake *et al.*, 2012). Brazil, with the largest area of savannahs in LAC (around 200 million ha), has 55 million ha of introduced pastures and 22 million ha under annual crops (Sousa, 2011). In general, the soils in this Biome are low in organic matter, very infertile, and acid throughout. Counteracting the acidity and infertility of these soil conditions requires the use of amendments including gypsum and limestone, fertilizers such as rock phosphate, and inoculants. Erosion has been the main threat, but following major research and extension programmes, conservation tillage is increasing and is having an impact on the problem. Another threat has been compaction, not only by farm machinery but also by overgrazing. In grasslands where minimal fertilizer is used, a problem of nutrient imbalance has been reported (Guimarães, 2013). In agricultural land too, there is a growing imbalance of nutrients, especially in recent years with the intensification of agriculture, where fertilizer use has fallen well short of the demand of the crops (Urquiaga *et al.*, 2014). This is a threat throughout LAC. However, inputs of fertilizer have increased in recent years.

Ecosystem services are affected both positively and negatively in this Biome. The rise in food production is the main positive effect. However, there are significant negative effects too: the carbon and nitrogen cycles are affected by the higher rate of organic matter decomposition; there is loss of water quality due to erosion and sediment movement; and there are biodiversity losses (Viglizzo and Frank, 2006; Viglizzo and Jobbagy, 2010).

Deserts and Xeric Shrublands occupy the third place amongst LAC Biomes in terms of area. They are characterized by low precipitation, high evaporation and quite windy conditions. The temperatures are variable: in the tropics, for example in northern Brazil, the coastal area of Peru and the Caribbean coast of Venezuela and Colombia, temperatures are quite warm; in northern and central Mexico and Chile, by contrast, a cold period occurs. In all cases, vegetation is dominated by cactuses and thorny shrubs that are very sparse and resistant to drought. The most typical soils are calcareous or gypsiferous, shallow, saline or sodic and reflect the very limited amount of leaching. Calcisols, Gypsisols, Arenosols, Regosols and Solonchaks are common soil groups.

Because of its aridity, this Biome requires irrigation for most forms of agricultural development. Irrigation requires not only a source of water, but also water of good quality and a good drainage system to leach the soluble salts common in this Biome. Preventing salinization is difficult but restoring salinized soils is even harder. The use of waste water from cities may bring additional problems, particularly contamination with heavy metals. Due to the sparse vegetation coverage and the presence of strong winds, it is common for wind erosion to increase after human intervention.

The principal ecosystem service that may be affected is the productivity of the soils due to salinization. Water quality may worsen for the same reason.

Temperate Grasslands, Savannahs and Shrublands have been widely studied and documented by various authors (Paruelo, Guerschman and Verón, 2005; Satorre, 2005; Viglizzo and Frank, 2006; Álvarez *et al.*, 2009; Lavado and Taboada, 2009; Viglizzo and Jobbágy, 2010). This Biome is predominantly located in the Argentinian Pampas. Its central plains are dominated by grasses on flat to gently sloping lands, with a temperate climate, and rains ranging from 1 500 mm in the northeast to 400 mm in the southwest. These areas have some of the most fertile soils of the world, the Phaeozems, although more than 13 million ha with natural saline-sodic soils (e.g. Solonetzs) also appear in this Biome.

Despite the wide adoption of no-tillage, intensive annual cultivation (largely of soybean) and the lack of rotation with other crops or pastures have resulted in soil degradation by wind and water erosion, waterlogging, compaction, sealing/capping, and soil fertility depletion (Satorre, 2005; Lavado and Taboada, 2009; Alvarez *et al.*, 2009; Sainz Rozas *et al.*, 2011; Viglizzo and Jobbagy, 2010). Clearing of forested lowlands to produce annual crops (soybean, cotton etc.) has also led to salinization or sodification in areas where the groundwater table has risen (Paruelo, Guerschman and Verón, 2005; Viglizzo and Jobbagy, 2010).

Dry Tropical and Subtropical Broadleaf Forest occurs in many different zones in LAC. These zones share common characteristics: they are all situated below 1 000 m elevation, all experience high temperatures, and all have at least one dry season when the trees lose their leaves. These forests are more common in hilly and mountainous landscapes with soils of medium fertility such as Luvisols, and Cambisols. Large populations live in these areas and deforestation and annual burning are common. These forests occur in western Mexico, Costa Rica, Cuba, the north of Venezuela and Colombia, the coasts of Ecuador and Peru, and central and the north east of Brazil.

This Biome is very attractive for agricultural development. The soils are fertile and the climatic conditions are favourable for human habitation and for the growth of many crops, including corn, beans, potatoes, sugarcane, fruits and coffee. Large areas of grassland are used for extensive pasturing. Deforestation is the major threat, and is in practice irreversible as the dry period makes it difficult for natural regrowth to occur. As the accumulation of organic matter in these soils is medium or low, deforestation also brings the threat of organic carbon loss, and the increase of soil temperature in bare soils accelerates decomposition. Steeper slopes are hard to cultivate and here the most common land use after deforestation is extensive pasture. After a few years, soil compaction develops, particularly along the small terraces where the animals pass. This compaction also reduces rainfall infiltration and consequently increases water erosion.

Ecosystem services affected are principally the carbon and nitrogen cycles and climate regulation due to the decrease in organic matter. Water production and quality may also decrease due to compaction by overgrazing and farm machinery which increases erosion. As erosion reaches high levels in many slopes, landscape stability can also be affected.

Montane Grasslands and Shrublands are present at high altitudes throughout the Andes, but mostly in Peru and Bolivia. Locally they are called Punas or Paramos. They occur above 3 000 m or even higher, and are dominated by grasses and small shrubs. Temperatures during the day may be quite high but frosts can occur at night. In general these area are quite dry. The topography may be mountainous or large mesas or high plains. Soils are generally rich in organic matter, but rather shallow. Predominant groups are Regosols, Leptosols and, in the drier zones, Solonchaks.

Most of the land remains bare or is used for extensive pastures. Most of the pastures are natural, but some have been introduced and have adapted to the conditions. In some parts of LAC, especially in northern South America, some areas are also used for intensive horticultural crops, including potatoes, carrots and quinoa. In these cases, conservation practices, irrigation and high inputs of organic and inorganic fertilizers are used throughout the year (Comerma, Larralde and Soriano, 1971). Although there are no data on impacts, it is suspected that contamination of soils and water from excessive use of fertilizers and other agrochemicals may be occurring. This can also affect soil biodiversity, which may be very important given the unique vegetation and fauna of this Biome.

This Biome is of high importance for certain ecosystem services, notably water production. It is located at the top of many watersheds and is a continuous source of pristine water, in many cases related to the process of thawing. Many soils in the Biome are rich in organic matter in the topsoil. Careful attention has to be paid to C and N cycles because of the services provided. The soil gene population, due to its unique nature, should also be studied and protected.

Tropical and Subtropical Coniferous Forests occur at high and medium altitudes, mostly in Mexico and south to Nicaragua. Small patches also exist in the Dominican Republic, southern Brazil and along the Chile-Argentine border. These forests occur mostly in mountainous landscapes and on many different geologic materials, including volcanic ashes. Most of the soils are Umbrisols, Luvisols, Leptosols and Andosols. The vegetation is dominated by many types of conifer with a diverse understory.

Deforestation for the establishment of pastures and wood harvesting are major land uses. In many cases these land uses are accompanied by burning. Deforestation and burning create the threats of reduced organic carbon and of water erosion. The main ecosystem services affected are water production/ regulation, C and N cycling, and landscape stability.

Temperate Broadleaf and Mixed Forest is confined to a temperate permanent forest in southern Chile. The forest occurs mainly in valleys and on the slopes of high mountains. The climate is very humid, cold and with few variations during the year. Soils are shallow and mostly Alisols, Andosols, Cambisols, and Histosols.

Because of the low temperatures and the rough relief, the land is largely a protected area, and only a few valleys are used for pasture. As human intervention is very low, threats and impacts for ecosystem services are very limited.

Flooded Grasslands and Savannahs in LAC occur in tropical alluvial plains, mostly in Brazil, Bolivia and Paraguay. The Pantanal, which stretches across all three countries, is the world's largest tropical wetland, and is a highly productive environment. The area is recognized by UNESCO as a World Natural Heritage Site and Biosphere Reserve. Other important areas occur in Venezuela and Colombia on flat alluvial plains. They all have in common the predominance of native pastures adapted to flooding, few trees at higher elevations, and a seasonal period of flooding alternating with a dry season. The predominant soils are Gleysols, Stagnosols, Vertisols, Plinthosols and Histosols. The most widespread use of this biome in LAC is as pasture for bovine and, more recently, bubaline cattle. Its strategic importance in the production system is in the supply of green pastures for the dry season.

Economic development in the Pantanal region, especially on the plateau of the Rio Taquari Basin, has intensified the input of sediments to the Pantanal lowlands, causing serious social, economic and environmental impacts on the region (Galdino, Vieira and Pellegrin, 2006).

The main ecosystem service is the provision of food and fiber, interacting with the service of water regulation. The ecosystem is unique and rich in flora and fauna. Soil biodiversity is of prime importance and should be investigated and protected.

Mediterranean Forest and Shrubs occupy a small strip of Chile near the coast. The Biome has a warm temperate climate, dry during the warm period and rainy in the winter time. The vegetation, relief and soils are very heterogeneous, as they represent a transition between tropics and temperate, and between dry and humid. The most common soil groups are Regosols, Leptosols and Andisols. The degree of human intervention is so complete that there are few remnants of the original vegetation.

Because of its Mediterranean climate, the natural productivity of the area is very high. This has been utilized by Chileans to create very important agricultural development areas, especially in the valleys. Irrigation, tillage and large quantities of fertilizers and other agrochemicals are used to obtain high yields. Deforestation has resulted in significant erosion threats; high levels of input use have led to contamination; and in the drier zones salinization is a threat due to the quality of irrigation water.

The ecosystem service positively affected is the provision of food. Negative effects are on water quality and possibly on human health, and loss of biodiversity related to contamination.

Mangroves are the smallest Biome in LAC in terms of area. They occur along the coastlines of all LAC countries and are particularly associated with the deltas of important rivers like the Amazon and the Orinoco. They are located where there is the mixture of fresh and saline water. Mangroves provide many ecological services and are considered very fragile as disturbances can produce irreversible consequences. Soils are mostly Histosols, Gleysols and Acid Sulphate soils.

Mangrove areas are mostly protected because they are hard to drain and the soils are poor, with predominance of reduced or organic soils. In cases where development projects have been implemented, the main land uses are pasture, rice, or plantation crops such as oil palm or bananas. Especially when mangroves are underlain by marine sediments, it is very common that after drainage, oxidation of Iron Sulphide (pyrite) will occur, producing extreme acidification and the formation of acid sulphate soils. This is an extreme case of the threat of acidification and reclamation is extremely difficult. After drainage, organic carbon is lost and the land surface subsides.

The ecosystem service principally affected is the reduction of productivity, especially if acid sulphate soils are formed. The Carbon and Nitrogen cycles and water quality will also be affected. Cultural heritage will also be affected, as local tribes – for example, the Waraos in the Orinoco Delta – have been living on the natural products extracted from this ecosystem for more than 4000 years.

12.3. General soil threats in the region

In recent years, the onset of climate changes, notably more intense and concentrated rainfall events and higher evaporation, has begun to bring change in the pace and intensity of threats such as erosion, flooding and desertification (IPCC, 2014). At the same time, other threats to soils and ecosystem services have also increased, including threats to organic carbon, biodiversity, crop production, and water quantity and quality.

12.3.1 | Erosion by water and wind

This threat is considered one of the most important in the region, because it has an impact on very large populations, particularly those concentrated in the mountainous regions of the Andes, Central America, Mexico and the Caribbean. Water erosion and landslides occur mainly on steep slopes that have been deforested or in dry mountain areas which are used as pastures and which have been overgrazed. Water erosion also occurs on the gentler slopes of the cerrados as well as in parts of the pampas subject to intensive cultivation. Erosion and landslides remove fertile topsoil, affecting crop productivity, making tillage more difficult, and producing sediments that affect fields and infrastructure downstream and cause flooding in flat areas. The threat is considered in more detail in Section 12.4.1.

12.3.2 | Soil organic carbon change

Changes in organic carbon occur mostly if carbon supplied by vegetation decreases, as happens with deforestation, or if mineralization is increased as happens with ploughing (Sánchez, 1981). LAC contains about half of the world's tropical forest, and until recently, it had the highest rate of deforestation, which drastically reduced organic inputs to soils. The region also has some of the best soils in the world, for example in the Pampas. These soils are rich in organic matter content and very fertile, but continuous cropping has increased mineralization and reduced organic carbon. In both cases, soils are becoming less productive, limiting their ecological services (Lavado and Taboada, 2009; Viglizzo and Jobbagy, 2010; Gardi *et al.*, 2014). A recent study (D'Accunto, Semmartin and Ghersa, 2014) found that uncropped agricultural borders are highly effective in the mitigation of soil organic carbon losses. The details of changes in SOC are outlined in Section 12.4.2.

12.3.3 | Salinization and sodification

Natural or primary salinization and sodification are quite common in the arid and semiarid regions of LAC, including in Mexico, Cuba, northern South America, Peru, Northeast Brazil and southern Argentine. Human-induced threats are also present in these regions because irrigation is common and the quality of the water used and the lack of drainage induce salinization. Even though it may not occupy large areas, salt accumulation is an important threat because it severely reduces crop productivity. It is very difficult to prevent and even more difficult to reclaim soils once salinized. It has been estimated (AQUASTAT, 1997) that 18.4 million ha in LAC are affected by salinization caused by irrigation. The problem also appears in humid climates, where topsoil in large plains with high and saline groundwater and sodic soils (e.g. Solonetzes) may be salinized by upward soluble salt rises promoted by land clearing and overgrazing (Taboada, Rubio and Chaneton, 2011, Bandera, 2013, Di Bella *et al.*, 2015). A country-by-country analysis is provided in Section 12.4.3.

12.3.4 | Nutrient imbalance

The largest proportion of soils in LAC are acid and have naturally low fertility. As a result, amendments and fertilizers are required for sustainable production. These inputs also serve to boost productivity where areas for agriculture and animal production are already limited and expansion of production requires intensification. In recent years, contrasting cases of nutrient imbalance have emerged as a result of the high levels of N-fertilizers (ammonia) applied in high-input production systems (Espinosa and Molina, 1999). These N-fertilizers increase the acid ion sources in the soils, even in very fertile soils. Nutrient impoverishment has been documented in coffee and sugar cane plantations on Andisols, reducing yields (Bertsch *et al.*, 2002). Nutrient imbalance could also appear in the highly fertile Pampas region, where farmers have historically applied low amount of fertilizers (Lavado and Taboada, 2009). The imbalance could be greater were it not for the significant contribution of biological nitrogen fixation (BNF) in agriculture and livestock production of the region (Urquiaga *et al.*, 2012; Franzluebbers, Sawchika and Taboada, 2014). The contribution of BNF is particularly important for soybean in the Pampas and for legume pastures in the Southern Cone (Alves, Boddey and Urquiaga, 2003; Campillo *et al.*, 2003, 2005).

12.3.5 | Loss of soil biodiversity

In LAC, conservation areas can provide a bank of original soil biodiversity (Alegre, Pashansi and Lavelle, 1996, White *et al.*, 2005; Urquiaga *et al.*, 2014). One study (Ferraro and Ghersa, 2007) found that the community of microarthopods was highly sensitive to crop management and resulting soil conditions. Microbiological indices and enzymatic activity are useful new indicators and have been used in different studies in the region (Balotta *et al.*, 2004; Sicardi, Garcìa-Préchac and Frioni, 2004; Nogueira *et al.*, 2006; Green *et al.*, 2007; Franchini *et al.*, 2007; de Moraes Sa and Lal, 2009; Romaniuk *et al.*, 2012; Henríquez *et al.*, 2014).

12.3.6 | Compaction

In LAC there are two main causes of compaction: livestock production and machinery transit. Overgrazing by cattle and sheep causes degradation of pastures and increased erosion as found in mountainous regions, the Cerrado, coffee plantations and the arid Patagonia (Bertiller, Ares and Bisigato, 2002; Henríquez *et al.*, 2011; Taboada, Rubio and Chaneton, 2011; Pais *et al.*, 2013). The widespread use of farm machinery, especially in the Cerrados and the Pampas has caused shallow soil compaction and poor structural conditions in topsoil, especially associated with soybean mono-cropping and long winter fallow periods (Taboada *et al.*, 1998; Botta, Tolon-Becerra and Melcon, 2009; Botta *et al.*, 2010; Alvarez *et al.*, 2014; Franzluebbers *et al.*, 2014).

12.3.7 | Waterlogging

Many flat areas are affected by episodic human-induced waterlogging and ponding. This can result from poor topsoil structural and drainage conditions which limit infiltration rates. It may also be associated with extreme rainfall events linked to the periodic climate phenomena of 'El Niño'. Waterlogging has been documented in both agricultural and pasture soils throughout the region. There are increasing problems of catastrophic flooding, landslides and sedimentation (Pla, 2003, 1996a, 2011; Restrepo *et al.*, 2006).

12.3.8 | Soil acidification

Natural acidification is a common process in the soils of LAC and is very intense in tropical areas of the region, because of the high rainfall. Acidic soil parent materials are also widespread (Kämpf, Curi and Marques, 2009; Fassbender and Bornemisza, 1987). As industrialization is not intensive and widespread in the region, soil pollution from industrial sources is not common. Anthropogenic acidification in soil could also appear where there is excessive N-fertilization on crops like banana, vegetables and oil palm and under intensive coffee systems (Sánchez, 1981; Espinosa and Molina, 1999).

12.3.9 | Soil contamination

Different human activities may result in the pollution of soils and adjoining water bodies caused by fertilizers and agrochemicals used in high-input agriculture, and from mining and oil spills (Nriaugu, 1994; Malm, 1998; Mol *et al.*, 2001). Residues of herbicides such as glyphosate have been observed in soils and groundwater in fields devoted to no-till farming (Ometo *et al.*, 2000; Christoffoleti *et al.*, 2008; Cerdeira *et al.*, 2011; Aparicio *et al.*, 2013).

The use of mercury compounds in mining activities and the use of huge amounts of water for shale oil exploitation are causing downstream pollution in soils and waters (Nriaugu, 1994; Malm, 1998; Mol *et al.*, 2001). The increasing use of agricultural by-products and sludges increase N concentrations in groundwater and cause eutrophication of lagoons (Torri and Lavado, 2008).

12.3.10 | Sealing

According to the United Nations[1] , the population of Latin America is currently 630 million, 8.6 percent of the world's population. This figure is expected to increase by 25 percent by 2050. Most Latin American countries are experiencing relatively low death rates and declining birth rates, resulting in slower population growth over time.

Today 79.8 percent of the population of Latin America live in urban areas, and they account for 12.7 percent of the world's urban population. Forecasts for 2050 indicate a further increase of the share of the urbanized population in LAC to 86.2 percent. The degree of urbanization is well above the global average of 54.0 percent. Figure 12.2 illustrates the extent of urban areas and of urbanization for LAC countries. In relation to population, Brazil, Argentina and Mexico have the largest urban areas, while the highest rates of urbanization are associated with small states with high population densities.

1 http://www.un.org/en/development/desa/population/events/ot her/10/index.shtml

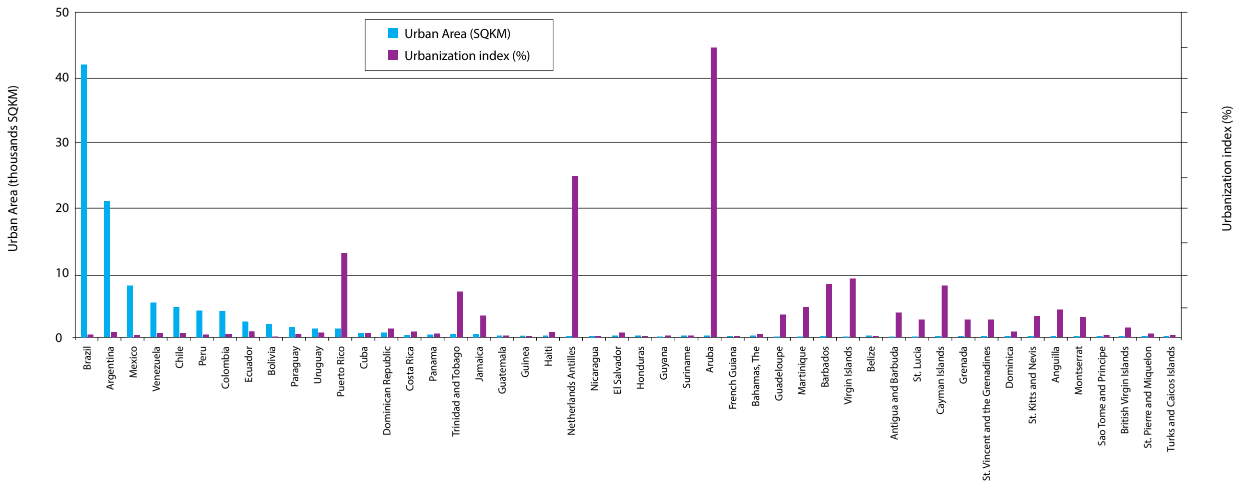


Figure 12.2 | Extent of the urban area and the urbanization index for Latin American and Caribbean countries.

12.4 | Major threats to soils

Among the general threats to soil that occur in the region, the three most important ones in LAC will be discussed here in more detail.

12.4.1 | Soil erosion

Soil erosion by water is the main soil degradation process worldwide and in LAC as well. Wind erosion is also prevalent in specific areas with arid and semiarid climates (rainfall lower than 600 mm).

A high proportion of the land in LAC is on steep slopes, and the main limitation for its agricultural use is water erosion (Alegre, Felipe-Morales and La Torre, 1990; Pla, 1993, 1996a; Duvert *et al.*, 2010; PNUMA-CEPAL, 2010). However, the problems of accelerated water erosion are not confined to steep slopes, but are also widespread in agricultural areas with more gentle slopes. In general, the increasing trend of erosion in LAC is mainly due to the fast growth of the human population and to pressures put on the land by deforestation and over-grazing, and by inappropriate agricultural practices in both subsistence and large-scale high-input commercial agriculture (Pla, 1996a).

Erosion in LAC is mostly caused by water. Some estimates suggest that 42 percent of flood events contribute to 70 percent of sediment export (Duvert *et al.*, 2010). In drier areas of Mexico and Argentina on the other hand, wind erosion prevails. Estimations of areas affected by erosion in the region vary, but a conservative figure is around 15 percent for South America and 26 percent for Central America (Oldeman, 1991b).

Although there is clear evidence that large and increasing areas of land are being affected by different processes of soil erosion, most of the existing evaluations of the type, extent and intensity of soil erosion at country or regional level are not very precise or objective. Mass and landslide erosion processes are usually not differentiated from surface erosion problems (Hincapié and Ramirez, 2010), leading to an often faulty identification of the origin of erosion processes (Pla, 1992, 1993, 2011). The magnitude of the soil erosion problems is highly variable, with estimated values for average soil losses in the Andes and Central America ranging between 5 and 50 percent of the area or from 100 to 1 000 Mg km^{-2} yr^{-1}. Probably almost half of agricultural lands are negatively affected by surface soil erosion at different levels, with 15-25 percent, depending on the region, strongly affected (Oldeman *et al.*, 1991a, b).

New regional information has been generated largely through indirect or remote sensing, usually without sufficient ground-truthing (Bai *et al.*, 2008; Nachtergaele, Petri and Biancalani, 2011). Some indirect recent evaluations based only on soil cover and slope show a reduction in the area with risks of soil erosion to only 5 percent (EC, 2013). In those evaluations, the processes of mass erosion have not received any attention or have been confused with the very different processes of surface erosion (Pla, 2011).

The control of soil erosion and of the derived effects depend on appropriate land use and management planning supported by appropriate soil governance (FAO, 2012). In many LAC countries, the application of conservation measures is limited by lack of integration between conservation and development, the lack of legislation or ways to implement it, and the shortage of basic local information, trained personnel and financial resources (Pla, 1996b). With few isolated exceptions, there have not been policies and well-targeted subsidies or incentives through marketing prices and credits to induce sustainable land management. In addition, agricultural research has been oriented towards increasing productivity through the use of inputs rather than to sustainable land use. This has contributed indirectly to growing environmental problems, including soil erosion.

A main objective of research on soil erosion in LAC must therefore now be to collect and evaluate data to generate technology for prediction and control of soil erosion. An understanding of the basic erosion processes in each particular situation is required in order to select an effective technology and transfer it to farmers (Pla, 2003). Some empirical models to predict erosion (USLE, RUSLE) are currently used indiscriminately, without scientific evidence of their applicability to a particular situation. The uncritical use of results from these models has led to gross errors in planning land use and management (Pla, 2011). However, the RUSLE equation has been successfully adopted as a basis for soil use regulations in some countries (Alegre, Felipe-Morales and La Torre, 1990).

One key area for research is the study of existing indigenous practices. An understanding of the biophysical and human factors behind these practices might indicate how they can be adopted or adapted to the present socio-economic situation.

Finally, institutional support is essential for developing and assuring continuity of the required research in soil erosion and conservation practices in LAC countries (Pla, 2003).

12.4.2 | Soil organic carbon change

Soil organic carbon is the main component of humus. It is an organic compound with a stable C:N ratio which varies between 10 and 13 in most soils of LAC and of the world as a whole (Palm and Sanchez, 1990; Sisti *et al.*, 2004; Jantalia *et al.*, 2007). It is therefore only possible to increase the SOC content if the N content also increases to maintain the relationship (Boddey *et al.*, 2012 a, b, 2014). The reverse is also true: increasing one unit of soil N availability due to the mineralization of soil organic nitrogen will release between 10 and 13 units of C, equivalent to around 40 kg of CO_2.

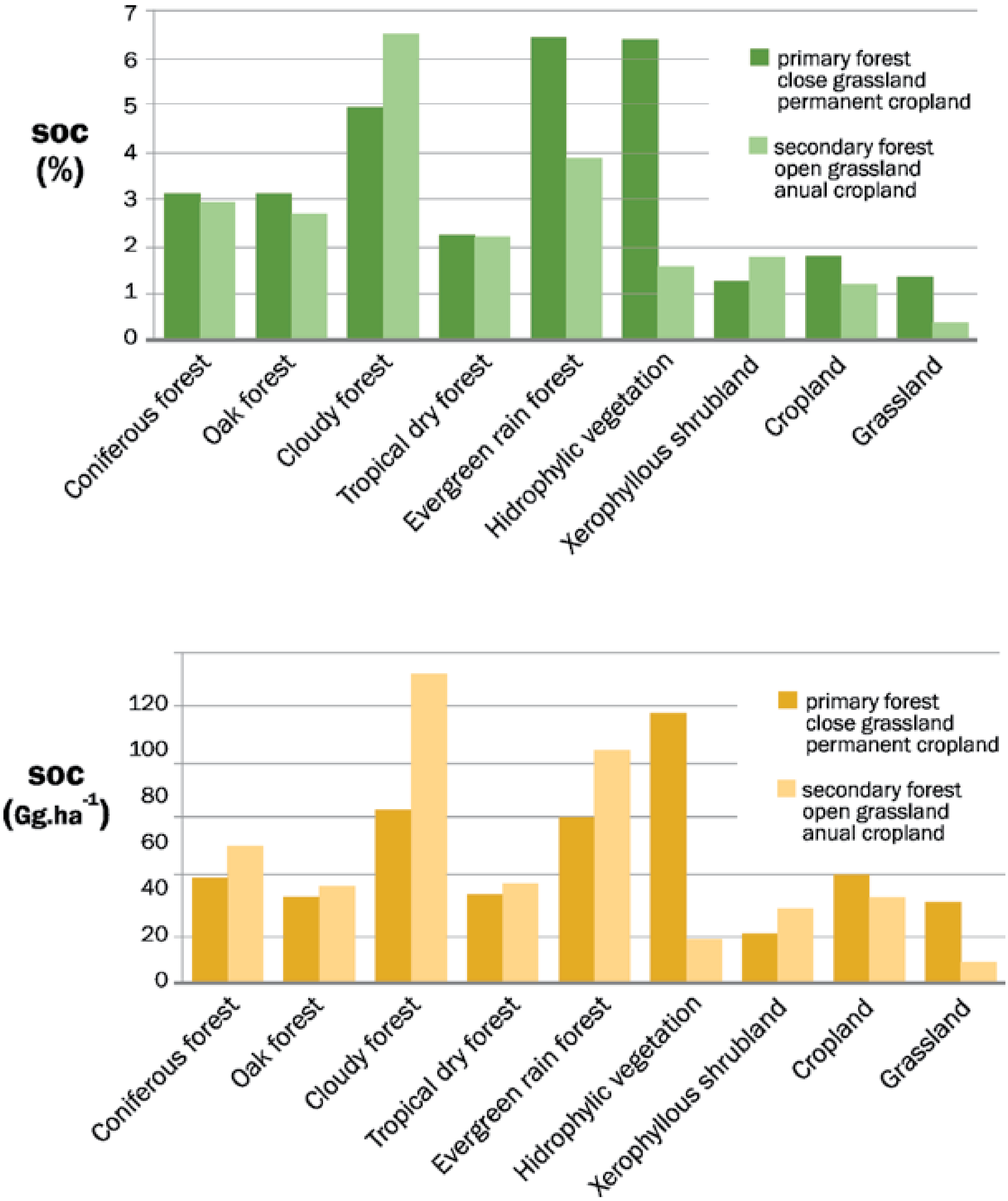


Figure 12.3 | shows soil organic carbon contents and stocks (taking into account soil bulk density) in different Mexican ecosystems.

Carbon concentrations (left) and carbon stocks (right) in the main ecosystems of Mexico. In both cases the bars with the strongest tone indicate a primary forest, closed pasture or permanent agriculture. Bars with the softer tone indicate a secondary forest, open pasture or annual agriculture. Source: Cruz-Gaistardo, 2014.

Primary forests have higher carbon concentration (in percentage) than secondary forests. However, the latter are better sinks because they have a greater capacity for conversion of CO_2 to biomass since they are in a more active phase of growth (Vargas *et al.*, 2008). About 185 Pg of organic carbon is stored at 1 m depth in LAC soils (16 percent of the world's soil carbon reserve). Half (95 Pg) of this amount is stored in the soils of the Amazon region, and about 52 percent of this carbon pool is held in the top 0.3 m of the soil profile (Batjes and Dijkshoorm, 1998).

LAC biomes where there is high risk of carbon losses – and also biodiversity losses – are the Amazon and the Atlantic Forest of Brazil, the Pampas of Argentina, the west coast of Colombia and the core of the Sierra Madre del Sur and Sierra Madre Oriental areas in Mexico (Hansen *et al.*, 2013). Satellite images reveal that more than half a million square kilometers of Amazon rainforest was destroyed between 1984 and 2005 and replaced by agriculture and the introduction of more than 240 million head of cattle (Gardi *et al.*, 2014).

The richest organic carbon soils in LAC, with stocks higher than 250 tonnes per hectare, are located in the sedimentary region of the Carso Huasteco and the Peninsula of Yucatán in Mexico, the tropical forests of Guatemala and Costa Rica, the region of Cauca and Magdalena in Colombia, the Orinoco delta in Venezuela, in the eastern Amazon, the Uruguayan savannas, the Valdivia forests in Chile, and the wet grasslands and steppes of Patagonia in Argentina (Gardi *et al.*, 2014). In these regions the highest proportion of Histosols, Andosols and Gleysols with high concentrations of soil carbon is found. The lowest soil carbon stocks are found in arid regions of LAC (the deserts of Mexico, Peru and Chile, as well as the drier regions of Brazil and Argentina). In these arid regions, it is essential to preserve the scarce carbon (less than 20 tonnes ha^{-1}) due to the fragility of the ecosystems found there (Figure 12.4).

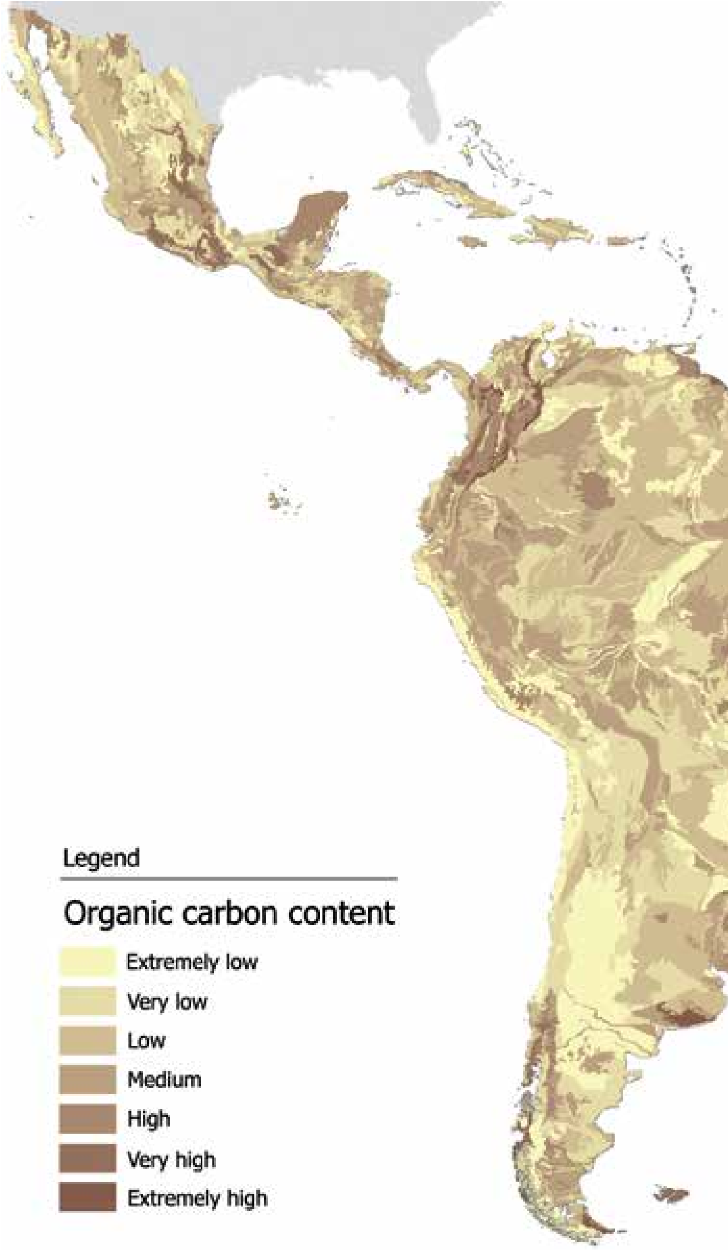
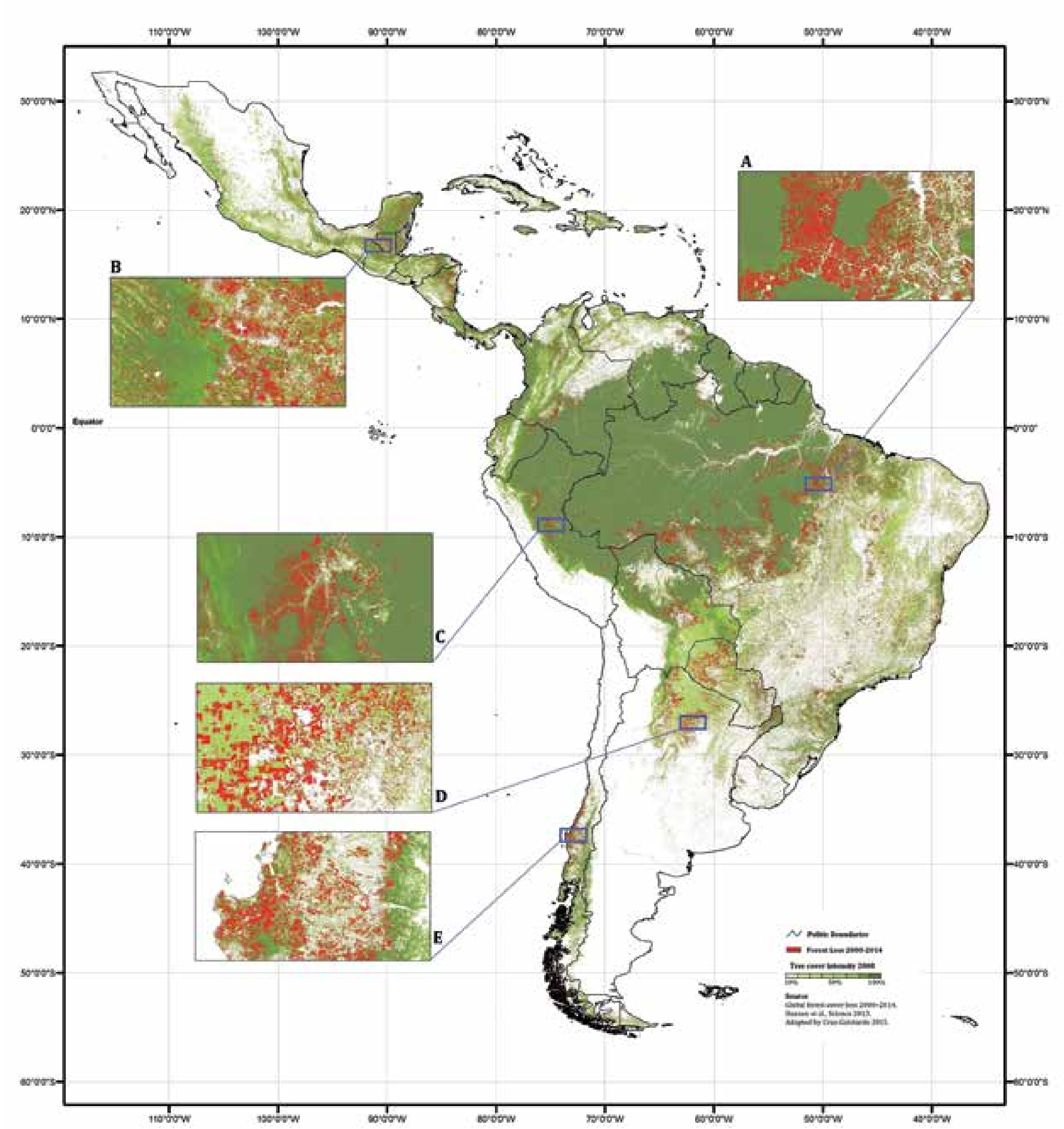


Figure 12.4 | Organic carbon stock (or density) in soils of Latin America and the Caribbean, expressed in Gigagrams per hectare. Source: Gardi et al., 2014.

Land use changes from forestry to urban or livestock use cause the greatest loss of soil carbon in LAC (Lal, 2005, 2006; van der Werf *et al*., 2010; INPE, 2010). The loss of ground cover due to deforestation exposes the soil to direct precipitation that could cause erosion and compaction of the soil surface microstructure in addition to carbon loss. Deforestation of tropical forests prevents the return to the soil of about 15 tonnes of organic inputs per ha each year. Agricultural soils return on average only 2 tonnes of residues per ha each year (Hughes, Kauffman and Jaramillo, 1999).

Forest cover loss in the global tropical rainforest biome accounts for about one third (32 percent) of all global forest cover loss, and nearly half of this loss of tropical rainforest occurs in South American rainforests. Several studies have been carried out and information is documented in the region related to deforestation and reforestation (CIAT, 2014; Gardi *et al*., 2014). The tropical dry forests of South America had the highest rate of tropical forest loss, due to deforestation dynamics in the Chaco woodlands of Argentina, Paraguay and Bolivia (Grau and Aide, 2008; Viglizzo and Jobbagy, 2010). Brazil is a global exception in terms of forest policy change, with a dramatic policy-driven reduction of deforestation in the Amazon Basin (INPE, 2010, 2013; Spavorek, 2012) (Figure 12.5).



Prepared by C. Cruz-Gaistardo

Figure 12.5 | Tree cover in the tonne 2000 and forest loss in the period 2000-2014. (A) Brazil, centered at 5.3°S, 50.2°W; (B) Mexico and Guatemala, centered at 16.3°N, 90.8°W and (C) Perú, centered at 8.7°S, 74.9°W; (D) Argentina, centered at 27.0°S, 62.3°W and (E) Chile, centered at 72.5°S, 37.4°W. Source: Hansen et al., 2013.

The litter on the ground also influences carbon fluxes to the mineral soil. A conserved forest can accumulate up to 10 tonnes of litter per hectare; however, when this forest is degrading its accumulation rate drops to 2 tonnes per ha (Cruz-Gaistardo, Díaz and Martínez, 2010). In an extreme case of erosion, litter would completely disappear. The loss of soil carbon results in the loss of natural fertility and of the existing biodiversity. These factors lead to extreme soil degradation, and could affect the local and regional economy (Smith *et al.*, 2007).

In the temperate Pampas of Argentina and Uruguay, conversion of grassland and pastures to agriculture caused soil organic carbon stocks to decrease from 27 Mg ha^{-1} to 19-20 Mg ha^{-1} in the first 20 cm of soil profile (Alvarez *et al.*, 2009).

Across LAC, there are initiatives to improve soil health and the environment. The large cities in LAC, such as Mexico City, Lima, Buenos Aires, São Paulo, Rio de Janeiro, Santiago and Bogota, manage their urban wastes, which are rich in organic carbon, in order to reduce pollution, especially in the surrounding rural areas which typically receive the waste (Da Silva and Donini, 2008). In the region of Santiago de Chile, for example, about 200 Gg of urban sludge and muds are produced each day by the seven million inhabitants (CEPAL, 2010). These wastes are currently being evaluated as possible fertilizers for farmland. However, this type of sludge may need up to three years to be mineralized, dissolved and available to plants. There is a risk that the concentration of metals (mainly zinc and copper) may increase in the soil (González, 2008). In Brazil, there are initiatives to grow rice without flooding the soils (so reducing methane emissions) and to cut sugar cane without resorting to direct burning. In Colombia and Cuba the use of organic fertilizers is widely promoted (Willer and Lukas, 2009).

International policy initiatives such as the United Nations Framework Convention on Climate Change (UNCCCD) and the programme for Reducing Emissions from Deforestation and Forest Degradation (REDD) often lack the institutional, investment and scientific capacity to begin implementation. In effect, policy is sometimes ahead of operational capabilities, even when the necessary information is available. By contrast, Brazil's use of Landsat data in documenting trends in deforestation was crucial to its policy formulation and implementation. To date, only Brazil produces and shares spatially explicit information on annual forest extent and change (UNFCCC, 2013; INPE, 2010, 2013).

Most soil sampling sites in LAC have been selected to address questions related to fertility or taxonomy. They were not selected to specifically quantify carbon in different pools (above- and below-ground). Consequently, they cannot be used for payment for environmental services. To overcome this, Mexico and Brazil are currently conducting national soil surveys designed to quantify soil carbon, increasing the number of sites within a systematic grid and using state-of-the-art instrumentation. The surveys are employing field spectrophotometers and micrometeorological towers to measure carbon fluxes between the land and atmosphere (Vargas *et al.*, 2013). In the case of Mexico, there are currently 11 000 sites with systematic information on all carbon reservoirs and coordinated programs for re-sampling and continuous monitoring.

12.4.3 | Soil salinization

Salt-affected soils are found mainly in the arid and semi-arid regions. Salinization caused by irrigation affects 18.4 million ha in LAC, particularly in Argentina, Brazil, Chile, Mexico and Peru (AQUASTAT, 1997). Many of the large plains of the continent are affected by natural salinization, but land use changes and overgrazing have also caused topsoil salinization (Taboada *et al.*, 2011; Bandera, 2013; Di Bella *et al.*, 2015).

Around 85 million ha are affected by excess of salts and sodium in **Argentina**, including arid and semi-arid zones (Szabolcs, 1979). Information provided by Bandera (2013) indicates that Argentina has approximately 600 000 ha of irrigated soils affected by salinity, which is the third largest area in a single country after Russia and Australia. In Argentina there are also areas of soils affected by salts in humid and sub humid climates, where salts come from groundwater. This is the case of the Pampa Deprimida (Depressed), the Buenos Aires East-Center (9 million ha), Buenos Aires Northwest (2.5 million ha), and the Submeridional Lowlands of Chaco and Santa Fe provinces (3 million ha).

The irrigated agricultural area of the northeast region of **Brazil** is around 500 thousand ha, and 25–30 percent is in the process of salinization (Heinze, 2002). Irrigation-induced salinization is an important land degradation process that affects crop yield in the Brazilian semi-arid region. Gypsum has been used as a corrective measure for saline soils in these areas (Moreira *et al.*, 2014). The addition of gypsum to irrigation water improves soil physical and chemical properties and can be considered as an alternative for the reclamation of saline-sodic and sodic alluvial soils in Northeast Brazil (Silveira *et al.*, 2008).

The salinity of soils on the cultivated land in the coastal valleys in northern **Chile** (Azapa and Lluta valleys) is generated by the salinity of irrigation water.

In **Colombia**, soils susceptible to salinization cover approximately 86 592 km^2, of which 90 percent are located in dry regions (Casierra-Posada, Pachón and Niño-Medina, 2007). Areas susceptible to salinization are located in the Caribbean region and inter-Andean valleys and highlands.

In **Cuba**, the soils with primary salinity only occupy restricted areas, often near to sea coasts. Secondary salinity, however, affects the majority of Cuban soils. The main causes of secondary salinity are: increase of the level of saline groundwater, deforestation of hilly lands, and use of saline water for irrigation (Alvarez *et al.*, 2008). There are 160 000 ha under rice, of which approximately 100 000 ha have problems of salinity and/or sodicity, in varying degrees (Borroto and Castillo, 1986).

In **Ecuador**, approximately 8.1 million ha have soils suitable for agriculture, of which about a quarter are currently used for agriculture. This includes: production of short cycle crops such as rice, maize, cassava, soy, watermelon, melon, tomato, pepper; cultivation of perennial crops, including cocoa, coffee, sugarcane and banana; and, on 63 percent of the area, natural and artificial pastures. One of the most serious problems facing this ecosystem is the increase in salinity. This is caused by irrigation with waters of medium quality and by excessive use of fertilizers in fert-irrigation. Soil salinity constitutes one of the main causes of crop yield reduction, and a significant part of Ecuadorian highland soils are considered as having high-salinity. The main causes are the pyroclastic nature of the soils, the effects of erosion, and the poor use of irrigation water (Jaramillo, Arahana and Torres, 2014).

In **Mexico**, approximately 20 percent of irrigated agricultural land (6 million ha) is affected by salinity and sodicity problems. In 1964 it was estimated that, in the irrigation districts of Culiacán, Fuerte River, Río Mayo, and Rio Yaqui which have a total area of 610 701 ha, over one third (218 495 ha, 36 percent) of the area had some kind of salt problem (Palacios-Vélez, 2012).

The lack of complete feasibility studies prior to implementing irrigation projects in Peru has caused increased drainage and salinity problems on 2 500 km^2 in the Coastal Area. In fact, all irrigation projects on the coast have experienced drainage and salinity problems within a few years after starting to irrigate (Cornejo, 1970). More than 300 000 ha (40 percent of the cultivated land in Peru's Coastal Valleys) were affected by soil salinity in the 1970s. About 25 percent (roughly 190 000 ha) were characterized by light to extreme salinity (>4 dS m^{-1}), enough to have negative effects on crop productivity (World Bank, 2007). Currently, there is little interest from the government to stop degradation of land. Even up-to-date information is lacking and what there is dates back to the 1970s (Santayana, 2012).

In **Venezuela**, the existence of problems of salinity was recognized in soils in the arid and semi-arid areas of the States of Zulia, Falcón, Lara, Anzoátegui and Trujillo (Villafañe, 1995). Falcon was one of the major vegetable producing states in Venezuela until soil degradation by salts had negative effects on yields and caused the abandonment of farms in this area.

12.5.1 | Argentina

Argentina is the eighth largest country in the world, with an area of 2 780 400 km^2. Seventy percent of its territory has an arid, semiarid or dry sub-humid climate, and the remaining 30 percent has a humid or sub humid climate. Regions under humid and sub humid temperate and subtropical climates (e.g. Pampean, Chaco and Mesopotamia) concentrate on the production of cereals, oilseeds, industrial crops, forages, forest plantations, domestic livestock and dairy products (SIIA, 2015).

Agriculture in Argentina began in earnest at the end of 19th century with the arrival of European immigrants and government colonization policies (Barski and Gelman, 2001; Viglizzo and Jobbagy, 2010). Agriculture and livestock grazing expanded until the mid-20th century by bringing new lands into production, largely employing low intensity production practices (Viglizzo and Jobbagy, 2010). This resulted in moderate to severe land degradation, not only in agricultural areas but also in dryland areas (SAGyP-CFA, 1995). In the second half of the 20th century agriculture intensified, especially in the Pampean region, with the use of more productive cereal varieties, hybrids and genetically modified crops, fertilizers and no till farming (Paruelo, Guerschman and Verón, 2005; Satorre, 2005; Viglizzo and Jobbagy, 2010).

In recent years, the agricultural frontier has expanded to the north-east, the north-west and the west (Figure 12.6), moving into areas with drier climates and/or less fertile soils (Paruelo, Guerschman and Verón, 2005; Viglizzo and Jobbagy, 2010). As a result, the cultivated area increased from about 15 to 32 million ha from 1988 to 2010, and bulk grain production shot up from about 20 to nearly 100 million tonnes in the same period (SIIA, 2015). At the same time, the ratio of crops produced changed. In 1990, the mix was: 37 percent wheat, 30 percent soybean and 13 percent maize. Twenty four years later (in 2014), production was 61 percent soybean, 19 percent maize and only 11 percent wheat (SIIA, 2015). This shift was driven by export demand for oil and biofuel soybean (Gobierno Argentino, 2014). Although a success in terms of fuel saving and adoption by farmers, this move towards a soybean monoculture appears to have driven many smaller farmers out of business (Pengue, 2005).

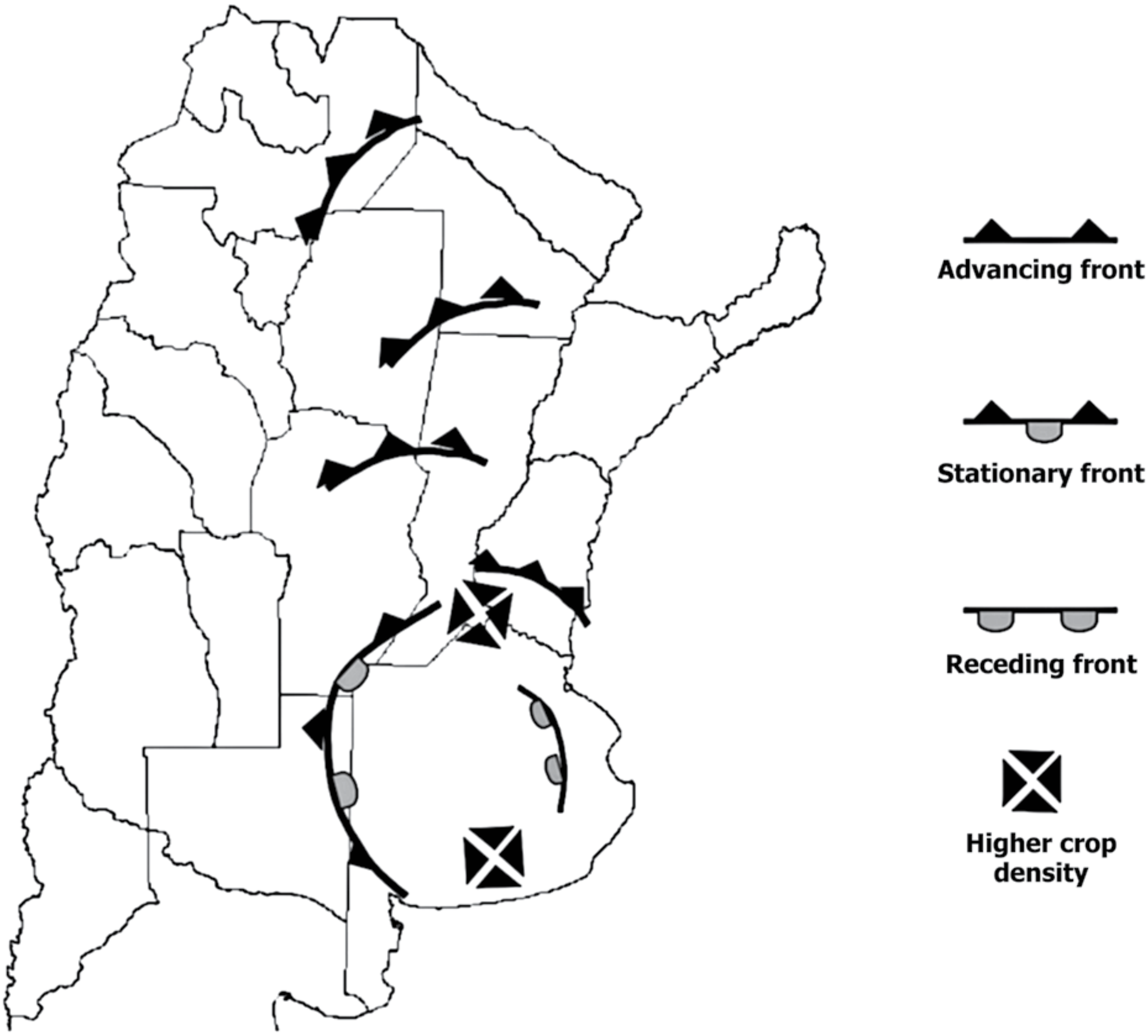


Figure 12.6 | Expansion of the agricultural frontier under rainfed conditions in the north of Argentina.
Source: Viglizzo & Jobbagy, 2010.

It has been estimated that agriculture in Argentina during the 20th century decreased soil carbon stocks by 27-50 percent. The causes included the turning over of grasslands and prairie soils rich in organic C with mould board and disc ploughs, and the loss of crop diversity (Studdert, Echeverría and Casanovas, 1997; Viglizzo and Jobbagy, 2010; Sainz Rozas, Echeverria and Angelini, 2011; Caride, Piñeiro and Paruelo, 2012; Milesi Delaye *et al.*, 2013). The potential of no-till farming to increase soil organic C stocks is still under discussion. In a review of 42 no-till vs plough till data sets from field experiments conducted in the Pampean region, Steinbach and Álvarez (2006) concluded that a 2.76 Mg ha^{-1} organic C increase was observed in no-till systems compared with tilled systems, with a relatively higher increase of organic C in areas of low organic matter level. However, Álvarez *et al.* (2009) observed no significant change in topsoil organic C content after adoption of continuous no-till in previously conventionally tilled loam, silty loam, and silty clay loam soils in well-managed farms of the region.

No-till farming is considered to improve topsoil physical properties, especially when combined with suitable crop rotations and pastures (Álvarez *et al.*, 2014). However, the prevalence of soybean mono-cropping in the Pampean and Chaco regions promoted unfavourable topsoil physical conditions such as laminar and massive aggregation, shallow compaction and decreased infiltration rates (Sasal, Andriulo and Taboada, 2006; Álvarez and Steinbach, 2009; Álvarez *et al.*, 2009, 2014). These structural forms were found to decrease soybean yields under rainfed conditions (Bacigaluppo *et al.*, 2011).

Fertilizer use in Argentina was quite low until the mid-1990s and despite the eight-fold increase in fertilizer use in the last two decades, negative budgets of nitrogen, phosphorus and sulphur still persist in the Pampean region (Lavado and Taboada, 2009; Viglizzo and Jobbagy, 2010). Nutrient removal by grains was found to have exceeded application by 2.3-3.2, 1.4-2.0 and 2.0-2.6 times for N, P and S over a four year period, respectively. Phosphorus is a special issue, as it has decreased to 'deficient' levels in several areas (Sainz Rozas, Echeverria and Angelini, 2012).

During the 20th century, more than 1.2 million ha (32 percent of the agricultural area) were affected by moderate to severe soil water erosion, characterized by 47 to 131 Mg ha^{-1} yr^{-1} soil losses and by 5 to 20 cm surface horizon thickness decreases (SAGyP-CFA, 1995). According to estimations by Viglizzo and Frank (2010), the widespread adoption of no-tillage helped to control erosion losses, which was reported to have decreased to only 7 Mg ha^{-1} yr^{-1}. However, more field data are needed to check these estimations.

Salt affected soils cover more than 70 million ha in Argentina, the third largest area in any country in the world (FAO, 1976). At least 600 000 ha of irrigated soils under arid and semiarid climates are affected by anthropogenic salinization, often related to unsuitable drainage management and/or poor water quality. In sub humid and humid regions, there are about 12 million ha of naturally salinized soils. In these areas, anthropogenic salinization was also promoted by livestock grazing (Taboada, Rubio and Chaneton, 2011), supplementary irrigation (Andriulo *et al.*, 1998) deforestation (Nosetto *et al.*, 2012) or afforestation (Jobbagy and Jackson, 2004).

A detailed evaluation of the state of land degradation in the drier areas of Argentina came to the following conclusions (FAO, 2010):

- Eighty one percent, or more than 1 240 000 km^2 of the evaluated systems, present some process of degradation. The degree of degradation of dry lands, defined by the intensity of the process, varies from 'non-degraded' to 'extreme'. This result is obtained after adding all degradation processes identified, including water and wind erosion, salinization and loss of vegetative cover.
- Amongst the degrading processes analyzed, 50 percent of the area is affected largely by processes of biological degradation (variation of vegetative cover, loss of habitats, and decrease of biomass). Twenty 6 percent of the area presents a strong degree of degradation. The rate of degradation is slow in 40 percent of the areas with biological degradation, while almost 20 percent of these areas present a high rate of increase.
- Fifteen percent of the land use systems analyzed presented symptoms of physical degradation of the soil (compaction, crusting, flooding). Of these 60 percent present a high degree of degradation. Thirty percent are degrading at a moderate rate.
- Five percent of the area analyzed presented symptoms of degradation of water resources (decrease of the average moisture content of the soil, changes in volume, reduction of the quality of water, reduction of the capacity to capture and retain water). Eighty percent of the area presents a strong degree of degradation, with approximately 20 percent having a moderate rate of increase.
- Forty percent of the area analyzed is affected by wind erosion, with a degree of degradation that is mostly moderate to strong with a tendency to increase. Water erosion affects 40 percent of the area analyzed (Figure 12.7), with a similar degree of severity to wind erosion, although with a lower rate of increase.

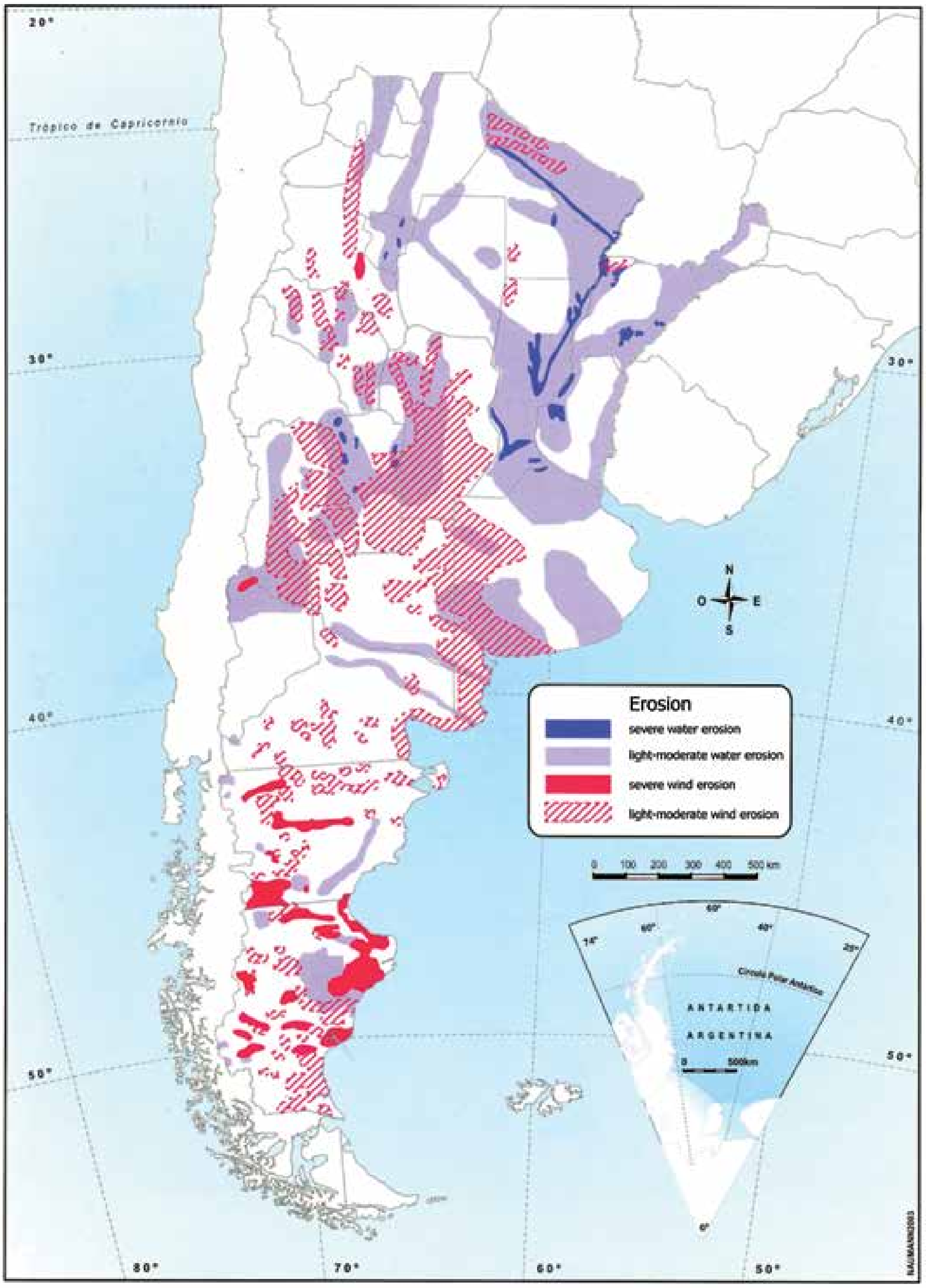


Figure 12.7 | Percentage of areas affected by wind (a) and water erosion (b) in Argentina.
Source: Prego et al., 1988.

12.5.2 | Cuba

The main island of Cuba is the largest island in the West Indies, with a total land area of 104 945 km². The area of the country as a whole, including the Isla de la Juventud (2 200 km²) and around 4 195 keys and small islands, is 110 860 km². The topography is mostly flat to rolling, with rugged hills and mountains in the southeast and south-central area. The Cuban mountain range system is formed by four massifs covering about 18 percent of the surface of the Cuban archipelago. The surface cover of Cuba consists of cropland and crop/natural vegetation mosaics (44 percent), shrub lands, savanna and grasslands (24 percent), forests (23 percent) and wetlands (9 percent).

Mean annual rainfall is 1 335 mm, with a pronounced seasonal variation between the driest and wettest months. Rainfall levels vary widely across the country, from 300 mm annually in the Guantánamo area of the south to more than 3 000 mm in the north. Mean annual temperature is 25°C (CUBA, 2014).

In recent decades, significant variations have been detected in the country's climatic patterns (Centella *et al.*, 1997). An overall increase in temperature has been accompanied by a reduction in annual rainfall totals of 10-20 percent and an increase in inter-annual variation in rainfall of 5-10 percent, with reduced rainfall in the rainy season and increased rainfall in the dry season (Lapinel, Rivero and Cutié, 1993; Goldenberg *et al.*, 2001). At the same time, the frequency of unseasonal droughts has increased.

The National Environment Strategy 2007/2010 is the guiding document for Cuban environmental policy. It defines the five main environmental issues in Cuba, which are: land degradation; factors affecting forest coverage; pollution; loss of biological diversity; and water scarcity. The Strategy proposes policies and instruments to tackle the five issues in order to improve environmental protection and the rational use of national resources. Land degradation is considered the most important of the five issues.

Cuba has a wide variety of soils, including those developed over sedimentary limestone (Nitisols, Ferralsols and Lixisols), and others over older rocks (Cambisols and Phaeozems). Cuba has complete soil studies and maps at cartographic scales of 1:250 000, 1:50 000 and 1:25 000 (Gardi *et al.*, 2014).

The most recent assessment of land degradation at local and national level was carried out between 2006 and 2010, using a standard methodology (Liniger *et al.*, 2011; Bunning, McDonagh and Rioux, 2011). In parallel, the sustainable land management practices applied to stop or reverse the degradation process in the country were inventoried in each land use system (AMA, 2010). A first set of 23 thematic maps at 1: 250 000 scale was produced in 2010 and these were updated in 2013 (IGT-AMA, 2014).

Fifteen different types of degradation, affecting the whole country to a lesser or more serious degree were recognized (Figures 12.8 and 12.9). The four main types of degradation in terms of extent are: (1) loss of topsoil by water erosion, which covers more than 30 000 km² and is present in all provinces; (2) the loss of vegetative cover, which has a similar extent and also occurs in every province; (3) processes of salinization and compaction, which cover between 10 000 to 20 000 thousand km² in 14 and 11 provinces respectively; and (4) loss of habitat condition and areas affected by fire, which occupy up to 5 000 km² each in nine provinces of the country. The other types of degradation such as aridity, loss of soil fertility, reduced organic matter content and reduced quality of surface and groundwater occupy areas that do not exceed 5 000 km².

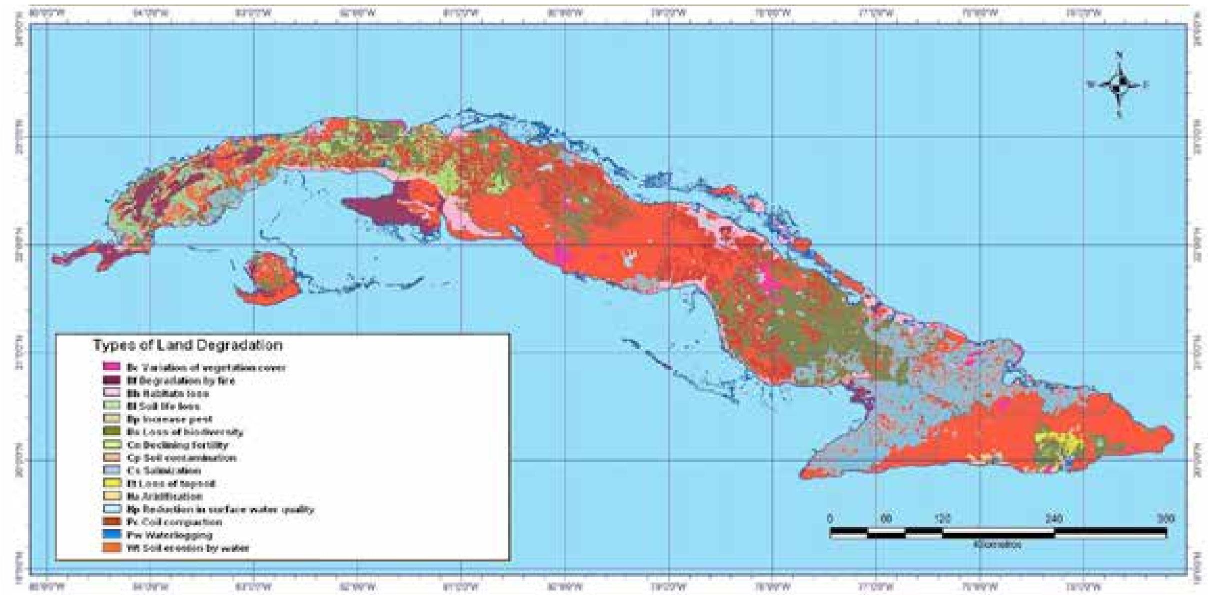


Figure 12.8 | Predominant types of land degradation in Cuba.
Source: FAO, 2010.

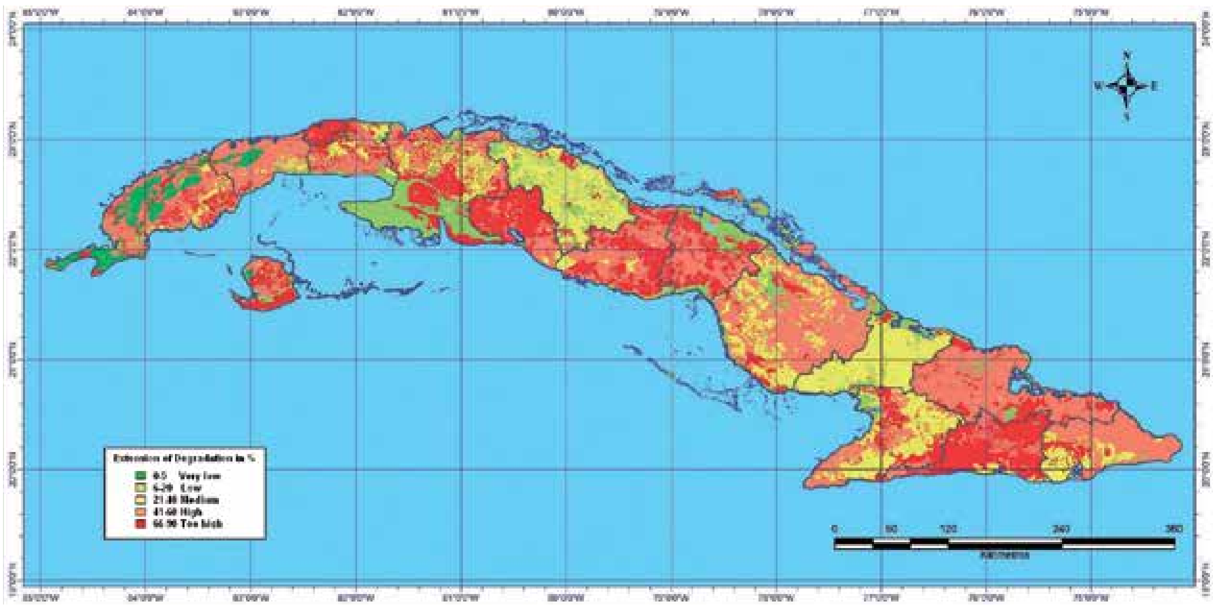


Figure 12.9 | Extent of land degradation in land use system units in Cuba.
Source: FAO, 2010.

As far as the intensity of degradation is concerned, 12 percent of Cuban territory is classified as grade 1 (slight), while 68 percent is grade 2 (slight or moderate intensity) and 19 percent is grade 3 (strong or intense degradation). The last two grades are mainly located in the central and eastern area (Figure 12.10).

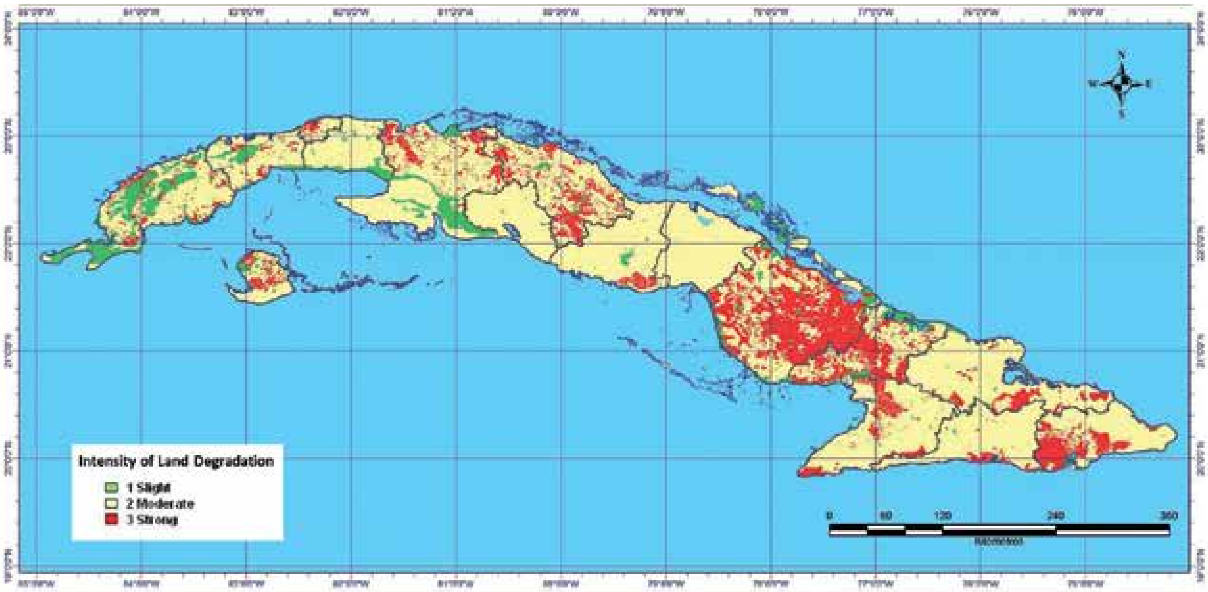


Figure 12.10: Intensity of land degradation in Cuba. Source: FAO, 2010.

A systematic local inventory of land degradation processes was carried out in the dryland areas of Cuba. The drylands occupy a total area of nearly 10 000 km², and occur mainly in the eastern region of the country, between Camagüey and Guantanamo, covering 6 provinces and 20 municipalities. Surveys were carried out in the most representative areas of Camaguey-Tunas, Granma and Guantanamo covering soil health, water quality and quantity, vegetation status and biodiversity, among others. Since 2001, the country has established different programmes and strategies based on sustainable soil management approaches in order to combat soil degradation. The most important of these is the National Program for Soil Improvement and Conservation, which is overseen by the Soil Institute. Over the last decade, at least 500 000 ha have benefited from these programs, which are financially supported by Cuban government (Instituto de Suelos, 2001).

12.6 | Conclusions and recommendations

LAC has a wide range of biomes and probably the largest potential area of cropland in the world. The soils of the region are subject to a number of threats, of which the three most important are: soil erosion, organic carbon losses and salinization. Other threats such as imbalance of nutrients, loss of biodiversity, compaction, waterlogging, contamination, and sealing and capping are also common (Table 12.1).

The most important ecosystem services affected in LAC are: climate regulation through the disturbance of the C and N cycles due to deforestation, and water regulation and food production on sloping lands. Deforestation and erosion caused by inappropriate land use are the initial causes of various anthropogenic threats to soil quality.

An enhanced natural resource information system is necessary in many countries of LAC, in order to perform a better diagnosis of soil conditions and their level of degradation. This will allow possible solutions to be identified, including land use planning and appropriate legislation.

A major effort is required to design and implement sustainable soil management in the region, taking into account the risks and threats assessed as well as the particular characteristics of each country. A participatory process is required if the final goal is to be reached, namely, protection of the soil resource for food security and for the production of ecosystem services for present and future human well-being.

=	↗↙	↗	↙			
Stable	Variable	Improving	Deteriorating	Evidence and consensus is **low**	Evidence and consensus are **limited**	**Adequate** high-quality evidence and **high level** of consensus

Table 12.1 | Summary of Soil Threats Status, trends and uncertainties in in Latin America and the Caribbean

Threat to soil function	Summary	Condition and Trend					Confidence	
		Very poor	Poor	Fair	Good	Very good	In condition	In trend
Soil erosion	Widespread across the region. Landslides are accelerated by land use in highland areas		↙				Adequate	Limited
Organic carbon change	Declines are caused by deforestation, intensive cultivation of grasslands and monoculture.		↙				Limited	Limited
Salinization and sodification	Caused by inadequate irrigation technology and water quality. Land use changes also promote salinization.		↙				Adequate	Adequate
Nutrient imbalance	Most countries have negative nutrient balances due to over-extraction. In some cases over fertilization also causes nutrient imbalance.		↙				Limited	Limited
Loss of soil biodiversity	Suspected to occur in deforested and over-exploited agricultural areas.			↗↙			Low	Limited
Compaction	Caused by overgrazing and intensive agricultural traffic.		↙				Limited	Limited
Waterlogging	Due to deforestation and poor structural conditions in agricultural areas.			=			Adequate	Adequate
Soil acidification	Soil acidification is limited to some areas with overuse of N fertilizers			↗↙			Low	Low
Contamination	Industrial sources cause soil contamination in some places. Non-point soil pollution prevails in sites with intensive agriculture (e.g. herbicides residues).			↗↙			Low	Limited
Soil sealing and land take	In some valleys and floodplains, urbanization has expanded onto fertile soils.			↗↙			Limited	Limited

References

Alegre, J.C., Felipe-Morales, C. & La Torre. 1990. Soil Erosion Studies in Peru. *Journal of Soil and Water Conservation*, 45: 417-420.

Alegre, J.C., Pashansi, B. & Lavelle, P. 1996. Dynamics of soil physical properties in Amazonina agroecosytem inocultated by earthworms. *SSSAJ*, 60: 1522-1529.

Álvarez, C.R., Taboada, M.A., Gutiérrez Boem, F.H., Bono, A., Fernández, P.L. & Prystupa, P. 2009. Topsoil properties as affected by tillage systems in the Rolling Pampa region of Argentina. *Soil Science Society of America Journal*, 73: 1242-1250.

Álvarez, C.R., Taboada, M.A., Perelman, S.B. & Morrás, H.J. 2014. Topsoil structure in no-tilled soils in the Rolling Pampas, Argentina. *Soil Research*, 52: 533-542.

Álvarez, R. & Steinbach, H. 2009. A review of the effects of tillage systems on some soil physical properties, water content, nitrate availability and crop yields in the Argentine Pampas. *Soil and Tillage Research*, 104: 1-15.

Alves, B.J.R., Boddey, R.M. & Urquiaga, S. 2003. The success of BNF in soybean in Brazil. *Plant & Soil*., 252: 1-9.

AMA. 2010. *Alcance, resultados e impactos del proyecto LADA en Cuba*. (Also available at http://www.educambiente.co.cu/Desercuba/index.php?option=com_content&view=article&id=45:lada&catid=36:miscelaneas)

Andriulo, A., Galetto, M.L., Ferreyra, C., Cordone, G., Sasal, C., Abrego, F., Galina, J. & Rimatori, Y.F. 1998. The effect of 11 years of supplementary irrigation on some soil properties. Physico-chemical properties. Actas XVI Congreso Argentino de la Ciencia del suelo, Villa Carlos Paz, 247 - 258. [in Spanish]

Aparicio, V.C., De Gerónimo, E., Marino, D., Primost, J., Carriquiriborde, P. & Costa, J.L. 2013. Environmental fate of glyphosate and aminomethylphosphonic acid in surface waters and soil of agricultural basins. *Chemosphere*., 93(9):1866-1873.

AQUASTAT. 1997. *Tablas resumen para America Latina y el Caribe*. FAO. (Also available at http/www.fao.org/GEO-2-199.)

Bacigaluppo, S., Bodrero, M.L., Balzarini, M., Gerster, G.R., Andriani, J. M., Enrico, J. M. & Dardanelli, J.L. 2011. Main edaphic and climatic variables explaining soybean yield in Argiudolls under no-tilled systems. *European Journal of Agronomy*. 35: 247-254.

Bai Z. G., Dent, D.L., Olsson, L. & Schaepman, M.E. 2008. Proxy global assessment of land degradation. *Soil Use and Management*, 24(3): 223-234.

Balotta, E.L., Colozzi Filho, A.A., Andrade, D.S. & Dick, R.P. 2004. Long-term tillage and crop rotation effects on microbial biomass and C and N mineralization in a Brazilian Oxisol. *Soil and Tillage Research*, 77(2): 137–145.

Bandera, R. 2013. *Rehabilitación de suelos salino-sódicos: evaluación de enmiendas y de especies forrajeras*. Tesis presentada para optar al título de Magíster de la Universidad de Buenos Aires, Área Recursos Naturales. 66 pp.

Barski, O. & Gelman, J. 2001. *Historia del agro argentino*. Desde la Conquista hasta fines del siglo XX. Buenos Aires, Grijalbo Mondadori. 460 pp.

Batjes, J.A. & Dijkshoorm. 1999. Carbon and nitrogen stocks in the soils of the Amazon Region. *Geoderma*, 89: 273–286.

Bertiller, M.B., Ares, J.O. & Bisigato, A.J. 2002. Multiscale indicators of land degradation in the Patagonia Monte, Argentina. *Environmental Management*, 30(5): 704-715.

Bertsch, F., Henríquez, C., Ramírez, F. & Sancho, F. 2002. Site-specific nutrient management in the Highlands of Cartago province (Costa Rica). *Better Crops International*, 16(1): 16-19.

Boddey, R.M., Alves, B.J.R, Jantalia, C.P., Machado, P.L.O.A., Soares, L.H.B. & Urquiaga, S. 2012a. Praticas mitigadoras das emissões de gases de efeito estufa na agropecuária brasileira. In R.M. Boddey, M.L. Lima, B.J.R. Alves, P.L.O.de A. Machado & S. Urquiaga, eds. *Carbon stocks and greenhouse gas emissions in Brazilian agriculture*, pp. 327-347. Brasilia, Embrapa, DF.

Boddey, R.M., Alves, B.J.R, Urquiaga, S., Jantalia, C.P., Martin –Neto, L., Madari, B.E., Milori, D.M.B. & Machado, P.L.O.A. 2012b. Estoques de carbono nos solos do Brasil. Quantidade e mecanismos de acúmulo e preservação. In R.M. Boddey, M.L. Lima, B.J.R. Alves, P.L.O.de A. Machado & S. Urquiaga, eds. *Carbon stocks and greenhouse gas emissions in Brazilian agriculture*, pp. 33-82. Brasilia, Embrapa, DF.

Boddey, R.M., Lima, M.L., Alves, B.J.R., Machado, P.L.O.de **A. & Urquiaga, S.** (eds.) 2014. *Carbon stocks and greenhouse gas emissions in Brazilian agriculture*. Embrapa, Brasilia, DF. 374 pp.

Borroto, M. & Castillo, D. 1986. *Ponencia sobre suelos salinos en Cuba y metodología para su mejoramiento*. La Habana, Ministerio de la Agricultura. 35 pp.

Botta, G.F., Tolon-Becerra, A. & Melcon, F.B. 2009. Seedbed compaction produced by traffic on four tillage regimes in the rolling Pampas of Argentina. *Soil Till. Res.*, 105(1): 128–134.

Botta, G.F., Tolon-Becerra, A., Lastra-Bravo, X. & Tourn, M. 2010. Tillage and traffic effects (planters and tractors) on soil compaction and soybean (Glycine max L.) yields in Argentinean pampas. *Soil Till. Res*., 110(1): 167–174.

Bunning, S., McDonagh, J. & Rioux, J. 2011. *Manual for local level assessment of land degradation and sustainable land management*. LADA Technical Report No 11. Rome, FAO.

Campillo, R., Urquiaga, S., Pino, I. & Montenegro, A. 2003. Estimación de la fijación biológica de nitrógeno en leguminosas forrajeras mediante la metodología del 15N. *Agricultura Tecnica*., 63: 169-179.

Campillo, R., Urquiaga, S., Undurraga, P., Pino, I. & Boddey, R.M. 2005. Strategies to optimise biological nitrogen fixation in legume/grass pastures in the southern region of Chile. *Plant and Soil*, 273: 57-67.

Caride, C., Piñeiro, G. & Paruelo, J.M. 2012. How does agricultural management modify ecosystem services in the Argentine Pampas? The effects on soil C dynamics. Agriculture, Ecosystems and Environment. 154: 23-33.

Casierra-Posada, F., Pachón, C.A. & Niño-Medina, R.C. 2007. Análisis bromatológico en frutos de tomate (Lycopersicon esculentum Mill.) afectados por salinidad por NaCl. *Revista U.D.C.A Actualidad & Divulgación Científica*, 10(2): 95-104.

Centella, A., Naranjo, L., Paz, L., Cárdenas, P., Lapinel, B., Ballester, M., Pérez, R., Alfonso, A., González, C., Limia, M. & Sosa, M. 1997. *Variaciones y cambios del clima en Cuba*. Informe Técnico. Centro Nacional de Clima, Instituto de Meteorología. Cuba, La Habana. 58 pp.

CEPAL. 2010. *Official web site*. Santiago de Chile, Vitacura. (Also available at http://www.cepal.org/en)

Cerdeira, A.L., Gazziero, D.L.P., Duke, S.O. & Matallo, M.B. 2011. Agricultural Impacts of Glyphosate-Resistant Soybean Cultivation in South America. *J. Agric. Food Chem*., 59(11): 5799–5807.

Christoffoleti, P.J., Galli, A.J.B., Carvalho, S.J.P., Moreira, M.S., Nicolai, M., Foloni, L.L., Martins, B.A.B. & Ribeiro, D.N. 2008. Pest Management Science. Special Issue. *Glyphosate-Resistant Weeds and Crops*, 64(4): 422–427.

CIAT. 2014. *Official web site*. Colombia, Cali. (Also available at https://ciat.cgiar.org/)

Comerma, J, Larralde, R. & Soriano, J. 1971. Aumento de la productividad agrícola a través de los trabajos de conservación de suelos. México I Seminario Latinoamericano FAO/PNUD/ México. Evaluación sistemática de los recursos de tierras y aguas. *Agronomía Tropical*, 23: 95- 113.

Cornejo, A. 1970. Resources of Arid South America. *In* H.E. Dregne, ed. *Arid Lands in Transition*, pp. 345-380. AAAS.

Cruz-Gaistardo, C. 2014. *Inventario de Gases de Efecto Invernadero 2014*. Reporte Bianual para el sector USCUSS, sector suelos. México, PNUD-FAO-CONAFOR.

Cruz-Gaistardo, C., Díaz, V. & Martínez, J. 2010. *Inventario Estatal Forestal y de Suelos*. México, Secretaría del Medio Ambiente del Estado de Aguascalientes.

CUBA. 2014. *Anuario Estadístico 2013*. Oficina Nacional de Estadísticas.

Da Silva, A. & Donini M.S. 2008. *Compostagem: A reciclagem da major parte dos residuos urbanos*. Brasil, Universidade Estadual Paulista.

D'Accunto, L, Semmartin, M. & Ghersa, C.M. 2014. Uncropped field margins to mitigate soil carbon losses in agricultural landscapes. *Agricultural, Ecosystems and Environment*, 183: 60-68.

de Moraes Sa, J.C. & Lal, R. 2009. Stratification ratio of soil organic matter pools as an indicator of carbon sequestration in a tillage chronosequence on a Brazilian Oxisol. *Soil and Tillage Research*, 103(1): 46–56.

Di Bella, C.R., Rodríguez, A.M., Jacobo, E.J., Golluscio, R.A. & Taboada, M.A. 2015. Impact of cattle grazing on temperate coastal salt marsh soils. *Soil Use and Management*. (in press)

Duvert, C., Gratiot, N., Evrard, O., Navratil, O., Némery, J., Prat, C. & Esteves, M. 2010. Drivers of erosion and suspended sediment transport in three headwater catchments of the Mexican Central Highlands. *Geomorphology*, 123: 243-256.

EC. 2013. *Cambio climático y degradación de los suelos en América Latina: escenarios, políticas y respuestas*. Programa EUROCLIMA, Dirección General de Desarrollo y Cooperación-EuropeAid, Comisión Europea. Bruselas. 188 pp.

Espinosa, J. & Molina, E. 1999. *Acidez y encalado de los suelos*. International Plant Nutrition Institute IPNI. 46 pp.

FAO. 1976. *Prognosis of salinity and alkalinity*. Soils Bulletin No. 31 Rome, FAO. (Also available at http://www.fao.org/docrep/x5871e/x5871e 03.htm#TopOfPage)

FAO. 2005. *Global Forest Resources Assessment 2005: Progress towards sustainable forest management*. FAO Forestry Paper 147. Rome, FAO Forestry Department. 320 pp.

FAO. 2010. *Proyecto Evaluación de la Degradación de la Tierra en zonas secas (Land Degradation Assessment in Drylands*). LADA. (Also available at http://www.fao.org/nr/lada/)

FAO. 2012. *Directrices voluntarias sobre la Gobernanza responsable de la tenencia de la tierra, la pesca y los bisques en el contexto de la seguridad alimentaria nacional*. CF. (Also available at http://www.fao.org/docrep/016/i2801s/i2801s.pdf)

Fassbender, H.& Bornemisza, E. 1987. *Química de suelos con énfasis en suelos de América Latina. 2nd edition*. San José Costa Rica, IICA. 420 pp.

Ferraro, D.O. & Ghersa, C.M. 2007. Exploring the natural and human-induced effects on the assemblage of soil microarthropods communities in Argentina. *Eur. J. Soil Biol.*, 109: 109–117.

Franchini, J.C., Crispino, C.C., Souza, R.A., Torres, E. & Hungria, M. 2007. Microbiological parameters as indicators of soil quality under various soil management and crop rotation systems in southern Brazil. *Soil and Tillage Research*, 92(1–2): 18–29.

Franzluebbers, A.J., Sawchik, J. & Taboada, M.A. 2014. Agronomic and environmental impacts of pasture-crop rotations in temperate North and South America. *Agriculture, Ecosystems and Environment*, 190: 18-26.

Galdino, S., Vieira, L.M. & Pellegrin, L.A. (eds.). 2006. *Impactos Ambientais e Socioeconômicos na Bacia do Rio Taquari* – Pantanal. Corumbá, Embrapa Pantanal, MS. 356 pp.

Gardi, C., Angelini, M., Barceló, S., Comerma, J., Cruz Gaistardo, C., Encina Rojas, A., Jones, A., Krasilnikov, P., Mendonça-Santos, M.L., Montanarella, L., Muniz Ugarte, O., Schad, P., Vara Rodríguez, M.I. & Vargas, R. (eds). 2014. *Atlas de suelos de América Latina y el Caribe*. Luxembourg, Comisión Europea, Oficina de Publicaciones de la Unión Europea, L-2995. 176 pp.

Garret, L.J. 1997. Challenges for the year 2020 in Latin America: nutrition and agriculture since 1970. Instituto internacional de investigaciones sobre políticas alimentarias. (IFPRI). Washington, DC. [in Spanish]

Gobierno Argentino. 2014. Soybean oil: Argentina is the vendor leader in the region. Argentina, Portal Público de Noticias. (available at: http://www.argentina.ar/temas/economia-y-negocios/29800-aceite-de-soja-argentina-lidera-ventas-en-la-region) [in Spanish]

Goldenberg, S.B., Landsea, C.W., Mestas –Nunez, A.M. & Gray, W.M. 2001. 'The recent increase in Atlantic hurricane activity: Causes and implications.' (El reciente incremento en la actividad de huracanes en el Atlántico: Causas e implicaciones). *Science*, 293: 474–479.

González, M.S. 2008. *Evaluación del uso agrícola de lodos de plantas de tratamiento de aguas servidas de la ciudad de Santiago de Chile*. INIA.

Grau, H.R. & Aide, M. 2008. Globalization and land-use transitions in Latin America. *Ecology and Society*, 13(2): 16.

Green, V.S., Stott, D.E., Cruz, J.C. & Curi, N. 2007. Tillage impacts on soil biological activity and aggregation in a Brazilian Cerrado Oxisol. *Soil and Tillage Research*, 92 (1–2): 114–121.

Guimarães, L.R. 2013. *Nutrient management challenges in Brazil and Latin America*. The second global conference on Land-Ocean Connections (GLOC-2), Montego Bay, 3 October.

Hansen, M.C, Potapov, P.V., Moore, R., Hancher, M., Turubanova, S.A., Tyukavina, A., Thau, D., Stehman, S.V., Goetz, S.J., Loveland, T.R., Kommareddy, A., Egorov, A., Chini, L., Justice, C.O. & Townshend, J.R.G. 2013. High-Resolution Global Maps of 21st-Century Forest Cover Change. *Science*, 342(6160): 850-853.

Heinze, B. 2002. *Importância da agricultura irrigada para o desenvolvimento da região nordeste do Brasil*, *Brasília* - DF. Monografia, MBA, Ecobusiness Scholl, FVG. Publicações IICA/braulioheinze.pdf.

Henríquez, C., Ortíz, O., Largaespada, K., Portuguéz, P., Vargas, M., Villalobos, P. & Gómez, D. 2011. Determinación de la resistencia a la penetración, al corte tangencial, densidad aparente y temperatura en un suelo cafetalero, Juan Viñas, Costa Rica. *Agronomía Costarricense*, 35(1): 175-184.

Henríquez, C., Uribe, L., Valenciano, A. & Nogales, R. 2014. Actividad enzimática del suelo – Deshidrogenasa, Glucosidasa, Fosfatasas y Ureasa- Bajo diferentes cultivos. *Agronomía Costarricense*, 38(1): 43-54.

Hincapié, E. & Ramírez, F. 2010. *Riesgo de erosion en suelos de ladera de la zona cafetera*. Avances Técnicos 400. Colombia, Cenicafé, Manizales. 8 pp.

Hughes, R.F., Kauffman, J.B. & Jaramillo, V.J. 1999. Biomass, Carbon, and nutrient dynamics of secondary forest in a humid tropical region of México. *Ecology*, 80: 1892–1907.

IGT-AMA. 2014. *Mapas CSMB Cartografia del Sistema de Monitoreo Biofísico*. Instituto de Geografía tropical y la Agencia de medio Ambiente. La Habana. (Also available at http://www.infogeo.cu/index.php/ct-menu-item-2/monitoreo-biofisico/si-proyectos/17-proyectos/34-mapas-de-lada)

INPE. 2010. *Deforestation estimates in the Brazilian Amazon*. Brazil São Jose dos Campos, SP, Instituto Nacional de Pesquisas Espaciais.

INPE. 2013. *Monitoring of the Brazilian Amazonian Forest by satélite*, 2002-2012. Brazil São Jose dos Campos, SP, Instituto Nacional de Pesquisas Espaciais.

Instituto de Suelos. 2001. Programa Nacional de Mejoramiento y Conservación de Suelos. Cuba, La Habana, Agrinfor. 39 pp.

IPCC. 2014. *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects*, by C.B. Field, V.R. Barros, D.J. Dokken, K.J. Mach, M.D. Mastrandrea, T.E. Bilir, M. Chatterjee, K.L. Ebi, Y.O. Estrada, R.C. Genova, B. Girma, E.S. Kissel, A.N. Levy, S. MacCracken, P.R. Mastrandrea, and L.L.White, eds. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. UK, Cambridge, Cambridge University Press &USA, New York, NY. 1132 pp.

Jantalia, C.P., Resck, D.V.S., Alves, B.J.R., Zotarelli, L., Urquiaga, S. & Boddey, R.M. 2007. Tillage effect on C stocks of a clayey Oxisol under a soybean-based cropo rotation in the Brasilian Cerrado region. *Soil & Tillage Research*, 95: 97-109.

Jaramillo, V., Arahana, V. & Torres, M. 2014. *Determination of the level of tolerance to salinity in in vitro conditions of the plants of tomate de árbol (Solanum betaceum) from different localities of the ecuatorian highlands. In* C. Zambrano, ed.

Jobbagy, E. & Jackson, R.B. 2004. Groundwater use and salinization with grassland afforestation. *Global Change Biology*, 10(8): 1299–1312.

Kämpf, N., Curi, N. & Marques, J. 2009. Weathering and minerals in the soil. In Vander de Freitas Melo & Luis Alleoni, eds. *Ing. Química e mineralogía do solo*. Parte I. Conceitos básicos, pp. 333-379. Sociedad Brasileira de Ciência do solo. [in Portuguese]

Laegreid, M., Bockman, O.C. & Kaarstad, E.O. 1999. *Agriculture, Fertilizers and the Environment*. Norks Hydro, CABI Publishing. 294 pp.

Lal, R. 2005. Forest soils and carbon sequestration. *Forest Ecology and Management*. 220: 242-258.

Lal, R. 2006. Soil Carbon Sequestration in Latin America. In: R. Lal, C.C. Cerri, M. Bernoux, J. Etcheverd & E. Cerri, eds. *Carbon Sequestration in Soils of Latin America*, pp. 49-64. USA, New, York & London, Oxford, The Haworth Press, Inc.

Lapinel, B., Rivero, R.E. & Cutié, V. 1993. *La sequía en Cuba: Análisis del período 1931-1990 (inédito).* Informe Científico-Técnico. Camagüey, Centro Meteorológico Territorial. 40 pp.

Lavado, R.S. & Taboada, M.A. 2009. The Argentinean Pampas: A key region with a negative nutrient balance and soil degradation needs better nutrient management and conservation programs to sustain its future viability as a world agroresource. *Journal of Soil and Water Conservation*, 65: 150-153.

Liniger, H.P., van Lynden, G., Nachtergaele, F., Schwilch, G. & Biancalani, R. 2011. *Questionnaire for mapping land degradation and sustainable land management v 2.0*. LADA Technical Report No 9. Rome, FAO.

Malm, O. 1998. Gold Mining as a Source of Mercury Exposure in the Brazilian Amazon. *Environmental Research*, 77(2): 73–78.

Milesi Delaye, L.A., Irizar, A.B., Andriulo, A.E. & Mary, B. 2013. Effect of Continuous Agriculture of Grassland Soils of the Argentine Rolling Pampa on Soil Organic Carbon and Nitrogen. *Applied and Environmental Soil Science*, 10: 1-17.

Miyake, S., Renouf, M., Peterson, A., McAlpine, C. & Smith, C. 2012. Land-use and environmental pressures resulting from current and future bioenergy crop expansion: A review. *Journal of Rural Studies*, 28(4): 650-658.

Mol, J.H., Ramlal, J.S., Lietar, C. & Verloo, M. 2001. Mercury Contamination in Freshwater, Estuarine, and Marine Fishes in Relation to Small-Scale Gold Mining in Suriname, South America. *Environmental Research Section* A., 86: 183-197.

Moreira, L., Dos Santos Teixeira, A. & Galvão, L. 2014. Laboratory Salinization of Brazilian Alluvial Soils and the Spectral Effects of Gypsum. **Remote Sens**., 6: 2647-2663.

Nachtergaele, F.O., Petri, M. & Biancalani, R. 2011. *Global Land Degradation Information System (GLADIS) manual v 1.0.* LADA Technical Report No 17. Rome, FAO.

Nogueira, M.A., Albino, U.B., Brandão-Junior, O., Braun, G., Cruz, F.M., Dias, B.A., Duarte, R.T.D., Gioppo, N.M.R., Menna, P., Orlandi, J.M., Raimam, M.P., Rampazo, L.G.M., Santos, M.A., Silva, M.E.Z., Vieira, F.P., Torezan, J.M.D., Hungria, M. & Andrade, G. 2006. Promising indicators for assessment of agroecosystems alteration among natural, reforested and agricultural land use in southern Brazil. *Agriculture, Ecosystems and Environment*, 115(1–4): 237–247.

Nosetto, M.D., Jobbágy, E.G., Brizuela, A.B. & Jackson, R.B. 2012. The hydrologic consequences of land cover change in central Argentina. *Agriculture, Ecosystems and Environment*, 154: 2–11.

Nriaugu, J. 1994. Mercury pollution from the past mining of gold and silver in the Americas. *Science of The Total Environment*, 149(3): 167–181.

Oldeman, L. R., van Engelen, V.W.P. & Pulles, J.H.M. 1991a. The extent of human-induced soil degradation. In L.R. Oldeman, R.T.A. Hakkeling & W.G. Sombroek, eds. *World map of the status of humam-induced soil degradation: an explanatory note*, pp. 27-33. Wageningen, International Soil Reference and Information Centre & Nairobi, United Nations Environment Programme.

Oldeman, L., Hakkeling, R. & Sombroek, W. 1991b. *World Map of the status of human- induced soil degradation*. Wageningen, GLASOD-ISRIC.

Olson, D.M., Dinerstein, E., Wikramanayake, E.D., Burgess, N.D., Powell, G.V., Underwood, E.C. & Kassem, K.R. 2001. Terrestrial Ecoregions of the World: A New Map of Life on Earth A new global map of terrestrial ecoregions provides an innovative tool for conserving biodiversity. *BioScience*, 51(11): 933-938.

Ometo, J.P.H.B., Martinelli, L.A., Ballester, M.V., Gessner, A., Krusche, A.V., Victoria, R.L. & Williams, M. 2000. Effects of land use on water chemistry and macroinvertebrates in two streams of the Piracicaba river basin, south-east Brazil. *Freshwater Biology*, 44(2): 327–337.

Pais, P.S.M., Dias Junior, M.C., Dias, A.C., Iori, P., Guimarães, P.T.G. & Santos, G.A. 2013. Load bearing capacity of a Red-Yellow Latosol cultivated with coffee plants subjected to different weed managements. *Ciênc. agrotec.*, 37(2): 145-151.

Palacios-Vélez, O. 2012. *Problemas de drenaje y salinidad en los Distritos de Riego de México: Panorámica. México*, Postgrado de Hidrociencias, Campus Montecillo del Colegio de Postgraduados en Ciencias Agrícolas

Palm, C. & Sanchez, P. 1990 Decomposition and nutrient release patterns of the leaves of three tropical legumes. *Biotropica*, 22(4): 330-338.

Paruelo, J.M., Guerschman, J.P. & Verón, S.R. 2005. Expansión agrícola y cambios en el uso del suelo. *Revista Ciencia Hoy*, 15(87): 14-23.

Pengue, W. 2005. Transgenic crops in Argentina: The ecological and social debt. *Bulletin of Science, Technology & Society*, 25(4): 314-322.

Pla, I. 1992. La Erodabilidad de los Andisoles en Latino América. *Suelos Ecuatoriales*, 22(1): 33-43.

Pla, I. 1993. Erosión de suelos de ladera del trópico Andino y Centroamericano. *In* F. Munevar, ed. *Manejo Integrado de Recursos Naturales en Ecosistemas Tropicales para una Agricultura Sostenible*, pp. 21-36. Colombia, Sta Fe de Bogotá, ICA.

Pla, I. 1996a. Degradación de suelos en zonas de ladera de América Latina. *In* F. Bertsch & C. Monreal, eds. *El Uso Sostenible del Suelo en Zonas de Ladera: El Papel Esencial de los Sistemas de Labranza Conservacionista*, pp. 28-49. Costa Rica, San Jose, FAO-MAG.

Pla, I. 1996b. Sistemas y prácticas de conservación de suelos y aguas. *In Planificación y Manejo Integrado de Cuencas Hidrográficas en Zonas Áridas y Semiáridas de América Latina*, pp. 131-160. Chile, Santiago de Chile, FAO-PNUMA.

Pla, I. 2003. Erosion research in Latin-America. In D. Gabriels & W. Cornelius, ed. 25 *Years of Assessment of Erosion*, pp. 19-27. Bélgica, Gent, Univ. of Gent.

Pla, I. 2011. Evaluación y Modelización Hidrológica para el Diagnóstico y Prevención de"Desastres Naturales". *Gestión y Ambiente*, 14(3): 17-22.

PNUMA-CEPAL. 2010. *Gráficos Vitales de Cambio Climático para América Latina y El Caribe*. Santiago de Chile, CEPAL.

Prego, A.J., Barnes, H.R. & Battioli, M.C. 1988. Environmental degradation in Argentina: soil, water, vegetation, fauna. Fundacion para la Educacion, la Ciencia y la Cultura (FECIC). Centro para la Promocion de la Conservacion del Suelo y del Agua (PROSA), Buenos Aires. 504 pp. [in Spanish]

Restrepo, J.D., Kjerfve, B., Hermelin, M. & Restrepo, J.C. 2006. Factors controlling sediment yield in a major South American drainage basin: the Magdalena River, Colombia. *Journal of Hydrology*, 316: 213–232.

Romaniuk, R., Giuffré, L., Costantini, A., Bartoloni, N. & Nannipieri, P. 2012. A comparison of indexing methods to evaluate quality of soils: the role of soil microbiological properties. *Soil Research*, 49(8): 733-741.

SAGyP-CFA. 1995. *El deterioro de las tierras en la República Argentina*. Alerta Amarillo. Secretaría de Agricultura, Ganadería y Pesca & Consejo Federal Agropecuario. 287 pp.

Sainz Rozas, H., Echeverria, H. & Angelini, H. 2012. Available phosphorus in agricultural soils of the pampa and extra-pampas regions of Argentina. *RIA, Revista de Investigaciones Agropecuarias*, 38: 33-39.

Sainz Rozas, H.R., Echeverria, H.E. & Angelini, H.P. 2011 Organic carbon and pH levels in agricultural soils of the pampa and extra-pampean regions of Argentina. *Ciencia del Suelo*, 29: 29-37.

Sánchez, P. 1981. *Suelos del trópico. Características y manejo*. Costa Rica, San José, Instituto Interamericano de cooperación para la agricultura IICA. 634 pp.

Santayana, S. 2012. *Problemas de drenaje y salinidad en la costa peruana*. XI Congreso Nacional de Ingeniería Agrícola. Perú, Lima.

Sasal, M.C., Andriulo, A. & Taboada, M.A. 2006. Soil porosity characteristics on water dynamics under direct drilling in Argiudolls of the Argentinean Rolling Pampas. *Soil and Tillage Research*, 87: 9-18.

Satorre, E.H. 2005. Cambios tecnológicos en la agricultura Argentina actual. *Revista Ciencia Hoy*, 15(87): 24-31.

Sicardi, M., Garcıa-Préchac, F. & Frioni, L. 2004. Soil microbial indicators sensitive to land use conversion from pastures to commercial Eucalyptus grandis (Hill ex Maiden) plantations in Uruguay. *Applied Soil Ecology*, 27(2): 125–133.

SIIA. 2015. Integrated system of agricultural information. Presidencia de la Nación, Ministerio de Agricultura, Ganadería y Pesca. (Also available at: http://www.siia.gov.ar/) [in Spanish]

Silveira, K., Ribeiro, M., De Oliveira, L., Heck, R. & Silveira, R. 2008. Gypsum-saturated water to reclaim alluvial saline sodic and sodic soils. *Sci. agric. (Piracicaba, Braz.)*, 65(1): 69-76.

Sisti, C.P.J. , dos Santos, H.P., Kochhann, R.A. , Alves, B.J.R. , Urquiaga, S. & Boddey, R.M. 2004. Change in carbon and nitrogen stocks in soil under 13 years of conventional or zero tillage in southern Brazil. *Soil & Tillage Research*, 76: 39-58.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, B. & Sirotenko, O. 2007. Agriculture. In B. Metz, O.R. Davidson, P.R. Bosch, R. Dave, L.A. Meyer, eds. *Climate Change 2007*: Mitigation Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. UK, Cambridge, Cambridge University Press & USA, New York, NY.

Sousa, D.M. 2011. *Solo: Recurso fundamental para o desenvolvimento da Agricultura na Savana tropical*. EMBRAPA-Cerrados. XIX Congreso Venezolano de la Ciencia del Suelo.Calabozo, Noviembre.

Spavorek, G., Bemdes, G., Barreto, A.G. & Klug, L. 2012. The revision of the Brazilian Forest Act: increased deforestation or a historic step towards balancing agricultural development and nature conservation? *Environ. Sci. Policy*, 16: 65-72.

Steinbach, H.S. & Álvarez, R. 2006. Changes in soil organic carbon contents and nitrous oxide emissions after introduction of no-till in Pampean agroecosystems. *Journal of Environmental Quality*, 35: 3-13.

Studdert, G.A.., Echeverría, H.E. & Casanovas, E.M. 1997. Crop-pasture rotation for sustaining the quality and productivity of a Typic Argiudoll. *Soil Science Society of America Journal*, 61(5): 1466-1472.

Szabolcs, I. 1979. *Review on Research of Salt Affected Soils*. Natural Resources Research XV. Paris, UNESCO.

Taboada, M.A., Micucci, F.G., Cosentino, D.J. & Lavado, R.S. 1998. Comparison of compaction induced by conventional and zero tillage in two soils of the Rolling Pampa of Argentina. *Soil and Tillage Research*, 49: 57–63.

Taboada, M.A., Rubio, G. & Chaneton, E.J. 2011. Grazing impacts on soil physical, chemical and ecological properties in forage production systems. *In* J.L. Hatfield & T.J. Sauer, ed. *Soil management: building a stable base for agriculture*, p. 301-320. American Society of Agronomy & Soil Science Society of America.

Torri, S.I. & Lavado, R.S. 2008. Dynamics of Cd, Cu and Pb added to soil through different kinds of sewage sludge. *Waste Management*, 28(5): 821-832.

UNDP. 2000. *A Guide to World Resources*. United Nation Development Programme. Washington, DC, World Resources Institute. 389 pp.

UNFCCC. 2013. *Non-Annex I national communications*. (Also available at http://unfccc.int/national_reports/non-annex_i_natcom/items/2979.php)

Urquiaga, S, Alves, B.J.R.**J,** Antalia, C.P.**, Martins, M.R. &** Boddey, R.M. 2014. A cultura de milho e seu impacto nas emissões de GEE no Brasil. *In Eficiência nas cadeias produtivas e o abastecimento global*. pp. 61-71.1 edition. Sete Lagoas, MG, ABMS.

Urquiaga, S., Xavier, R.P.**, de** Morais, R.F. **, Batista, R.B.,** Schultz, N.**,** Leite, J.M.**, Maia e Sá, J., Barbosa, K.P., de Resende, A.S., Alves, B.J.R &** Boddey, R. 2012. Evidence from field nitrogen balance and 15N natural abundance data for the contribution of biological N_2 fixation to Brazilian sugarcane varieties. *Plant and Soil*, 356: 5 – 21.

van der Werf, G.R., Randerson, J.T., Giglio, L., Collatz, G.J., Mu, M., Kasibhatla, P.S., Morton, D.C., DeFries, R.S., Jin, Y. & van Leeuwen, T.T. 2010. Global fire emissions and the contribution of deforestation, savanna, forest, agricultural, and peat fires (1997– 2009). *Atmos. Chem. Phys.*, 10: 11707–11735.

Vargas, R., Allen, M.F. & Allen, E.B. 2008. Biomass and carbon accumulation in a fire chronosequence of a seasonally dry tropical forest. *Global Change Biology*, 14: 109-124.

Vargas, R., Yepez, E.A., Andrade, J.L., Angeles, G., Arredondo, T., Castellanos, A.E., Delgado-Balbuena, J., Garatuza-Payan, J., Gonzales del Castillo, E., Oechel, W., Rodriguez, J.C., Sanchez-Azofeifa, G.A., Velasco, E., Vivoni, E.R. & Watts, C. 2013. Progress and opportunities for monitoring greenhouse gases fluxes in Mexican ecosystems: the MexFlux network. *Atmosfera*, 26: 325-336.

Viglizzo, E.F. & Frank, F.C. 2006. Land-use options for Del Plata Basin in South America: Tradeoffs analysis based on ecosystem service provision. *Ecol. Econ.*, 57: 140-151.

Viglizzo, E.F. & Jobbágy, E. (eds.). 2010. *Expansión de la frontera agropecuaria en Argentina y su impacto ecológico-ambiental*. Buenos Aires, Ediciones INTA.

Villafañe, R. 1995. Detección de suelos afectados por sales en áreas bajo riego de los estados Portuguesa, Barinas y Lara, Venezuela. *Agronomía Tropical*, 54(3): 445-456.

White, D., Arca, M., Alegre, J., Yanggen, D., Labarta, R., Weber, J.C., Sotelo, S. & Vidaurre, H. 2005. The Peruvian Amazon: Development Imperatives and challenges. In C.A. Palm, S.A. Vosti, P.A. Sanchez & P.J. Ericksen, eds. *Slash and Burn: the search for alternatives*, pp. 332-354. USA, New York, Columbia University Presss. 463 pp.

Willer, H. & Lukas K. (eds.). 2009. *The World of Organic Agriculture. Statistics and Emerging Trends 2009*. FIBL-IFOAM Report. Bonn, IFOAM,, Frick, FiBL &, Geneva, ITC.

World Bank. 2007. *Republic of Peru environmental sustainability: a key to poverty reduction in Peru*. Washington, DC. (Also available at http://documents.worldbank.org/curated/en/2007/06/7910058/republic-peru-environmental-sustainability-key-poverty-reduction-peru.

13 | Regional Assessment of Soil Changes in the Near East and North Africa

Regional Coordinator: Seyed Kazem Alavi Panah (ITPS/Iran)

Regional Lead Author: Mubarak Abdelrahman Abdalla (Sudan)

Contributing Authors: Hedi Hamrouni (Tunisia), Abdullah AlShankiti (ITPS/Saudi Arabia), Elsiddig Ahmed El Mustafa El Sheikh (ITPS/Sudan), Ali Akbar Noroozi (Iran).

13.1 | Introduction

The Near East and North Africa (NENA) region includes Tunisia, Algeria, Morocco, Libya, Egypt, Sudan, South Sudan, Jordan, Israel, Lebanon, Syria, Palestine, Iraq, Yemen, Saudi Arabia, Oman, Qatar, United Arab Emirates, Kuwait, Bahrain and Iran. The region has a land area of approximately 14.9 million km^2, nearly all of which is hyper-arid, arid or semi-arid. The region faces three climatic constraints: aridity, recurrent drought, and desertification, the latter also in part human induced. South Sudan is the only country in the region which falls within the dry sub-humid tropical zone. Large areas of Libya, Egypt, Bahrain, Kuwait, Qatar and the United Arab Emirates are entirely desert (FAO, 2013a).

The soils of the region are broadly as follows:

- The soils of the Maghreb region (Morocco, Tunisia, Libya and Algeria) fall into three broad divisions: (i) along the Mediterranean and Atlantic coasts productive Kastanozems (Xerolls) and Luvisols (Alfisols) occur (SEDENOT, 1999; MADRPM, 2000; Halitima, 1988); (ii) Leptosols (lithic subgroups) and Cambisols (Inceptisols) are found in the Atlas Mountains away from the coast (Yigini, Panagos and Montanarella, 2013); and (iii) Calcisols (Calcids), Gypsisols (Gypsids), Leptosols and Cambisols are found in the southern part (Jones *et al.*, 2013).
- Vertisols, Arenosols (Psamments), Fluvisols (Fluvents), Calcisols and Gypsisols (Aridisols) are the dominants soils in Sudan and Egypt.
- In the Mashreq region (Jordan, Syria, Lebanon, Iraq and Palestine), the soils of the valleys are Arenosols (Psamments) and Fluvisols (Fluvents). In the highlands, steppe and desert regions, the main orders are Calcisols (Calcids) and Cambisols (Aridisols), Arenosols (Psamments) and Leptosols (Lithic subgroups), and Vertisols which are calcareous in the subsoil horizons.
- In the Arabian Peninsula and the Gulf (Oman, Kingdom of Saudi Arabia, Kuwait, Bahrain, United Arab Emirates, Yemen, Iran and Qatar), there are alluvial soils rich in silt and desert soils, and sandy soils poor in organic carbon but in which evaporite Tertiary Formations played an important role in the formation of contemporary minerals (Abbaslou *et al.*, 2013).

Agriculture is an important source of income for many countries in the region. Arable land constitutes only 6.8 percent of the total land area, while about 26 percent is used for pasture and about 7 percent is under forest (Hamdallah, 1997). Based on the type of agriculture practiced, Dregne and Chou (1992) divided the productive land in the region into: irrigated land (0.7 percent, 8.5 million ha); rainfed (2.2 percent, 27.7 million ha); rangelands (40 percent, 495.6 million ha); and extremely arid land (57 percent, 705.2 million ha). Proportions vary considerably by country. Productive lands represent 30 percent of the total area in Syria and Lebanon, but only 3 percent in Egypt, Algeria and Sudan, and only 0.5 percent in Saudi Arabia, Oman and Mauritania (Mamdouh Nasr, 1999).

Natural resource degradation, especially where agriculture is practiced, is a real threat in all countries of the region and remains a major limitation to the reliable supply of food. In most countries, salinization, water and wind erosion, loss of vegetation cover, soil physical degradation (including compaction and surface crusting) are the main threats to the soil's capacity to provide ecosystem services. The expansion of agriculture into marginal lands has greatly aggravated water erosion and consequently soil degradation. In almost all countries in the region, extreme climatic conditions, overgrazing, unsuitable cropping patterns and accumulation of salts have rendered large areas of land unproductive (Abahussain *et al.*, 2002).

Mamdouh Nasr (1999) reported that rainfed cropland represents only 3 percent (about 30 million ha) of the region's total drylands, yet about 22 million ha of this total (73 percent of the cropland area) are estimated by UNDCPAC to be degraded. The extent of degradation of rainfed cropland is greatest in the countries of northern Africa: Algeria (93 percent), Morocco (69 percent), Tunisia (69 percent) and, exceptionally, Egypt (10 percent). The eastern sub-region (Iran, Iraq, Jordan, Lebanon, Sudan and Syria) is the part of the NENA region most affected by land degradation (FAO, 2004). The extent of degradation in the countries of the Middle East - Iraq (72 percent), Syria (70 percent) - is higher than that in the Gulf Countries: Oman (50 percent), Qatar (25 percent), and Bahrain (20 percent).

The Arab Centre for the Study of Arid Zones and Dry Lands (CAMRE/UNEP/ACSAD, 1996) has estimated that, overall, land degradation affects approximately 49 percent of farmland in the eastern sub-region; 29 percent in the Nile Valley of Egypt; 17 percent in North Africa; and 9 percent in the Gulf Cooperation Council Countries. More than one process of degradation can occur in a single farming system. For example, degradation is serious in one of the largest countries in the region, Sudan, a country with high agricultural potential (Ayoub, 1998). In Sudan, the 46 million ha lying in the semi-arid zone, where mixed farming of both animal husbandry and rainfed arable cropping are practiced, have experienced intensive soil degradation over the last 35 years, affecting production of field crops, gum Arabic, and livestock products.

In North Africa, causes of soil degradation are divided between overgrazing (68 percent), over-cultivation (21 percent), deforestation (10.5 percent) and overexploitation of natural vegetation for about 0.5 percent (Thomas and Middelton, 1994). In the newly created country of the region, South Sudan, there are very limited studies on land degradation. However, there are indications that land use changes have impaired the quality of the land in many places (Dima, 2006).

Land degradation in certain areas may affect adjacent areas. For example, degradation in rangelands where rainfall is low has negative effects on resources of rainfed farming areas. This is also one of the regions most vulnerable to climate change (FAO, 2011). Agriculture faces major losses due to land degradation, and yields are expected to decrease by the year 2050: rice yields by 11 percent; soybean yields by 28 percent; maize yields by 19 percent; and barley grain yields by 20 percent (FAO, 1994). Recent studies on the economic cost of land degradation in the region were reported by Hussein *et al.* (2008); they were estimated at US$9 billion yr^{-1} (2.1-7.4 percent of GDP).

This chapter will discuss the main soil threats - erosion by water and wind, salinity/sodicity, soil contamination, and organic C depletion. Major causes of soil degradation in the region are due to many factors, including: (i) excessive irrigation and poor drainage; (ii) wind and water erosion; (iii) waterlogging; (iv) deteriorated soil fertility; (v) over-grazing; (vi) loss of soil cover; (vii) land mis-management; (viii) sand encroachment; and (ix) overuse of herbicides, pesticides and chemical fertilizers (FAO, 2004).

Data correlating land degradation with yields are scarce at the global level. However, recent studies (2000-2010) show that land losses due to degradation in North Africa are the highest among selected countries worldwide, and that this has resulted in a considerable food gap of 0.6×106 metric tonnes (Wiebe, 2003). There have been many efforts to tackle the issue of land degradation. Experience has shown that a key factor for success is political will. In this respect, the 21st summit of African leaders in 2013 urged member states to place land degradation at the centre of the debate on the post-2015 development agenda, and to recognize it as one of the sustainable development goals. This is particularly important for NENA because the agricultural sector in the region contributes about 10 percent of the region's GDP, but is characterized by an exceptionally fragile and vulnerable resource base.

13.2 | Major land use systems in the Near East and North Africa

Land use systems in the region have been identified and broadly delimited based on a range of characteristics (Nachtergaele and Petri, 2011). Their geographical location is indicated in Figure 13.1. They can be combined in three major systems: irrigated crop based, rainfed mixed and livestock based. These systems are briefly discussed below (Dixon and Gulliver, 2001).

Irrigated land use systems

Given the arid and semiarid nature of much of North Africa and the Near East, irrigated farming has always been of crucial importance in generating much of the region's agricultural output. The 'irrigated farming system' in NENA contains both large and small-scale irrigation schemes with high population densities and generally very small farm sizes. The prevalence of poverty within both large and small segments of the system is moderate.

Traditionally, areas within the large scale irrigation sub-system have been linked primarily to perennial surface water resources, such as the Nile (Egypt and Sudan) and Euphrates (Syria and Iraq). However, the intensification of traditional karez or qanat systems has also led to the evolution of large-scale irrigated areas where sub-surface water is abundant. More recently, the availability of deep drilling and pumping technologies has permitted the development of new areas drawing entirely on subterranean aquifers. Large-scale schemes are found across all zones of the region and include high-value cash and export cropping and intensive vegetable and fruit cropping.

Patterns of water use vary greatly but throughout the region inappropriate policies on water pricing and centralised management systems have meant that water is seldom used efficiently. Significant economic and environmental externalities have arisen through excessive utilisation of non-recharged aquifers while, in a number of cases, the excessive application of irrigation water has resulted in rising groundwater tables, soil salinization and sodification problems.

The small scale irrigated sub-system also occurs widely across the region. Although not as important as the larger schemes in terms of numbers of people involved or in the amount of food and other crops produced, it is a significant element in the survival of many people in arid and remote mountain areas. This sub-system, examples of which are sometimes of considerable antiquity, typically develops along small perennial streams and at oases, or where flood and spate irrigation is feasible. It sometimes also draws on shallow aquifers and boreholes, although these rarely penetrate to the depths seen in the large schemes. The major crops grown within small-scale irrigation areas are mixed cereals, fodder and vegetables. These areas also provide important focal points for socio-economic activity, but intense local competition for limited water resources between small rural farmers and other users is becoming increasingly evident.

Rainfed mixed crop and livestock land use systems

The rainfed agricultural and livestock systems are the most important land use system in the region in terms of population engaged in agriculture. However, as these systems are practiced on less than 10 percent of the land area, population densities in these farming areas are moderately high. These systems covers two sometimes overlapping segments. The first segment occurs on high terraces and is dominated by rainfed cereal and legume cropping, with tree crops, fruits and olives on terraces, together with vines. In Yemen, higher reaches are reserved for qat trees and coffee, which are traditionally the most important tree crops in Yemen's mountain regions. The second segment is based primarily on the raising of livestock (mostly sheep) on communally managed lands. In some cases, both the livestock and the people who control them are transhumant, migrating seasonally between lowland steppes in the more humid winter season and upland areas in the dry season. This type of livestock keeping is still important in Iran and Morocco.

Poverty within this system is extensive, as markets are often distant, infrastructure is poorly developed and the degradation of natural resources is a serious problem. In the lowlands where rainfed production is feasible, an increasing area is now benefiting from the availability of new drilling and pumping technologies, which have made it possible to use supplementary winter irrigation on wheat and full irrigation on summer cash crops. There is some dry season grazing of sheep migrating from the steppe areas.

The more humid areas (with 600 to 1000 mm annual rainfall) that occur in the Caspian and Mediterranean coastal areas are characterised by tree crops (olives and fruit), melons and grapes. There is also some protected cropping with supplementary irrigation for potatoes, vegetables and flowers. Common crops are wheat, barley, chickpeas, lentils and fodder crops. Poverty in these more humid areas is moderate, but would be higher without extensive off-farm income from seasonal labour migration.

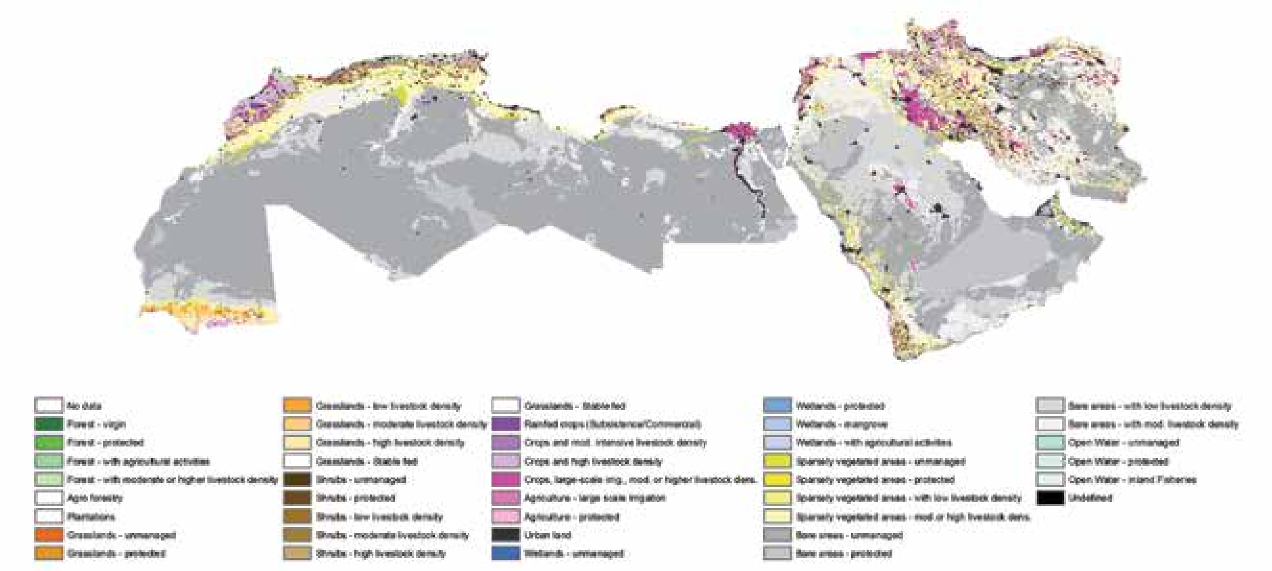


Figure 13.1 | Land use systems in the Near East and North Africa.
Source: FAO, 2010.

Livestock-based land use systems in sparsely vegetated areasù

The pastoral land use system, mainly involving sheep and goats but also with some cattle and camels, is practiced on large areas of semiarid steppe lands, and is characterised by low population densities, with more densely populated areas around irrigated settlements. There are irrigated croplands scattered throughout the system, thus boosting the agricultural population – and helping to support a cattle population. Strong linkages exist to other farming systems through the movement of stock, both through seasonal grazing of herds in more humid areas and through the sale of animals to large feedlots located around urban areas. Seasonal migration, which is particularly important as a risk minimisation measure, depends on the availability of grass, water and crop residues in neighbouring arable systems. Nowadays, pastoral herds are often partially controlled and financed by urban capital. Where water is available, small areas of crop production have been developed to supplement the diets and income of pastoral families. However, such sites are few and poverty within the system is extensive.

The sparse (arid) land use system covers more than 60 percent of the region and includes vast desert zones. People are concentrated in oases and on a number of irrigation schemes (notably in Tunisia, Algeria, Morocco and Libya). Part of the land is irrigated and utilised for the production of dates, other palms, fodder and vegetables. Pastoralists within this system also raise camels, sheep and goats. Following scattered storms and in good seasons, the system provides opportunistic grazing for the herds of pastoralists. The boundary between pastoral grazing and sparse agriculture systems is indistinct and depends on climatic conditions. Poverty within this system is generally low as population pressure is limited.

13.3 | Major threats to soils in the region

13.3.1 | Erosion

Water erosion

Water erosion is predominant in the part of the region which has sloping lands and where rainfed agriculture is practiced, although it may also occur in gently sloping areas. The degree of water erosion depends on the intensity and duration of the rainstorms, often enhanced by the terrain attributes and land use practices, particularly where these have reduced land cover. Water erosion results in the removal of fertile soil and in the reduction of irrigation efficiency and storage capacity. Considerable volumes of soil may be lost. Based on the GLASOD survey quoted by Abahussain *et al.* (2002), the total area affected by water erosion in NENA has been estimated at about 41 million ha. However, the extent varies significantly by country (Table 13.1.).

Table 13.1 | Land degradation caused by water erosion in the NENA region (1000 ha)
Source: Abahussain et al., 2002.

1 *Azimzadeh et al.*, 2008

Country	Area	Country	Area	Country	Area
Algeria	3 900	Lebanon	65	Sudan	17 300
Bahrain	0	Libya	1 300	South Sudan	n.d.
Egypt	Negligible	Morocco	3 600	Syria	1 200
Iran[1]	70 000	Oman	2 800	Tunisia	3 800
Iraq	1 150	Palestine	n.d.	United Arab Emirates	0
Jordan	330	Qatar	0	Yemen	5 600
Kuwait	0	Saudi Arabia	200		

Wind erosion

A review conducted by Abahussain *et al.* (2002) indicated that more than half of the total area in the region received annual rainfall of less than 150 mm. Consequently, large areas are without plant cover, or the cover is very sparse. This situation is aggravated further by high land-use pressure, both from human and animals, which ultimately causes severe topsoil disturbance. The resulting wind erosion is the most common environmental problem in the region and accounts for approximately 60 percent (135 million ha) of soil degradation. Countries differ in the extent they are affected, with Saudi Arabia the most affected (Table 13.2). Wind erosion has resulted in detrimental effects on land quality by removing the fertile top soils. In addition, the accumulation of eroded materials in irrigation canals, agricultural fields (sand encroachment) and water harvesting points affects the cropped areas in the region severely.

Table 13.2 | Soil degradation caused by wind erosion in the NENA region (1000 ha).
Source: Abahussain et al., (2002).

1 *Azimzadeh et al., 2008*

Country	Area	Country	Area	Country	Area
Algeria	12 000	Lebanon		Sudan	71 000
Bahrain	n.d.	Libya	24 000	South Sudan	n.d.
Egypt	1 400	Morocco	600	Syria	3 000
Iran[1]	20 000	Oman	4 000	Tunisia	4 000
Iraq	3 000	Palestine	n.d.	United Arab Emirates	1 100
Jordan	3 000	Qatar	200	Yemen	6 000
Kuwait	300	Saudi Arabia	50 000		

An example of the problems is provided by the largest irrigated scheme in the world – the Gezira scheme in Sudan – which has been badly affected by sand encroaching from surrounding areas. One study found that over the last decade, wind erosion has decreased the soil level outside the scheme by 10 cm, whereas soil depth increased inside the scheme by 30 cm - any increase in level beyond 20 cm prevents irrigation water flow. Inside the scheme, the topsoil texture has changed from clayey to more sandy. Dune displacement in the scheme was about 3-5 m month^{-1} during the season of active sand movement (Al-Amin, 1999). This situation has forced famers to take considerable land out of production because of problems irrigating, with sand filling the irrigation canals and altering slopes (Mohammed, Stigter and Adam, 1995).

The continental sands of Kordofan and Darfur (Sudan) which support the world's major source of gum Arabic (Acacia senegal L.) encounter severe wind erosion due to overgrazing and mismanagement, which endangers the dominance of the species in the area. Omar *et al.* (1998) also gave the example of sand encroachment into the cultivated areas of south Kuwait that has forced famers to take 35 percent of their total cultivable land out of production. Water and wind erosion are also prevalent in the western area of Libya (Jifara Plain and Jebal Naffusah) where agricultural development is dominant, and in El-Witia area in the southwestern Jifara Plain as observed by the Remote Sensing Center in Tripoli (Ben-Mahmoud, Mansur and AL-Gomati, 2000). For example, the area of fertile cultivated soils in El-Witia classified as class 1 has decreased (1986 to 1996) from 66 000 to 44 900 ha (a 31 percent reduction), whereas the area of soils low in fertility has increased from 97 000 to 134 000 ha (a 38 percent increase). In addition, the area covered with sand dunes has increased from 32 000 to 104 000 ha (a 52 percent increase).

The Nile Valley system comprising around 2.63 million ha represents the most fertile lands in Egypt - and probably in the whole region. However, wind erosion and salinity have degraded the soil resource (El-Kholei, 2012). Sand encroachment and mobile dunes are now estimated to cover more than 16 percent of the total area of Egypt. These conditions have led to active sand encroachment on the fringes of the cultivated areas in most areas of the country. An estimated area of 0.76 million ha has been reported to be affected by sand encroachment and active dunes, causing reduction in productivity of as much as 25 percent. The annual soil loss was estimated to be about 1.0 million ha.

In Jordan, the flat topography (<8 percent) and low annual rainfall (< 200 mm) are responsible for active and strong winds carrying sediments. The severity of wind erosion in the country was reported by Khresat, Rawajfih and Mohammad, (1998) who stated that erosion by wind and water is considered the major cause of soil degradation in the north-western part of Jordan.

13.3.2 | Soil organic carbon change

There is very little information in the region relating to changes in soil organic carbon (SOC). The prevailing arid and semi-arid conditions combined with high temperatures result in very low contents of SOC in most of the region. Although in many countries of the region, the reduction in topsoil SOC cannot be overlooked, the value of remedial measures has not yet been proven. One study examined a very long-term (more than 70 years) management practice of sewage sludge application aimed at increasing C in sandy to loamy sandy soils. However, this resulted in an increase in SOC of only 1.33 percent, which indicates a very slow SOC accumulation (Abd Elnaim *et al.*, 1987).

13.3.3 | Soil contamination

Soil contamination in the region is most prevalent in countries with high population, high oil production or heavy mining. The increase of population is coupled with huge increases in both solid and liquid wastes that are dumped on land or into water resources causing degradation through pollution. The overuse of chemical fertilizers and the residues of applied pesticides are also sources of pollution of soil and water resources.

In Egypt, the construction of the High Dam reduced soil fertility as the result of sediment load reduction. This in turn forced farmers to rely heavily on inorganic fertilizers which led to high levels of nitrogen and phosphorous in run-off and drainage water, causing an off-site impact on water quality (NAP, 2002). The discharge of industrial effluents, agricultural drainage water and effluent from navigation activities into the Nile and into main canals and drains contaminates the surface water resource. Industrial effluent also directly affects the water quality of the River Nile system. This comes from sugar factories; cement and fertilizer plants; and plants producing iron and steel, coke and chemicals (EEAA, 1994). Pollutants then accumulate in waterways where, at high concentrations (Biochemical Oxygen Demand and phosphates and total dissolved salts), they cause harmful chemical and biological impacts. Other sources of soil pollution are mainly due to the impacts of heavy mining, agro-chemical residues, and the oil and other industries.

13.3.4 | Soil acidification

Acidification is not commonly a problem in the region and is restricted to coastal areas with relatively high rainfall that tends to leach bases from the soil. Land use that removes all harvested materials may also result in soil mining and acidification. The excessive use of nitrogenous fertilizers, in particular the use of acidifying fertilizers (e.g. ammonium sulphate) has in some cases caused reduction in pH. However, in Vertisols with high cation exchange capacity (over 60 cmole kg^{-1}), this reduction is found to be temporary because such soils have very good buffering capacities which can restore the initial soil pH.

13.3.5 | Soil salinization/sodification

Previous analyses (Hussein, 2001) revealed that 11.2 percent of the region's soils are affected by various levels of soil salinization. Salt-affected soils vary in extent by country from 10-15 percent in Algeria to over 50 percent in Iraq. In the United Arab Emirates 33.6 percent of the area is salinized (EAD, 2009). About 50 percent of the reclaimed lands in the Euphrates plain in Iraq and Syria are seriously affected by salinization and waterlogging, and about 54 percent of the cultivated area in Saudi Arabia suffers from moderate salinization (CAMRE/UNEP/ACSAD, 1996). In Egypt, 93 percent of the cultivated lands are affected by salinization and waterlogging. The salt-affected area in Iran has increased from 15.5 Mha in 1960 to 18 Mha in 1980, to 23 Mha in 1990, and to more than 25 Mha today (Qadir, Qureshi and Cheraghi, 2008). In the United Arab Emirates, areas along the coast sabkha (salt marshes or lagoonal deposits) are considered highly degraded due to high levels (28.8 dS m^{-1}) of salinity (Abdelfattah, 2012). In the coastal region of the Abu Dhabi Emirate, salinity is more than 200 dS m^{-1} (Abdelfattah and Shahid, 2007).

Saline and sodic soils are influenced by climate, agricultural practices, irrigation methods and policies related to land management (FAO, 1997). Low annual precipitation and high temperatures have also contributed to problems of salinity. In many countries of the region, where irrigation completely depends on groundwater, excessive irrigation has caused the formation of a shallow water table leading to increased salinization and degradation of the soil resource base. Yield reduction due to salinization and/or waterlogging amounts to 25 percent in Egypt, and has led to a complete loss of productivity and abandoned agricultural lands in several countries. From the very scattered information on the extent and characteristics of salt-affected soils, salinity and sodicity in the region is rapidly increasing, both in irrigated and non-irrigated areas. Salinity, sodicity or the combination of both in some countries of the region are seriously affecting productive areas such as the Nile Delta of Egypt, and the Euphrates Valley in Iraq and Syria. The situation is further complicated by association with problems of waterlogging and high $CaCO_3$ (up to 90 percent in United Arab Emirates, Al Barshamgi, 1997).

13.3.6 | Loss of soil biodiversity

The impact of soil degradation on biodiversity has received little attention in the countries of the region and there is little information available. Nevertheless, it is estimated that the region is home to one-tenth of the recorded plant species worldwide or about 25 000 species of plants. Of these, 25 percent are endemic to the region, 10 percent are of medicinal value, and many are a source of food. This indicates the importance of the region as a store of genetic resources (Abahussain *et al.*, 2002). The lack of proper conservation practices, overgrazing of herds of ruminants, and deforestation for fuel are causing serious losses of plant cover and of valuable genetic resources, including below-ground biodiversity that is rarely quantified in this region. Proper and sustainable utilization of plant species which yield valuable products could boost incomes and help reduce poverty amongst nomads and local settled populations. However, thousands of plant species and varieties have disappeared, and a further 800 plant species are threatened with extinction (Al-Eisawi, 1998) and this loss of plant species is likely to result in changes in soil biodiversity.

Iran is renowned for having one of the richest plant reserves in the world. The country has some 12 000 species of plants, the majority of which are endemic (the Iranian National Action Programme). In the Elmalha area of Sudan, Bakheit (2011) studied the availability and distribution of famine foods and their role in times of famine. The study revealed endemic species that are considered as alternative foods in time of crisis but which are threatened by genetic erosion due to soil degradation. Soil degradation studies in South Sudan indicate the disappearance of palatable grasses such as Panicum turgidum and appearance of less palatable grasses such as Aristidia funiculata. The alien species now covers 40 percent of the pasture area, resulting in disappearance of many wild animals and decrease in biodiversity (Elfaig, Ibrahim and Jaafar, 2015). The study pointed out that drought, unsustainable use of forest and pasture, and increase in population pressure were the main causes of this environmental degradation.

In Tunisia, decades of open grazing in the Bou Hdma National Park have caused severe loss of perennials and increased density of annuals (Belgacem, Tarhouni and Louhaichi, 2013). The study reported that grazing has reduced total plant cover by 38 percent and the contribution of perennials by 72 percent, while annuals were affected 100 percent. Overgrazing of rangelands generally causes replacement of highly palatable species with less desirable plants. Along the sea coast of Egypt, overgrazing during the period from 1974 to 1979 decreased the total density vegetation by more than 15 percent, mainly due to a decrease in some perennial herbs, while at the same time the total cover increased by about 38 percent, due mainly to perennial shrubs and succulents. Also in Egypt, vegetation on Mount Elba and the surrounding valleys has been reduced, and the resulting increased runoff threatens the diversified natural plant communities in the valleys. Some of these plant species are considered to be of high value as genetic resources as they are adapted to the desert conditions. In Lebanon biodiversity is threatened by many factors, chief among them are erosion, urban development and overgrazing resulting in dominance of xerophytes at the expense of other species (Zahreddine *et al.*, 2007). Rising levels of poverty in the Ramallah area of Palestine have led to most farmers (83 percent) turning to the collection of medicinal plants for commercial use (Abu Hammad and Tumeizi, 2012).

13.3.7 Waterlogging

Waterlogging is a common constraint in irrigated areas of the region because of inadequate drainage. The problem is exacerbated by the dominant heavy textured alluvial soils and by seepage from the conveyance canals. Soil salinity, sodicity and water logging conditions have definite adverse impacts on soil productivity, estimated to be of the order of 30-35 percent of the potential productivity. In many areas of the old Nile Valley in Egypt, waterlogging has led to increased soil salinity and in certain areas to increased soil sodicity. In the Siwa oasis, for instance, the rate of water table rise during the period 1962–1977 was 1.33 cm yr^{-1}. Subsequently, the rate increased to 4.6 cm yr^{-1} and consequently subjected fertile soil to degradation (Misak, Abdel Baki and El-Hakim, 1997).

Waterlogging has also become a serious problem on many farms in the United Arab Emirates due to poor drainage caused by the presence of a strong and thick hardpan and by excessive use of irrigation water. In addition, sea water intrusion in many areas reaches the surface and causes complete vegetation failure (Abdelfattah, Shahid and Othman, 2008). In Tunisia, of the 410 000 ha of irrigated area, about 87 000 ha (22 percent) are affected in varying degrees by waterlogging. This hydromorphy affects most of the irrigated areas in the valley of Medjerda, from Ghardimaou up to Kalaat Andalous, and also affects the majority of oases. Overall, waterlogging affects 29-67 percent of irrigated areas in the north, 35 percent in the oases of Kibili and Toezure, and to a lesser extent the plains of Dorsal and irrigated areas of Gabes and Cap Bon (14-20 percent). It also affects some irrigated areas in the far north (Nefza, Sejnane and Mateur) and some irrigated areas of the centre.

13.3.8 | Nutrient balance change

The problem of nutrient-constrained agriculture is particularly acute in the region. It is associated with land use pressure and the consequent intensification of cropping systems and related soil degradation. Nutrient depletion is increasingly affecting land productivity in the region. In Sudan, continuous cultivation over nearly a century has decreased the base saturation percentage by 25 to 42 percent, indicating leaching with irrigation water of soluble anions and cations down the soil profile. Soil degradation due to nutrient depletion in Sudan is largely concentrated in the arid and semi-arid parts, particularly in southern Kordofan and Darfur, and in the dry sub-humid and moist sub-humid zones of south-western Sudan. This soil degradation is clearly related to agricultural activities and to deforestation.

13.3.9 | Compaction

Soil compaction and crusting are the most serious forms of physical degradation affecting several irrigated areas in Libya, especially in sandy soils (Ben-Mahmoud, Mansur and Al-Gomati, 2000). Most soil compaction in the region is caused by tillage practices. For example in Iran continuous tilling over more than 50 years has exposed surface soil to water run off due to an increase of up to 33 percent of soil bulk density. Surface compaction and crusting in the Arabian Peninsula, especially in the United Arab Emirates, is often due to land filling and levelling for infrastructure development. One study found that compaction increased from a bulk density of 1.15 to 1.66 g cm^{-3} (e.g. by 44 percent) and water infiltration dropped from 267 to 52 mm h^{-1} (e.g. by 81 percent). Soil compaction due to extensive tillage operations in the furrow slice (to 30 cm) is a major physical degradation in soils with high clay content where heavy mechanization is practiced. An example is documented (Biro *et al.*, 2013) in rainfed farming systems of eastern Sudan, where three decades (1979-2009) of cultivation have increased compaction in the 0-5 and 5-15 cm depth from 1.33 and 1.42 g cm^{-3} in woodland and from 1.37 and 1.56 g cm^{-3} in fallow land to 1.56 and 1.72 g cm^{-3} (e.g. 16 percent). These levels of compaction have contributed to the general decline in productive potential in Sudan. This increase in compaction is comparable to the increase in bulk density of 13 percent documented in Jordan during the half century following conversion of forest land to cultivation of wheat and barley (Khresat *et al.*, 2008).

Military activities in the Al Salmi area on the western border of Kuwait have resulted in huge disturbance and caused soil and vegetation degradation (Al-Dousari, Misak and Shahid, 2000). The soil pores in the area have sealed and the infiltration rate has declined by 19.5 to 64.4 percent. Bulk density has increased by 26-33 percent. In the Kabd area southwest of Kuwait City, pressure on land has resulted in compaction 20 percent higher in non-protected areas (bulk density of 1.8 g cm^{-3}) than in protected areas (bulk density of 1.5 g cm^{-3}; Misak *et al.*, 2002). More generally, it has been estimated that a wide range of activities in Kuwait – grazing, quarrying, camping, and agricultural and animal production – have increased compaction by 12.9 to 23.4 percent (Al-Awadhi, Al-Helal and Al-Enezi, 2005).

13.3.10 | Sealing/capping

The population of the region is approximately 6.2 percent of the world population. The region's fragile ecosystem is endangered by one of the highest rates of population increase (3 percent) in the world. This puts enormous pressure on the capacity of land resources to provide goods and services. Encroachment of human settlements on scarce good quality agricultural land or in areas of adequate rainfall for agriculture occurs in many countries of the region, jeopardizing the role of land as a source of food. For example, in Egypt the net population density in towns is more than double the recognized maximum threshold of 360 ha^{-1}. During 1987-2007, the cultivated land in the Delta and Nile Valley did increase (to about 7 260 000 ha), but at the same time human settlement and land allocated to roads and irrigation canals and drains also increased (by 33.6 percent and 40 percent, respectively). As a result, recent studies (ESCWA, 2007) have shown that urban encroachment on highly fertile agricultural land in Egypt is emerging as a significant problem. For example, in El-Mahalla El-Kobra in the Gharbiya Governorate, the rate of urbanization from 1950 to 1987 was 10 percent annually, but from 1987 to 1995 the rate shot up to 33 percent a year. In the 1950-1987 period, annual loss of agricultural land averaged 0.4 percent but it has subsequently risen considerably.

Iran has the largest urbanized area in absolute terms in the region, followed by Saudi Arabia and Iraq, while the highest Urbanization Index is recorded for Gaza Strip, Bahrain, Palestine, Israel and Lebanon (Figure 13.2).

Land sales also play an important role in the decline in the area of productive lands. In Jordan, the agricultural sector lost about 24.3 percent of its land during the period from 1997 to 2007. Rainfed cultivation, which represented 89 percent of total cultivated land in 1983, had lost 22.6 percent of its area by 1997.

The main causes of soil problems in Jordan are: (i) improper farming practices, such as failure to use contour ploughing, or over-cultivation of the land; (ii) overgrazing; (iii) the conversion of rangelands to croplands in marginal areas where rainfall is insufficient to support crops in the long term; and (iv) uncontrolled expansion of urban and rural settlement at the cost of cultivable land.

Urban populations are growing at 8 percent a year as opposed to just 1 percent in rural areas. In some countries of the region nearly the whole population is urban (e.g. Kuwait, 97 percent; Bahrain, 90 percent; Saudi Arabia, 83 percent; and United Arab Emirates, 84 percent). This high rate of urbanization has been accompanied by conversion of agricultural lands into urban areas. In Libya, over 25 percent of highly fertile lands have been taken over by the expansion of urban areas.

The rapid urban population growth in the region increases the pressure on the natural resources (AOAD, 2004). An example of dramatic urban expansion is found in Lebanon where a study by Darwish and Khawlie (2004) showed that during the period from 1962 to 2000, urban areas expanded by 208 percent while agricultural lands decreased by 35 percent. Much of the area converted to settlements was highly productive agricultural land on Fluvisols, Luvisols and Cambisols. Some 32 percent of class 1 (prime land) and 26 percent of class 2 land were converted into urban areas.

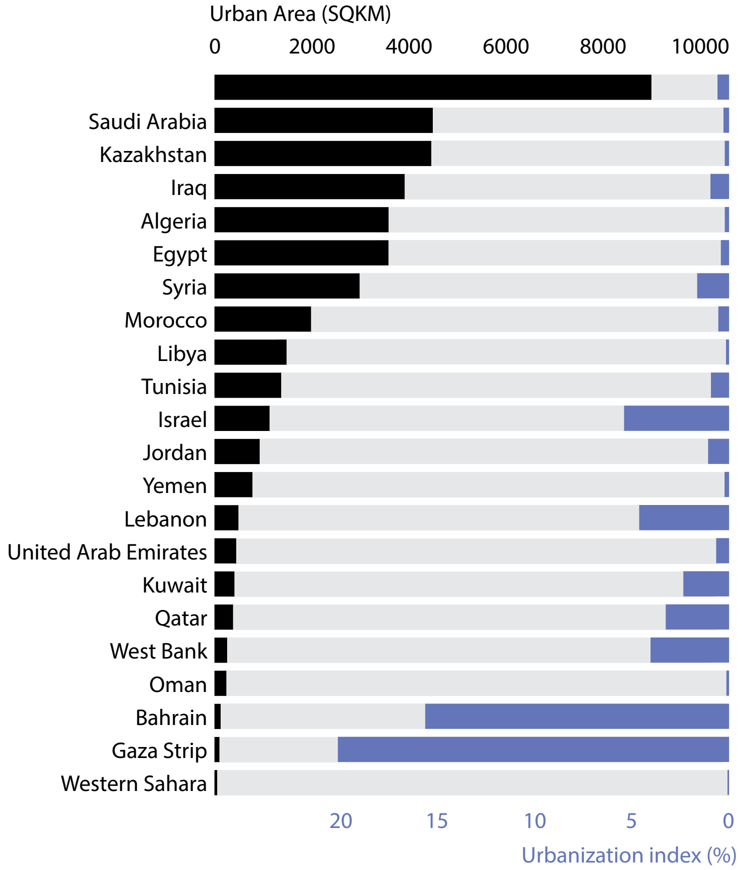


Figure 13.2 | Extent of the urban areas and Urbanization Indexes for the Near East and North African countries. Source: Schneider, Friedl and Potere, 2009.

13.4 | Major soil threats in the region

Most of the land area of the region falls in the hyper-arid, arid and semi-arid climatic zones. About 87 percent of the region is predominantly desert. Major soil threats are: erosion, salinity/sodicity, pollution, and soil C loss. Main causes of soil degradation are: mismanagement coupled with poor policies; use of inappropriate technology; increased levels of traffic movements and road construction; industrial activities and mining; urban expansion; deforestation, overgrazing and inappropriate cultivation practices; and dumping of hazardous wastes.

13.4.1 | Water and wind erosion

Wind and water erosion in the region are due to a complex interaction of factors related to the resilience of land resources, land use and management, and socio-economic conditions. In this section, a link is made to the prevailing land uses in the different countries of the region, and the consequences and responses are discussed.

Erosion caused by rainfed farming

FAO (2004) reported that the annual soil loss in Iran due to erosion is 1-2 billion tonnes yr^{-1}, and that 76 percent of the total area is under erosion threat. The area affected by wind erosion in Iran is about 12 percent of the total country surface area, six times the global rate of 1.8 percent. Other studies also report soil erosion as a serious problem in Iran, up to 20–30-times acceptable levels (Jalalian, Ghahsareh and Karimzadeh, 1996). In the north-west of Yazd in the Yazd-Ardakan plain, total soil mass transported was measured at 220.93 kg m^{-1} yr^{-1} and soil loss at 1.356 kg m^{-2} or about 13.56 tonnes ha^{-1} yr^{-1} causing topsoil C reduction at an average rate of 4 percent $month^{-1}$ (Azimzadeh *et al.*, 2008). The Iranian National Action Programme (NAP) reported in 2004 a lack of access for farmers to inputs as an additional cause of soil erosion. This is due to various driving forces like poverty, lack of security and awareness, inadequate extension, absence of technical knowledge and financing.

Elsewhere, driving forces for erosion processes were reported to be interrelated and to result in different degrees of degradation. In the El Bayadh region of Algeria, soils on limestone covers were classified as moderately vulnerable, vulnerable and highly vulnerable to degradation as a function of their vegetation cover (Belaroui, Djediai and Megdad, 2014). The study describes how the Algerian steppe has in recent years become the scene of an ecological and climatic imbalance.

More generally in the southern part of the Mediterranean region, overgrazing and cultivation of vulnerable land in arid and desert regions have induced severe wind erosion. In Morocco, erosion is a serious agro-environmental threat and was found to cause soil losses generally between 12 tonnes ha^{-1} yr^{-1} and 14 tonnes ha^{-1} yr^{-1}. In some areas of the Rif Mountains these values reach 30 to 70 $tonnes^{-1}$ ha^{-1} yr^{-1} (Benmansour *et al.*, 2013). Using radioisotopes (137Cs), these authors found that the tillage process on sloping lands over the last half-century had resulted in significant translocation of soils within the field. A study by Dahan *et al.* (2012) found that soil erosion in Morocco is a result of several factors. The most important factors have been: increased population pressure on limited natural resources; over exploitation of forestry assets; removal of natural vegetation from sloping lands; overgrazing; cultivation of vulnerable lands in arid and desert regions; and inappropriate land management, mainly tillage practices. They also reported that water erosion accelerated by human intervention is the main cause in Morocco of soil degradation and of the deterioration of water quality that it entails. Soil erosion in Morocco affects up to 40 percent of its territory with the total annual soil loss evaluated at 100 million tonnes, equivalent to 50 million m^3 annual reduction in dam storage capacity.

In El Bayadh region of Algeria, the Sirocco (a hot, dry wind blowing northwards from the Sahara) with a speed of 1.1 to 2.9 m s^{-1} is a dust-blowing wind which causes significant wind erosion (Belaroui, Djediai and Megdad, 2014). A recent study (Houyou *et al.*, 2014) showed that 20 million ha of steppe lands faced a high risk of wind erosion due to low rainfall and poorly rooted vegetation on sandy soils. This area is threatened by yet more intensive erosion because of newly adopted government policies favouring extensive rainfed cereal cropping. The study showed that this decision has led to very high erosion rates (74.4 Mg ha^{-1} yr^{-1}). Clearly this cropping system is unsustainable and the policy should be revised.

In Tunisia, overall soil loss due to water erosion has been estimated to be equivalent to 23 000 ha yr^{-1} in the isohyets above 200 mm. In some areas of Syria, soil loss due to water erosion has been estimated to range from 10 to 60 kg ha^{-1} (under forest), from 200 to 2 550 kg ha^{-1} (under burned forest) and as high as 960 to 3 280 kg ha^{-1} (under agricultural land). High population growth 1984-1999 (4.0 percent annually against a national average of 3.0 percent) in the village of Im Mial in the north-west of Syria led farmers to practice continuous rainfed barley production with no fallow or rotation (Nielsen and Zöbisch, 2001). This resulted in a very sparse vegetation and extensive wind erosion.

Grazing and tillage practices generally contribute to vulnerability to water erosion. In one study in NENA, the highest soil loss was recorded in over-grazed mountains, which lost up to five times more soil than slopes under managed grazing (Shinjo *et al.*, 2000). However, soil loss on tilled slopes can be one to four times that from grazed areas. Estimates of areas under serious water erosion in northern Iraq showed an increase from 12 percent in 1954 (Gibbs, 1954) to around 22 percent in 1997 (Hussein *et al.*, 1998), suggesting a rate of increase of 0.2 percent yr^{-1}. The main reason was mismanagement of cropland and rangeland during the intervening four decades. In Yemen the surface runoff to the sea measured in some major wadis is estimated at 1430 million m^{-3} yr^{-1} (Al-Hemiary, 1999).

In Jordan, water and wind erosion are both problems. Water erosion in the more vulnerable areas of the country with annual rainfall more than 400 mm and a slope greater than 25 percent – occurs on just 2.5 percent of the total area of the country. However, water erosion also occurs in the Badia region with its very low rainfall. The Badia soils are subject to water erosion because they are bare and highly exposed to what rainfall there is. The consequent formation of a slowly permeable seal and crust has enhanced runoff and water erosion (Rawajfih, Khersat and Buck, 2005). In Palestine it has been demonstrated that soil conservation pays – net profit was found to 3.5 to 6 times higher than without conservation measures. However, farmers' willingness to adopt conservation measures was influenced by other factors too, including knowledge and perception, land tenure, and the type of landscape (Abu Hammad and Børresen, 2006).

Early studies estimated the risk of water erosion in Lebanon to be 50–70 tonnes ha^{-1} yr^{-1}, but subsequently this has increased to 150 tonnes ha^{-1} yr^{-1} (Bou Kheir, Cerdan and Abdallah, 2006) and may be as high as 317 tonnes ha^{-1} yr^{-1}. Some farmers in Lebanon are nonetheless reluctant to change their cultivation practices, particularly ploughing up and down slopes of more than 20 percent. Farmers are also expanding cultivation on steep slopes, even though they may be aware that this aggravates water erosion. They justify up and down ploughing as necessary for tractor performance, and steep lands are the only available land for extending cultivation (Zurayk *et al.*, 2001). In general, human activities have been identified as causes of water erosion in Lebanon including: encroachment into agricultural land, cultivation of fragile soils, over-grazing, deforestation and overexploitation of woodland resources, uncontrolled use of fire for agricultural and forest clearing, unsustainable agricultural practices, poor irrigation practices and inefficient water use, and chaotic urban sprawl into fertile lands and forests (NAP, 2003).

In Sudan, studies showed that in areas affected by water erosion, about 74 percent of respondent households are exposed to food shortages that are sometimes severe (Akuot Gareng Apiu Anyar, 2006). The expansion of mechanized rainfed agriculture (1970–1990) at the expense of rangelands and forests led to land degradation by enhancing soil erosion. Subsequently crop yields declined sharply due to the decreased fertility. A principal driver has been the rapid growth of the population (2.6 percent per annum^{-1}) which increased demand for agricultural land. Mechanized rainfed agriculture expanded from 500 ha in the 1940s to 2.3 million ha in 2003. In the coastal part of Sudan, although total annual rainfall is low (75 mm), the sandy texture of the soils makes them very highly erodible (Elagib, 2011). With the observed increasing seasonality and intensity of rainfall, high runoff and erodibility in these areas could be expected to cause heavy soil degradation through soil loss.

Studies on factors contributing to wind erosion in Sudan showed that in rainfed agricultural zones, deep ploughing and leveling of the surface soil caused an increase in its susceptibility to wind erosion, which, in turn, has led to a severe decline in its fertility and, in some places, to the formation of sand dunes. The fragility index (degraded land in ha divided by population) is a good measure of the extent of growing pressure in fragile ecosystems. In Sudan, this value is very high in the hyper arid zone (31.1 percent), the arid zone (30.5 percent), and the semi-arid zone (22.5 percent). The value is low in the dry sub-humid zone (7.9 percent) and in the moist sub-humid zone (8 percent) (Ayoub, 1998).

Erosion caused by other land uses

In Lebanon, increasing demand for construction materials has led to extensive unregulated mining activities, including a large number of open quarries. Darwish *et al.* (2011) reported that, during the period from 1989 and 2005, quarries increased by 63 percent, covering an area of 5 267 ha. Many quarries were established on sloping lands (62.2 percent), triggering acceleration of water erosion processes. The high population density in countries like Kuwait (120 person km^{-2}) has a profound influence on soil disturbance through uncontrolled human activities. The Nabkha (stabilized aeolian landform developed as result of the deposition of wind-driven sediments around desert shrubs) along the coastal plain in Kuwait is used as a land degradation indicator (Khalaf and Al-Awadhi, 2012). The average annual sand drift rate in Kuwait is about 20 m^3 (m width)$^{-1}$ yr^{-1} and negatively affects farms causing adverse environmental and economic impacts (Khalafa and Al-Jjimi, 1993). Al-Awadhi and Cermak (1998) calculated the average sand movement in Kuwait as 7.8×104 kg (m width)$^{-1}$ yr^{-1}. In the Jalal-Alzor (Kuwait), human activities such as the unregulated use of off-road vehicles has resulted in soil disturbance and accelerated soil erosion. There has been a consequent increase in the rate of deflated sand from 220.5 kg m^{-1} width in 1989 to 400 kg m^{-1} in 2007, a total rate of increase of 81 percent in two decades (cited by Al-Awadhi, 2013).

Grazing was found to enhance soil loss by water. In the Matash Mountains of the Alborz Mountain range in Talesh Region, Iran (slope of 16 percent and 1 286 mm of precipitation per annum), soil loss due to open grazing was more than 26 times that in rainfed agriculture (Sadeghi *et al.*, 2007). In Sudan the dominant type of housing is the traditional hut made from forest products. These buildings need to be renewed every two years on average. A study found that this practice exacerbates the process of soil erosion. Conserving and restoring the vegetation cover in these areas was achieved through adoption of mud huts. As a result, the stocking density of trees around the villages went up compared to the control (FAO, 2013b).

Deforestation is also a factor inducing soil erosion. Fuel wood or charcoal production for domestic use is one element in this deforestation. Deforestation for agriculture in semi-arid lands around settlement areas is also a cause of soil degradation, for instance in the Jifara Plain of Libya.

Dust storms

Dust storms are frequent in the region and widely reported. A dust storm carries toxic elements like Pb with concentrations as low as 20 to 288 mg kg–1 in Oman to higher levels of 742 mg kg–1 in Bahrain and 1762 mg kg–1 in Saudi Arabia (Madany, Akhter and Al-Jowder, 1994; Al-Rajhi, Seaward and Al-Aamer, 1996). The problem of sand drifting and dune migration is of special concern in some countries of the region such as Saudi Arabia, where approximately one-third of the country is covered by moving sand. Al-Harthi (2002) found that these storms have resulted in dune movement of 9.9 m yr^{-1} (for 4.9 m-high dune) up to 16.5 m yr^{-1} (for 1.9-m high dunes). This problem is exacerbated by human activities such as overgrazing or other activities that may destroy the desert pavement which protects the loose sand underneath.

In Kuwait, the sand drift potential was found to be as high as 354 vector units (Al-Awadhi, Al-Helal and Al-Enezi, 2005). Sandstorms are very frequent in summer especially when the wind speed exceeds 6 m s^{-1}. An annual amount of sand drift can measure 7.8×104 kg m^{-1} width (Al-Awadhi and Misak, 2000).

As cited by Goudie and Middleton (2001), estimates of rates of dust deposition exist for a number of sites at varying distances from the Sahara. The dust originates from southern Algeria, the Nubian Desert in southern Egypt and Northern Sudan. Volumes carried to Western Europe are less than 1 g m^{-2}. Up to 5.1 g m^{-2} may reach Spain, while over Sardinia, Corsica, Crete and the south-east Mediterranean, most values are between 10 and 40 g m^{-2}. Long-range transport of Saharan dust to the central Mediterranean is characterized by events lasting two to four days, compared to an average duration of just one day for events reaching the Eastern Mediterranean from the Arabian Desert (Dayan *et al.*, 1991).

Several studies reported the frequency of dust storms in the Arabian Peninsula and found that the average quantity of dust falling on Kuwait and Riyadh were 191 and 392 tonnes km^{-2} $year^{-1}$ (cited by Ibrahim and El-Gaely, 2012). A recent study (Jish Prakash *et al.*, 2014) on the impact of dust storms on the Arabian Peninsula and the Red Sea reported that strong winds (velocities exceeding 15 m s^{-1}) entrained large quantities of dust particles into the atmosphere with sources including the lower Tigris and Euphrates in Iraq, areas of Kuwait, Iran and the United Arab Emirates, and the basin of the Arabian desert (which includes the Rub' al Khali, An Nafud and Ad Dahna). The study also reported that the frequent dust outbreaks and dust storms each year in the NENA region have profound effects on all aspects of human activity and natural processes. The total amount of dust generated by the storms is estimated at 93.76 million tonnes, of which 80 percent is deposited within the area, around 6 percent (5.3 million tonnes) is deposited in the Arabian Sea, the Gulf received 15 percent (1.2 million tonnes), and the Red Sea roughly 6 million tonnes. In the Middle East, more than 60 dust storms occurred during the period 2003–2011 with significant impact on the countries of the region (Hamidi, Kavianpourl and Shao, 2013). Some countries are worst affected. Iraq, for example, experiences on average about 122 dust storms and 283 dusty days each year. Some experts expect this may increase to 300 dusty days and dust storms a year within the next ten years (Kobler, 2013).

Consequences of soil erosion

Erosion processes remove the fertile part of the soils and thus reduce the effective depth to be exploited by roots and the amount of water available to plants. This is considered a major constraint limiting productivity in Morocco (Dahan *et al.*, 2012). In the Maghreb region (Morocco, Algeria, Tunisia), uncontrolled runoff from terraces has reached the stage of gully formation. More generally, sheet and gully erosion has become common due to increased population, deforestation, overgrazing, and expansion of cultivation on steep land (Dregne, 2002). There are few studies on the effects of erosion on land productivity in the region. However, one research study on barley in Aridisols of Egypt (Afifi *et al.*, 1992; Wassif, Atta and Tadros, 1995) found a declining yield of 2 kg ha^{-1} Mg^{-1} of soil erosion which is equivalent to 0.21 percent Mg^{-1} soil erosion. A study in Iran found soil erodibility by water was negatively correlated with wheat yields (Vaezi, 2012). The agricultural productivity of oases in countries like Sudan and Egypt is threatened by the adverse impacts of sand encroachment and mobile dunes.

Responses to soil erosion

Ways to contain erosion are very diverse and location-specific. In the sandy depression of El-Farafra, Egypt, which suffers from wind erosion, Sallam, Elwan and Rabi, (1995) reported that mixing sandy soils with grey shale at a ratio of 15 percent (w/w) improved the quality of these soils. Organic manures, compost and synthetic soil conditioners have been used to contain wind erosion. Compost alone or compost combined with hydrogel conditioners was found to decrease erosion by 58 to 74 percent (El-Hady and Abo-Sedera, 2006). One study found that water erosion could be stemmed in Morocco by increasing levels of C in the topsoil using conservation measures (Mrabet *et al.*, 2001). Irrespective of rotation, conservation measures were found to increase topsoil C by 44 percent. The study concluded that systems with increased C are generally characterized by diminished erosion.

Government policy responses can play a vital role. For example, public policy in Egypt and Iraq has been determinant in increasing green cover in desert soils in those countries (Nielsen and Adriansen, 2005). The Iraqi case is a negative one, illustrating the effects of deliberate government policies in draining the marshlands, which resulted in significant land degradation. Recent efforts have been devoted to the re-establishment of these marshes. They are being re-flooded and vegetation is returning. The Egyptian government promoted land reclamation after the 1952 revolution aiming at increasing agricultural production. This reclamation was executed through internationally funded developmental projects and with local funding and involved distributing small areas of lands to graduates.

Efforts at containing water erosion in rainfed farming in Yemen depend on the establishment and maintenance of terraces as conservation structures, a highly labour intensive task. However, labour shortages and lack of profitability have been constraints, and the degradation of these structures has continued. In Jordan, the use of polyacrylamide (PAM) at application rates of 10 to 30 kg ha^{-1} was found to be very effective in reducing runoff and soil loss by up to 23 percent and 53.9 percent, respectively. As a result, dry matter crop yield went up by 35 to 56 percent (Abu-Zreig, Al-Sharif and Amayreh, 2007). Interestingly, this study developed an increased threshold runoff value of 0.56 mm rainfall in control sites to 1.11 mm in PAM plots (e.g. close to 100 percent). Also Abu-Zreig (2006) pointed out that PAM with more surface area (30 percent) reduced soil loss by approximately 46 percent compared to the 24 percent reduction with less surface area (20 percent).

Soil conservation to mitigate erosion has also been done by planting trees and grass along wadis. Construction of diversion banks and dams across watercourses has been carried out on slopes and stream beds in Libya. One study (Mohammed, Stigter and Adam, 1996) found that windbreaks reduced wind speed by about 20 percent and in turn limited sand deposition. However, sand deposition also occurred within the belt which after some time started to act as a zero permeability wind break. This study suggested that control of sands in the source area using shelterbelts may not be sufficient in the long term.

Some countries have introduced policies to reverse land degradation due to erosion. In the mechanized rainfed projects of Sudan, farmers are required to leave 10 percent of the cultivated area for forest trees. To protect soil against erosion after fire in Lebanon, a ministerial decision (181/98) imposed a five year ban on grazing on public land after fires to enhance land cover recovery.

13.4.2 | Soil salinization/sodification

Distribution of salt-affected soils in the region varies geographically with climate, agricultural activities, irrigation methods and policies related to land management. These soils are mainly confined to irrigated farming systems in the arid and semi-arid zones. The salts present are either of intrinsic origin (typical of Egypt, Sudan and Iran) or are the result of sea water intrusion in coastal regions or of irrigation with brackish or saline groundwater. In the irrigated zones of Morocco, continuous irrigation has resulted in soil salinization. Secondary salinization due to irrigation with saline water is also reported in the NENA region. In Libya, Sudan, Iran, Iraq and United Arab Emirates, large tracts of lands have been degraded due to heavy irrigation with groundwater. Salinity, sodicity or the combination of both are seriously affecting productive areas like the Nile Delta of Egypt and the Euphrates Valley in Iraq and Syria. The situation is further complicated by association with problems of waterlogging and high $CaCO_3$ (up to 90 percent in the United Arab Emirates, Al Barshamgi, 1997). In Kuwait and the United Arab Emirates, soil salinization is mainly confined to coastal areas, but also occurs on irrigated farms.

Local soil conditions can worsen the situation. For instance in the southern part of the Jordan Valley, the soil is characterized by high salt content, poor permeability and high gypsum content. The degradation is worse when low quality irrigation water, for example treated waste water, replaces fresh water. In Libya, El-Tantawi (2005) reported that the soils of the Jifara Plain are usually calcareous and often shallow, with huge areas of calcrete outcrops developed during the Pleistocene epoch. Gypsum encrustation is commonplace in the drier parts where annual precipitation is below 200 mm.

Salinity problems in the region also stem from inadequate irrigation water management (Al-Hiba, 1997). In almost all countries of the region with coastlines, heavy extraction of groundwater has led to intrusion of seawater into aquifers, thereby raising the content of salts in the water. An example is the Batinah aquifer of the Sultanate of Oman where seawater is intruding at an alarming pace (Naifer, Al-Rawahy and Zekri, 2011). The cause was the expansion of agriculture since the 1980s which accelerated the overuse of groundwater, disturbing the water balance and ultimately leading to water intrusion from the sea. In the Jifara Plain of Libya, increased human pressure on aquifers has induced seawater intrusion in the coastal zones and a combination of over-irrigation and inefficient drainage causes waterlogging and secondary salinization.

The main causes of build-up in salinity in the region are: (1) improper functioning or absence of drainage systems; (2) a rise in groundwater salinity combined with high rates of evapotranspiration; and (3) high salinity in irrigation water. Siadat, Bybordi and Malakouti (1997) recognized natural factors and humaninduced factors that cause salinity in Iran. The natural causes of soil salinity in Iran are geological conditions, climatic factors (evaporation, rainfall and wind), salt transport by water, and intrusion of saline bodies of water into the coastal aquifers. However, of greater concern and importance is humaninduced salinity. This type of salinity can stem from a number of causes, including: poor water management, over-grazing, improper land levelling, and overuse of groundwater leading to saline water intrusion.

Secondary salinization due to irrigation with saline water is also reported in the region. In the Tadla irrigated perimeter in Morocco, soil degradation through secondary soil salinity and sodicity are caused by heavy irrigation with ground- and surface water together with agricultural intensification. In Algeria, secondary salinization affects 10 to 15 percent of the total irrigated land. About 90 percent of the agricultural farms in Al Ain in the United Arab Emirates are affected by salinity (Abdelfattah, Shahid and Othman, 2009). It is clear that in some countries of the region, secondary salinization due to irrigation does not develop. One example is in the Vertisols of Sudan, despite more than 80 years of irrigation. This is mainly due to the low salt content of Nile water (EC of 120 to 220 $\mu S\ cm^{-1}$).

Consequences of salinization

Salinity in the region has badly affected cropping systems and in many cases has significantly reduced crop yields. For example, soil salinity in the Jifara plain in Libya has caused wheat yields to decrease from 5 tonnes ha^{-1} in the 1980s to just 0.5 tonnes ha^{-1} by 1987. In Iran the annual economic losses due to salinity are estimated at more than US$ 1 billion (Qadir, Qureshi and Cheraghi, 2008). The coastal area is one of the most highly populated regions of Oman, especially the Batinah area where about 52 percent of land is under cultivation and suffers from salinity. Naifer, Al-Rawahy and Zekri, (2011) showed that when salinity increases from low (less than 2.5 dS m^{-1}) to medium (7.5 dS m^{-1}) and to high (more than 7.5 dS m^{-1}) levels, losses are equivalent to US$ 1 604 ha^{-1} and US$ 2 748 ha^{-1}, respectively. Soil salinity, sodicity and waterlogging conditions have definite adverse impacts on soil productivity, in the range of 30-35 percent of potential productivity.

Responses to salinization

There are many responses in the region to contain the salinity threat such as: (1) direct leaching of salts, (2) planting salt tolerant varieties, (3) domestication of native wild halophytes for use in agro-pastoral systems, (4) phytoremediation or bioremediation, (5) chemical amelioration, and (6) the use of organic amendments. In Iraq and Egypt, surface and subsurface drainage systems have been installed to control rising water tables and arrest soil salinity. In Iran, Syria and other Gulf countries, crop-based management and fertilizers are used to combat salinization (Qadir, Qureshi and Cheraghi, 2007). In Iran, Haloxylon aphyllum, Haloxylon persicum, Petropyrum euphratica and Tamarix aphylla are potential species for saline environments (Djavanshir, Dasmalchi and Emärarty, 1996). Atriplex is a fodder shrub adapted to arid lands which can bring annual income as high as US$200 ha^{-1} (Koocheki, 2000; Tork Nejad and Koocheki, 2000). Breeding salt tolerant varieties of crops (e.g. wheat, barley, alfalfa, sorghum) is also a response to saline environments, although most results so far are based on controlled environments rather than on actual yields from the field.

The use of organic amendments in Egypt showed that the mixed application of farmyard manure and gypsum (1:1) significantly reduces soil salinity and sodicity (Abd Elrahman *et al.*, 2012). Recently, phytoremediation or plant-based reclamation has been introduced in the region. In Sudan there are very good responses for control of sodicity relying on phytoremediation, superior to results from the gypsum amendment traditionally used. The production of H^+ proton in the rhizosphere during N-fixation from some legumes like hyacinth bean (Dolichos lablab L.) removed as much Na^+ as did gypsum application which indicates its importance in calcite dissolution of calcareous salt affected soils (Mubarak and Nortcliff, 2010).

13.4.3 | Soil organic carbon change

Information on soil carbon changes in the region is scarce. Estimates of soil C sequestration are basically confined to the work on the drylands ecosystems of West Asia-North Africa (WANA) carried out by Lal (2002). These data are nonetheless very useful and can be applied across NENA. Lal's study indicated that the total loss of soil-C from the WANA region could be about 6 to 12 Pg. Despite the low soil C levels in the region (generally less than 5 g kg^{-1}), with effective control measures of degraded soils, the region could sequester C at the rate of 0.1 to 0.2 Mg ha^{-1} yr^{-1} (for irrigated crop land) and 0.05 to 0.1 Mg ha^{-1} yr^{-1} for both rainfed and rangeland. In other words, with desertification control, reclamation of salt-affected soils, and intensification of agriculture on undegraded soils, the soils of the region have the potential to sequester 24 to 31 percent (168-380 Tg yr^{-1}) of the total global drylands soil C (710-1220 Tg yr^{-1}). The potential annual sequestration rate could reach values between 0.2 and 0.4 Pg C yr^{-1}, compared to the 1.0 C yr^{-1} in total global drylands (e.g. 20 to 40 percent).

SOC change in rainfed farming systems

The Century model was used by Poussart, Ardö and Olsson, (2004) in a study of the Arenosols of Kordofan (Sudan) to estimate soil C levels and changes with reference to values prior to known human interaction. Changes were due to pastoral activities combined with cultivation of rainfed crops like Pennisetum typhoideum), sesame (Sesamum indicum), sorghum (Sorghum vulgare), and groundnuts (Arachis hypogaea). The base scenario modelled indicates that the land management practices continued for more than a century have led to a loss of C of about 180 g m^{-2} (1.8 tonnes C ha^{-1}) which is equal to 1.6 gm^{-2} yr^{-1} or approximately equivalent to half of the historical level of C in the year 1890. Additionally, Ardö and Olsson (2003) modelled C changes in north Kordofan in the top 20 cm during the period 1800–2100, with mostly Arenosol and Vertisols soil types. They found that C estimates in cropped land have dropped from 16.64 million tonnes in the year 1800 to 9.16 million tonnes (e.g. a 45 percent reduction), whereas C in other land uses such as shrublands, savannah, grassland, or barren or sparsely vegetated land remained almost constant. Another study in north Kordofan found that rapid population growth has caused huge soil C loss (73 percent at rate of 16.9 g C m^{-2} yr^{-1}) in the top 0-20 cm from 851 g C m^{-2} in 1963 to 227 g C m^{-2} in 2000 (Ardö and Olsson, 2004).

The effects of cultivation of heavy textured soils on C change in some countries of the Mashreq region (Jordan, Syria, Lebanon, Iraq, Palestine) have been found to be broadly similar to results from the Maghreb (Morocco, Tunisia, Algeria, Libya). Masri and Ryan (2006), in a study on Syria, compared the effects of more than twenty years of wheat cultivation in rotation with lentil (Lens culinaris), chickpea (Cicer arietinum), medic (Medicago sativa), vetch (Vicia sativa) and watermelon (Citrullus vulgaris) with continuous wheat grown in montmorillonitic thermic Chromic Calcixert. The trial showed that, apart from the rotation of wheat with medic which has higher C (8.0 g kg^{-1}), C content in continuous wheat cultivation (6.3 g kg^{-1}) or in other rotations (6.6-7.0 g kg^{-1}) differs little from fallow (6.6 g kg^{-1}). This suggests that conventional farming systems in the Vertisols of the region may not be depleting soil C.

In the western part of Jordan, the change in C is a function not only of a land use system but also of the human impact in a specific land use. For example, cultivating tobacco and clearing residues, and growing irrigated cereals retain almost similar amounts of C (6-7 g kg^{-1}), whereas ploughing rainfed cereals and grazing with compaction consequences retain C contents of only about 1.1 g kg^{-1} (Khresat *et al.*, 1998). In the Lorestan Province of Iran, management practices on rangeland in areas with slopes of 10 to 26 percent caused pronounced reduction in C, sometimes twice levels found in dryland farming. When wheat was subsequently grown, wheat dry matter, grain yield and grain weight all reduced relative to the control dry farming (by 14, 33 and 21 percent respectively), which indicates the degrading effects of using these slopes as rangeland (Asadi *et al.*, 2012).

SOC changes in irrigated farming systems

Irrigated soils in the southern Mediterranean region can be considered as incubators providing optimal conditions (humidity and temperature) for microbial activity and thus a rapid degradation of organic carbon. A mean annual variation rate of organic matter of -0.09 percent yr^{-1} over a decade was established in the Doukkala region of Morocco (Badraoui, 1998). This decrease of organic matter is attributed to the non-incorporation of crop residues into the soils. Crop residues contribute about 30 percent of the total forage consumption in Morocco. Countries like Morocco experience rapid organic matter decomposition and turnover and hence very low levels of organic matter are converted into humus. The reasons for this rapid decomposition include: the high temperature, widespread of use of tillage, clean fallow, overgrazing, no practice of residue recycling, and climatic conditions.

Rates of C change are influenced not only by land use but also by soil type. Clay soils tend to counteract decomposition and hence reduce chances of C loss. For example, in clay soils of Morocco, Bessam and Marbet (2003) reported that in seven years of continuous tilling, soil SOC reduced in the 0-20 cm profile by 17 percent (e.g. 0.33 g kg^{-1} yr^{-1}). As cultivation of such soils advances, it was possible over time (an eleven year period was monitored) to store more C (by 3.7 percent) in the entire 0-20 cm depth but not the top 2.5 cm due to incorporation. The decrease in topsoil C noted increases vulnerability to degradation by erosion due to reduction in topsoil buffering capacity or resilience.

It is apparent that continuous cultivation does not always consume soil C. There are exceptions, however, for example the case of the irrigated Vertisols of the Gezira Scheme of Sudan which have been continuously cropped for more than 80 years. The carbon content in a furrow slice (0-10 and 10-35cm) of permanent fallow (4 and 4.61 g kg^{-1}) was lower than that in cultivated plots (6.35 and 5.44 g kg^{-1}) by about 37 and 15 percent respectively which indicates that C loss due to cultivation in such soils is not a small problem (Elias and Alaily, 2002). Physical protection of C in heavy textured soils under long-term cultivation (about 30 years) and with intensive tillage decreased C in the top 0-15 cm by 15 to 24 percent relative to wood and fallow lands (Biro *et al.*, 2013).

In desert soils carbon could likely be increased by irrigation of alfalafa rotated with wheat or wheat rotated with fallow. One study showed this to have increased C in the top 0-10, 10-20 and 20-30 cm depths by factors of 5.1-6.8, 3.0-4.6 and 6.8-11.6 respectively (Fallahzadeh and Hajabbasi, 2012a). This rotation could thus stabilize soil aggregates and consequently reduce the vulnerability of desert soils to erosion by either wind or water. However, C could also be decreased by 24 to 47 percent during bio-remediation (land farming processes) of soils contaminated with hydrocarbon of petroleum origin where microbial activities are greatly enhanced due to aeration (Besalatpour *et al.*, 2011).

SOC change and forest clearing

Tree plantations have been found to be the best system to conserve C in the region. Other land use systems practiced on light soils in the region seem to result in C loss. For example, in the sandy soils of western Sudan, taking tree C in the 0-30 cm depth as a reference, three years of sole cropping or cropping mixed with trees caused about 41 to 47 percent C reduction (El Tahir *et al.*, 2008).

In Jordan, land resources have been affected by rapid land use change, accelerated by socio-economic factors including high population growth, urbanization and agricultural intensification. Farmers converted a forest located in Ajloun to cultivation of wheat (Triticum spp.) and barley (Hordeum spp). After more than 50 years of cultivation, Khresat *et al*. (2008) reported that soil C in the top 0-20 cm of mostly Inceptisols, Mollisols and Vertisols had decreased from 6.73 g kg^{-1} to 4.70 g kg^{-1} (e.g. a 30 percent reduction). In Iran, the conversion of forest or pasture to arable lands in five cultivation sites where tillage has now been practiced for 40-50 years was found to have caused 30-68 percent loss of topsoil (0-20 cm) SOC, specifically from 3.86-9.84 to 3.25-8.06 kg m^{-2} (Golchin and Asgari, 2008). Farming practices had also mixed SOC down the profile, leaving less SOC on the topsoil for surface protection against degradation by erosion and also reducing the capacity of the soil to retain nutrients.

Also in Iran, change from pasture to dryland farming on Inceptisols was found to have degraded soil and reduced SOC by about 67 percent. Recovery times can be very long indeed. For example, a study found that 30 to 60 years will be required to restore soil C to the initial content of rangeland ecosystems of Chaharmahal and Bakhtiari Province in Central Iran. After more than a century of cultivation, SOC in the top 0-15 and 15-30 cm had reduced by 25-34 percent (Chigani, Khajeddin and Karimzadeh, 2012).

The change in SOC depends on soil type. For example, continuous cultivation of Cambisols in northern Iran tends to decrease C by 29 to 74 percent as compared to grass or forest use respectively, whereas the equivalent decline for Vertisols is only 11 to 59 percent.

Another study on soil quality indicators potentially sensitive to land degradation due to land use change was carried out in a rangeland pasture in Iran's Isfahan Province that had been protected for more than two decades (Nael, Khademi and Hajabbasi, 2004). The study showed that C content in the protected range was close to double (1.7 times) that of areas of uncontrolled grazing. This study clearly indicated that in the dryland of this region, decisions that control grazing are favourable for soil C storage.

In another study, four decades after conversion from oak forest (Quercus brantii Lindl.) to vegetable cultivation (tomato and snap bean), soils had lost almost 53 percent of C (Fallahzadeh and Hajabbasi, 2012b). A further study found that twenty years after oak forest (Quercus brantii) in the Lordegan region of Iran's Zagros mountains was converted into either wheat or barley cultivation or agroforestry uses, soil C in the 0-30 cm depth decreased by 52 and 61 percent, respectively (Hajabbasi *et al.*, 1997).

13.4.4 | Soil contamination

Land degradation due to accumulation of contaminants is concentrated in either oil producing countries or those that are heavily populated. In agricultural soils, contamination is generally restricted to irrigated farming systems. In some instances, the over-use of chemicals (fertilizers, pesticides and herbicides) has sharply increased the amount of chemical nutrients in the drainage water, causing water eutrophication (NAP, 2002). Since the construction of the Aswan High Dam of Egypt in 1970, fertilizers and pesticides have been heavily used in order to substitute for the loss of fertile sediments. FAO (2012) reported that during the period from 1950 to 1990, chemical fertilizer use increased more than fourfold, from 2 143 tonnes in the 1950s up to 11 700 tonnes in 1990. These chemical fertilizers and the residues of applied pesticides have caused the contamination of soil and water resources in the Nile Delta.

In Iran, contamination of soil is increasing from a variety of sources: petroleum hydrocarbons spilled during transportation, leakage from tanks, accidental spillage, pipeline ruptures, or dumping of oil landfill. This contamination threatens soil functions. It may decrease seed germination of grasses by more than 50 percent (Besalatpour *et al.*, 2008) – although this may not necessarily affect their subsequent performance. It also reduces dry matter accumulation in sunflower and safflower by 50 and 73 percent, respectively (Besalatpour *et al.*, 2008). The two Gulf wars in 1990 and 1991 contributed to contamination of this kind through the detonation of oil wells (Al-Senafy *et al.*, 1997; Misak, Khalaf and Omar, 2009).

Case study: Kuwait experience in remediation of oil contamination

After the Iraq war, over 300 oil lakes covering an area of 46 km^2 were formed within Kuwait. The lakes were up to two meters deep, and the oil penetrated the soil to varying depths. To restore areas degraded by oil, the Kuwait Institute for Scientific Research and the Japan Petroleum Energy Centre began in 1994 to devise biological technologies for remediation and rehabilitation (Figure 13.3).

In a small scale pilot (1 920 m_2) and in field demonstrations, heavily oil contaminated soil was remediated over a 12-18 month period, using bioremediation techniques involving enhanced land farming techniques, windrow composting piles, and static bioventing piles. The programme resulted in 80-90.5 percent reduction in the total petroleum hydrocarbons and total alkanes (Al Awadhi, 1996; Balba *et al.*, 1998). This technology is considered to be economical, energy efficient, and environmentally friendly with minimal residue disposal problems. However, the volatilization of airborne volatile organic compounds in the atmosphere during the process of degradation may lead to serious human health risk (Hejazi, Hussain and Khan, 2003).

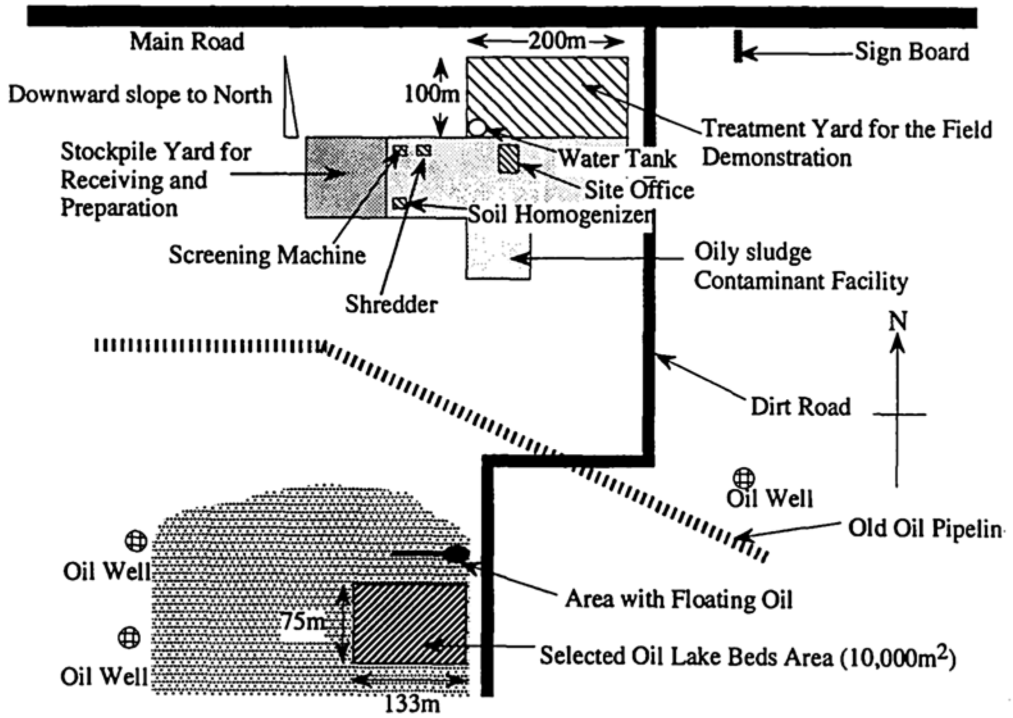


Fig. 1. Layout of the project site.

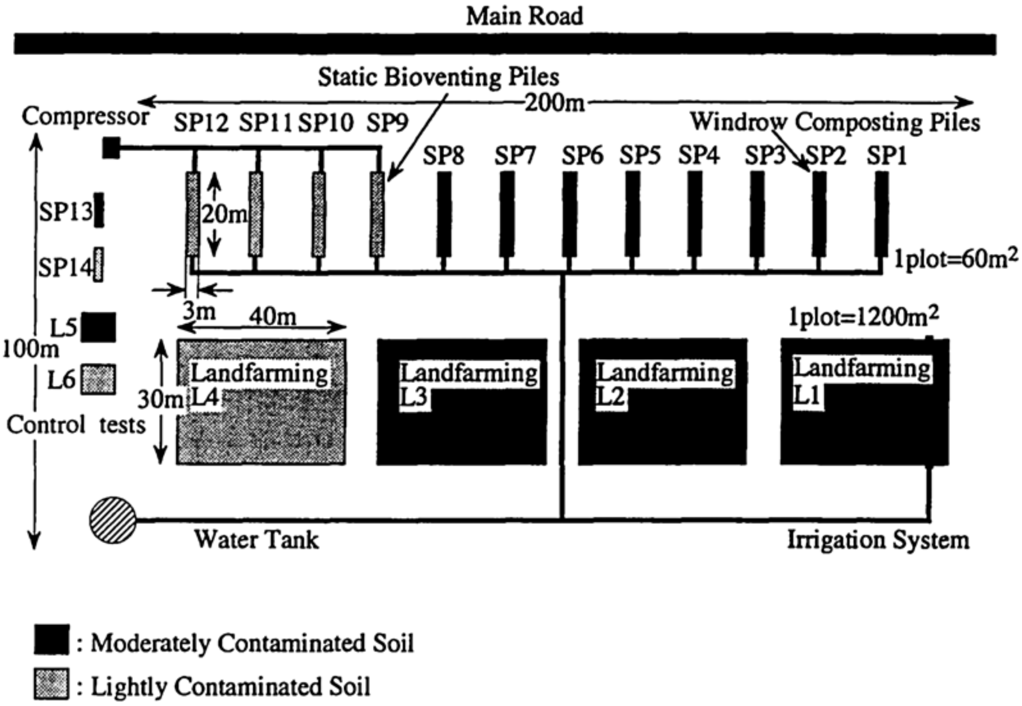


Fig. 2. Conceptual design and layout of bioremediation system.

Figure 13.3 | Layout of the project site source (a) and conceptual design and layout of bioremediation system (b).
Source: Balba et al., 1998.

Contamination of soil by heavy metals is also an issue (Misak, Khalaf and Omar, 2009). In central Iran, farmers are extensively using sewage sludge as a fertilizer for vegetable production and, in the absence of regulation, heavy metals tend to accumulate in the soil (Afyuni, Rezaeinejad and Schulin, 2006). The so-called 'global dust belt' that extends from the west coast of North Africa, through the Middle East into Central Asia (see Section 7.4) transports mineral dust in the region. This dust may carry contaminants and this has been in fact the main source of soil pollution with heavy metals in the Arabian Peninsula. The dust was found to carry high levels of lead (65 mg kg^{-1} in Muscat, 742 mg kg^{-1} in Bahrain and 1762 mg kg^{-1} in Riyadh, Saudi Arabia) and nickel (43 to 3033 mg kg–1 in Muscat) (Yaghi and Abdul-Wahab, 2004).

In general, soil contamination depends on the distribution of contaminants influenced by high intensity rainfall of short duration that results in short runoff, by dust storms, and by human induced factors such as mixing residual oil with soil, transport to new areas and dumping in selected sites.

Responses to soil contamination

Regional policies for combating desertification

Most NENA countries have policies and programmes for protection of natural resources, including combating desertification. However, many of these policies and programmes (e.g. in Jordan) emphasize protective measures, and do not adequately consider rehabilitation or the dimension of the economic and social cost of land degradation (Al Karadsheh, Akroush and Mazahreh, 2012).

Iran has implemented nine strategies for sustainable development. Egypt has strategies for each agro-ecological zone. Lebanon has initiated a large-scale reforestation program and is very active in fighting the root causes behind land degradation, mainly by promoting the development of rural areas and reducing regional disparities. The national efforts to combat desertification in Oman have concentrated on development and conservation of water resources, improvement of land capability and rehabilitation of rangeland. Saudi Arabia has programmed an array of activities including capacity building, controlling urbanization, sustainable agricultural development, improvement of water sector, legislation, rehabilitation of degraded rangelands, forest development and sand dune stabilization. Sudan is integrating strategies for poverty alleviation with programmes to combat desertification and these include activities for improvement of land resources, production systems and protection of the environment. Syria has implemented many projects aimed at expansion of plant cover, controlling desert invasion, establishment of protected areas and green oases, sand dune fixation and afforestation. United Arab Emirates has ambitious programmes to improve degraded ecological systems, conserve biodiversity, mitigate climate change effects, and combat desertification.

13.5 | Case studies

13.5.1 | Case study: Iran

Soil nutrient changes

Change in land use and the use of fertilizers are the main factors affecting soil nutrient change in Iran (Shiranpour, Bahrami and Shabanpour, 2011). A study in Gilan Province compared the status of forest soils and the same soils turned into tea gardens over a period of 10 to 40 years and showed a significant decline in the amounts of nitrogen, potassium, phosphorus, calcium and exchangeable magnesium. Deforestation effects on soil nutrient losses have been studied in Kajoor watershed in Sari city where results indicated significant losses of organic matter and phosphorus. Land uses and different management types were compared in the Taleghan area. The study showed the negative effect of irrigation and monoculture on soil nutrients, while horticulture and pasture land uses scored better (Sohrabi and Zehtabian, 2012).

Soil pollution changes

In Iran, surveys carried out in areas where soil pollution occurs indicate that heavy elements make up the majority of soil contaminants. These pollutants originate from a range of sources, including geological and mining sources, industrial pollution, petroleum spills, sewage sludge application and excessive usage of fertilizers on agricultural soils.

Losses and sequestration of soil carbon

Loss of organic matter in soils of Iran is among the most important consequences of soil erosion. Greening barren land and improving soil management can significantly increase soil carbon sequestration. Forest ecosystems in equilibrium, with both trees and other vegetation cover, are the principal reservoir of organic carbon. Varamesh *et al.* (2010) assessed the effects of reforestation in Tehran Cheetgar Park on carbon sequestration and soil characteristics. The study indicated that soil carbon sequestration of Acacia senega.is equal to 78 tonnes ha^{-1}, and of Conifer Species 57 tonnes ha^{-1}. In general, the carbon sequestration process led to improvement of soil and water quality, increased fertility and an improved soil hydrology system as well as preventing erosion and reducing nutrient loss.

Salinity changes

Yazdani-Nejad and Torabi-Golsefidi (2013) examined the spatial variations and salinity zoning of agricultural soil in an area of southern Tehran. About 30.4 percent of the area, covering 20 000 ha, was found to be without any salinity problem. These lands were located mainly in areas where irrigation water from deep wells was used. A further 42.4 percent of the land had an electrical conductivity of 2 to 4 dS m^{-1}. In these sections, irrigation was with water from deep wells but wastewater was also used for irrigation due to water use restrictions. Zones with conductivity of 4 to 8 dS m^{-1} occupied 22.6 percent of the area, located in the flat plains in the southern part. These zones were frequently irrigated with waste water and water from shallow wells, and also with low quality water from the downstream sections of the Kan River. Finally, 4.5 percent of the land, located in low lying areas, had high electrical conductivity of 8 to 13 dS m^{-1}. The overall finding of the study was that it was the position of the land in the landscape, the depth to the water table and the quality of the irrigation water that determined the degree of salinization of the land. Other factors that may play a role are the length of the interval between irrigations and the texture of the soils.

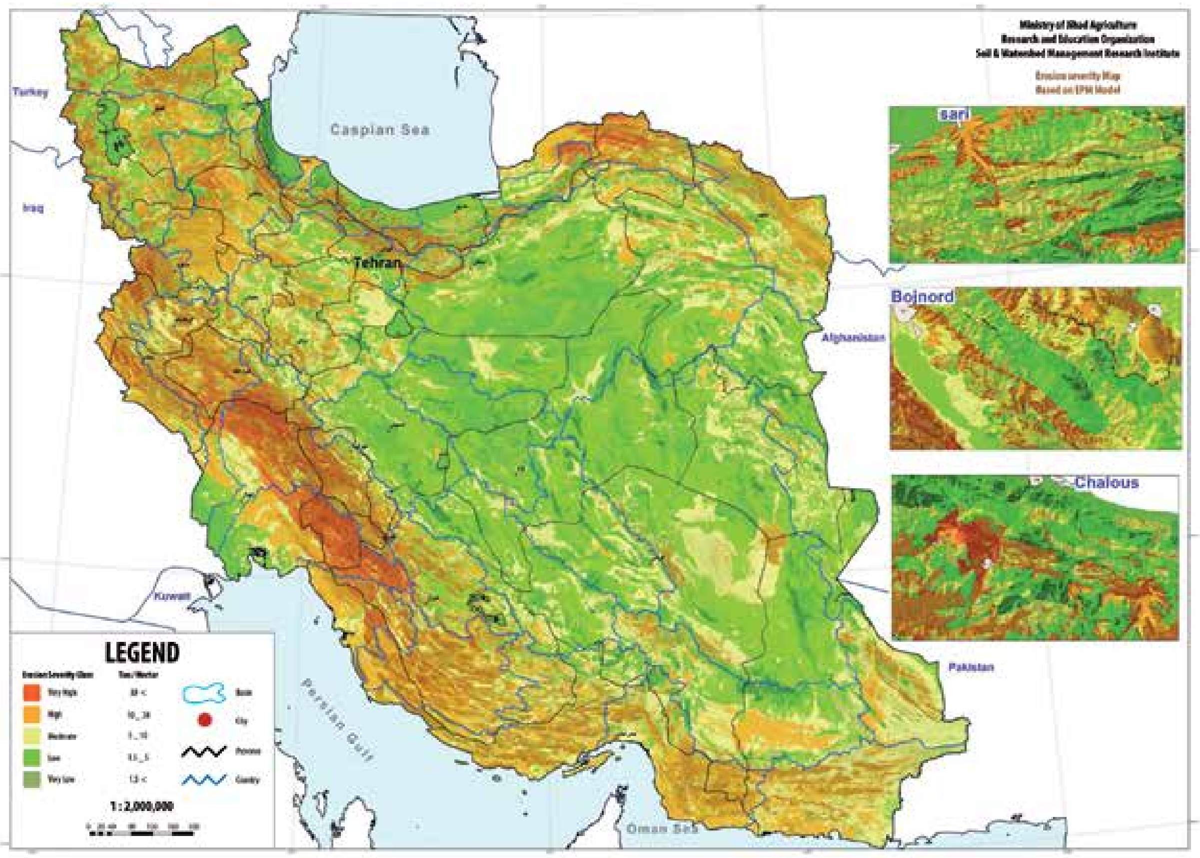


Figure 13.4 | Rate of water erosion in Iran.
Source: Soil Conservation and Watershed Management Research Institute.

Water erosion

In Iran about 40 percent of the country experiences a low erosion rate (less than 10 tonnes ha^{-1}), 25 percent of the area has a moderate rate of water erosion (10 to 20 tonnes ha^{-1}), 23 percent area of the country has a high erosion rate (20 to 50 tonnes ha^{-1}), and 12 percent has a very high erosion (more than 50 tonnes ha^{-1}). Figure 13.4 shows the erosion rates in different regions of Iran.

Results of recent research conducted in the watersheds of the country in recent years suggest an increased rate of erosion according to various models used (Hosseinkhani, 2013; Kavian *et al.*, 2014; Karam, Safarian and Hajjeh Froshnia, 2010; Zomorodian and Rahimi, 2012; Bayat *et al.*, 2011; Naderi, Karimi and Naseri, 2010; Zare Bidaki and Badri, 2014). Land-use change is one of the most important factors that exacerbate erosion in basins (Mohamadzade, Charm and Eskandari, 2014; Ajami, Khormali and Ayoubi, 2012).

Wind erosion

In Iran's deserts and arid areas, rapid changes in temperature cause pressure gradients in different parts which result in constant strong winds. Due to these strong winds and to the lack of moisture and vegetation, both small and large soil particles are transported, leading to soil erosion and deposition (Mehrshahy and Nakoonam, 2009). Sand dunes are estimated to cover 12 million ha. Half of these dunes are active or semi-active (Refahi, 2004). The density of air deposits measured over a 40 year period shows a growing trend in wind erosion in recent years. Sediment textures have also changed, with a significant increase in evaporated deposits such as salts and gypsum.

Studies show that both climatic and human factors play an important role in the development of wind erosion. The most important climatic factors are rainfall patterns, rising temperatures, intense evaporation from the playa, and reduced intake of water. High wind strengths also reduce the moisture level of the soil and enhance wind erosion rates. The human factor is related to land use and concerns the reduction of water entering the playa because of dam construction and excessive pressure on pastures and agricultural land in recent years (Hosseinzade, Khaneabad and Bargi, 2011).

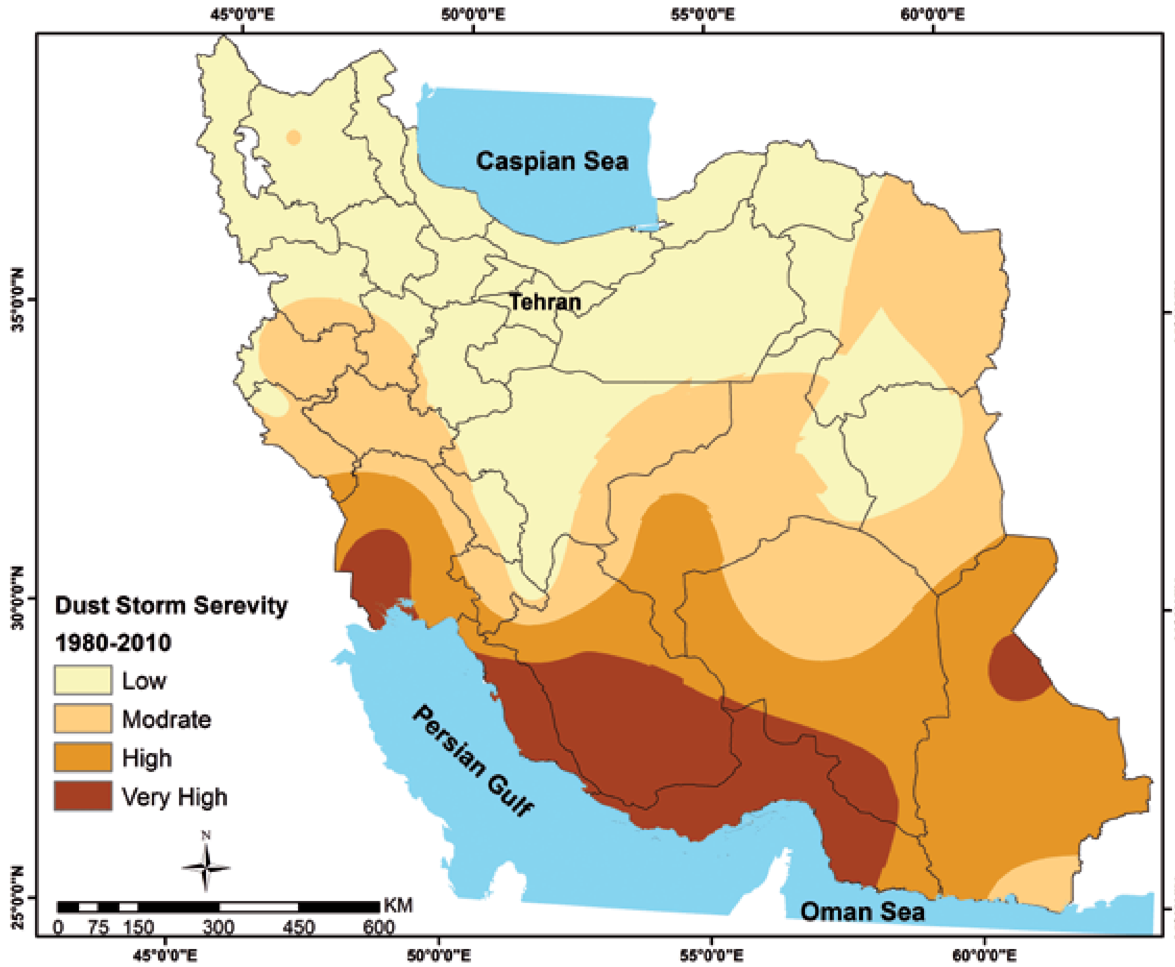


Figure 13.5 | shows days with dust storms in 2012, while Figure 13.6 shows the origin of dust storms in 2012.

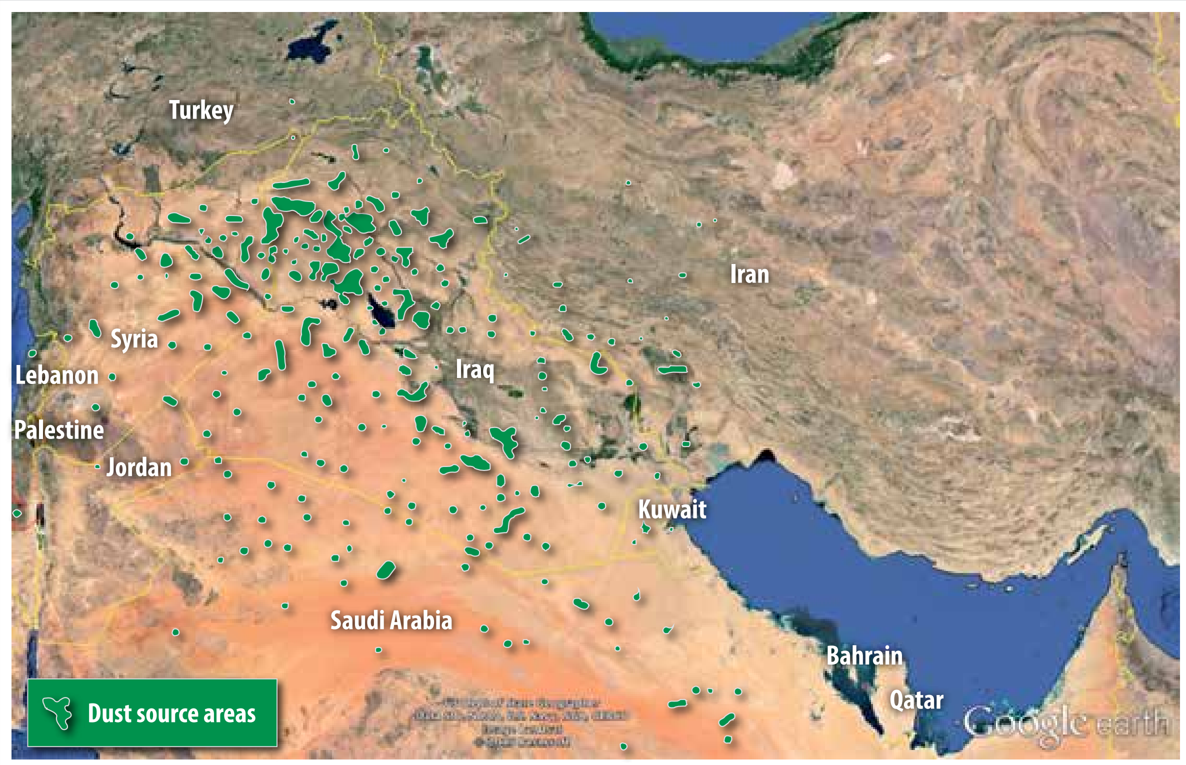


Figure 13.6 | Internal and external dust sources in recent years in Iran.
Source: University of Tehran, 2013.

The origin of dust in dust storms in Iran is both internal and external to the country. Since 2006 considerable dust has affected the west and south west of the country, originating from Iraq and Syria. Other areas such as parts of Jordan, Kuwait and northern Saudi Arabia are also involved in the creation of dust in Iran (Jalali, Bahrami and Darvishi Bolurani, 2012). Although many recent dust storms in Iran have foreign origins, this does not mean that domestic sources play no part in this phenomenon (Fattahia, Noohia and Shiravand, 2012).

13.6.2 | Case Study: Tunisia

Between 2006 and 2010 a detailed study was undertaken to assess land degradation in Tunisia. A standardized methodology was adopted to map in an interdisciplinary way all aspects of land degradation (type, extent, degree, trends and impact on ecosystem services) at a national scale. The exercise involved a large number of national institutions and stakeholders, together with international expertise (FAO, 2011a). In addition, an inventory was made of successful local practices that combatted land degradation (FAO, 2011b). This study was complemented by three investigations at local level (in Kasserine, Siliana and Médenine) that refined the identification of the socio-economic pressures and drivers behind land degradation (FAO, 2011c). The results have subsequently been expanded and refined.

Major outcomes of the investigation, relevant for the present assessment of soil change and its impact on ecosystem goods and services were:

1 - The preparation of a national land use system map

The scale of this map is 1:500 000. It is based on a rasterized database at 30 arc seconds. It was prepared using a standard methodology (Nachtergaele and Petri, 2011). The draft map was validated by regional institutions in the country and was later refined by simplifying the pastoral classes and by introducing a specific unit that concerned alfalfa areas which make up 170 000 ha of the country.

2 - Land degradation assessment mapping

The assessment used a standard methodology (Liniger *et al.*, 2011) that allowed the participatory mapping of the major types of ecosystem degradation (soil, water and biological) and sub-types (for instance, water erosion, compaction, decline in ground water quality or reduction of vegetative cover). At the same time the intensity (degree) and the trend of the ongoing ecosystem change was evaluated on the basis of arbitrary classes (typically ranging from none to severe or slow to fast). The evaluation was conducted in a participatory way involving various stakeholders in the assessment and using hard data where they were available. The direct pressures (for instance improper soil management, deforestation or natural causes) and the indirect socio economic causes (for instance lack of knowledge or investment) were also determined in the same participatory way, as was the impact on ecosystem goods and services. Examples illustrating the outcome of the assessment are given in the following maps which evaluate water erosion (Figure 13.7a) and wind erosion (Figure 13.7b). Differences in the extent of ongoing degradation processes in the country could also be mapped.

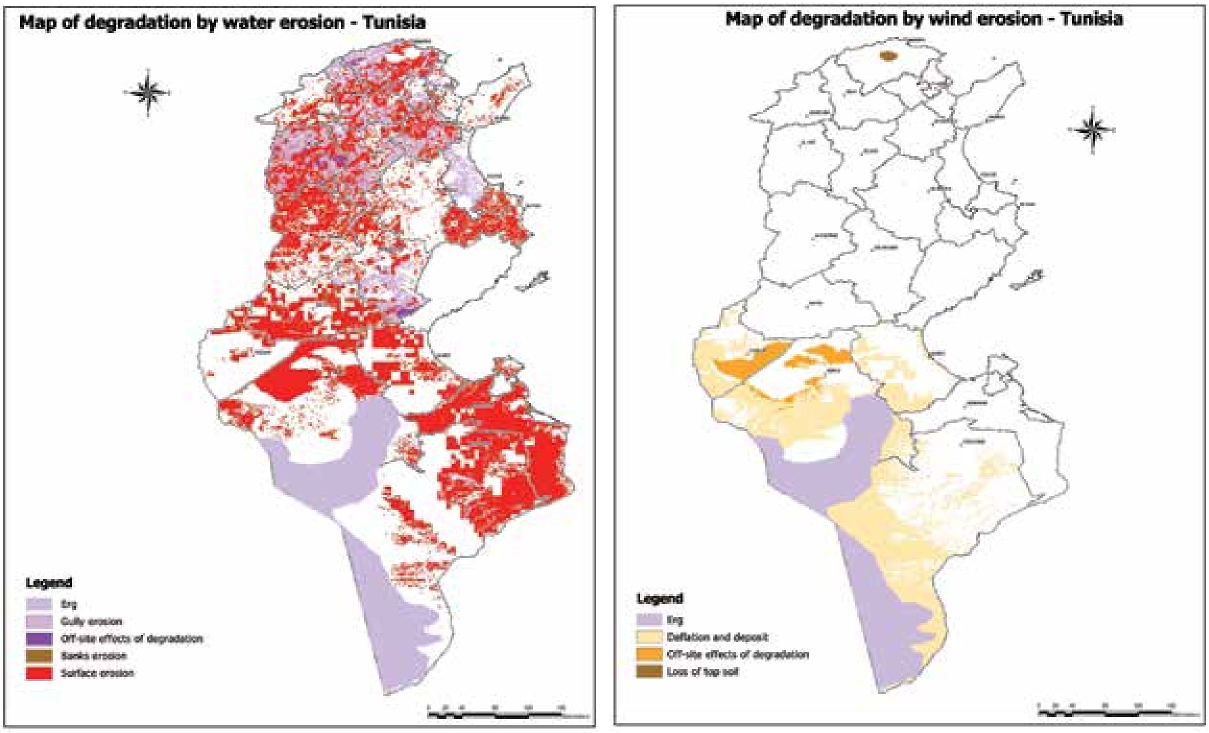


Figure 13.7 | Assessment of Water (a) and Wind Erosion (b) in Tunisia

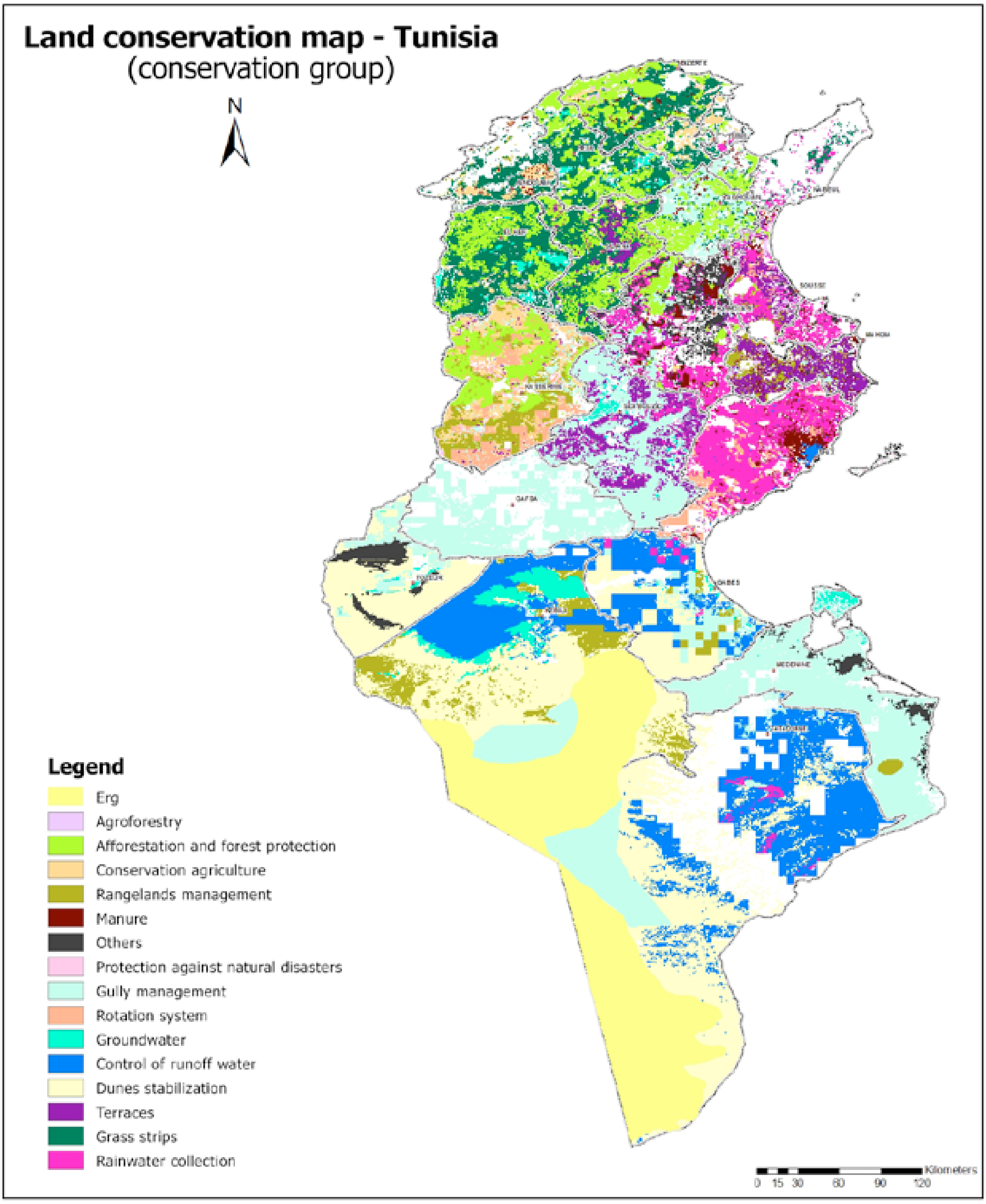


Figure 13.8 | Soil Conservation in Tunisia

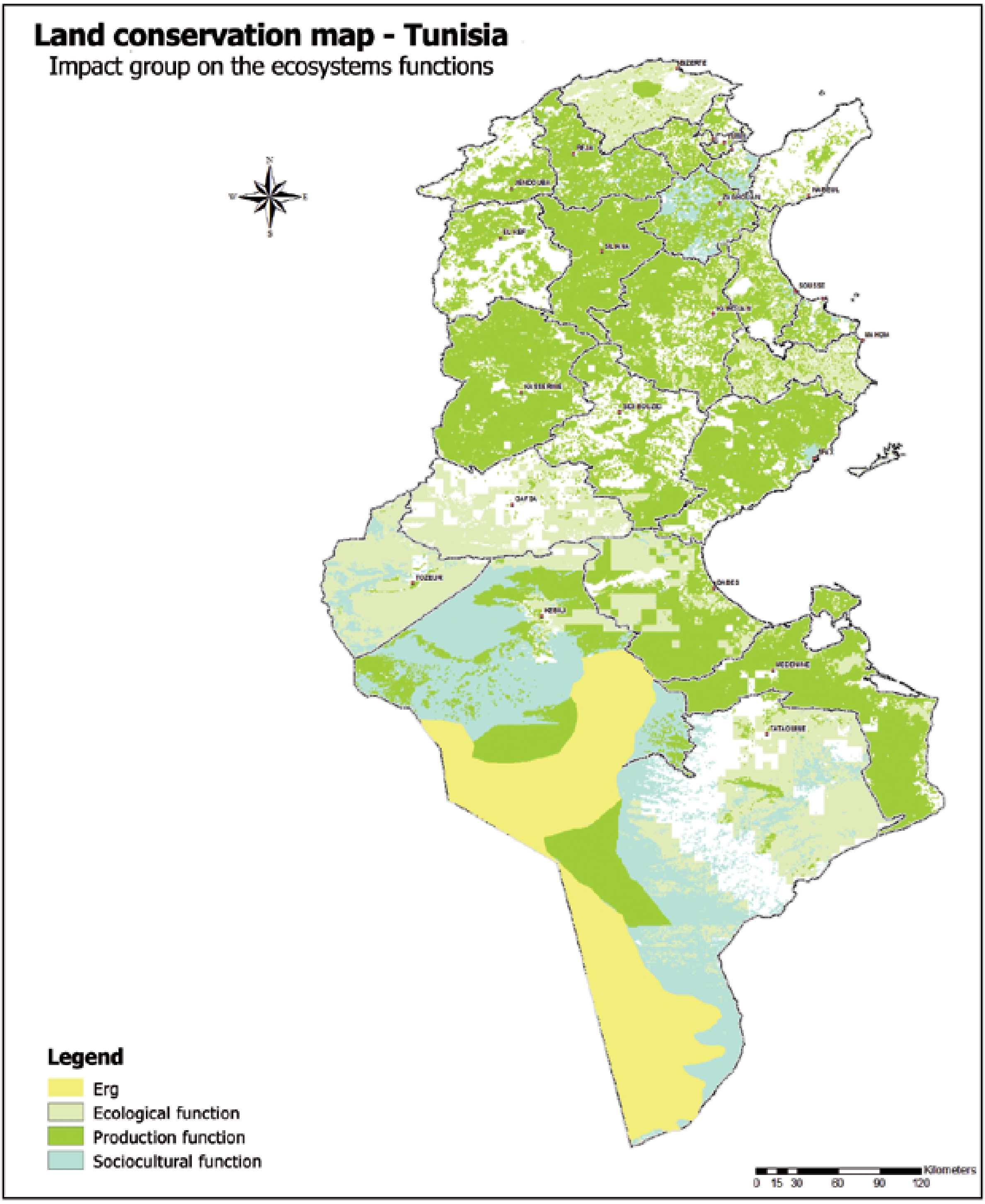


Figure 13.9 | Type of ecosystem service most affected.

3 - Sustainable land management mapping.

In each unit of the national map, the prevailing land management practices were inventoried using a standard methodology (Liniger *et al.*, 2011). This allowed the characterization of practices in terms of type of intervention (agronomic, vegetative etc) and in terms of objectives (prevention, mitigation or rehabilitation). At the same time, the extent, trend and efficiency of the conservation practices were assessed. Examples of outputs are given for the main conservation types used in the country (Figure 13.8.) and for the impact of soil degradation on ecosystem goods and services (Figure 13.9.).

13.6 | Conclusions

Although there is a wealth of local and national studies on soil change in the region, a systematic and standardized approach is lacking. Results on the extent and intensity of soil change processes still refer to the GLASOD study carried out in the late 1980s.

The degradation of natural resources in arable lands is considered as one of the main threats to agricultural production in all countries of the region. Ecosystem service quality and capacity is greatly reduced by degradation caused by salinity, erosion, contamination and poor management that leads to a loss of soil organic matter. Water erosion is predominant in that part of the region which has sloping lands. Where rainfed agriculture is practiced, water erosion may even occur in gently sloping areas. Wind erosion is also a causative factor of topsoil removal. Population increase has resulted in soil disturbance due to uncontrolled human activities such as mining and open quarries that have triggered and accelerated erosion processes. Degradation due to salinity and sodicity varies geographically with climate, agricultural activities, irrigation methods and land management policies and is mainly restricted to irrigated farming systems. Causative factors are of intrinsic origin, seawater intrusion or irrigation from groundwater with elevated salt content. Degradation due to contamination is mainly found in countries with high population, high oil production or heavy mining. In irrigated farming systems with overuse of chemicals, the load of toxic elements in groundwater is increased. Salinity has greatly reduced crop yields and increased economic annual losses across the region to nearly US$1 billion, equivalent to as much as US$1 604 ha^{-1} to US$2 748 ha^{-1}. In some countries the reduction in soil productivity was estimated to be in the range of 30-35 percent of the potential productivity.

Responses to degradation caused by erosion include improving soil resilience by increasing C inputs. This can be achieved using organic manures, compost and synthetic soil conditioners and soil conservation measures on sloping lands. Policies and regulation and socio-economic factors at individual country level were found to help reverse land degradation due to erosion. Ways of reclaiming salt-affected soils include: salt leaching and drainage interventions, crop-based management, chemical and organic amendments, fertilizers, salt tolerant plants, crop management and phytoremediation. Measures to contain degradation caused by oil contamination include farming techniques that partly eliminate hydrocarbons through decomposition, and bio-remediation using some grass species. With effective desertification control, the potential annual C sequestration rate could reach values between 0.2 to 0.4 Pg C yr^{-1}, compared to the 1.0 Pg C yr^{-1} in drylands worldwide. Ranking of soil threats is given in Table 13.3.

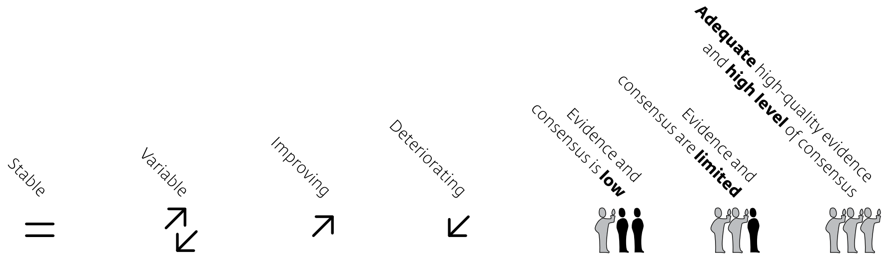


Table 13.3 | Summary of soil threats: Status, trends and uncertainties in the Near East and North Africa

Threat to soil function	Summary	Condition and Trend					Confidence	
		Very poor	Poor	Fair	Good	Very good	In condition	In trend
Soil erosion	Wind erosion and dust storms are a problem throughout the region. Sand stabilization in source areas is difficult and expensive to undertake. Water erosion can be controlled with adaptive management.	↙						
Salinization and sodification	Salinization is a widespread problem in the region due to the high temperatures, inappropriate irrigation practices and sea water intrusion in coastal areas. There is adequate research and technical knowledge in the region to counteract the problem. Socio-economic conditions hamper widespread implementation in some countries.		↙					
Organic carbon change	High temperatures throughout most of the region result in a very high turnover of soil organic Carbon. SOC change is sensitive to soil management changes.		↙					
Contamination	Contamination is locally a significant problem in the region particularly in urbanized areas that produce waste dumped on the land and in oil producing areas.	↙						
Sealing	Substantial expansion of housing, quarrying and infrastructures is a concern. There are no reliable data on sealing and land take.	↙						
Compaction	Compaction is a problem where heavy clay soils are intensively tilled (e.g. rainfed and irrigated Vertisols) and to a lesser extent is caused by off-road vehicles.		↙					

Loss of soil biodiversity	The extent of loss of soil biodiversity due to human impact is largely unknown in the NENA region. More studies need to be undertaken to understand the scope of the problem.		↙					
Soil acidification	Given the dry conditions throughout most of the region, acidification is restricted to some coastal areas with higher rainfall.					=		
Nutrient imbalance	Nutrient imbalances occur in areas with continuous cultivation where nutrients are lost in harvested crops and no engagement in fallowing, manuring or mineral fertilizer application.				=			
Waterlogging	Waterlogging is a very localized problem in the region limited to flash floods, heavily irrigated areas and excessive rise in subsoil water level.				=			

References

Abahussain, A.A., Abdu, A.Sh**., Al-Zubari, W.K., El-Deen, N.A. & Abdul Raheem, M.** 2002. *Desertification* in the Arab region: analysis of current status and trends. *J. Arid Environ.*, 51: 521–545.

Abbaslou, H., Abtahi, A., Peinado, F.J.M., Owliaie, H. & Khormali, F. 2013. Mineralogy and Characteristics of Soils Developed on Persian Gulf and Oman Sea Basin, Southern Iran: Implications for Soil Evolution in Relation to Sedimentary Parent Material. *Soil Sci.*, 178: 568-584.

Abd Elnaim, E.M., Omran, M.S., Waly, T.M. & E 1 Nashar, B.M.B. 1987. Effects of prolonged sewage irrigation on some physical properties of sandy soil. *Biological Wastes*, 22: 269-274.

Abd Elrahman, S.H., Mostafa, M.A.M., Taha, T.A., Elsharawy, M.A.O. & Eid, M.A. 2012. Effect of different amendments on soil chemical characteristics, grain yield and elemental content of wheat plants grown on salt-affected soil irrigated with low quality water. *Annals Agric. Sci.*, 57: 175-182.

Abdelfattah, M.A. & Shahid, S. A. 2007. A Comparative characterization and classification of soils in Abu Dhabi Coastal area in relation to arid and semi-arid conditions using USDA and FAO Soil Classification Systems. *Arid Land Res. and Manage.*, 21: 245-271.

Abdelfattah, M.A. 2012. Land degradation indicators and management options in the desert environment of Abu Dhabi, United Arab Emirates. *Soil Survey Horizon*, 50: 3-10.

Abdelfattah, M.A., Shahid, S.A. & Othman, Y.R. 2008. A model for salinity mapping using remote sensing and geographic information systems-A case study from Abu Dhabi, United Arab Emirates. *In Forum handbookm 2nd*, p. 92. International Salinity Forum Adelaide, Australia.

Abdelfattah, M.A., Shahid, S.A. & Othman, Y.R. 2009. Soil Salinity Mapping Model Developed Using RS and GIS – A Case Study from Abu Dhabi, United Arab Emirates. *European Journal of Scientific Research*, 26(3): 342-351.

Abu Hammad & Børresen, T. 2006. Socioeconomic factors affecting farmers' perceptions of land degradation and stonewall terraces in central Palestine. *Environ. Manage.*, 37: 380-394.

Abu Hammad & Tumeizi, A. 2012. Land degradation: Socioeconomic and environmental causes and consequences in the Mediterranean. *Land Degrad. Develop.*, 23: 216–226.

Abu-Zreig, M. 2006. Control of rainfall-iinduced soil erosion with various types of polyacrylamide. *J. Soils and Sedi.*, 6: 137-144.

Abu-Zreig, M., Al-Sharif, M. & Amayreh, J. 2007. Erosion control of arid land in Jordan with two anionic polyacrylamides. *Arid Land Res. Manage.*, 21: 315–328.

Afifi, M.Y., Genead, A.Y., Atta, S.Kh**. & Aly, A.A.** 1992. Impact of rainfall erosion and management practices on properties and productivity of Maryut soil. *Desert Institute Bulltin Egypt*, 42: 173–184.

Afyuni, M., Rezaeinejad, Y. & Schulin, R. 2006. Extractability and plant uptake of Cu, Zn, Pb and Cd from a sludge-amended Haplargid in Central Iran. *Arid Land* Res.and *Manage.*, 20: 29-41.

Ajami, M., Khormali, F. & Ayoubi, S. 2012. Soil quality indexes changes by effects of land-use change in different slopes of loss land in Western part of Golestan. *Iranian journal of Water and Soil Research*, 39(1): 15-30. [in Persian]

Akuot Gareng Apiu Anyar. 2006. Land and Environmental degradation in South Kordofan State (Case study on Dilling area). PhD Thesis. University of Khartoum.

Al Barshamgi, A. 1997. *Soil Resources in the UAE*. Proceedings of the Regional Workshop on Management of Salt-Affected Soils in the Arab Gulf States. 29 October-2 November, 1997. UAE, Abu-Dhabi.

Al Karadsheh, E., Akroush, S. & Mazahreh, S. 2012. *Land Degradation in Jordan-Review of knowledge resources*. ICARDA.

Al-Amin, N.K.N. 1999. *The physical potential of indigenous vegetation and other means to suppress sand movement in a secondary desertification source area near the White Nile in Gezira region (Sudan).* Sudan, Wad-Medani, University of Gezira. (PhD thesis)

Al-Awadhi, J.M. & Cermak, J. 1998. Sand trap for field measurements of aeolian sand-drift rate in the Kuwaiti desert. In S. Omar, R. Misak & D. Al-Ajmi, eds. *Sustainable Development in Arid Zones*, pp. 177–188. Proceedings of the International Conference on Desert Development in the Arab Gulf Countries, Kuwait. The Netherlands, Rotterdam, A.A. Balkema

Al-Awadhi, J.M. & Misak, R.F. 2000. Field assessment of aeolian sand processes and sand control measures in Kuwait. *Kuwait J. Sci. & Eng.*, 27: 156-176.

Al-Awadhi, J.M. 2013. A case assessment of the mechanisms involved in human-induced land degradation in northeast Kuwait. *Land Degrad. Develop.*, 24: 2–11.

Al-Awadhi, J.M., Al-Helal, A. & Al-Enezi, A. 2005. Sand drift potential in the desert of Kuwait. *J. Arid Environ.*, 63: 425-438.

Al-Dousari, A. M., Misak, R. & Shahid, S. 2000. Soil compaction and sealing in Al-Salmi area, western Kuwait. *Land Degrad. Develop.*, 11: 401-418.

Al-Eisawi, D. 1998. *Medicinal plants and biodiversity*. Paper presented at the Workshop on National Biodiversity Planning, Arabian Gulf University, 12–14 October, 1998, Bahrain. 9 pp.

Al-Harthi, A.A. 2002. Geohazard assessment of sand dunes between Jeddah and Al-Lith, western Saudi Arabia. *Environ. Geology*, 42: 360-369.

Al-Hemiary A.M. 1999. Yemeni Experience in the Watershed Management. YEM/97/200. Sana'a ROY.

Al-Hemiary, A.A. 2001. *The state of land and water resources in Yemen.* Regional Land Resources Information Systems Workshop. Cairo, Egypt 3-6 September, 2001.

Al-Hiba, M. 1997. *Improvement of irrigation and drainage systems for soil salinity in the Arab Region*. Regional Workshop on Management of Salt-affected Soils in the Arab Gulf States. Abu-Dhabi, UAE, 29 October-2 November, 1997.

Al-Rajhi, M.A., Seaward, M.R.D. & Al-Aamer, A.S. 1996. Metal levels in indoor and outdoor dust in Riyadh, Saudi Arabia. *Environ. Int.*, 22: 315-324.

Al Senafy, M.N., Viswanathan, M.N., Senay, Y. & Sumait, A. 1997. Soil contamination from oil lakes in northern Kuwait. *J. Soil Contamin.*, 6: 481-494.

Arab Organization for Agricultural Development (AOAD). 2004. Also available at http://www.aoad.org/e-mail_en.htm

Ardö, J. & Olsson, L. 2003. Assessment of soil organic carbon in semi-arid Sudan using GIS and the CENTURY model. *J. Arid Environ.*, 54: 633-651.

Ardö, J. &. Olsson, L. 2004. Soil carbon sequestration in traditional farming in Sudanese drylands. *Environ. Manage.*, 33: 318-329.

Asadi, H., Raeisvandi, A., Rrabie[i], B. & Ghadiri, H. 2012. Effect of land use and topography on soil properties and agronomic productivity on calcareous soils of a semiarid region, Iran. *Land Degrad. Develop.*, 23: 496–504.

Ayoub, A.T. 1998. Land degradation, rainfall variability and food production in the Sahelian zone of the Sudan. *Land Degrad. Develop.*, 10: 489-500.

Azimzadeh, H.R., Ekhtesasib, M.R., Refahic, H.Gh**., Rohipourd, H. & Gorji, M**. 2008. Wind erosion measurement on fallow lands of Yazd-Ardakan plain, Iran. *Desert*, 13: 167-174.

Badraoui, M. 1998. Effects of intensive cropping under irrigati on on soil quality in Morocco. In Badraoui, ed. *Proceedings of the 16th world Congress of Soil Science.* B 5-Post Congress tour in Morocco. Rabat, AMSSOL.

Bakheit, S.A. 2011. *Availability and Contents of Famine Foods in Northern Darfur.* University of Khartoum. (MA thesis)

Balba, M.T., Al-Daher, Al-Awadhi, N., Chino, H. & Tsuji, H. 1998. Bioremediation of oil-contaminated desert soil: The Kuwaiti experience. *Environ. Int.*, 24: 163-173.

Bayat, H., Neyshabouri, M.R., Mohammadi, K. & Nariman Zadeh, N. 2011. Estimating water retention with pedotransfer functions using multiobjective group method of data handling and ANNs. *Pedosphere*, 21(1): 107–114.

Belaroui, K., Djediai, H. & Megdad, H. 2014. The influence of soil, hydrology, vegetation and climate on desertification in El-Bayadh region (Algeria). *Desalin. Water Treat.*, 52: 2144-2150.

Belgacem, A.O., Tarhouni, M. & Louhaichi, M. 2013. Effect of protection of Mediterranean arid zone of southern Tunisai: A case study from Bou Hedma National Park. *Land Degrad. Develop.*, 24: 57-62.

Ben-Mahmoud, R., Mansur, S. & AL-Gomati, A. 2000. *Land degradation and desertification in Libya.* Land Degradation and Desertification Research Unit. Libya, Tripoli, Libyan Center for Remote Sensing and Space Science.

Benmansour**, M.,** Mabit**, L.,** Nouira**, A.,** Moussadek**, R.,** Bouksirate**, H. ,** Duchemin**, M. &** Benkdad**, A.** 2013. Assessment of soil erosion and deposition rates in a Moroccan agricultural field using fallout 137Cs and 210Pbex. *J. Environ. Radioact.*, 115: 97-106.

Besalatpour A, Hajabbasi, M.A., Khoshgoftarmanesh, A.H. & Dorostkar, V. 2011. Land farming process effects on biochemical properties of petroleum-contaminated soils. *Soil and Sed. Contamin.*, 20: 234-248.

Besalatpour, A., Khoshgoftarmanesh, A.H., Hajabbasi, M.A. & Afyuni, M. 2008. Germination and Growth of Selected Plants in a Petroleum Contaminated Calcareous Soil. *Soil and Sed. Contamin.*, 17: 665-676.

Bessam, F. & Marbet, R. 2003. Long-term changes in soil organic matter under conventional tillage and no-tillage systems in semiarid Morocco. *Soil Use Manag.*, 19: 139-143.

Ben-Mahmoud, R., Mansur, S. & Al-Gomati, A. 2000. Land degradation and desertification in Libya, Land Degradation and Desertification Research Unit, Libyan Center for Remote Sensing and Space Science, Tripoli, Libya.

Biro, K.B., Pradhan, M., Buchroithner & Makeschin, F. 2013. Land use/land cover change analysis and its impact on soil properties in the northern part of Gadarif region, Sudan. *Land Degrad. Develop.*, 24: 90–102.

Bou Kheir, R., Cerdan, O. & Abdallah, C. 2006. Regional soil erosion risk mapping in Lebanon. *Geomorphology*, 82: 347–359.

CAMRE/UNEP/ACSAD. 1996. *State of Desertification in the Arab Region and the Ways and Means to deal with it.* Council of Arab Ministers Responsible for the Environment (CAMRE), United Nations Environment Programme (UNEP), Arab Center for Studies of Arid Zones and Drylands (ACSAD). Syria, Damascus.. 444 pp. [in Arabic with English summary]

Chigani, H.K., Khajeddin, S.J. & Karimzadeh., H.R. 2012. Soil-vegetation relationship of three arid land plant species and their use in rehabilitating degraded sites. *Land Degrad. Develop.*, 23: 92-101.

Dahan, R., Boughlala, M., Mrabet, R., Laamari, A., Balaghi, R. & Lajouad, L. 2012. A review of available knowledge on land degradation in Morocco. International Center for Agricultural research in the Dry Areas (ICARDA), Aleppo, Syria.

Darwish, T., Khater, C., Jomaa, I., Stehouwer, R., Shaban, A. & Hamze, M. 2011. Environmental impact of quaaries on natural resources in Lebanon. *Land Degrad. Develop.*, 22: 345–358.

Darwish, T.G.F. & Khawlie, M. 2004. Assessing Soil Degradation by Landuse-Cover Change in Coastal Lebanon. *Lebanese Sci. J.*, 5: 45-59.

Dayan, U., Heffter, J., Miller, J. & Gutman, G. 1991. Dust intrusion into the Mediterranean basin. *J. Applied Meteorology*, 30: 1185-1199.

Dima, S J. 2006. *Land use systems in South Sudan and their impacts on land degradation*: A paper presented at the Conference on Environmental Management Plan in Post Conflict South Sudan, Juba Raha Hotel, 31 October - 03 November 2006.

Dixon, J. & Gulliver, A. 2001. *Farming Systems and Poverty*. Rome, FAO & Washington, DC, World Bank.

Djavanshir, K., Dasmalchi, H. & Emararty, A. 1996. Ecological and ecophysiological survey on sexual Euphrate poplar and Athel trees in Iranian deserts. *Journal of Biaban*, 1: 67–81.

Dregne, H. & Chou, N. 1992. Global desertification dimensions and cost. In H.E. Dregne, ed. *Degradation and Restoration of Arid Lands Texas*, pp. 249-282. USA, Texas, Texas University Press.

Dregne, H.E. 2002. Land Degradation in the Drylands. *Arid Land Res. Manage.*, 16: 99-132.

EAD. 2009. *Soil survey of Abu Dhabi Emirate*. Vol. 5. United Arab Emirates, Environment Agency Abu Dhabi.

EEAA. 1994. *Industrial Wastewater Pollution Abatement in Kafr El-Zayat*. Pre- Feasibility Study. Technical Cooperation Office for the Environment.

El Tahir, B.A., Ahmed, D.M., Ardö, J., Gafaar, A.M. & Salih, A.A. 2008. Changes in soil properties following conversion of Acacia senegal plantation to other land management systems in North Kordofan State, Sudan. *J. Arid Environ.*, 73: 499-505.

Elagib, N.A. 2011. Changing rainfall, seasonality and erosivity in the hyper-arid zone of Sudan. *Land Degradation and Development*, 22: 505-512.

Elfaig, A., Ibrahim, O. & Jaafar, M. 2015. Environmental degradation in the Sahel: A study from Western Sudan-ghubaysh area. *International Journal of Current Research*, 7(2): 12588-12596.

El-Hady, O.A. & Abo-Sedera, S.A. 2006.Conditioning Effect of Composts and Acrylamide Hydrogels on a Sandy Calcareous Soil. II-Physico-bio-chemical Properties of the Soil. *Int. J. Agric. Biol.*, 8: 876–884.

Elias, E.A. & Alaily, F. 2002. Effects of long-term irrigation on soluble salts and exchangeable cations in Vertisols from Gezira. *Geograficky Casopis*, 54: 343-353.

El-Kholei, A. 2012. *Country Study on Status of Land Tenure, Planning and Management in Oriental Near East Countries*. Food and Agriculture Organization of the United Nations, Regional Office for the Near East (RNE), Oriental Near East Sub-Region (SNO).

El-Tantawi, A.M.M. 2005. *Climate Change in Libya and Desertification of Jifara Plain Using Geographical Information System and Remote Sensing Techniques*. der Johannes Gutenberg-Universität. (PhD thesis)

ESCWA. 2007. The Millennium Development Goals in the Arab Region: A youth lens. UN-ESCWA.

Fallahzadeh, J. & Hajabbasi, M.A. 2012a. Land Use change effects on carbohydrate fractions, total and particulated organic matter of forest soils in central Zagros Mountain, Iran. *J. App. Sci.*, 12: 387-392.

Fallahzadeh, J. & Hajabbasi, M.A. 2012b. The effects of irrigation and cultivation on the quality of desert soil in central Iran. *Land Degrad. Develop.*, 23: 53-61.

FAO. 1994. *Land degradation in South Asia, its severity, causes and effects upon the people*. World Soil Resources Report No. 78. Rome.

FAO. 1997. *Regiomal Workshop on Management of Salt-affected Soils in the Arab Gulf States*. 29 October-2 November 1995. UAE, Abu-Dhabi.

FAO. 2004. *Regional Workshop on Promoting LADA Program in western Asia and the Near East*. 25-28 July 2004. Syria, Damascus.

FAO. 2010. *Land Degradation Assessment in Drylands* (LADA). Rome, FAO. (Also available at http://www.fao.org/nr/lada/)

FAO. 2011. *Regional Priority Framework for the Near East*. Food and Agriculture Organization of the United Nations, FAO Regional Office for the Near East.

FAO. 2011a. Pour une évaluation de la degradation des terres en Tunisie. Cadre institutionnel et législatif Information et données disponibles Etat des connaissances. Ministère de l'Agriculture et des Ressources Hydrauliques DG/ACTA and FAO (Eds). LADA Country Report. Rome, FAO.

FAO. 2011b. Gestion durable des terres en Tunisie – Bonnes pratiques agricoles. LADA Country Report – Tunisia. Rome, FAO.

FAO. 2011c. Manual for Local Level Assessment of Land Degradation and Sustainable Land Management. LADA Technical Report 11 vol. 1 and 2. Rome, FAO.

FAO. 2012. *Country Study on Status of Land Tenure, Planning and Management in Oriental Near East Countries*. Rome, FAO.

FAO. 2013. *Statistical Analysis of Select Food and Agricultural Indicators for the Near East and North Africa*. Rome, FAO.

FAO. 2013. *The 3rd Regional Multi-StakeholderWorkshop on Food Security and Nutrition*. 4-6 November 2013, Tunis, Tunisia. Egypt, Cairo, Regional Office for the Near East and North Africa, FAO.

Fattahia, E., Noohia, K. & Shiravand, H. 2012. Study of Dust Storm Synoptical Patterns in Southwest of Iran. DESERT 17 (2012) 49-55

Gibbs, G.K. 1954. *Report to the Government of Iraq on Soil Conservation*. Report No. 242. Rome, FAO.

Golchin, A. & Asgari, H. 2008. Land use effects on soil quality indicators in north-eastern Iran. *Australian Journal of Soil Research*, 46: 27–36.

Goudie, A.S. & Middleton, N.J. 2001. Saharan dust storms: nature and consequences. *Earth-Science Reviews*, 56: 179-204.

Hajabbasi, M.A., Jalalian, A. & Karimzadeh, H.R. 1997. Deforestation effects on soil physical and chemical properties, Lordegan, Iran. *Plant Soil*, 190: 301-308.

Halitima, A. 1988. *Sols des régions arides*. Alger, OPU. 384 pp.

Hamdallah, G. 1997. *An overview of the salinity status of the Near East Region*. Proceedings of the Regional Workshop on Management of salt-affected soils in the Persian Gulf Status. 29 October – 2 November 1995. UAE, Abu Dhabi.

Hamidi, M., Kavianpour 1, M.R. & Shao, Ya. 2013. Synoptic Analysis of Dust Storms in the Middle East. *Asia-Pacific J. Atmos. Sci.*, 49(3): 279-286.

Hejazi, R.F., Hussain, T. & Khan, F.I. 2003. Landfarming operation of oily sludge in arid region: human health risk assessment. *J. Hazard. Mat.*, 99: 287–302.

Hosseinkhani, H. 2013. Assessment the erosion risk and potential of sedimentation in Shahriar dam catchment, using GIS techniques and EPM model. *Journal of Iran geology*, 7(26): 96-87.

Hosseinzade, S.R. Khaneabad, M. & Bargi, M. 2011. Evaluation of wind erosion in the Boshroiye region since 1975 to 2010. *The first national conference of iranian association of geomorphology*.

Houyou, Z., Bielders, C.L., Benhorma, H.A., Dellal, A. & Boutmedjet, A. 2014. Evidence of strong land degradation by wind erosion as a result of rainfed cropping in the Algerian Steppe: A case study at Laghout. *Land Degrad. Develop.*

Hussein, H. 2001. *Development of environmental GIS database and its application to desertification study in Middle East*. Japan, Chiba University, Graduate School of Science and Technology. (PhD thesis)

Hussein, M.A. 2008. Costs of environmental degradation: An analysis in the Middle East and North Africa region. *Manag. Environ. Qual*., 19: 305-317.

Hussein, M.H. 1998. Water erosion assessment and control in Northern Iraq. *Soil Till. Res.*, 45: 161–173.

Ibrahim, M.M. & El-Gaely, G.A. 2012. Short-term effects of dust storm on physiological performance of some wild plants in Riyadh, Saudi Arabia. *Afric. J. Agric. Res.*, 7(47): 6305-6312.

Jalali, M. Bahrami, H A. & Darvishi Bolurani, A. 2012. Evaluation of climatic parameters and vegetation changes in the dust generating regions using satellite imagery. Thesis of MSc soil science, Tarbiat Modares University. 114 pp.

Jalalian, A., Ghahsareh, A.M. & Karimzadeh, H.R. 1996. *Soil erosion estimates for some watersheds in Iran*. The First International Conference on Land Degradation 10–14 June. Turkey, Adana.

Jish Prakash, P., Stenchikov, G., Kalenderski, S., Osipov, S. & Bangalath, H. 2014. The impact of dust storms on the Arabian Peninsula and the Red Sea. Atmos. Chem. *Phys. Discuss*., 14: 1918-19245.

Jones, A., Breuning-Madsen, H., Brossard, M., Dampha, A., Deckers, J., Dewitte, O., Gallali, T., Hallett, S., Jones, R., Kilasara, M., Le Roux, P., Micheli, E., Montanarella, L., Spaargaren, O., Thiombiano, L., Van Ranst, E., Yemefack, M. & Zougmore R. (eds.). 2013. *Soil Atlas of Africa*. Luxembourg, European Commission, Publications Office of the European Union. 176 pp.

Karam, A. Safarian, A. & Hajjeh Froshnia, S.H., 2010. Estimation and zoning of soil erosion in the Mamlou catchment (East of Tehran) using the modified universal equation of soil erosion and analytic hierarchy process. *Earth science research*, 1(2): 73-86.

Kavian, A. Zabihi, M. Safari, A. & Mohammadi, M.A. 2014. Estimating soil erosion using the universal soil erosion equation (USLE) (Case study: Watershed of Nekarood). 13th Iranian *Soil Science Congress*. Shahid Chamran University of Ahvaz.

Khalafa, F.I. & Al-Jimi, D. 1993. Aeolian processes and sand encroachment problems in Kuwait. *Geomorphology*, 6: 111-134.

Khresat, S., Al-Bakir, J. & Al-Tahran, R. 2008. Impacts of land use/cover change on soil properties in the Mediterranean region of northwestern Jordan. *Land Degrad. Develop*., 19: 397-407.

Khresat, S.A., Rawajfih, Z. & Mohammad, M. 1998. Land degradation in north-western Jordan: causes and processes. *J. Arid Environ*., 39: 623–629.

Kobler, M. 2013. *Dust storms of Iraq*. UN Secretary General for Iraq, A ministerial meeting. Kenya, Nairobi. (Also available at http://www.term123.com/dust-storms-of-iraq/#mh32BcOB4S6cRkIG.99)

Koocheki, A. 2000. Potential of saltbush (Atriplex spp.) as a fodder shrub for the Arid Lands of Iran. In G. Gintzburger, M. Bounejmate & A. Nefzaoui, eds. *Fodder shrub development in arid and semi-arid zones*, pp. 178- 183. ICARDA.

Lal, R. 2002. Carbon sequestration in dryland ecosystems of West Asia and North Africa. *Land Degrad. Develop*., 13: 45–59.

Liniger, H.P. van Lynden, G., Nachtergaele, F., Schwilch, G. & Biancalani, R. 2011. Questionnaire for mapping land degradation and Sustainable land management (QM). LADA Technical Report # 9. FAO, Rome.

Madany, I.M., Akhter, M.S. & Al-Jowder, O.A. 1994. The correlations between heavy metals in residential indoor dust and outdoor street dust in Bahrain. *Environ. Int.*, 20: 483-492.

MADRPM. 2000. *Atlas de l'Agriculture. Colloque National de l'Agriculture et du Développement Rural*. Ministère de l'Agriculture du Développement Rural et des Pêches Maritimes. Maroc, Rabat.

Mamdouh Nasr. 1999. *Assessing Desertification and Water Harvesting in the Middle East and North Africa*: Policy Implications, ZEF Discussion Papers On Development Policy No. 10. Bonn, Center for Development Research. 59 pp.

Masri, Z. & Ryan, J. 2006. Soil organic matter and related physical properties in a Mediterranean wheat-based rotation trial. *Soil Till. Res.*, 87: 146–154.

Mehrshahy, D. & Nakoonam, Z. 2009. Amazing face of wind erosion in Iran deserts. *Geography education development*, 88(24): 3-9.

Misak, R.F., Abdel Baki, A.A. & El-Hakim, M.S. 1997. On the causes and control of the waterlogging phenomenon, Siwa Oasis, northern Western Desert, Egypt. Journal of arid environments, 37(1): 23-32.

Misak, R.F., Al Awadhi J.M., Omar S.A. & Shahid S.A. 2002. Soil degradation in Kabad area, southwestern Kuwait City. *Land Degradation & Development*, 13(5): 403-415.

Misak, R.F., Khalaf, F.I. & Omar, S.A.S. 2009. Managing the hazarads of drought and shifting sands in dry lands: The cast study of Kuwait. In S.A. Shahid, F.K. Taha & M.A. Abdelfattah, eds. *Developments in Soil Classification, Land Use Planning and Policy Implications: Innovative Thinking of of Soil Inventory for Land Use Planning and Management of Land Resources*, pp. 731-752. Springer.

Mohamadzade, M. Charm, M. & Eskandari, M. 2014. Evaluation of land use change effect on sediment production, using WEPP model. 13th *Iranian Soil Science Congress*. Shahid Chamran University of Ahvaz.

Mohammed, A.E., Stigter, C.J. & Adam, H.S. 1995. Moving sand and its consequences on and near a severely desertified environment and a protective shelterbelt. *Arid Soil Res. Rehabilit.*, 9: 423-435.

Mohammed, A.E., Stigter, C.J. & Adam, H.S. 1996. On shelterbelt design for combating sand invasion. Agric., *Ecosyst. Environ.*, 57: 81-90.

Mrabet, R., Ibno-Namr, K., Bessam, F. & Saber, N. 2001. Soil chemical quality changes and implications for fertilizer management after 11-years of no-tillage wheat production systems in semiarid Morocco. *Land Degrad. Develop.*, 12: 505-517.

Mubarak, A.R. & Nortcliff, S. 2010. Calcium Carbonate Solubilization through H-Proton Release from some Legumes Grown in Calcareous Saline-Sodic Soil. *Land Degrad. Develop.* 21: 29-39.

Nachtergaele, F. & Petri, M. 2011. *Mapping land use systems at global and regional scales for land degradation assessment analysis*. Technical Report No.8. Version 1.1. LADA. FAO.

Naderi, F A., Karimi, H. & Naseri, B., 2010. Soil erosion potential zoning in Aseman Abad watershed by erosion index. *Watershed Management Researches Journal*, 89: 44-51.

Nael M., Khademi H. & Hajabbasi M.A. 2004. Response of soil quality indicators and their spatial variability to land degradation in central Iran. *Appl. Soil Ecol.*, 27: 221-232.

Naifer, A., Al-Rawahy, S.A. & Zekri, S. 2011. Economic Impact of Salinity: The Case of Al-Batinah in Oman. Int. J. Agric. Res., 6: 134-142.

NAP. 2002. *National Action Plan for Combating Desertification*. Egypt, Cairo.

NAP. 2003. *Causes and Effects of Desertification. In National Action Programme to Combat Desertification.* Lebanon, Ministry of Agriculture. pp. 99-109.

Nielsen, T.L. & Zöbisch, M.A. 2001. Multi-factorial causes of land-use change: land-use dynamics in the agro-pastoral village of Im Mial, northwestern Syria. *Land Degrad. Develop.*, 12: 143-161.

Nielsen, T.T. & Adriansen, H.K. 2005. Government policies and land degradation in the Middle East. *Land Degrad. Develop.*, 16: 151–161.

Omar, S.A.S., Madouh, T., El-Bagouri, I., Al-Mussalem, Z. & Al-Telaihi, H. 1998. Land degradation factors in arid irrigated areas: the case of Wafra in Kuwait. *Land Degrad. Develop.*, 9: 283-294.

Poussart, N., Ardö, J. & Olsson, L. 2004. Effects of Data Uncertainties on Estimated Soil Organic Carbon in the Sudan. *Environ. Manage*., 33: 405-415.

Qadir, M., Qureshi, A.S. & Cheraghi, M.S. 2007. Extent and characterization of Salt-prone land resources in Iran and strategies for amelioration and management. *Land Degrad. Develop*., 19: 214-227.

Qadir, M., Qureshi, A.S. & Cheraghi, S.A.M. 2008. Extent and characterisation of salt-affected soils in Iran and strategies for their amelioration and management. *Land Degrad. Develop.*, 19: 214-227.

Rawajfih, Z., Khersat, S.A. & Buck, B. 2005. Arid soils of the Badia region of northeastern Jordan: Potential use for sustainable agriculture. *Arch. Agron. Soil Sci.*, 51: 25–32.

Refahi, H. 2004, Wind erosion and its control. Tehran University, Iran. 320 pp.

Sadeghi, S.H.R., Ghaderi Vangah, B. & Safaeeian, N.A. 2007. Comparison between effects of open grazing and manual harvesting of cultivated summer rangelands of northern Iran on infiltration, runoff and sediment yield. *Land Degrad. Develop*., 18: 608–620.

Sallam, A.S., Elwan, A.A. & Rabi, F.H. 1995. Shale Deposits of El- Farafra Depression (Egypt) as Soil Conditioner for Sandy Soils. *Arid Soil Res. Rehabilit*., 9: 209-217.

Schneider, A., Friedl, M.A. & Potere, D. 2009. A new map of global urban extent from MODIS satellite data. *Envir. Res. Lett*., 4: 044003.

SCWMRI. 2015. *Identification and monitoring of Dust storm in period 1980-2010*. Soil Conservation and Watershed Management Research Institute.

SEDENOT. 2000. *Sustainable feeding systems through local resources in small farms in the North West of Tunisia*. Inter-University Belgium & Tunisia Co-operation Project.

Shinjo, H., Fujita, H., Gintzbuger, G & Kosaki, T. 2000. Impact of grazing and tillage on water erosion in northeastern Syria. *Soil Sci. Plant Nutr.*, 46: 151-162.

Shiranpour, B., Bahrami, A. & Shabanpour, M. 2011. Effects of Converting Forest to Tea Garden on Soil Fertility in Guilan Province. *Water and Soil Journal*, 26: 826-831.

Siadat, H., Bybordi, M. & Malakouti, M.J. 1997. *Salt- affected soils of Iran: A country report.* International symposium on Sustainable Management of Salt Affected Soils in the Arid Ecosystem. Egypt, Cairo.

Sohrabi, T. & Zehtabian, G.H. 2012. The role of agriculture in chemical soil degradation of Taleghan. *Iranian Journal of Range and Desert Research*, 19(46): 17-31.

Thomas, D.S.G. & Middelton, N.J. 1994. *Desertification: exploring the myth*. UK, Chichester, Jhon Willey & Sons Ltd. 194 pp.

Tork Nejad, A. & Koocheki, A. 2000. Economic aspects of fourwing saltbush (Atriplex canescens) in Iran. In G. Gintzburger, M. Bounejmate, & A. Nefzaoui, eds. *Fodder Shrub Development in Arid and Semiarid Zones*. Vol. 1, pp. 184–186. Aleppo, International Center for Agricultural Research in the Dry Areas (ICARDA).

University of Tehran. 2013. External dust storm sources report. *Geoinformatic Institute*, University of Tehran.

Vaezi, A.R. 2012. *Soil Degradation and Wheat Yield in Dry-Farming Lands in A Semi-Arid Region*. Iran.

Varamesh, S., Hosseini S.M., Abdi, N. & Akbarinia, M. 2010. Effects of afforestation on soil carbon sequestration in an urban forest of arid zone in Chitgar forest park of Tehran. *Nauka za Gorata*, 47(3): 75–90.

Wassif, M.M., Atta, S.Kh**. & Tadros, S.F.** 1995. Water erosion of calcareous soil and its productivity under rainfed agriculture of Egypt. *Egypt. J. Soil Sci*., 35: 15–31.

Wiebe, K. 2003. *Linking Land Quality, Agricultural Productivity and Food Security*. Agricultural Economic Report No. 823. Department of Agriculture, Resource Economics Division. USA, Economic Research Service.

Yaghi, B. & Abdul-Wahab, S.A. 2004. Levels of heavy metals in outdoor and indoor dusts in Muscat, Oman. Int. *J. Environ. Stud*., 61: 307-314.

Yazdani Nejad, F. & Torabi Gol Sefidi, H. 2013. Evaluation of Spatial variability and zoning of salinity in agricultural land in south of Tehran, using kriging and GIS. *Soil and Water Research of Iran*, 44(3): 255-262.

Yigini, Y., Panagos, P. & Montanarella, L. (eds). 2013. Soil Resources of Mediterranean and Caucasus Countries. JRC Technical Report. Luxembourg: Publications Office of the European Union.

Zahreddine, H.G., Barker, D.J., Quigley, M.F., Sleem, K. & Struve, D.K. 2007. Patterns of woody plant species diversity in Lebanon as affected by climatic and soil properties. *Lebanese Sci. J.*, 8: 21-44.

Zare Bideki, R. & Badri, B. 2014. Zoning of flood potential in the Pardingan basin of Chaharmahal Bakhtiari province. 13th *Iranian Soil Science Congress*. Shahid Chamran University of Ahvaz.

Zomorodian, M.J. & Rahimi, R. 2012. Quantitative and Qualitative Analysis of Erosion on Southern River Basins Adjacent to Mashhad and its Environmental Impact. Geography and Development 10nd Year - No. 28: 44-46

Zurayk, R., El-Awar, F., Hamadeh, S., Talhouk, S., Sayegh, C., Chehab, A. & Al Shab, K. 2001. Using indigenous knowledge in land use investigations: a participatory study in a semi-arid mountainous region of Lebanon. *Agric., Ecosyst. Environ.*, 86: 247–262.

14 | Regional Assessment of Soil Changes in North America

Regional Coordinator: Hempel, J. (ITPS/United States)

Regional Lead Author: D. Pennock (ITPS/Canada)

Contributing Authors: Adams, M-B. (United States), Basiliko, N. (Canada), Bedard-Haughn, A. (Canada), Bock, M. (Canada), Cerkowniak, D. (Canada), Cruse, R. (United States), Dabney, S. (United States), Daniels, L. (United States), Drury, C. (Canada), Fanning, D. (United States), Flanagan, D. (United States), Grayston, S. (Canada), Harrison, R. (United States), Hempel, J. (ITPS/United States), Lobb, D. (Canada), Parikh, S. (United States), Reid, D.K. (Canada), Sheppard, S. (Canada), Smith, C.A.S. (Canada), Watmough, S. (Canada).

14.1 | Introduction

Although Canada and the United States of America have a long history of collaborative research activity in soil science, there have been no previous attempts at a regional assessment of threats to soil functions. Nor are there any ongoing institutional arrangements that coordinate soil assessment or management across the two countries, unlike trans-border water issues – which are adjudicated by an International Joint Commission – or atmospheric issues, such as when cross-border problems with acidification were the focus of the Air Quality Agreement of 1991. Cross-border coordination is further complicated by the lack of a common soil classification system between the two countries.

In the absence of a regional reporting system for threats to soil functions, this chapter draws on the appropriate national reporting systems and on the expertise of leading soil scientists where national assessments do not exist.

The main data source used for Canada is the Agri-Environmental Indicators report series, which was developed by Agriculture and Agri-Food Canada (AAFC). This series began in 1993 with the intent of producing science-based environmental indicators specific to the agriculture and agri-food sector. The work presented in this chapter is drawn from the forthcoming 4th Agri-Environmental Indicators report (Clearwater *et al.*, 2015). The series estimates change in the indicators in five-year periods beginning (for most indicators) in 1981. Indicators that assess primary agriculture are calculated using mathematical models or formulas that integrate information on soil, climate and landscape, mainly derived from the Soil Landscapes of Canada (SLC) (Soil Landscapes of Canada Working Group, 2007), with information on crops, land use, land management and livestock from the Census of Agriculture and other custom data sets from provincial agencies, the private sector, remote sensing, etc. Information on the specific indicators is available in AAFC (2013).

The 4th Agri-Environmental Indicators report provides information for soil erosion, change in SOC, and nutrient imbalance, and this information is featured in Sections 4.3 and 4.4 of this chapter. Leading Canadian scientists selected by the Canadian Society of Soil Science provided information on the remaining threats.

The major information source used for the United States is the National Resources Inventory (USDA, 2013a). This report provides a range of land use and management statistics and national estimates for sheet and rill erosion and for wind erosion. The report provides data for the period 1982-2010. Data are gathered annually by the National Resources Conservation Agency and major reports are released at five-year intervals. Information on specific threats such as salinization was also provided by the $STATSGO_2$ database (Soil Survey Staff, 2014). Leading United States soil scientists selected by the Soil Science Society of America also provided information for the United States.

14.2 | Regional stratification and soil threats

14.2.1 | Regional stratification and land cover

The spatial framework used in this chapter is the multi-level Ecological Regions of North America developed by the Commission for Environmental Cooperation (Commission for Environmental Cooperation, 1997). The Level II ecoregions are used as a consistent geographical reference (Figure 14.1).

The contiguous 48 United States, Hawaii, Puerto Rico, and the United States Virgin Islands cover almost 88 Bha of land and water. About 71 percent of this area is non-Federal rural land – nearly 57 billion ha (USDA, 2013a). The non-Federal rural lands of the United States are predominantly rangeland (165 million ha), forest land (166 million ha), and cropland (146 million ha) with smaller areas composed of developed land, pastureland and water.

Rangeland is dominant in the western half of the United States in the Cold Deserts, Warm Deserts, Great Plains, South Central Semi-Arid Prairies and Western Cordillera ecoregions. Forest is the dominant land cover in the northwest, north central and eastern one-third of the country, in the Western Cordillera, Mixed Wood Plains, Ozark, Ouachita-Appalachian Forests ecoregions.

Cropland is the dominant land cover in the Central United States Plains, South Eastern United States Plains, the Mississippi Alluvial and Southeast United States coastal plains, Temperate Prairies, West Central Semi-Arid Prairies, South Central Semi-Arid Prairies and Mediterranean California ecoregions.

Cropland in the United States increased by about 1 million ha from 2007 to 2010, following a steady decline in area in the previous 25 years. These gains can be attributed to land withdrawn from the Conservation Reserve Program as grain prices reached near-record levels. This led to an increased threat to the soil resource from erosion and loss of terrestrial C. The National Resources Inventory tracks the cropland area for specific conservation measures such as terracing. However, according to the Conservation Technology Information Center there has not been a national survey of crop residue management practices since 2004; at that time 45.6 million ha was in conservation tillage out of a total of 112 million ha of cropland (Conservation Technology Information Center, 2014).

Total farmland in Canada increased from 1981 (65.9 million ha) to 2006 (67.6 million ha) (all values from AAFC, 2013). The largest agricultural region (54.7 million ha) occurs in the Temperate Prairies and West-Central Semi-Arid Prairies ecoregions in southern Alberta, Saskatchewan, and Manitoba. In 2006 approximately 29 million ha of farmland in this region was cropped, primarily to cereal grains, oilseeds and pulse crops, and 18 million ha was in pasture. Approximately 1.4 million ha was in tillage summer fallow, where land is fallowed during the growing season with weed suppression by one or more tillage operations. The area under tillage summer fallow has declined greatly from 5.3 million ha in 1991 and this decline has reduced soil degradation in this region.

Farmland in Ontario and Quebec (8.9 million ha) is concentrated in the Mixed Wood Plains ecozone. In 2006 5.6 million ha of this was cropped, primarily to forages, maize, cereal grains, and oilseeds (soybeans). The area of pastures in this region has declined from 1.7 million ha in 1991 to 1.1 million ha in 2006. Tillage practices in this region have also undergone a major shift, with the percentage of cropland under conventional tillage decreasing from 80 percent in 1991 to 50 percent in 2006, and conservation tillage and no-tillage increasing from 16 percent to 26 percent and from 4 percent to 24 percent respectively over the same period.

Farmland in British Columbia (2.8 million ha in 2006) is dominated by forage production and pasture dispersed through the Western Cordillera and Cold Desert ecoregions. Finally the Atlantic Highlands and Mixed Wood Plains ecoregions in Atlantic Canada have 1.1 million ha of farmland, dominantly in forages but with significant areas of potato production in New Brunswick and Prince Edward Island.

Overall the greatest shift in Canadian agriculture has been the adoption of no-till and reduced tillage systems. The 2011 Census of Agriculture (Statistics Canada, 2015) reports that of 29.6 million ha seeded in 2011, 16.7 million ha were in no-till and a further 7.2 million ha were in tillage systems that left most residue on the soil surface; only 5.6 million ha were in conventional tillage e.g. with most residue turned into the soil.

Wetlands are extensive in both Canada and the United States and are subject to considerable alteration by human activity. Bridgham *et al.* (2006) estimate that the historical area of wetlands in Canada of 150 million ha has been reduced to 130 million ha by land-use conversion. In the contiguous United States, the same study estimated a reduction from 90 million ha historically to current levels of 43 million ha.

The area of forested land in Canada is estimated at 348 million ha (Natural Resources Canada, 2012). Forest is the dominant land cover in the Northern Forests, Boreal Plain, Boreal Cordillera, Western Cordillera and Marine West Coast Forest ecoregions. Forest harvest activity has decreased in the period from 2004, and approximately 0.7 million ha were harvested in 2010 (Natural Resources Canada, 2012). Canada's National Forest Inventory shows that gross deforestation rates were typically in the order of 64 400 ha yr^{-1} circa 1990, and decreased to 44 800 ha yr^{-1} by 2010, corresponding to about 0.02 percent of Canada's forest area (Natural Resources Canada, 2012). Deforestation minus afforestation (according to UNFCCC definitions) amounts to a net loss of 35 000 ha yr^{-1}. Overall, there is a definite decrease in total deforestation rate from the 1990s to present, which is expected to continue in coming years, but at a lower rate of decrease (Masek *et al.*, 2011). The expansion of agriculture is the largest source of forest conversion, accounting for two thirds of gross deforestation. Urban and industrial development is the next largest driver at approximately 17 percent, followed by forestry at half that rate. The Boreal Plains ecozone spanning central Alberta, Saskatchewan, and Manitoba (generally termed the Prairie provinces) was the dominant location of deforestation over the 1990–2008 time period, contributing just under half the nation's deforestation for most years, largely due to agricultural conversion (Masek *et al.*, 2011).

The Tundra (233 million ha), Taiga (200 million ha) and Arctic Cordillera (21.7 million ha) ecoregions are vulnerable to effects of climate change. Permafrost soils in these regions are estimated to contain 39 percent of all organic C in Canada (Tarnocai and Bockheim 2011) and hence the interaction of soils and climate in these regions is a major concern.

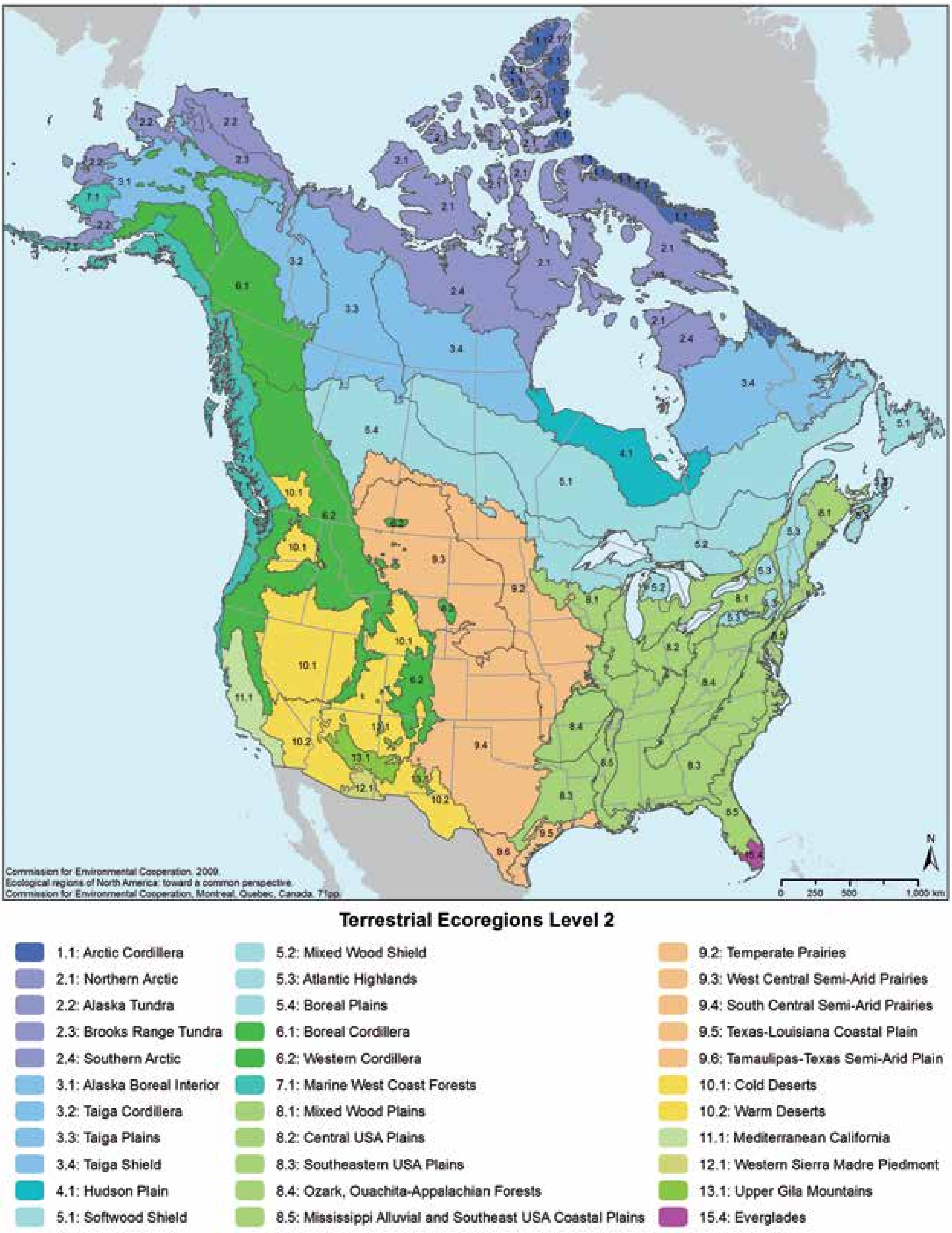


Figure 14.1 | Level II Ecological regions of North America.
Source: Commission for Environmental Cooperation, 1997.

14.3 | Soil threats

This section focuses on the status and trends for six threats to soil functioning: acidification, contamination, salinization, sealing/capping, compaction, and waterlogging. Four threats that are judged to be the most serious (erosion, nutrient imbalance, carbon change, and soil biodiversity) are discussed in greater detail in the following section.

14.3.1 | Soil acidification

While many native forest communities are well-adapted to strongly acidic (pH < 5.5) soil conditions, most managed agricultural and horticultural plants suffer from enhanced metal phytotoxicity (Al, Fe, Mn, etc.), reduced N and P availability, and decreased microbiological activity because soils have become acidified below their optimum range. Excessive soil acidification also poses environmental risks of enhanced surface water acidification, sediment losses due to loss of vegetation, and increased loadings of soluble metals into groundwater. Soil acidification is enhanced by a range of anthropogenic effects, including excessive inputs of acidic atmospheric deposition, intensive removal of aboveground biomass, and exposure of sulfidic materials by mining, construction, dredging, and other disturbances.

Acidic deposition from rain that has low pH and significant amounts of sulphate and nitrate contributes to base cation depletion and soil acidification in industrialized regions of the world (Meinz and Seip, 2004). These effects, however, have been documented only rarely in the United States. Coarse-textured soils in high-altitude forests that were originally low in pH and base saturation are particularly susceptible to degradation and loss of function. Finer-textured and more highly buffered soils are much more resistant to the negative effects of acidic deposition. The Clean Air Act Amendments of 1990 resulted in significant declines in sulphate emissions (http://nadp.sws.uiuc.edu/ntn/), although emissions of nitrogen oxides have remained elevated due to transportation and agricultural sources. Some lakes in the Adirondacks have shown significant improvement in water quality as a result of the Clean Air Act Amendments. Research by Driscoll *et al.* (2001) has shown, however, that tighter controls on atmospheric emissions will be needed if soil and stream chemistry in this region is to return to pre-industrial levels in a reasonable time range. Alternatively, aerial application of calcium sources such as wollastonite ($CaSiO_3$) to acidifed catchments can accelerate the return of base saturation to pre-industrial levels (Johnson *et al.*, 2014). However, the widespread applicability of this reclamation approach is limited. Localized, highly acidic deposition from heavy metal smelter and other industrial facilities also impacted large areas of land such as the Copper Basin in Tennessee, where many square miles of land were denuded by open air smelting of metal sulphide ores. Current concerns centre on the effects on soil fertility and acidification from the interaction between intensive biomass harvesting and acidic deposition on forest soils (Adams *et al.*, 2000).

Sulfidic materials (as defined by Soil Taxonomy (Soil Survey Staff, 1999)) are routinely exposed by mining, construction, and dredging activities. This exposure can rapidly lower the pH of local soils and water to < 4.0 via sulfuricization processes (Kittrick, Fanning and Hossner, 1982). While most active mining operations now isolate sulfidic materials from contact with groundwater and surface water via the application of appropriate acid-base accounting procedures (Skousen *et al.*, 2002), construction-related impacts have become increasingly common since the 1970s due to larger scale excavations and the construction industry's lack of recognition of risk (Fanning *et al.*, 2004; Orndorff and Daniels, 2004). Once exposed, sulfidic materials require large inputs of liming agents (e.g. ~ 31 Mg of agricultural lime per 1000 Mg material for 1 percent pyritic-S) to become properly neutralized and stabilized.

In Canada, the major risks associated with soil acidification occur in forested areas. As in the United States, areas most at risk from acidification are in regions dominated by coarse-textured soils that have low base cation weathering rates and that receive high levels of acid deposition (Ouimet at al., 2006). These areas of coarse-textured soils include much of the Softwood Shield and Mixed Wood Shield ecoregions and the southern, coastal parts of British Columbia (Aherne and Posch, 2013) in regions where glacial parent materials were derived from igneous rocks. Both the loss of essential base cations and the mobilization of metals such as Al and Mn can have adverse impacts on forest vegetation. It has been proposed that the ratio of base cations or Ca to Al (e.g. base cations (BC/Al or Ca/Al) in soil solution is a useful indicator of the potential risk to trees from soil acidification (Cronan and Grigal, 1995; Sverdrup and Warfvinge, 1993). Critical loads are also increasingly used to estimate the risk of soil acidification (Whitfield *et al.*, 2010).

Aherne and Posch (2013) estimated critical loads for acid deposition for upland forest soils in Canada (~2 600 000 km^2). They reported that in 2006, because of acid deposition levels, 4.5 percent of the mapped area (~100 000 km^2) was at risk based on a BC/Al ratio of 1.0; and 20.3 percent (~500 000 km^2) of the area was at risk based on a BC/Al ratio of 10.0. Exceedance of the critical load was primarily driven by elevated anthropogenic emissions from large point sources, such as the activities in the Athabasca Oil Sands region and in ore smelting near Sudbury, Ontario. In addition, exceedance in central and eastern Canada was associated with long-range (transboundary) air pollution and emissions from shipping along the St. Lawrence River.

In Canada, national emissions of SO_2 and NO_x decreased by 21 percent and 3 percent, respectively, between 2008 and 2010 (Canadian Council of Ministers of the Environment (CCME), 2103). This decrease reduces the risk of soil acidification in Canada. Emission reductions, however, vary by province. The bulk of the reduction in SO_2 and NO_x emissions has been in Ontario in central Canada. Minimal decreases or even increases in emissions, associated primarily with the oil and gas industry (CCME, 2013), have occurred in British Columbia and Alberta in western Canada. Currently, the risk of soil acidification caused by acid deposition is generally decreasing over much of eastern Canada but is unchanged or increasing in parts of western provinces, such as Alberta and British Columbia.

14.3.2 | Soil contamination

Soils can be compromised via industrial, mining, municipal, residential and agricultural activities. In North America, metals (Pb, Cd, Cr and As), salts (Na and K), pesticides (herbicides and insecticides), pathogens and nutrients (N and P) contaminate soils to varying degrees and with great spatial variation. There are also chemicals of emerging concern, including engineered nanoparticles, pharmaceuticals and personal care products. Perfluorinated compounds are also of concern: they occur in small concentrations but, because of their high reactivity or potential to be endocrine disrupting, they may pose significant risks to human health and the environment.

In the United States, there are thousands of organic- and metal-contaminated sites of varied scope and significance. To address, monitor and remediate the myriad of sites, the United States Environmental Protection Agency oversees the Superfund programme, which is charged with the clean-up of the nation's hazardous waste sites. This effort follows the National Priorities List (NPL), which defines the known releases or threatened releases of contaminants in the United States and its territories (EPA, 2014a). As of 29 September 2014, there are 1 322 final sites on the NPL with 1 163 having completed measures to address the contamination threat and an additional 49 proposed sites (Figure 14.2). In addition, there are vast areas of low-level soil contamination across the United States which are not monitored by the EPA.

Figure 14.2 | Map of Superfund sites in the contiguous United States Yellow indicates final EPA National Priorities List sites and red indicates proposed sites.
Source: EPA, 2014a.

In Canada, because it has a huge expanse of soil and a relatively small population, soil contamination in a spatial context is a relatively minor issue. However, the most agriculturally productive soils and greatest density of population and industry occur concomitantly along the narrow region close to the southern border. This is also the region where there is the greatest potential for soil contamination. In addition, the hinterland has widely dispersed petroleum and mineral resource industries that form hot spots of soil contamination (Doyle *et al.*, 2003).

More insidious is non- point- source, dispersed contamination (Chan *et al.*, 1986). For example, field crop soils surveyed throughout the Mixed Wood Plains ecoregions of southern Ontario in Canada showed elevated levels of Ba, Cd, Mo, Pb, Sb, Se, Nb, U and Zn, which were speculatively attributed to non- specific urban sources such as road dust (Sheppard *et al.*, 2009). Watmough and Hutchinson (2004) came to a similar conclusion about Pb in forest soils of Southern Ontario. Toxicity in soil from such sources, however, is a relatively remote possibility.

There is concern about soil contamination by agricultural activities, especially as farms increase in size and effectively become industrial point sources. For example, soils in areas of livestock facilities have been found to have metal levels that exceed Canadian soil quality guidelines (Sheppard and Sanipelli, 2012). Some of these metals came from livestock pharmaceuticals (e.g. Bi in teat dips). The contribution of livestock manures containing antibiotic residues to the development of antibiotic-resistant genes in the environment is a growing public concern. In a few cases, the naturally occurring, trace element bioavailability of some Canadian soils has resulted in food crops with amount of elements that exceed guideline concentrations. The most notable are spatially isolated cases of Cd in durum wheat and sunflowers (Grant *et al.*, 1998).

As industrialization and urbanization increase, concomitant with increased agricultural activities in decreasing land areas, the potential for soil contamination remains an important issue. Although soil contaminants in the United States and Canada are ubiquitous in areas close to human populations, the specific threats posed by these contaminants to human health and environmental quality are not well defined. There is a need for improvements in assessments of soil contamination to better protect human health and environmental quality and ensure food safety and security.

14.3.3 | Soil salinization

Soil salinization is a serious threat to the ecosystem services provided by the soil resource with regard to food and fibre production in many parts of North America. The movement and accumulation of salts that cause saline conditions in the soil are affected by the soil water balance. Processes such as climate shifts, improper irrigation and drainage, farming and management practices affect this balance. Soil salinity is a dynamic soil condition and can spread or become more severe in areas that are already saline, especially if the land is not managed properly.

These concerns are especially prevalent in the western portions of the United States (Figure 14.3) (Soil Survey Staff, 2014). Similar threats to food and fibre production are also associated with sodic conditions in the soil. In the United States, saline soils occupy approximately 2.2 million ha of cropland and another 31 million ha are at risk of becoming saline (USDA, 2011).

In Canada, a Risk of Soil Salinization (RSS) Indicator has been developed as part of the Agri-Environmental Indicators programme to assess the state and trend of the risk of dryland soil salinization on the Canadian Prairies as a result of changing land use and management practices. Two of the primary conditions required for dryland salinization - water deficits and an inherent salt content in the soil and/or groundwater - occur to a significant extent only in the Prairie region of Canada. The risk of salinization on other agricultural lands in Canada is negligible. The risk of soil sodicity is not assessed as part of this index and is not believed to be a major risk in western Canada. The RSS is derived by calculating a unit-less Salinity Risk Index (SRI) which considers a combination of factors that control or influence the salinization process (Wiebe, Eilers and Brierley, 2010).

In terms of the state of soil salinity in Canada, approximately 1 million ha of surface soils on the Prairies are affected by moderate to severe soil salinity (Wiebe, Eilers and Brierley, 2006). In 2011, 85 percent of the land area in the agricultural region of the Canadian Prairies was rated as having a very low risk of salinization (Figure 14.4).

From 1981 to 2011, the trend has been a 19 percent increase in the land area in the Very Low and Low risk classes. Over the same 30-year period, the land area in the Moderate, High and Very High risk classes decreased from 15 percent to 8 percent. These improvements were largely attributed to the reduction in tillage summer fallow, mentioned above, and to a 4.8 million ha increase of permanent cover (a 14 percent increase from 1981 to 2011). A reduction in risk has been observed in all Prairie provinces. The greatest decline was recorded in Saskatchewan, where the area of summer fallow decreased by more than 5 million ha and the area of permanent cover increased by more than 3 million ha. Changes in land use and management practices have reduced the risk of salinization and indicate a trend towards improved soil health and agri-environmental sustainability.

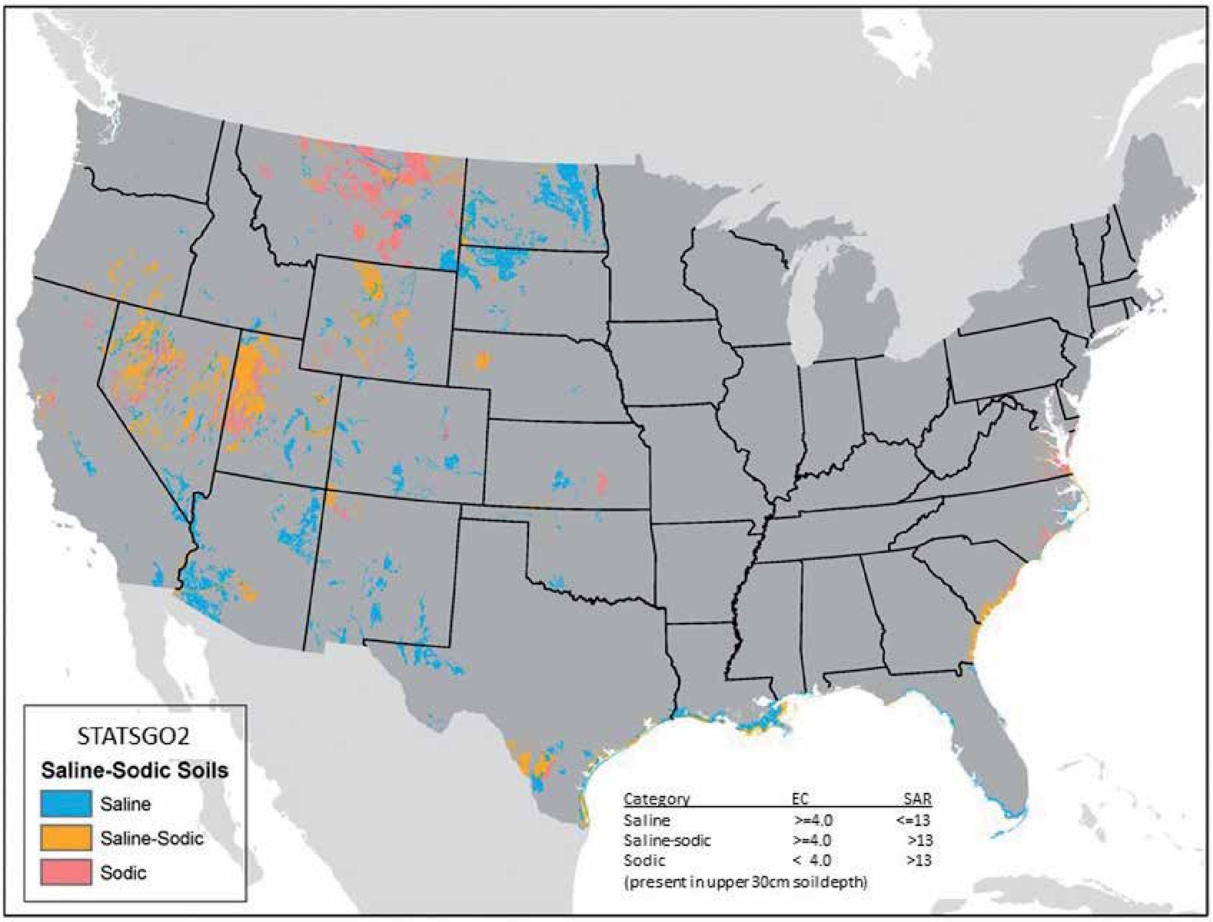


Figure 14.3 | Areas in United States threatened by salinization and sodification.
Source: NRCS[1]

1 http://www.nrcs.usda.gov/wps/portal/nrcs/site/national/home/

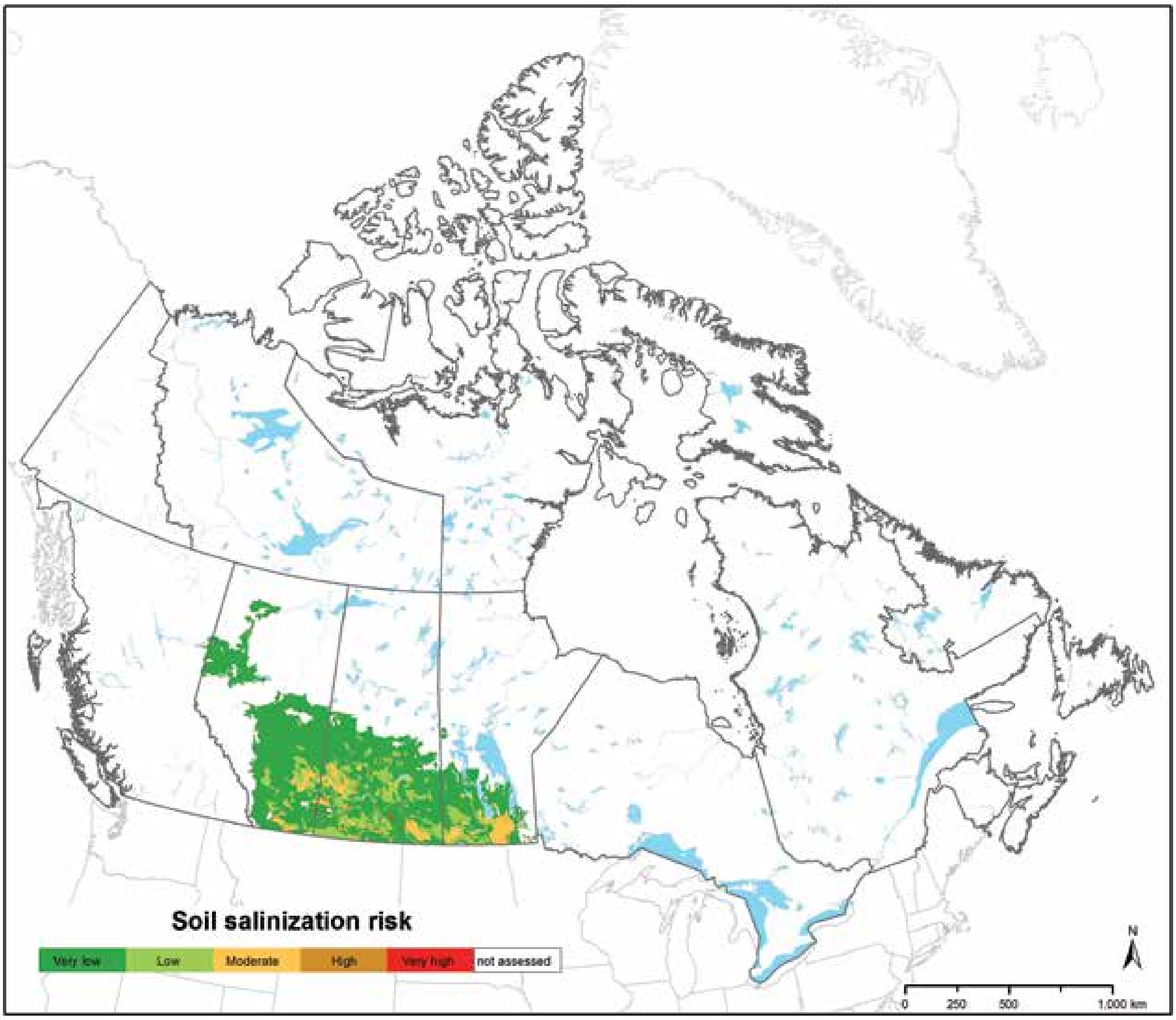


Figure 14.4 | Risk of soil salinization in Canada 2011.
Source: Clearwater et al., 2015.

14.3.4 | Soil sealing/capping

The population of North America is approximately 5 percent of the world population (340 million) and it has been growing at a rate of 0.9 percent a year, during the last decade. In the United States, the threat of loss of soil due to soil sealing and capping consequent on expansion of settlements and infrastructure is significant and has been steadily increasing since 1982, as documented by the USDA Natural Resources Conservation Service's National Resources Inventory Program (NRI) (USDA, 2013a). Between 1982 and 2007, it is estimated that 16.5 million ha of land were developed for urban or transportation uses. By 2007, the United States had an estimated total of 45 million ha developed into urban uses. Of the newly developed land, 41 percent was previously forest land, 27 percent was cropland, 17 percent was pasture, and 13 percent was rangeland. In regard to land categories, 35 percent of the land in the United States that was developed into urban uses during the period of 1982 to 2007 was classified as prime farmland. Prime farmland is land that has the best combination of soil physical and chemical characteristics for producing food, feed, forage, fibre and oilseed crops and has the soil quality, growing season, and moisture supply needed to economically produce sustained high yields of crops when treated and managed at high levels. Prime farmland is also the most economically viable land to develop as it typically has the lowest degree of limitations for conversion into urban development. This increases the pressure on land with the best soil. Using the 2007 NRI information as a base, the trajectory of prime farmland (cropland portion) conversion and the loss of potential food production is immense over the next 25 years. With this rate of loss, the United States will lose the equivalent of 10M metric tonnes of corn in 2022 due to sealing/capping activities.

The drivers for soil sealing in Canada are very similar to those in the United States. The growth of metropolitan centers has been particularly rapid since the 1990s in areas surrounding Toronto, Kitchener-Waterloo, Ottawa, and Vancouver and, most recently, in areas surrounding urban centers in Saskatchewan and Alberta. Between 1996 and 2006, urban land increased by more than 10 percent nationally. It now totals nearly 0.13 million ha (Statistics Canada, 2009), or 0.25 percent of the total land area in Canada. Given land-cover data extrapolated from a suburban United States city (specifically, San Jose with 59 percent impervious surface by area (Xiao *et al.*, 2013) and the fact that suburbs now make up the majority of Canada's metropolitan population (Gordon and Janzen, 2013), it is estimated that more than 1 300 km^2 of soil was capped through urban expansion between the 1990s and 2000s. It is noteworthy that this expansion has occurred largely on highly productive soils (Francis *et al.*, 2012; Hofmann, Filoso and Schofield, 2005). Major interregional highway construction and expansion (e.g. highway twinning) have also been ongoing, particularly in Ontario, British Columbia, and Alberta. Total road length (in two lane equivalents) in Canada increased by more than 17 percent between 1990 and 2009 (Transport Canada, 2012; United States Department of Transportation, 2014), and, assuming a conservative average road width of 10 m, roads now cover more than 10 000 km^2. However, it is important to note that a portion of this impervious area is also included in the estimated urban area described above.

In the past few decades, areas used for agriculture and forest harvest have decreased (Natural Resources Canada, 2014; Francis *et al.*, 2012; Hofmann *et al.*, 2005). Thus, it is assumed that sealing as a result of new construction for agriculture service and forest industrial roads is relatively minor. The overall existing extent of unpaved roads in Canada that serve the needs of these industries is substantial.

14.3.5 | Soil compaction

Soil compaction is an acknowledged threat to the ability of the soil resource to provide a wide range of essential ecosystem services, including food and fibre production and maintenance of good water quality. Compaction decreases the water infiltration capacity of the soil, increases runoff and erosion, reduces plant growth, and reduces the penetration, size, and distribution of roots. It restricts water and air movement in the soil. It also causes nutrient stresses and slow seedling emergence. Soils in North America that are most susceptible to the threat of compaction are located in managed agricultural and forested regions.

The most common cause of soil compaction (or the formation of hardpans) is agricultural traffic, including tractors, harvesting equipment and implement wheels on moist soils where soil moisture is at or above field capacity. Wheeled traffic compacts soil aggregates, in some cases destroying the aggregates completely, which results in a dense soil with few large pores and limited aeration. Compaction can reduce yields up to 50 percent in some areas, depending upon the depth of compaction and its severity (Wolkowski and Lowery, 2008).

The type and condition of a soil have an effect on the potential of compaction to occur. Soils low in organic matter tend to be more susceptible to compaction because their ability to form strong aggregates is decreased. Soils high in clay content compact more easily because clay particles adhere to water, which makes it easier for them to move against each other. In the United States, Ultisols in the Coastal Southeast United States Coastal Plains ecozone are especially susceptible to compaction due to their highly weathered state and low organic carbon levels (Simoes *et al.*, 2009).

In Canada, there has not been a national-scale assessment of compaction since McBride, Joosse and Wall, (2000). Generally, fine-textured soils in the Mixed Wood Plains ecozone under cropping systems with a high potential for soil structure degradation and compaction (e.g. those for the production of maize, soybeans, vegetable and root crops) were judged to have the highest risk of compaction. In the period from 1981 to 1996, the area of fine-textured soils under high-risk cropping practices grew substantially (e.g. by as much as 61 percent in Ontario). This area expansion of potentially compaction-inducing cropping practices has continued to the present day. Agricultural soils in western Canada and the United States are generally believed to have a lower risk for soil compaction. The exception is irrigated soil in this region, where the wetter soil conditions can contribute to higher compaction levels. In addition, adoption of conservation tillage, which limits compaction, has also been lower on irrigated land.

Soil compaction also occurs in forestry operations throughout Canada and the United States, especially where roads and landings have been constructed. Generally, finer-textured soils are at the greatest risk, but the amount of organic matter in coarser textured soils can form a strong negative relationship with the potential for compaction (Krzic *et al.*, 2004). Compaction has been shown to reduce regeneration of species such as aspen (Populus tremuloides Michx.) and white spruce (Picea glauca [Moench] Voss) (Kabzems, 2012), but little is known about its general impact on soil functions.

14.3.6 | Waterlogging and wetlands

The threat of waterlogging to the sustainable use of soil resources is difficult to assess because waterlogging is not considered a threat in all cases. Wetland areas that are nearly or permanently waterlogged provide many positive benefits to the environment. Wetlands are some of the most biologically diverse habitats on earth and are of great benefit to many species of wildlife. They also act as a filter, trap sediments, improve water quality, are a carbon sink, and reduce peaks of floodwater runoff. Until 2014, the Wetland Reserve Program in the United States was a voluntary programme offered to landowners to protect, restore, and enhance wetlands on their property. Nearly 1 million ha has been enrolled in this programme (USDA, 2014).

Much of the drainage of wetlands in North America has been concentrated on freshwater mineral wetlands (Bridgham *et al.*, 2006), which are wetlands dominated by Gleysolic soils rather than organic soils. Bridgham *et al.* (2006) estimate that both the United States and Canada have experienced over 50 percent conversion of this class of wetlands (e.g. a reduction from 36 million ha to 16 million ha in Canada and from 76 million ha to 31 million ha in the United States).

In North America, the primary driver of large-scale waterlogging on non-wetland soils is flooding due to dam construction for hydroelectric power, to flood control measures and to mining activities (Maynard *et al.*, 2014). This includes both upstream flooding associated with dam construction and high spring water levels and downstream flooding associated with controlled releases during high-water periods and for hydroelectric power generation during the winter.

Another indirect driver is deforestation, which can reduce infiltration and/or evapotranspiration. It has been proposed that reduced evapotranspiration in northern Alberta contributes to a significant rise in the water table when deforestation is coupled with wet climatic periods (Carrera-Hernandez *et al.*, 2011).

Concern about waterlogging has been growing due to the substantial increase in the frequency and severity of extreme precipitation events in recent years in some regions (Brimelow *et al.*, 2014), even though this may not indicate an overall increase in mean annual precipitation for most regions. Waterlogging related to all of the above causes is exacerbated by increased precipitation.

14.4 | Major soil threats

Four threats to soil functions were selected as major threats and are covered in more detail in this section. The main criterion for their selection was the area of land affected by these threats – all four of these threats operate in most agricultural (and many non-agricultural) landscapes, whereas the threats covered in Section 14.3 tend to be more locally focused. Data on these four threats for Canada are covered in more detail in the case study in the following section (14.5). The present section focuses on North American-scale drivers and specific results for the United States of America.

14.4.1 | Soil erosion

Soil erosion in the United States and Canada accelerated after the arrival of European settlers, who cleared extensive areas for agriculture and subsequently ploughed and overgrazed the land (Montgomery, 2008). Soils rapidly degraded and erosion increased as settlement spread from east to west. In the United States, erosion was greatest on the east coast in the early 1800s, in the mid-south during the early 1900s, and in the Great Plains during the 'Dust Bowl' era in the 1930s. Some badly degraded lands were abandoned and then reverted to secondary growth forests, a process that slowed erosion rates. Wind erosion was very significant in Prairie Canada during the 1930s. Soils that were badly degraded due to wind erosion were subsequently stabilized and converted to permanent pasture.

Agricultural mechanization, commercial nitrogen availability, and federal policies encouraging maximum crop production led to cash-crop intensification throughout the middle of the 20th century in both the United States and Canada. Forage-based rotations were shortened or eliminated, field sizes were increased by the removal of hedgerows and fences, and tillage intensity remained high. As a result, the potential for soil erosion increased during this period. In the late 20th and early 21st centuries higher yielding varieties and improved herbicide technology supporting the adoption of conservation tillage helped reduce the potential for water and wind erosion. Federal farm programmes in the late 20th and early 21st centuries had both favourable and unfavourable impacts on soil erosion rates. In Canada, the most significant cropping change was the major reduction in summer fallow (e.g. the practice of leaving land fallow for one growing season and suppressing weed growth by one or more tillage events) in the two Prairie ecoregions in Canada. This change substantially reduced the risk of wind erosion.

In the United States, the National Resources Inventory (NRI) has reported statistically robust estimates of water (sheet and rill) erosion and wind erosion on privately owned cropland, since 1982 at five year intervals (USDA, 2013a), based on a wide monitoring network and an assumed historic average climate for each location. The estimated decrease in sheet and rill erosion between 1982 and 2002 was 39 percent, and that between 1982 and 2010 was 41 percent. In the same periods, wind erosion decreased by 41 percent and 46 percent, respectively.

In 2010, the most intense sheet and rill erosion was in the Temperate Prairie and Mixed Wood Plains ecoregions of the Midwest United States, in the adjacent area of the South-eastern United States Plains ecoregion and in the Palouse (Cold Desert ecoregions in the state of Washington). These areas and the West-Central Semi-Arid Prairies ecoregion also had the highest wind erosion rates.

If 2010 NRI estimates of sheet and rill erosion plus wind erosion are averaged across all United States cropland, the average annual rate is about 10 Mg ha^{-1}, and 57 percent of this is due to sheet and rill erosion (USDA, 2013a). Tolerable annual soil erosion rates ('T') used in the United States typically range from 2 to 11 Mg ha^{-1}, depending upon the soil type. The criteria used to calculate T have been widely criticized (Johnson, 1987) and the overall average soil loss rate is one order of magnitude greater than estimated soil renewal rates, which are less than 1 Mg ha^{-1} per year (Alexander, 1988; Montgomery, 2007). In addition, erosion varies spatially and may greatly exceed T at any specific NRI point, due to the soil, slope, management and climate at that location.

The state of soil erosion in Canada and the drivers of change are covered in more detail in Section 14.5 of this chapter. Soil erosion on undisturbed forested land in Canada is generally believed to be low (Maynard *et al.*, 2014) and no national estimates exist for rates of erosion on disturbed forest land.

Not all forms of erosion are considered in the national estimates and this leads to considerable overall uncertainty in estimates. In the United States, the NRI erosion assessment does not include ephemeral gully or tillage erosion. Tillage erosion is within-field soil redistribution by tillage implements, which has been extensively documented in both the United States and Canada. Both of these processes may result in degradation rates comparable to those from wind and water erosion. The best estimate of soil erosion rates must include estimates of these processes. When these two processes are included, average United States cropland soil erosion rates exceed published soil development rates by more than one order of magnitude. In the future, an increased frequency of extreme rainfall events due to climate change will likely increase the water soil erosion threat in many parts of the United States and Canada.

In Canada, the national Agri-Environmental Indicators programme does include an assessment of tillage erosion, which is known to be of equal or greater significance than wind and water erosion on some landscapes. Gully erosion is not included in the Canadian monitoring system, but its incidence in Canada is believed to be limited. Although accelerated erosion associated with forest harvest is a concern, there are no recent national-level surveys of its incidence in Canada.

14.4.2 | Nutrient imbalance

Many regions of North America have experienced and continue to experience nutrient applications in excess of plant requirements. These surpluses lead to elevated levels of N and P in soils, which cause a range of environmental problems and are a source of considerable societal concern throughout North America.

The greatest issue with nutrient imbalance in North America is the impact of elevated N and P levels in soil from past and present agricultural activities on water quality. The linkage of elevated soil N and P levels to water quality problems ranges from algal blooms due to eutrophication in Lake Winnipeg in Manitoba (Schindler, Hecky and McCullough, 2012), at the northern edge of the agricultural zone, to the seasonal hypoxia in the shallow coastal waters of the Louisiana shelf in the northern Gulf of Mexico, at the southern end of the agricultural zone (Alexander *et al.*, 2008).

Estimates on excess nutrient levels presented by Foley *et al*. (2010, Supplementary Information Maps S6e and S6f) show that excess application of N continues in many regions of North America whereas little excess application of P occurs. Excess N application of between 60 to 100 kg ha^{-1} occurs in much of the Temperate Prairie and Mixed Wood Plains ecoregions in both the United States and Canada, throughout the Mississippi River valley, and in pockets in the Southeast United States Coastal Plains ecoregion. Hence over-application of N is an on-going issue whereas elevated P levels may be largely due to historical over-application.

The linkage between agricultural practices and N and P loads in waterways has been shown by many studies. For example, recent studies on the Missouri River (Brown, Sprague and Dupree 2011) and the entire Mississippi River basin (Alexander *et al.*, 2008) using the Spatially Referenced Regressions On Watershed Attributes (SPARROW) model by United States Geological Survey (2014) researchers found that the majority of both total N and total P in these waterways is from agricultural land. Specifically, 52 percent of total N reaching the Gulf of Mexico was from maize-and soybean-producing land with a further 14 percent from all other crops in the basin. Some 37 percent of total P was from rangeland/pasture land with a further 25 percent from maize- and soybean-producing land. Considerable regional variations occur. For example, in the western sections of the Missouri River basin where cattle grazing is the dominant land use, as much as 34 percent of total N was from manure whereas in the Mississippi River basin as a whole, only 5 percent of total N was from rangeland and pasture land.

The relationship between soil properties, management, and particular N and P fractions is complex (Sharpley and Wang, 2014; Harmel *et al.*, 2006). Harmel *et al.* (2006) examined the relationship between soil and site attributes and N and P fractions in nutrient loads from watersheds in 15 United States states and two provinces of Canada. Particulate N and P loss contributed, on average, three times as much as dissolved forms to loads, indicating the overriding effect of soil erosion and transport on N and P loads. Median particulate N loads were greater in areas of conventional tillage (which experience higher erosion rates on average) and lower in areas of conservation tillage and no-till land, although no differences were observed for particulate P. There was a weak relationship between soil test P and all forms of P load, again indicating the importance of existing or legacy soil P content. Dissolved N and P loads were highest in areas of no-till land. The build-up of P at the soil surface in no-till systems was also implicated in the increase since 1995 of dissolved P load in the Maumee River system (which drains into Lake Erie), although the response of this system to management changes was very complex (Sharpley and Wang, 2014).

Excess soil nitrogen can also leach from the soil as nitrate. This threat is most severe in situations where shallow aquifers are used as potable water sources in humid or sub-humid climates with coarse-textured soils utilized for intensive agricultural production. Examples in Canada include Prince Edward Island, the Abbotsford aquifer, BC and Kings County, Nova Scotia. The indicator of risk of water contamination by nitrogen (IROWC-N) in the Canadian Agri-Environmental Indicators doubled from ~5 mg N L^{-1} in 1981 to a value approaching the Canadian drinking water guideline for nitrate (10 mg N L^{-1}) in 2006 in the Atlantic Highlands and in the Canadian portions of the Mixed Wood Highlands (AAFC, 2013).

Country-specific information for nutrient imbalance in Canada is given in Section 14.5 of this report.

N levels in excess of plant requirements in soils are also linked to other environmental issues, especially the enhanced release of the potent greenhouse gas, N_2O, from soils. In both Canada and the United States, agriculture accounts for 6 to 7 percent of total GHG emissions (EPA, 2014b; Environment Canada, 2013b). Emissions of N_2O from agricultural soils account for 75 percent of the agricultural total in the United States and 65 percent in Canada. The highest N_2O emissions occur under anaerobic conditions and hence are intimately linked to changes in waterlogging in agricultural landscapes. Health concerns are also linked to forms of N in groundwater and fertilizer application, although the direct link to human health can be difficult to ascertain (Manassaram, Backer and Moll, 2006).

14.4.3 | Soil organic carbon change

Assessment of soil organic carbon (SOC) in the United States and Canada currently includes a number of approaches aimed at either directly measuring or modelling SOC change over time. Both countries utilize national-scale modelling of SOC change for GHG emissions inventories and reporting.

Models of SOC changes currently show increases in the United States. These increases in SOC are primarily due to less intensive agriculture (McLauchlan, Hobbie and Post, 2006) and reduced tillage intensity (West *et al.*, 2008). Models generally predict that converting areas from native vegetation (e.g. prairies and native forest) to cropland and forest plantations results in decreased SOM; however, in some cases, high-productivity crops and SOC-conserving management may enhance SOC. For instance, Ogle *et al.* (2010) estimated that SOC in United States croplands increased by 14.6 and 17.5 Tg yr^{-1} during 1990-1995 and 1995-2000, respectively, primarily due to reductions in tillage intensity.

For agricultural soils in Canada, national-level modelling indicates that improvements in farm management have resulted in a dramatic shift from stable SOC levels (e.g. additions are equal to losses) during the mid-1980s, to a situation where the majority of cropland had increasing SOC levels in the mid-1990s through to 2011 (discussed in more detail in Section 14.5 of this chapter).

There is currently high uncertainty associated with SOC models (Ogle *et al.*, 2010) such as CENTURY (National Resource Ecology Laboratory, 2007), the SOC model most commonly used in the United States and Canada to predict SOC change. Improving model parameterization and adding additional SOC measurements over time could help reduce uncertainty. This information could be used to better calibrate, estimate and potentially reduce model uncertainty as well as to directly track SOC change (Jandl *et al.*, 2014). National-level modelling is also limited by the lack of data on SOC stocks and change deeper in the soil profile. Models and researchers generally consider subsurface SOC as a relatively stable pool. Little direct evidence has been provided, however, that SOC stabilized under previous conditions will remain stable with changing conditions, such as with climate change. Some studies have shown that previously stable SOC may be rapidly converted to CO_2 (Fontaine *et al.*, 2007; Fang *et al.*, 2005).

There are currently two major United States-wide efforts that sample soil over time, the Forest Inventory and Analysis (FIA) by Gillespie, 1999, covering United States forests, and the Rapid C assessment (USDA, 2013b), which includes all vegetated areas. The FIA samples soil to a maximum depth of 20 cm, so its utility in monitoring whole-profile SOC over time is limited (Waltman *et al.*, 2010; Jandl *et al.*, 2014). The Rapid C assessment (USDA, 2013b) samples soil profiles to 100 cm depth. It is currently uncertain what depth is required to truly understand SOC and potential changes in SOC (Harrison, Footen and Strahm, 2011). Some studies have shown that results of monitoring SOC change vs. land management depend more on maximum soil sampling depth than on treatments (Liebig *et al.*, 2005; Khan *et al.*, 2007; Harrison, Footen and Strahm, 2011).

Field data on losses of SOC in Canadian soils after conversion from native land to cropland, and for different tillage, crop rotation and fertilizer management practices were compiled from a total of 62 studies by VandenBygaart *et al.* (2003). They demonstrated that 24 ± 6 percent of the SOC was lost after native land was converted to agricultural land. In the past two decades, no-till (NT) increased the storage of SOC in Mollisols (Chernozems) of the two Prairie ecoregions by 2.9 ± 1.3 Mg ha–1; however, in the moister soils of central and eastern Canada, conversion to NT did not increase SOC. More recent studies using meta-analyses (Congreves *et al.*, 2014) of long-term agricultural management effects on SOC in Ontario indicate trends towards higher SOC with NT than under conventional tillage practises. Crop rotation was found to lead to higher SOC than when continuous maize was grown, and the application of N fertilizer led to an increase in SOC compared to when no N fertilizer was applied.

Carbon change in managed forests (232 million ha) in Canada is assessed as part of the National Inventory, primarily using the CBM-CFS model (Kurz *et al.*, 2009). Carbon emissions from the dead organic matter and soil pools are lumped together. Results showed a small increase in emissions from 2000-2007 due to the short-term effect of past disturbances, especially insect infestation. However, the values decreased from 2008 until 2012 and have returned to long-term levels. Freshwater mineral wetland soils in the Prairie ecoregions are also important carbon reservoirs, and approximately 70 percent of these were impacted by agricultural activities in 2005 (Bartzen *et al.*, 2010); however there is no regional estimate of associated carbon change.

Permafrost soils are classified as Cryosols in Canada and cover 2.5 million km^2; they are estimated to contain 39 percent of all organic carbon in Canadian soils (Tarnocai and Bockheim, 2011). The greatest driver of change in these soils is climate change. The IPCC 5th Assessment Report (Clais *et al.*, 2013) states that there is high confidence that reductions in permafrost due to warming will cause thawing of some currently frozen carbon, but there is low confidence on the magnitude of CO_2 and CH_4 emissions to the atmosphere due to the complexity of the biogeochemical processes involved.

14.4.4 | Soil biodiversity

Soil biodiversity refers to the myriad of organisms living in the soil, ranging from the smallest microorganisms (e.g. bacteria, archaea and fungi) to soil invertebrates. Up until the advent of molecular biology and its use in soil science in the 1990s, the assessment of soil biodiversity was done using morphological methods. Methods to study soil biodiversity are improving constantly with the application of sequencing technologies, complementing the morphological assessments.

As a result of our paucity of knowledge, assessing threats to soil biodiversity is very difficult. There is no reference baseline data for these organisms, nor do scientists have the ability to estimate the true numbers of soil organisms, particularly microorganisms. Many organisms have not been described and overall we need a better understanding of the biogeography of soil organisms in North America (Nunez and Dickie, 2014). Advances, however, have been made. For example, Taylor *et al.* (2014) completed a comprehensive survey of fungi in black spruce (Picea mariana) sites in Alaska and recorded 1 002 taxa in this system. They reported a fungus: plant ratio of 17:1.

However, as soil organisms are so intricately tied to aboveground plant species, threats to plant species such as habitat loss are also liable to affect soil organisms (Wardle *et al.*, 2004). Evidence of this exists. For example, it has been shown that removal of logging residues from harvested forest sites is one of the major threats to forest fungi and insects (Berch, Morris and Malcolm, 2011). Similarly, invasive plant species and their mutualistic microbes pose a threat to native mutualist communities (Nunez and Dickie, 2014). Invasive alien species are entering North America with increasing frequency due to the growing volume of trade, the broadening of trading partners, and the increases in travel and tourism that accompany globalization (Environment Canada, 2013a).

Regional or national-level programmes that monitor soil biodiversity are lacking in North America. In Europe, Gardi *et al.* (2013) used modelling of data from 20 experts to demonstrate that the main pressures on soil biodiversity were intensive land exploitation (such as agriculture intensification using tillage, crop rotations, and additions of pesticides and herbicides) and changes in land use (such as the reduction in forests that have the highest soil biodiversity and the increase in soil sealing) combined with decreasing amounts of soil organic matter and increasing numbers of invasive species.

These modelling results are increasingly being supported by field studies using emerging identification and community analysis techniques. Crowther *et al.* (2014) assessed deforestation effects on soil biodiversity at eleven sites in the United States and found that forest removal was generally associated with reductions in fungal and bacterial microbial biomass and increases in diversity of taxa. The magnitude of differences due to deforestation varied drastically between sites and was best explained by differences in soil texture: the effects were greatest in coarse-textured soils and least in fine-textured soils. Crowther *et al.* (2014) suggest that the relationship between soil biodiversity and soil texture offers the potential for mapping regional and national patterns of the susceptibility of total (fungal, bacterial, and archaeal) soil biomass to changes in vegetation (see Figure 4 in Crowther *et al.*, 2014). Studies based on a meta-analysis of crop rotation found that adding one or more crops in rotation increased microbial biomass carbon by 20.7 percent and microbial biomass nitrogen by 26.1 percent, indicating the sensitivity of soil microbes to the quantity and biochemistry of crop inputs (McDaniel, Tiemann and Grandy, 2014). Other studies, such as those by Wagg *et al.* (2014), are explicitly examining the relationship between decreasing biodiversity in the soil community and changes in ecosystem functions, such as carbon sequestration and nitrogen and phosphorus leaching.

In addition, climate change is considered a threat to soil biodiversity. Unfortunately, there is a scarcity of data on this issue. To determine this threat we need to predict soil biodiversity patterns, which is not possible with our current knowledge. However, there are indications that soil biodiversity will be reduced by climate change. Studies in the Canadian Arctic show that extreme ecosystems contain many unique organisms that may become extinct with permafrost melting (Vincent *et al.*, 2009). Wildfire events, which are another threat to soil biodiversity, are also predicted to increase because of climate change (Krawchuk *et al.*, 2009).

At present, no national or regional assessment on loss of biodiversity can be made for North America, but there are signs that such an assessment may be possible in the future. The Global Soil Biodiversity Initiative (GSBI) was launched in 2011 and provides a welcome platform for the coordination of research in this area (GSBI, 2014).

14.5 | Case study: Canada

As discussed in the Introduction (14.1), there are no existing regional maps for threats to soil functions for North America. In this section the maps produced under the Canadian Agri-Environmental Indicators programme (Clearwater *et al.*, 2015) for wind and water erosion, SOC change and nutrient imbalance (residual soil N and P-source risk classes) are presented so that the linkage between drivers and threats to soil functions can be illustrated. The maps focus on agricultural impacts on soil functions. No comparable products exist for other soil threats.

The major changes to drivers in Canada have been discussed in detail in Section 14.3 above. There are distinct differences across the main ecoregions of Canada that experience concentrated human impact. The main drivers can be summarized as:

- Intensification of agriculture e.g. reduced pasture area, increased area of maize and soybean production, higher fertilizer inputs in the Mixed Wood Plains ecoregion in southern Ontario and Quebec and in the agricultural areas of eastern Canada e.g. in New Brunswick, Nova Scotia, and Prince Edward Island.
- Significant reductions in the area of tillage summerfallow in the Temperate Plains and West-Central Semi-Arid ecoregions.
- Widespread adoption of conservation tillage practices in most cropping systems in Canada.

14.5.1 | Water and wind erosion

In Canada, the Soil Erosion Risk Indicator was used as part of the Agri-Environmental Indicators programme to assess the risk of soil erosion from the combined effects of wind, water and tillage on cultivated agricultural lands (Figure 14.5). This indicator and its component indicators for wind, water and tillage erosion reflect the characteristics of the climate, soil and topography and correspond to changes in farming practices over the 30-year period from 1981 to 2011. Wind and water erosion are the primary focus of this summary. For details on calculation of the indicators, see Li *et al.* (2008), McConkey, Li and Black, (2008), and Huang and Lobb (2013).

The erosion indicator calculation estimates the rate of soil loss. These values are reported in five classes. Areas in the very low risk class are considered capable of sustaining long-term crop production and maintaining agri-environmental health under current conditions. The other four classes represent the degrees of risk of unsustainable conditions that call for soil conservation practices to support crop production over the long term and to reduce risk to soil quality.

The risk of soil erosion on Canadian cropland has steadily declined between 1981 and 2011. The majority of this change occurred between 1991 and 2006. In 2011, 61 percent of cropland area was in the very low risk class overall, a considerable improvement over 1981 when only 29 percent was in this risk class. This decrease in water and wind erosion risk was most pronounced in the Temperate Prairie and West-Central Semi-Arid ecoregions in Alberta and Saskatchewan (Figure 14.5 and 14.6). Much of the improvement in erosion risk is from reductions due to the reduction in tillage summer fallow. A second driver is the increased adoption of direct seeding and conservation tillage, which is largely responsible for the decrease in tillage intensity and soil erosion.

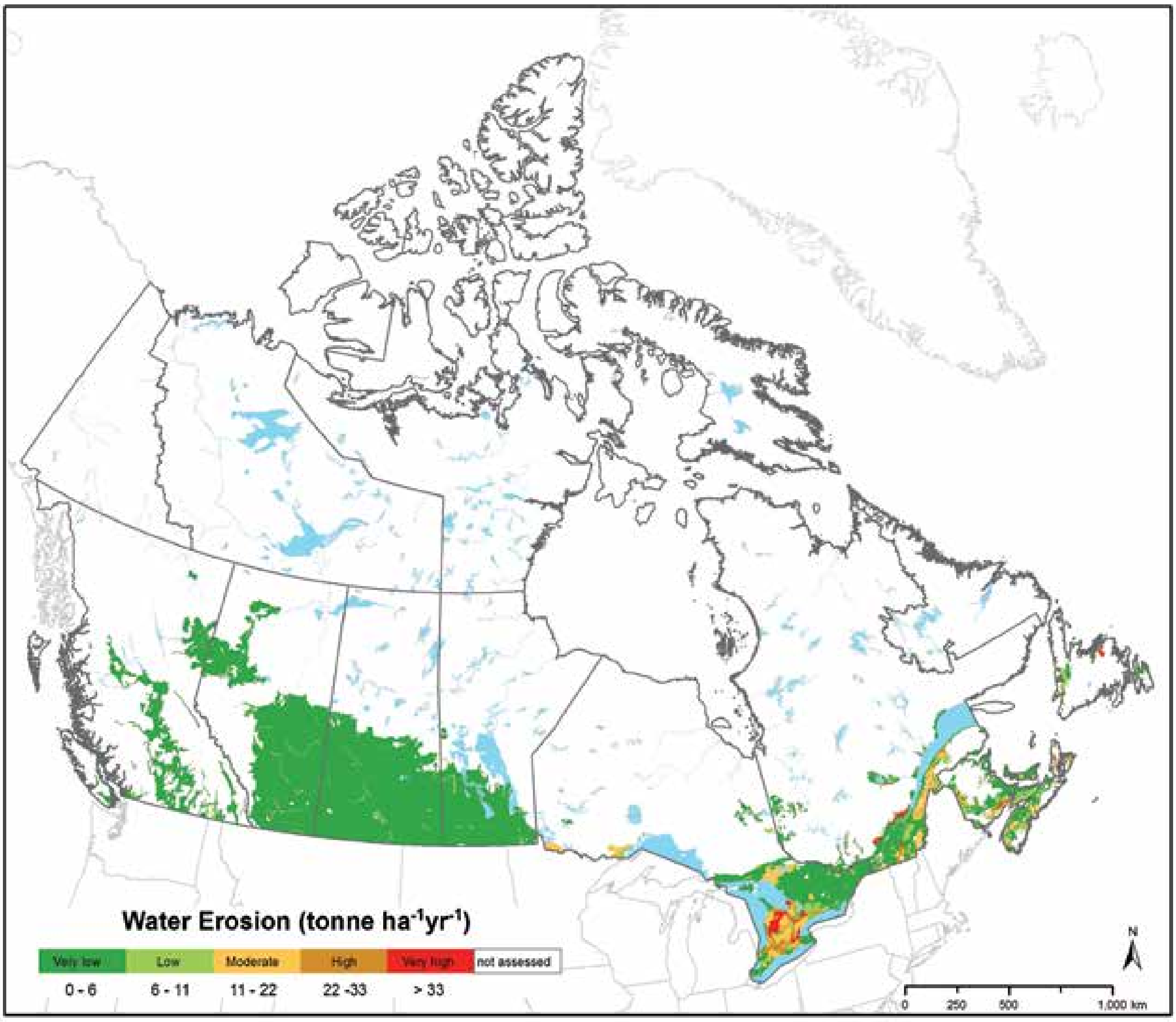


Figure 14.5 | Risk of water erosion in Canada 2011.
Source:Clearwater et al., 2015.

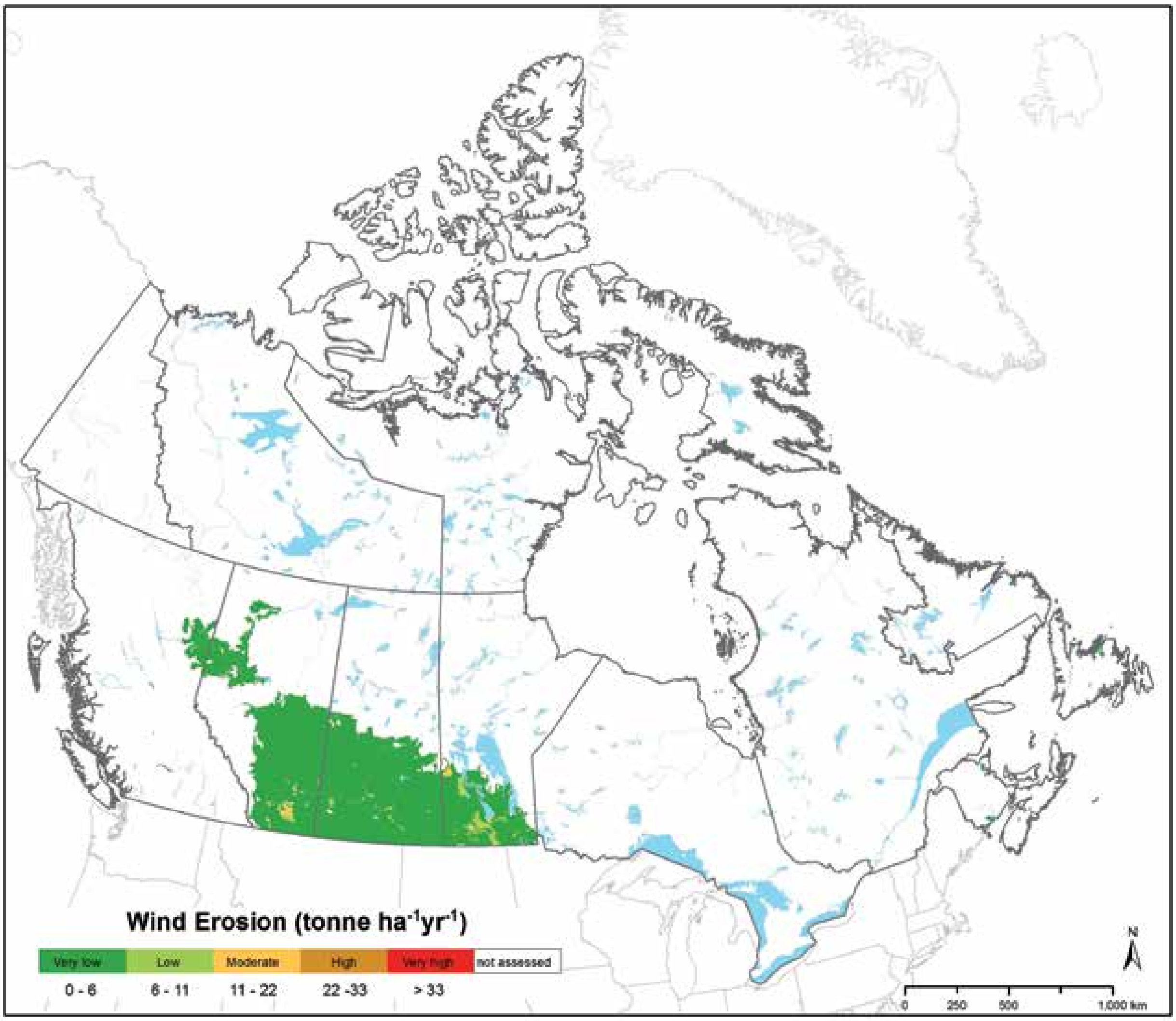


Figure 14.6 | Risk of wind erosion in Canada 2011.
Source: Clearwater et al., 2015.

Of the cropping systems across Canada, the risk of soil erosion by water is greatest under potato production in central and eastern Canada. In these areas there is intensive tillage and little opportunity to reduce the intensity through conservation tillage practices (Figure 14.5). The cropping system with the next greatest risk of erosion is the production of maize and soybeans under conventional tillage; however, there is a significant opportunity to reduce this erosion risk with conservation tillage. Of all soil landscapes across Canada, the risk of soil erosion by water is greatest in areas with maximum slopes of 10 percent or more, especially those located in central and eastern Canada where the risk of water erosion is inherently high due to climate (Figure 14.5).

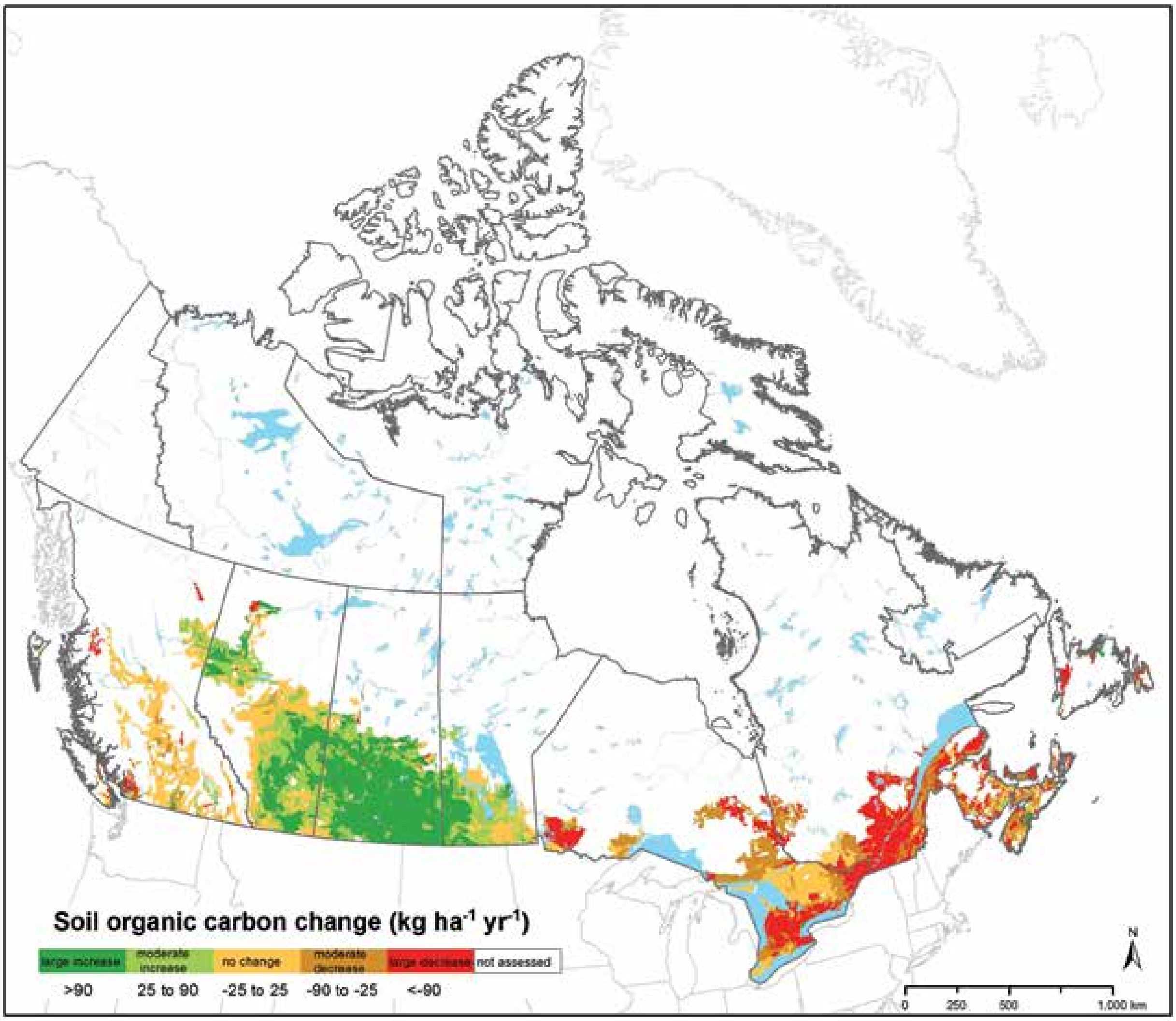


Figure 14.7 | Soil organic carbon change in Canada 201.
Source: Clearwater et al., 2015.

14.5.2 | Soil organic carbon change

The soil organic carbon (SOC) change indicator used in the national Agri-Environmental Indicators programme assesses how organic C levels are changing over time in Canadian agricultural soils. The indicator is based on the method used for the Canadian National Inventory Report (Environment Canada, 2014). The indicator uses the Century model (NREL, 2007) to predict the rate of change of organic C content in Canada's agricultural soils due to the effects of land management change since 1951. These include changes in tillage and summer fallow frequency, and change between annual crops and perennial hay or pasture. It includes land use changes such as clearing forests for agriculture or breaking native grass for cropland, but does not include the loss of C from the above-ground forest biomass.

No changes in SOC were assumed if there were no indicated changes in land use or land management. The SOC change indicator does not consider soil erosion.

The SOC change indicator results are presented (Figure 14.7) as the percentage of total cropland that falls into each of five SOC change classes expressed in kilograms per ha per year (kg ha^{-1} yr^{-1}). Negative values represent a loss of SOC from the soil and positive values represent a gain of SOC.

For the Boreal Plains ecoregions from Ontario eastward, there was an overall loss of SOC from 1981 to 2011 due to the reduction in the area of hayland and pasture and the corresponding increase in the area of annual crops (Figure 14.7). This shift in land use reflects a reduction in the demand for feed associated with the declining cattle populations in those provinces. The losses in Ontario and Quebec have been offset to a limited degree as a result of the adoption of conservation tillage. However, conservation tillage has not been implemented to the same extent in provinces in eastern Canada due to their cooler and wetter climatic conditions.

The two agricultural ecoregions in the Prairie provinces have seen major increases in SOC over time due to reductions in tillage intensity and in summer fallow. These changes are responsible for the overall net gain in SOC in Canada (see Section 14.4.3 above).

14.5.4 | Nutrient imbalance

The assessment of nutrient imbalance in the Agri-Environmental Indicators programme focuses on N and P, and assesses both the N and P status of soils and the risk to water quality associated with the soil stores. The risk to water quality involves coupling hydrological and climate data with the land surface information for each region. This section focuses on the N and P status of soils.

The residual soil nitrogen (RSN) indicator used in the National Agri-Environmental Indicators program provides an estimate of the amount of inorganic N that is left in the soil at the end of the growing season which may be susceptible to loss (Drury *et al.*, 2007, 2010). The RSN indicator is estimated as the yearly difference between the total N input to agricultural soils and the output in harvested crops and gaseous losses including ammonia, nitrous oxide and dinitrogen. The major categories of N inputs into soil include fertilizer addition, manure application, biological nitrogen fixation by leguminous crops and free-living bacteria, and atmospheric wet and dry deposition. Nitrogen outputs include N removal in the harvested crop and gaseous N emissions via ammonia volatilization (NH_3), nitrification (N_2O) and denitrification (N_2O, N_2).

The RSN on Canadian agricultural land has steadily increased from a low of 9.4 kg N ha^{-1} in 1981 to a maximum of 25.3 kg N ha^{-1} in 2001 (a year where many regions experienced drought conditions and were unable to use the applied N). The latest figure is 23.6 kg N ha^{-1} in 2011, the most recent census year. On a national basis, N inputs have almost doubled over the 30 years from 44.4 kg N ha^{-1} to 80.8 kg N ha^{-1} whereas N outputs have only increased by 63 percent from 35 kg N ha^{-1} in 1981 to 57.2 kg N ha^{-1} in 2011.

The RSN map (Figure 14.8) for Canadian farmland in 2011 generally shows that there are high or very high residual N contents in farmland in many areas across Canada. Considerable change has occurred since 1981; for example, the Temperate Prairies in Manitoba were primarily in the very low and low risk classes in 1981 whereas the great majority of the province is now in a very high risk class in 2011 (Figure 4.4). The Mixed Wood Plains in central and eastern Canada are also currently predominantly in the very high risk group, which again a considerable increase since 1981.

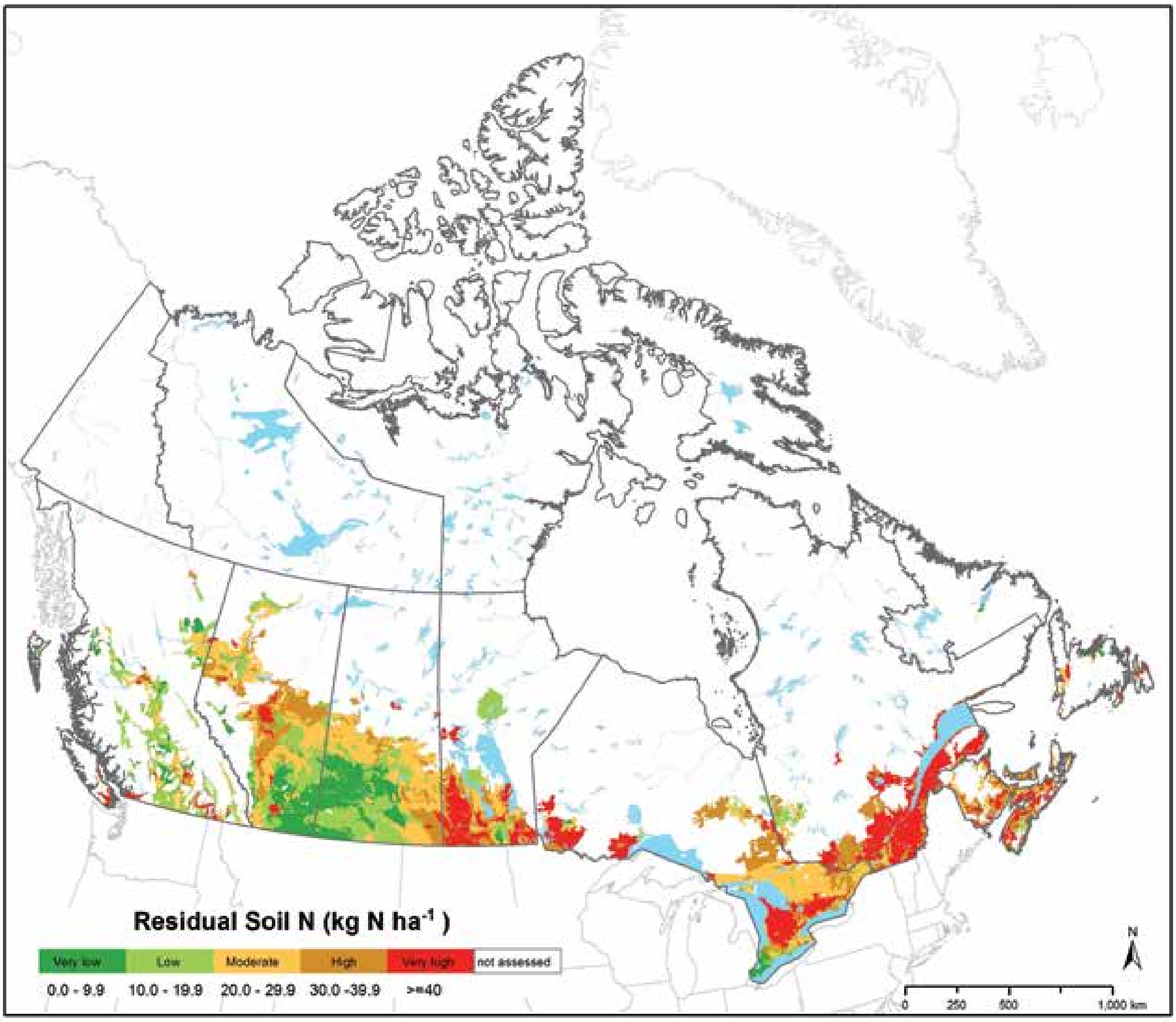


Figure 14.8 | Residual soil N in Canada 2011.
Source: Clearwater et al., 2015.

Changes to management practices are required especially in the more humid regions in the Canada (notably Ontario, Quebec and eastern Canada) to reduce N losses from soils, to increase fertilizer use efficiency and to better synchronize N application with crop N demand. Further, the use of cover crops, especially in years with reduced yields, may help to reduce N losses from soil.

The IROWC-P Indicator was developed to assess the status and trends over time for the risk of surface water contamination by P from Canadian agricultural land and is reported for agricultural watersheds (van Bochove *et al.*, 2010). The initial stage in calculating IROWC-P involves the estimation of the annual amount of dissolved P that may potentially be released from agricultural soils (P source). P source is estimated as a function of cumulative P additions and removals (P-balance) over a 35-year period up to 2011 and the resulting degree of soil P saturation.

There has generally been an increasing trend in the P-source levels in the surface of agricultural soils in Canada since 1976 as intensified agricultural practices have resulted in the application of P in excess of crop uptake (also called positive annual P balance) and have therefore increased soil P saturation. In 2011, very high concentrations of P (more than 4 mg of P per kg, or >4 mg P kg^{-1}) at risk for release by storm events were located in regions where the agricultural production has been historically intensive and where soils have reached high P saturation values. High risk of water- contamination by P occurs around Abbotsford, British Columbia in the Marine West Coast Forest ecoregions, in the Temperate Prairies around Lethbridge in Alberta, and in portions of southern Saskatchewan and Manitoba (Figure 14.9). Intensive livestock operations near Abbotsford and

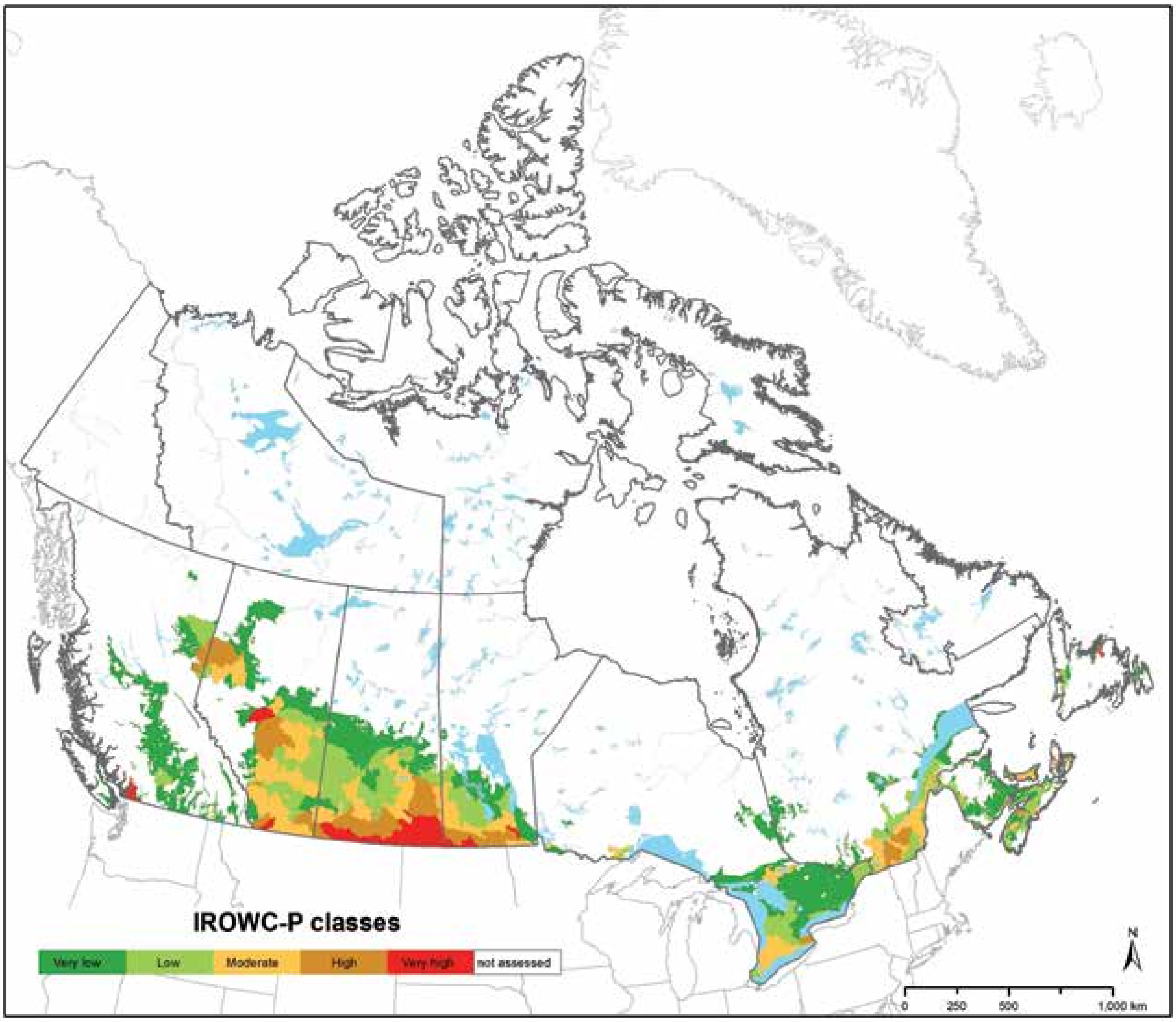


Figure 14.9 | Indicator of risk of water contamination by phosphorus (IROWC-P) in Canada in 2011.
Source: Clearwater et al., 2015.

Lethbridge are major local sources of high P loadings. The Mixed Wood Shield ecoregion of central Ontario and the Mixed Wood Plains ecoregion in Quebec, New Brunswick and Prince Edward Island are also dominated by very high or high P source risk.

Implications of soil threats for soil functions

Clearly the national assessment of the threats to soil functions shows a distinct separation between the agricultural systems of the Prairie provinces and those of Ontario, Quebec, and Atlantic Canada. Both the state and the trend of soil change in the Prairie provinces are generally positive, especially in soil erosion (notably wind erosion) and carbon change. The greatest risk in this region lies in the high residual nutrient levels in areas like Manitoba and in the possible contribution of these high residual levels to eutrophication in lakes in this region (Schindler, Hecky and McCullough, 2012).

The level of the threats in the Mixed Wood Plains of central and eastern Canada is very different. There are generally high or very high levels of threat for soil organic carbon change, erosion by water, and nutrient imbalance. The well-integrated drainage system and higher precipitation levels than in the Prairie provinces lead to significant sediment and nutrient delivery to waterways. There is thus a risk of serious impact of agricultural land management on water quality (Clearwater *et al.*, 2015). The combined effects of soil organic carbon loss and water erosion presumably also reduce services and products delivered by the soil, but these have been poorly documented in this region.

14.6 | Conclusions and recommendations

Overall there has been significant progress made in reducing threats to soil functions in North America. Threats from acidification and contamination have been reduced due to the imposition of a stronger regulatory framework. The greatest change in cropping practices - reduced tillage - has largely come about through adoption by individual producers supported by government and private sector extension agents.

However, major areas of concern remain. Erosion rates are still above what are believed to be tolerable levels in the Temperate Prairies ecoregion of the United States and throughout the Mixed Wood Plains ecoregion of Canada and the United States. Transport of soil-derived N and P to waterways is a major problem, and excess application of N continues throughout much of the cropland in the United States and in central and eastern Canada.

Although a wide variety of best management practices for optimum nutrient application and erosion control have been developed and promoted, the problems of erosion and nutrient imbalance persist.

Salinization, contamination, and acidification affect smaller areas in North America, and in the case of the latter two threats the current regulatory framework limits the expansion of the affected area. Waterlogging is little studied in North America, and we recommend that future reports include assessments of loss of wetlands as an important metric for sustainable management in this and other regions.

The loss of agricultural land to soil sealing is not perceived as a major issue in North America. However, the paucity of data on this threat needs to be addressed for a more informed assessment to be made.

Changes in carbon stocks in North America have been extensively modeled as part of national reporting programs on greenhouse gas emissions, but only in a few landscapes are the models adequately supported by field observations of SOC change. The greatest uncertainty surrounding SOC change lies in the response of carbon in permafrost soils to climate change in northern Canada and Alaska and improved monitoring of this response is essential.

Like the SOC models, the agri-environmental indicators approach used in the Canadian case study allows estimation of the change in threats to soil function over time. However, several criticisms can be made of the approach. First, there is a lack of ongoing monitoring of important soil physical, chemical and biological properties at relevant scales through time and hence it is difficult to assess the performance of the models that underlie the indicators. Second, the interaction between indicators (for example, between erosion and carbon) is not considered, and this can bias the assessment of soil change. Third, it is difficult to assess the variability associated with modelled results and therefore to evaluate overall the confidence that can be placed in the results. Overall there is a need to revise the models in light of new scientific advances and to develop and refine scientifically credible programs to assess model performance.

The greatest uncertainty overall in our knowledge about the threats to soil functions lies in our limited understanding of the changes in soil biodiversity in the past and present and the implications of these changes for sustainable soil management.

Based on the above finding, a provisional assessment is made of the status and trend of the 10 soil threats in order of importance for the region. At the same time an indication is given of the reliability of these estimates (Table 14.1)

Stable	Variable	Improving	Deteriorating	Evidence and consensus is **low**	Evidence and consensus are **limited**	**Adequate** high-quality evidence and **high level** of consensus
=	↗↙	↗	↙			

Table 14.1 | Summary of soil threats status, trends and uncertainties in North America

Threat to soil function	Summary	Condition and Trend					Confidence	
		Very poor	Poor	Fair	Good	Very good	In condition	In trend
Soil erosion	Reduced tillage and improved residue management have lowered erosion rates in regions such as the Great Plains in Canada but water erosion rates continue to be too high in the northern Mid-West of the U.S. and agricultural areas of central and Atlantic Canada.			↗				
Nutrient imbalance	Excess application of fertilizers in many regions causes significant degradation of surface water quality and increased emissions of nitrous oxide to the atmosphere. Contamination of surface water is strongly linked to high erosion rates, and occurs in the same regions (northern mid- west U.S., Mississippi River Basin, and agricultural regions of central Canada).		↙					
Organic carbon change	The majority of cropland in the U.S. and Canada has shown improvements in SOC stores due to the wide-spread adoption of conservation agriculture (i.e., reduced tillage and improved reside management). There is a lack of field validation sites to support the national-level modelling results. Loss of SOC from northern and Arctic soils due to climate change is a major concern.			↗				

Loss of soil biodiversity	The extent of loss of soil biodiversity due to human impact is largely unknown in North America. The effects of increasing agricultural chemical use, especially pesticides, use on biodiversity is a major public concern. Known level of carbon loss suggests similar loss in biodiversity.			↗↙				
Compaction	Compaction continues to be a low-level issue, especially in regions with texture-contrast (Luvisol, Alfisol, Ultsol) soils. The regional-scale impact of compaction on plant growth is largely unknown.			=				
Sealing and land take	Substantial expansion of housing and infrastructure in areas of high quality farmland continues in both countries but is (incorrectly) not perceived as a concern. Neither country has reliable data on sealing and land take.			↙				
Salinization and sodification	Salinization is believed to be increasing in parts of the northern Great Plains in the U.S.A. but the risk of salinization is decreasing in western Canada.			↗↙				
Contamination	Although many legacy contamination sites exist, improved regulatory systems in both countries has limited the creation of new areas of contamination. Large-scale land disturbance due to resource extraction activities continues to be a significant issue.				↗			
Soil acidification	Trans-national environmental legislation has significantly reduced soil acidification in forested areas of eastern and central North America. Localized areas of acidification in agricultural land managed through lime application.				↗			
Waterlogging	Waterlogging is not believed to be a significant threat in North America. Localized flooding has occurred due to a wider amplitude of precipitation events in the past decade. Loss of wetlands is a more significant threat in North America.				=			

Acknowledgements

Jenny Sutherland of NRCS in Lincoln, Nebraska very capably edited the material on the soil threats. Tim Martin of Agriculture and Agri-Food Canada provided access to the updated Canadian agri-environmental indicators material used in this chapter. The chapter was reviewed by D. Burton, F. Walley, C. Rice, E. Gregorich, C. van Kessel, T. Moore, and F. Larney and their suggestions for revisions are gratefully acknowledged.

References

AAFC. 2013. *Environmental Sustainability of Canadian Agriculture: Agri-Environmental Indicator Report Series*. Report No. 3. Agriculture and Agri-Food Canada.

Adams, M.B., Burger, J.A., Jenkins, A.B. & Zelazny, L.W. 2000. Impact of harvesting and atmospheric pollution on nutrient depletion of eastern USA hardwood forests. *For. Ecol. Manage.*, 138: 301–319.

Aherne, J. & Posch, M. 2013. Impacts of nitrogen and sulphur deposition on forest ecosystem services in Canada. *Current Opinion in Environmental Sustainability*, 5: 108–115.

Alexander, E.B. 1988. Rates of soil formation: Implications for soil-loss tolerance. *Soil Sci.*, 145: 37-45.

Alexander, R.B., Smith, R.A., Schwarz, G.E., Boyer, E.W., Nolan, J.V. & Brakebill, J.W. 2008. Differences in phosphorus and nitrogen delivery to the Gulf of Mexico from the Mississippi River basin. *Environmental Science and Technology*, 42: 822–830.

Bartzen, B.A., Dufour, K.W., Clark, R.G. & Caswell, F.D. 2010. Trends in agricultural impact and recovery of wetlands in prairie Canada. *Ecological Applications*, 20: 525-538.

Berch, S.M., Morris, D. & Malcolm, J. 2011. Intensive forest biomass harvesting and biodiversity in Canada: A summary of relevant issues. *The For. Chron.*, 87: 478–487.

Bridgham, S., Megonigal, J.P., Keller, J.K., Bliss, N.B. & Trettin, C. 2006. The carbon balance of North American wetlands. *Wetlands*, 20: 605-615.

Brimelow, J., Stewart, R., Hanesiak, J., Kochtubajda, B., Szeto, K. & Bonsal, B. 2014. Characterization and assessment of the devastating natural hazards across the Canadian Prairie Provinces from 2009 to 2011. *Nat. Hazards*, 73: 761–785.

Brown, J.B., Sprague, L.A. & Dupree, J.A. 2011. Nutrient sources and transport in the Missouri River basin, with emphasis on the effects of irrigation and reservoirs. J.Amer. *Water* Res.Assoc., 47: 1030-1065.

Carrera-Hernandez, J.J., Mendoza, C.A., Devito, K.J., Petrone, R.M. & Smerdon, B.D. 2011. Effects of aspen harvesting on groundwater recharge and water table dynamics in a subhumid climate. *Water Resour. Res.*, 47: 18.

CCME. 2013. *Progress report on the Canada-wide acid rain strategy for post-2000*. Canadian Council of Ministers of the Environment. (Also available at http://www.ccme.ca/en/resources/air/acid_rain.html).

Clais, P., Sabine, C., Bala, G., Bopp, L., Brovkin, V., Canadell, J., Chhabra, A., DeFries, R., Galloway, G., Helmann, M., Jones, C., Le Quere, C., Myeni, R.B., Piao, S. & Thornton, P. 2013. Carbon and other Biogeochemical Cycles. *In Stocker T.F. ed. Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, pp. 465-570. UK, Cambridge & USA, New York, NY, Cambridge University Press.

Clearwater, R.L., Martin, T., Hoppe, T. & Kalff, S. 2015. Environmental sustainability of Canadian agriculture: Agri-environmental indicator report series. Report No. 4. Ottawa, ON, Agriculture and Agri-Food Canada. (DRAFT)

Commission for Environmental Cooperation. 1997. *Ecological regions of North America. Toward a Common Perspective*. (Also available at http://www.cec.org/Page.asp?PageID=122&ContentID=1329).

Congreves, K.A., Smith, J.M., Németh, D.D., Hooker, D.C. & Van Eerd L.L. 2014. Soil organic carbon and land use: Processes and potential in Ontario's long-term agro-ecosystem research sites. *Can. J. Soil Sci.*, 94(3): 317-336.

Conservation Technology Information Center. 2014. *Crop Residue Management Survey*. (Also available at http://www.conservationinformation.org/CRM/.)

Cronan, C.S. & Grigal, D.F. 1995. Use of calcium/aluminum ratios as indicators of stress in forest ecosystems. *Journal of Environmental Quality*, 24: 209–226.

Crowther, T.W., Maynard, D.S., Leff, J.W., Oldfield, E.E., McCulley, R.L., Fierer, N. & Bradford, M.M. 2014. Predicting the responsiveness of soil biodiversity to deforestation: A cross-biome study. *Global Change Biology*, 20: 2983–2994.

Chan, W.H., Tang, A.J.S., Chung, D.H.S. & Lusis, M.A. 1986. Concentration and deposition of trace metals in Ontario. *Water Air Soil Pollut.*, 29: 373-389.

Doyle, P.J., Gutzman, D.W., Bird, G.A., Sheppard, M.I., Sheppard, S.C. & Hrebenyk, D. 2003. An ecological risk assessment of air emissions of trace metals from copper and zinc production facilities. Human *Ecol. Risk Assess.*, 9(2): 607-636.

Driscoll, C.T., Lawerence, G.B., Bulger, A.J., Butler, T.J., Cronan, C.S., Eager, C., Lambert, K.F., Likens, G.E., Stoddard, J.L., & Weathers, K.C. 2001. Acidic deposition in the northeastern United States: Sources and inputs, ecosystem effects, and management strategies. *Bioscience*, 51: 180-198.

Drury, C.F., Yang, J.Y., De Jong, R., Huffman, T., Yang, X.M. & Reid, K. 2010. Residual Soil Nitrogen. In W. Eilers, R. MacKay, L. Graham, & A. Lefebvre, eds. *Environmental Sustainability of Canadian Agriculture: Agr-Environmental Indicator Report Series - Report No. 3.*, pp. 74-80. Ottawa, Ont, Agriculture and Agri-Food Canada.

Drury, C.F., Yang, J.Y., De Jong, R., Yang, X.M., Huffman, E., Kirkwood, V. & Reid, K. 2007. Residual soil nitrogen indicator for Canada. Can. *J. Soil Sci.*, 87: 166-177.

Environment Canada. 2013a. *National Inventory Report*. 1990-2001. Part 3. (Also available at http://unfccc.int/national_reports/annex_i_ghg_inventories/national_inventories_submissions/items/7383.php)

Environment Canada. 2013b. *Invasive Alien Species Program*. (Also available at http://www.ec.gc.ca/nature/default.asp?lang=En&n=B 008265C-1#_Toc 315269282.)

Environment Canada. 2014. *National Inventory Report: Greenhouse Gas Sources and Sinks in Canada 1990-2012. Submission to the United Nations Framework Convention on Climate Change*. (Also available at http://unfccc.int/national_reports/annex_i_ghg_inventories/national_inventories_submissions/items/8108.php).

EPA. 2014a. *National Priorities List* (NPL). (Also available at http://www.epa.gov/superfund/sites/npl.)

EPA. 2014b. *Overview of Greenhouse Gas Emissions*. (Also available at http://epa.gov/climatechange/ghgemissions/gases/n2o.html.)

Fang, C., Smith, P., Moncrieff, J.B. & Smith, J.U. 2005. Similar response of labile and resistant SOM pools to changes in temperature. *Nature*, 433: 57–59.

Fanning, D., Rabenhorst, M., Coppock, C., Daniels, W. & Orndorff, Z. 2004. Upland active acid sulfate soils from construction of new Stafford County, Virginia, USA, airport. *Aust. J. Soil Res.*, 42: 527–536.

Foley,J.A., Ramankutty,N., Brauman, K.A. Cassidy E.S., Gerber, J.S., Johnston, M., Mueller, N.D., O'Connell, C., Ray, D.K., West, P.C., Balzer, C., Bennett, E.M., Carpenter S.R., Hill,J., Monfreda C., S. Polasky, S., Rockström, J. Sheehan, J., Siebert, S., Tilman, D. & Zaks, D.P.M. 2011. Solutions for a cultivated planet. *Nature*, 478: 337-342.

Fontaine, S., Barot, S., Barre, P., Bdioui, N., Mary, B. & Rumpel, C. 2007. Stability of organic carbon in deep soil layers controlled by fresh carbon supply. *Nature*, 450: 277–281.

Francis, C.A., Hansen, T.E., Fox, A.A., Hesje, P.J., Nelson, H.E., Lawseth, A.E. & English, A. 2012. Farmland conversion to non-agricultural uses in the USA and Canada: Current impacts and concerns for the future. *International Journal of Agricultural Sustainability*, 10: 8–24.

Gardi, C., Jeffery, S. & Saltelli, A. 2013. An estimate of potential threats levels to soil biodiversity in EU. *Glob. Ch. Biol.*, 19: 1538–1548.

Gillespie, A.J.R. 1999. Rationale for a National Annual Forest Inventory Program. J. *Forestry*, 97: 16-20.

Gordon, D.L.A. & Janzen, M. 2013. Suburban nation? Estimating the size of Canada's suburban population. *Journal of Architectural and Planning Research*, 30: 197–220.

Grant, C.A., Buckley, W.T., Bailey, L.D. & Selles, F. 1998. Cadmium accumulation in crops. *Can. J. Plant Sci.*, 78: 1–17.

GSBI. 2014. *Official web site*. Global Soil Biodiversity Initiative. (Also available at http://www.globalsoilbiodiversity.org/)

Harmel, D., Potter, S., Casebolt, P., Reckhow, K., Green, C. & Haney, R. 2006. Compilation of measured nutrient load data for agricultural land uses in the United States. *J. Amer. Water Res. Assoc.*, 42: 1163- 1178.

Harrison, R.B., Footen, P.W. & Strahm, B.D. 2011. Deep soil horizons: Contribution and importance to soil C pools and in assessing whole-ecosystem response to management and global change. *Forest Science*, 57: 67-76.

Hofmann, N., Filoso, G. & Schofield, M. 2005. The loss of dependable agricultural land in Canada. *Statistics Canada's Rural and Small Town Canada Analysis Bulletin*, 6(1).

Huang, Q. & Lobb, D.A. 2013. *Uncertainty analysis for the soil erosion risk indicators. Technical supplement*. Canada, Ottawa, ON, Agriculture and Agri-Food Canada.

Jandl, R., Rodeghiero, M., Martinez, C., Cotrufo, M.F., Bampa, F., van Wesemael, B., Harrison, R.B., Guerrini, I.A., Richter, D.D., Rustad, L., Lorenz, K., Chabbi, A. & Miglietta, F. 2014. Current status, uncertainty and future needs in soil organic carbon monitoring. *Science of the Total Environment*, 468: 376-383.

Johnson, C.E., Driscoll, C.T., Blum, J.D., Fahey, T.J., & Battles, J.J. 2014. Soil chemical dynamics after calcium silicate addition to a Northern Hardwood forest. *Soil Sci. Soc. Am. J.*, 78(4): 1458-1468.

Johnson, L.C. 1987. Soil loss tolerance: Fact or myth? *J. Soil & Water Conservation*, 42: 155–160.

Kabzems, R. 2012. Aspen and white spruce productivity is reduced by organic matter removal and soil compaction. *The Forestry Chronicle*, 88: 306–316.

Khan, S.A., Mulvaney, R.L., Ellsworth, T.R. & Boast, C.W. 2007. The myth of nitrogen fertilization for soil carbon sequestration. *J. Environ. Qual.*, 36: 1821–1832

Kittrick, J.A., Fanning, D.S. & Hossner, L.R. 1982. *Acid sulfate weathering*. Special Pub. No. 10. USA, Madison, WI, Soil Sci. Soc. Am. 234 pp.

Krawchuk, M.A., Moritz, M.A., Parisien, M-A., Van Dorn, J. & Hayhoe, K. 2009. Global pyrogeography: The current and future distribution of wildfire. PLOS One, 4: e 5102.

Krzic, M., Bulmer, C.E., Teste, F., Dampier, L. & Rahman, S. 2004. Soil properties influencing compactability of forest soils in British Columbia. *Canadian Journal of Soil Science*, 84: 219–226.

Kurz, W.A., Dymond, C.C., White, T.M., Stinson, G., Shaw, C.H., Rampley, G.J., Smyth, C., Simpson, B.N., Neilson, E.T. & Trofymow, J.A. 2009. CBM-CFS 3: A model of carbon-dynamics in forestry and landuse change implementing IPCC standards. *Ecological Modelling*, 220: 480–504.

Li, S., McConkey, B.G., Black, M.W. & Lobb, D.A. 2008. *Soil erosion risk indicators: Technical supplement*. Canada, Ottawa, ON, Agriculture and Agri-Food Canada.

Liebig, M.S., Johnson, H.A., Hanson, J.D. & Frank, A.B. 2005. Soil carbon under switchgrass stands and cultivated cropland. *Biomass Bioenergy*, 28:347–354.

Manassaram, D.M., Backer, L.C. & Moll, D.M. 2006. A review of nitrates in drinking water: Maternal exposure and adverse reproductive developmental outcomes. *Environ. Health Persp.*, 114: 320-327.

Masek, J.G., Cohen, W.B., Leckie, D**.,** Wulder, M.A**.,** Vargas, R**.,** de Jong, B**.,** Healey, S**.,** Law, B**.,** Birdsey, R**.,** Houghton, R.A**,** Mildrexler, D**.,** Goward, S**. &** Smith, W.B. 2011. Recent rates of forest harvest and conversion in North America. *J. Geophys. Res.*, 116: G 00K 03.

Maynard, D.G., Pare, D., Thiffault, E., Lafleur, B., Hogg, K.E. & Kishchuk, B. 2014. How do natural disturbances and human activities affect soils and tree nutrition and growth in the Canadian boreal forest? *Environ. Rev.*, 22: 161–178.

McBride, R.A., Joosse, P.J. & Wall, G. 2000. Indicator: Risk of soil compaction. In T. McRae, C.A.S. Smith & L.J. Gregorich, eds. *Environmental Sustainability of Canadian Agriculture: Report of the Agri Environmental Indicator Project. A Summary*, p. 16. Ottawa, Ontario, Agriculture and Agri-Food of Canada.

McConkey, B.G., Li, S. & Black, M.W. 2008. *Wind Erosion Risk Indicator (WindERI.) Methodology in Soil erosion risk indicators: Technical supplement*. Ottawa, ON, Agriculture and Agri- Food Canada.

McDaniel, M.D., Tiemann, L.K. & Grandy, A.S. 2014. Does agricultural crop diversity enhance soil microbial biomass and organic matter dynamics? A meta-analysis. *Ecological Applications*, 24: 560–570.

McLauchlan, K.K., Hobbie, S.E. & Post, W.M. 2006. Conversion from agriculture to grassland builds soil organic matter on decadal timescales. *Ecological Applications*, 16: 143–153.

Meinz, F.C. & Seip, H.M. 2004. Acid rain in Europe and the United States: An update. *Environ. Sci and Policy*, 7: 253–265.

Montgomery, D. 2007. Soil erosion and agricultural sustainability. *Proceedings of the National Academy of Science*, 104: 13268–13272.

Montgomery, D.R. 2008. *Dirt*. USA. Berkeley, Calif, Univ. of Calif. Press.

Natural Resources Canada. 2012. *The State of Canada's Forests: Annual Report 2012.* (Also available at http://cfs.nrcan.gc.ca/pubwarehouse/pdfs/34055.pdf.)

Natural Resources Canada. 2014. *Annual harvest of timber relative to the level of harvest deemed to be sustainable*. (Also available at http://www.nrcan.gc.ca/forests/canada/sustainable-forest-management/criteria-indicators/.)

NREL. 2007. *Century*. National Resource Ecology Laboratory. (Also available at http://www.nrel.colostate.edu/projects/century/).

Nunez, M.A. & Dickie, I.A. 2014. Invasive belowground mutualists of woody plants. *Biol. Invas.*, 16: 645–661.

Ogle, S.M., Breidt, F.J., Easter, M., Williams, S., Killian, K. & Paustian, K. 2010. Scale and uncertainty in modeled soil organic carbon stock changes for USA croplands using a process-based model. *Global Change Biology*, 16: 810-822.

Orndorff, Z.W. & Daniels, W.L. 2004. Evaluation of acid-producing sulfidic materials in Virginia highway corridors. *Environ. Geol.*, 46: 209–216.

Ouimet, R., Arp, P.A., Watmough, S.A., Aherne, J. & DeMerchant, I. 2006. Determination and mapping of critical loads and exceedances for upland forests in eastern Canada. *Water, Air and Soil Pollution*, 172: 57–66.

Schindler, D.W., Hecky, R.E. & McCullough, G.K. 2012. The rapid eutrophication of Lake Winnipeg: Greening under global change. *J. Great Lakes Res.*, 38: 6-13.

Sharpley, A. & Wang, X. 2014. Managing agricultural phosphorus for water quality: Lessons from the ISA and China. *J. Environ. Sci.*, 26: 1770-1782.

Sheppard, S.C. & Sanipelli, B. 2012. Trace elements in feed, manure and manured soils. *J. Environ. Qual.*, 41: 1846-1856.

Sheppard, S.C., Grant, C.A. & Drury, C.F. 2009. Trace elements in Ontario soils – mobility, concentration profiles, and evidence of non-point-source pollution. *Can. J. Soil Sci.*, 89: 489-499.

Simoes, R.P., Raper, R.L., Arriaga, F.J., Balkcom, K.S. & Shaw, J.N. 2009. Using conservation systems to alleviate soil compaction in a southeastern United States Ultisol. *Soil & Tillage Research*, 104: 106–114.

Skousen, J., Simmons, J., McDonald, L. & Ziemkiewicz, P. 2002. Acid-base accounting to predict post-mining drainage quality on surface mines. *J. Env. Qual.*, 31: 2034–2044.

Soil Landscapes of Canada Working Group. 2007. *Soil Landscapes of Canada* v 3.1.1 Agriculture and Agri-Food Canada. (digital map and database at 1:1 million scale) (Also available at http://sis.agr.gc.ca/cansis/nsdb/slc/index.html).

Soil Survey Staff, 2014. **USA General Soil Map** (STATSGO 2). Natural Resources Conservation Service, United States Department of Agriculture.

Soil Survey Staff. 1999. *Soil taxonomy: A basic system of soil classification for making and interpreting soil surveys. 2nd edition*. Natural Resources Conservation Service. USA Department of Agriculture Handbook. 436 pp.

Statistics Canada. 2009. *Total and urban land area, 1996, 2001 (modified) and 2006*. (Also available at http://www.statcan.gc.ca/pub/92f0138m/2008001/t/4054949-eng.htm.)

Statistics Canada. 2015. *Census of Agriculture, tillage practices used to prepare land for seeding*. (Also available at http://www.statcan.gc.ca/cansim/a 26?lang=eng&retrLang=eng&id=0040205&pattern=0040200..0040242&tabMode=dataTable&srchLan=-1&p 1=1&p 2=50).

Sverdrup, H. & Warfvinge, P. 1993. *The effects of soil acidification on the growth of trees, grass and herbs as expressed by the (Ca+Mg+K)/Al ratio*. Reports in Ecology and Environmental Engineering 2. Sweden, Lund, Lund University, Department of Chemical Engineering II.

Tarnocai, C. & Bockheim, J.G. 2011. Cryosolic soils of Canada: Genesis, distribution, and classification. *Canadian Journal of Soil Science*, 91: 749-762.

Taylor, D.L., Hollingsworth, T.N., McFarland, J.W., Lennon, N.J., Nusbaum, C. & Ruess, R.W. 2014. A first comprehensive census of fungi in soil reveals both hyperdiversity and fine-scale niche partitioning. *Ecol. Monographs*, 84: 3-20.

Transport Canada. 2012. *Road transportation*. (Also available at http://www.tc.gc.ca/eng/policy/anre-menu-3042.htm.)

United States Department of Transportation. 2014. *Domestic physical system extent—Bureau of Transportation statistics*, (Also available at http://www.rita.dot.gov/bts/sites/rita.dot.gov.bts/files/publications/north_american_transportation_in_figures/html/table_11_1.html.).

United States Geological Survey. 2014. *Irrigation water use*. (Also available at http://water.usgs.gov/edu/wuir.html.)

USDA. 2011. *Resource Conservation Act (RCA) appraisal – Soil and Water Resources Conservation Act.* Chapter 3. The State of the Land.

USDA. 2013a. *Summary Report: 2010 National Resources Inventory*. Washington, DC, Natural Resources Conservation Service and Ames, Iowa, Iowa State University, Center for Survey Statistics and Methodology.

USDA. 2013b. *Rapid carbon assessment (RaCA) methodology.* Sampling and Initial Summary. USDA, 24 May 2013.

USDA. 2014. *Restoring America's wetlands: A private lands conservation success story*. National Resources Conservation Service.

van Bochove, E., Thériault, G., Denault, J.-T., Dechmi, F., Rousseau, A.N. & Allaire, S.E. 2010. Risk of Water Contamination by Phosphorus (IROWC-P). In W. Eilers, R. MacKay, L. Graham & A. Lefebvre, eds. *Environmental Sustainability of Canadian Agriculture, Agri-Environmental Indicator Report Series, Report No.3*, pp. 87—93. Canada, Ottawa, Agriculture and Agri-Food Canada.

VandenBygaart, A.J., Gregorich, E.G. & Angers, D.A. 2003. Influence of agricultural management on soil organic carbon: A compendium and assessment of Canadian studies. Can. *J. Soil Sci.*, 83: 363-380.

Vincent, W.F., Whyte, L.G., Lovejoy, C., Greer, C.W., Laurion, I., Suttle, C.A., Corbeil, J. & Mueller, D.R. 2009. Arctic microbial ecosystems and impacts of extreme warming during the International Polar Year. *Polar Sci.*, 3: 171–180.

Wagg, C., Bender, S.F., Widmer, F. & van der Heijden, M.G.A. 2014. Soil biodiversity and soil community composition determine ecosystem multifunctionality. *Proceedings of the National Academy of Science*, 111: 5266–5270.

Waltman, S.W., Olson, C., West, L., Moore, A. & Thompson, J. 2010. *Preparing a soil organic carbon inventory for the United States using soil surveys and site measurements*. IUSS World Congress of Soil Science. Why carbon stocks at depth are important, 1–6 August 2010.

Wardle, D., Klironomos, J.N., Setala, H., Putten, W.H. & Wall, D. 2004. Ecological linkages between aboveground and belowground biota. *Science*, 304: 1629–1633.

Watmough, S.A. & Hutchinson, T.C. 2004. The quantification and distribution of pollution Pb at a woodland in rural south central Ontario, Canada. *Environ. Pollut*, 128: 419-428.

West, T.O., Brandt, C.C., Wilson, B.S., Hellwinckel, C.M., Tyler, D.D., Marland, G., De La Torre Ugarte, D.G. & Larson, J.A. 2008. Estimating regional changes in soil carbon with high spatial resolution. *Soil Science Society of America Journal*, 72: 285–294.

Whitfield, C.J., Aherne, J., Watmough, S.A. & McDonald, M. 2010. Estimating the sensitivity of forest soils to acid deposition in the Athabasca Oil Sands Region, Alberta, Canada. *Journal of Limnology*, 69(1): 201–208.

Wiebe, B.H., Eilers, R.G. & Brierley, J.A. 2006. *The presence and extent of moderate to severe soil salinity on the Canadian Prairies*. Proceedings of the Manitoba Soil Science Society Annual Meetings. Canada, Winnipeg, MB.

Wiebe, B.H., Eilers, W.D. & Brierley, J.A. 2010. Soil salinity. In W. Eilers, R. MacKay, L. Graham & A. Lefebvre, eds. *Environmental sustainability of Canadian agriculture: Agri-environmental indicator report series – Report No. 3*, pp. 66-71. Ottawa, Ontario, Agriculture and Agri-Food Canada.

Wolkowski, R. & Lowery, B. 2008. *Soil compaction: Causes, concerns and cures.* (Also available at http://www.soils.wisc.edu/extension/pubs/A 3367.pdf.).

Xiao, Q., Bartens, J., Wu, C., McPherson, G., Simpson, J. & O'Neil-Dunne, J. 2013. *Urban Forest Inventory and Assessment Pilot Project phase two report*. USA, California, City of San Jose.

15 | Regional Assessment of Soil Change in the Southwest Pacific

Regional Coordinator/Lead Author: N.J. McKenzie (ITPS/Australia)

Contributors: J.A. Baldock (Australia), M.R. Balks (New Zealand), M. Camps Arbestain (ITPS/New Zealand), L.M. Condron (New Zealand), M. Elder-Ratutokarua (Secretariat Pacific Community/Fiji), M.J. Grundy (Australia), A.E. Hewitt (New Zealand), F.M. Kelliher (New Zealand), J.F. Leys (Australia), N.J. McKenzie (ITPS/Australia), R.W. McDowell (New Zealand), R.J. Morrison (Fiji/Australia), N.R. Schoknecht (Australia).

15.1 | Introduction

The Southwest Pacific region includes the 22 island nations of the Pacific[1], New Zealand and Australia (Figure 15.1). The landscapes of the region are very diverse ranging from a large continental land mass through to tens of thousands of small islands across the enormous expanse of the southwest Pacific Ocean. There are extensive ancient flat lands through to some of the youngest and most tectonically active landscapes on the planet. Temperature and rainfall ranges are large because of the breadth of latitudes and elevations. As a consequence, the soils of the region are also diverse. The strongly weathered soils in humid tropical areas and the vast expanses of old soils across the Australian continent are particularly susceptible to disturbance and this is where some of the more intractable problems of soil management occur today.

15.2 | The major land types in the region

The major land types in the region owe their origin to the relative movement of the Earth's lithospheric plates, and in particular to the interaction between the Australian and Pacific Plates. The breakup of the supercontinent of Gondwana included the separation, around 96 million years ago, of the Australian Plate from the Antarctic Plate. The Australian Plate includes the present day island continent of Australia, Papua New Guinea, a small part of the South Island of New Zealand, and some islands including New Caledonia and Norfolk Island. The Australian Plate is moving northwards at approximately 70 mm yr^{-1}, colliding with the Eurasian Plate. This has created the mountains of Indonesia and Papua New Guinea, the highest peaks being Puncak Jaya (4 848 m) and Mt Wilhelm (4 509 m) respectively. The Pacific Plate is moving westwards towards the Australian Plate. Movement along the transform boundary of these plates has created the Southern Alps of New Zealand (Aoraki/Mt Cook 3 724 m) and related geological activity has resulted in substantial volcanic activity around the aptly named Pacific Rim of Fire. New Zealand straddles the boundary of the Australian and Pacific Plate. Most of the island nations of the Pacific are volcanic in origin and exist today as either hilly or mountainous features formed by the volcanoes themselves, or as atoll islands.

1 These countries are American Samoa, Cook Islands, Federated States of Micronesia, Fiji, French Polynesia, Guam, Kiribati, Marshall Islands, Nauru, New Caledonia, Niue, Northern Mariana Islands, Palau, Papua New Guinea, Pitcairn Islands, Samoa, Solomon Islands, Tokelau, Tonga, Tuvalu, Vanuatu, and Wallis and Futuna.

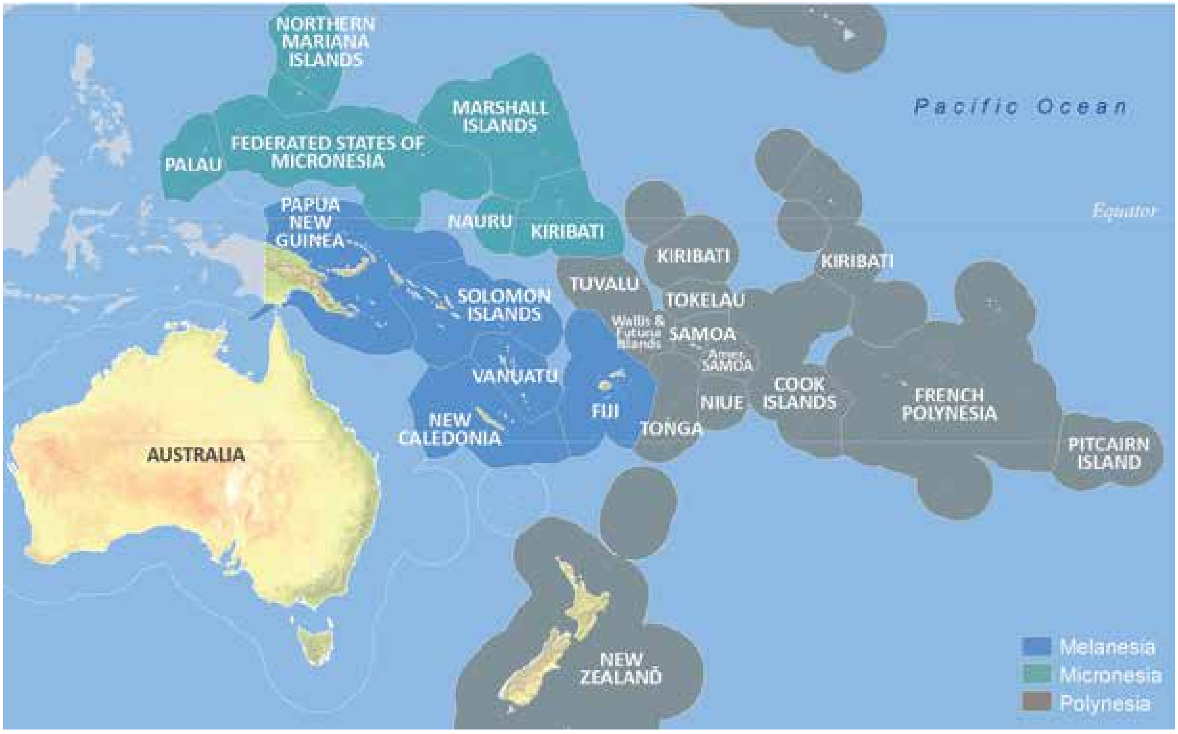


Figure 15.1 | Nations in the Southwest Pacific region and the extent of Melanesia, Micronesia and Polynesian cultures.
Figure based on base map imagery: exclusive economic zone boundaries (EEZ)v 8 2014, Natural Earth 11 3.2.0

Ancient landscapes

After separating from Antarctica, the Australasian landmass moved northwards into warmer latitudes at the same time that the globe was cooling during the Pleistocene. The development of the circumpolar Southern Ocean further moderated the climate of the Australasian landmass. The resulting relative stability meant that biological evolution and soil development occurred on similar timescales and without major phases of interruption. This is in contrast to much of the Northern Hemisphere where repeated glaciations renewed landscapes and ensured that large areas have relatively young soils. This long history of soil development has many implications today for land management in the region.

Low-relief landscapes of Australasia

The western two-thirds of the Australian continent are dominated by ancient landscapes and strongly weathered soils. Some of these soils bear the imprint of previous climates with some unexpected patterns of soil distribution – very acid leached soils (normally associated with humid regions) now occur in deserts, and deeply weathered soil profiles (tens of metres deep) occur in Mediterranean climates with limited leaching. Vast areas of sandy soils have formed from the predominantly acid-igneous and sedimentary parent materials. Nutrient status is very poor and micronutrient deficiencies are common. In the south, substantial quantities of salt have accumulated in the low relief landscapes. As a consequence, sodic and saline soils are widespread and human-induced salinity is a major land management problem.

Uplifted and eroded continental margins

The soils and landscapes of the eastern third of Australia are dominated by the influence of the Great Escarpment (Ollier, 1982) – a landscape feature that extends for 3 000 km from Northern Queensland to Victoria. The Great Escarpment was formed by uplift associated with the passive continental margin to the east. Inland of the Escarpment, the soils and landscapes tend to be older but more clay-rich than in the west of the country. The undulating and low relief landscapes also have saline and sodic soils. Large sedimentary basins (e.g. Murray–Darling Basin) have moderately fertile soils although the current climate is highly variable.

The elevated tablelands adjacent to the Great Escarpment have significant areas of basalt and their associated alluvial landscapes have some of Australia's best soils for agriculture. Likewise, the dissected landscapes to the east and south of the Great Escarpment where most Australians live are much younger. The coastal river systems have fertile alluvial landscapes.

The remnants in New Caledonia, Papua New Guinea and New Zealand

The areas of the Australian Plate in New Caledonia, Papua New Guinea and New Zealand share ecosystems with a common evolutionary origin (e.g. Antarctic flora including Araucaria and Nothofagus). They typically have high levels of biodiversity although the above-ground systems are much better documented that those in the soil. These remnant areas are on the more tectonically active fringes of the Australian Plate and they have greater relief as a result.

Young active landscapes

New Zealand

The tectonically active landscapes of New Zealand can be broadly divided into the axis of high mountain ranges, the basin and range provinces on either side of these ranges, the Taupo Volcanic Zone on the North Island, and the lowlands and sedimentary basins. Only 15 percent of the land area is flat. The climate of the country ranges from sub-tropical in the north to sub-Antarctic in the far south. Sixty percent of New Zealand is >300 m above sea-level and there are about 3 000 mid-latitude mountain glaciers.

A climate sequence of Luvisols, dystric Cambisols and Podzols cover 69 percent of the country and are derived from sedimentary rocks (greywacke, sandstone, siltstone and mudstone as colluvium, alluvium and loess). Significant soils in the remaining areas include vitric Andosols derived from rhyolytic tephra (North Island) and silandic Andosols, Nitisols and Ferralsols derived from andesitic tephra and basalt (mainly on the North Island).

Papua New Guinea

Papua New Guinea has five major landscape regions (Löffler, 1977, 1979). The Southern Plains and Lowlands, up to 400 km wide, are in the west of the country. Most of this region is less than 30 m above sea level and includes the extensive alluvial plains of the Fly River. There are two mountainous regions that occupy the majority of the country. The Central Ranges run the length of the mainland and have very high relief with many peaks between 3 000 m and 4 000 m. The Northern Ranges run parallel and descend to a discontinuous and narrow coastal plain. Between these ranges are the plains, lowlands and wetlands of the Inter-montane Trough, many of which are associated with the Sepik River. The Islands Region is diverse and includes active volcanoes (e.g. New Britain) particularly along the Northern Bismarck Island Arc. Fringing coral reefs and raised coral limestone landforms are common in this tectonically active area.

Island landscapes and atolls

As noted earlier, most of the smaller islands in Melanesia, Micronesia and Polynesia are volcanic in origin. Those with hilly or mountainous features were formed by the volcanoes themselves and they may have fringing coral reefs, elevated coral platforms, or both. The low lying atoll islands have been formed by corals growing on extinct seamounts or volcanoes that have eroded or subsided. Many of the atoll islands are only a few meters above sea level and are therefore vulnerable to sea-level rise.

15.3 | Climate

The climate of the region is strongly influenced by circulation patterns and processes in the Pacific Ocean (the Southern Oscillation) that bring La Niña conditions associated with floods and cyclones and El Niño conditions that are associated with droughts. A similar circulation pattern in the Indian Ocean (the Indian Ocean Dipole) influences drought across southern Australia. The Southern Annular Mode of the Southern Ocean affects weather and climate in New Zealand and southern Australia. These large-scale circulation processes interact with the landscape (e.g. through orographic processes and in accord with the scale of the land mass) to exert a strong control on land use and management across the region. Some of the most significant features are as follows.

- Australia has a very high year-to-year rainfall variability, and major droughts and wet periods occur on a decadal scale. Resilient systems of land and water resource management are essential to deal with this level of climate variability.
- Landscapes in the tropics and sub-tropics experience cyclones and very high intensities of rainfall especially in coastal areas. Maintenance of surface cover is essential to avoid extremely high rates of soil erosion.
- The wet-dry tropics of northern Australia and some Pacific Islands receive most of their rainfall in three consecutive months and the remainder of the year is severely water-limited. This restricts options for land use unless some form of irrigation is possible. The harsh climate, in conjunction with the ancient and strongly weathered soils, largely accounts for the dramatic difference in land use and population density of Northern Australia when compared to nearby Indonesia and Papua New Guinea.

In some parts of the region, there is a sensitive interplay between rainfall, evaporation and the capacity of soil to store water. Many soils in southern Australia have a limited capacity to store water and this makes them especially vulnerable to small changes in the distribution and amount of rainfall. As a consequence, relatively small changes in rainfall and temperature caused by climate change are having a significant impact on farming systems and water resources (Reisinger *et al.*, 2014). Likewise, sea-level rise caused by global warming (Nurse *et al.*, 2014) creates an immediate and serious threat to thousands of low lying atoll islands in the region.

15.4 | Land use

15.4.1 | Historical context

The history of human settlement has occurred in several distinct waves over a very long period. In every case the impact on soils has been substantial and in some cases catastrophic. The earliest records indicate that human arrival occurred in Papua New Guinea and Australia at least 45 000 years BP. At this time, sea levels were much lower and a land bridge connected the two countries forming the single continent of Sahul. This continent was widely colonized by 35 000 years BP (O'Connell and Allen 2004).

It is difficult to assess the impact on soils caused by the initial colonization of the region, particularly by the Australian Aborigines, because it coincides with a period of rapid climate change towards the end of the Pleistocene. In Australia, major changes in fire, vegetation, and wildlife occurred (Roberts, Jones and Smith, 1990; Roberts *et al.*, 1994; Bowler *et al.*, 2003; Turney *et al.*, 2001). The cumulative effect on soils caused by humans and other environmental drivers during this time may rival the later direct impact caused by Europeans, although the latter has been concentrated into a very short period.

The broad area of Near Oceania (including present-day Papua New Guinea, Solomon Islands, Vanuatu, New Caledonia and Fiji) was occupied by peoples that were to remain relatively isolated for more than 25 000 years giving rise to the remarkably diverse cultures and languages of Melanesia – for example, more than 800 languages are still spoken today in Papua New Guinea. It is likely that agriculture was invented in the New Guinea highlands at about the same time (10 000 years BP) as it appeared in other parts of the world, and that agricultural development and plant domestication in New Guinea was independent of what happened elsewhere in the world until about 3 500 years BP (Denham *et al.*, 2003).

The Polynesian and Micronesian peoples arrived in relatively recent times and their ancestors were primarily from East Asia. The development of sailing approximately 3 000 years BP enabled their colonization of the islands in Remote Oceania as far east as Tonga and Samoa where Polynesian culture then developed (Friedlaender *et al.*, 2008).

When the Polynesian Maoris first arrived in New Zealand c. 1200 CE, most of the land was covered by dense forest which they began to clear by burning to facilitate settlement and hunting (McGlone, 1989). Most of the forest clearance occurred between 1350 and 1550, with impacts on soils and landscapes that were greatest in the drier east coast regions of both main islands. By 1840 just prior to large scale European settlement, the forest cover had been reduced from 85 percent (23 million ha) to 53 percent (14.3 million ha) (Condron and Di, 2002).

15.4.2 | Nineteenth and twentieth centuries

European colonization was widespread across the region during the nineteenth and twentieth century. The impact on soils in many districts, particularly in Australia and New Zealand, was profound and in some areas initially catastrophic. In Australia, the severity of soil degradation, particularly in the 100 years after 1850, was extreme, culminating in the dust bowl years of the 1930s and 1940s. In New Zealand, the clearing of steep hill country led to widespread erosion and sedimentation in river systems.

Other countries in the region (e.g. Papua New Guinea, Fiji, French Polynesia) went through distinct phases of land use in this period. Subsistence shifting-agriculture was disrupted by European commercial farming (e.g. pastoralism, coconut plantations, cotton and coffee) which often involved widespread clearing of the tropical lowland forests. Sugar cane industries were established in some countries and the logging of the indigenous timber resource was widespread. Urban areas expanded and in some countries agriculture spread onto more marginal lands with intensification of land use being widespread.

The legacy of the early and destructive phases of land use is still evident throughout the region. Many landscape processes take decades to stabilize after a change in land use and as a consequence, the full extent of soil change caused by prior land use across the region has yet to be fully expressed. The most significant and potentially irreversible causes of soil change across the region are discussed in the following paragraphs.

Clearing: In Australia, the removal of deep-rooted native vegetation and its replacement with annual crops and perennial pastures has in most cases led to a net loss of organic carbon, nutrients and less efficient use by plants of the available rainfall. This can lead to rising groundwater levels and, in some cases, to dryland salinity. The time between initial disturbance and the longer term equilibrium for different soil properties can range from a few decades (e.g. organic carbon in light-textured soils (Sanderman, Farquharson and Baldock, 2010) through to many decades or centuries (e.g. dryland salinity in regional-scale groundwater systems (NLWRA, 2001b)). Conversely, in New Zealand, carbon and nitrogen stocks increase (in particular in hill country with low stocking rates) while C:N ratios decline when converting from native vegetation to pasture (Schipper and Sparling, 2011; Sparling *et al.*, 2014).

Mining: Major disruptions to landscapes were caused by mining, beginning with the gold-rush era (from the 1850s onwards) in Australia and New Zealand. Alluvial landscapes in particular were dug up, turned over, sluiced and degraded. Enormous quantities of sediments were generated and soils and river systems were damaged irreparably. The environmental controls around mining operations in these two countries have improved in more recent decades but this has not been the case elsewhere in the region.

Mining operations on various Pacific islands have had major environmental and social impacts. Examples include the Ok Tedi (see below) and Porgera mines in Papua New Guinea, extensive strip mining for phosphorus in the small country of Nauru where >90 percent of the original soils have been removed (Morrison and Manner, 2005), and the ill-fated Bougainville Copper mine.

During the last decade, the area of land used for mining and extraction of energy resources (e.g. coal and coal seam gas) has expanded significantly in eastern Australia. This has generated conflicts and difficult policy decisions on land use because some of the highest potential areas for coal and coal seam gas coincide with high-quality agricultural land (Chen and Randall, 2013).

Excessive cultivation and compaction: This was widespread during the first half of the 20th century and it is still a problem in some parts of the region. The economic cost of compaction caused primarily by heavy machines and animals is likely to be large but it remains unquantified.

Grazing: Excessive grazing damages soil quickly because removal of cover leads directly to erosion. It can also permanently remove nutrients unless replenished with fertilizer. Large areas across the Australian rangelands have been affected (Bastin, 2008). In New Zealand, conversion of forested hill country to pastures has had a large impact on soil erosion.

Rabbits and other feral animals: Wild grey rabbits were introduced to Australia in 1860 and within a decade they reached plague proportions. Their main effect was to exacerbate over-grazing caused by stock, leading to bare ground and erosion by water and wind. Rabbit plagues continued throughout the first half of the 20th century and were particularly devastating during the 1940s. Numbers were reduced greatly when in 1950 the myxomatosis virus was introduced. The pest gradually increased again in the following decades but was again reduced when the calici virus was released in the 1990s. Rabbits and other feral animals (e.g. goats and pigs) degrade soils in other parts of the region. They are a significant problem in New Zealand and on sub-Antarctic islands such as Macquarie Island which has been severely eroded in recent decades because of rabbits.

15.4.3 | Contemporary land-use dynamics

At the end of the twentieth century, diverse systems of land use operated throughout the Southwest Pacific ranging from traditional systems (e.g. subsistence shifting-agriculture in Papua New Guinea and some of the Pacific Islands) through to technologically advanced systems (e.g. highly mechanized agriculture in Australia and New Zealand). While the general patterns of land use have been relatively stable for several decades, significant changes are continuing. Those with the greatest impact on soil resources are summarised in Table 15.1. There are large differences in the degree of economic development in the region (Table 15.2) as the region contains some of the world's poorest nations and some of the richest. There are also large differences within nations. Twenty percent of Australia is owned and managed by economically disadvantaged Indigenous Australians (SOE, 2011). Approximately 5.6 percent of New Zealand is Maori land, a major contributor to New Zealand's economy. About 26 percent of Maori businesses are in the primary sector, principally farming, forestry and horticulture. In some countries (e.g. Solomon Islands and Vanuatu) increasing rural populations have led to significantly greater use of productive land in the subsistence shifting-cultivation systems with much reduced fallow periods and subsequent declines in soil quality.

Primary industries play a major role in most countries. The development of mineral and energy resources has been a key driver of economic growth, particularly in Australia and Papua New Guinea. New Zealand and Australia are major exporters of agricultural products.

Table 15.1 | Summary of current primary drivers of land-use and the associated implications for soil resources in the Southwest Pacific region.

Country and land-use driver	Implication for soil resources
Australia	
Commodity prices	Disruption to, or intensification of, current systems of land use and management
Cessation of land clearing	Carbon stocks in areas under native vegetation are protected. Intensification of land use occurs elsewhere
Expansion of mining and energy industries	Loss of productive soils used for agriculture
Agricultural land management	Intensification of land use may increase some threats (e.g. acidification, nutrient imbalance) but improved systems of soil management are reducing others (e.g. compaction, carbon loss, contamination)
Population growth and urban expansion	Sealing and capping of land
Drying of southwest Western Australia	Risks to farming systems, forestry and other plantations
Land degradation	Decreased viability of current systems of land use (e.g. southern rangelands of Australia)
New Zealand	
Commodity prices	Disruption to, or intensification of, current systems of land use and management
Intensification of agriculture	Intensification of land use may increase some threats (e.g. nutrient imbalance) but improved systems of soil management are reducing others (e.g. compaction, carbon loss, contamination) and other off-site impacts are likely (e.g. eutrophication of waterways, increased emissions of GHG)
Expansion of forestry	Reduced risk of erosion and potential changes in carbon stocks
Urban expansion	Sealing and capping of land
Papua New Guinea	
Rapid population growth	Reduced rotation length in traditional subsistence systems leading to fertility decline
Unsustainable logging	Erosion, fertility decline, reduced carbon stocks, loss of biodiversity
Mining	Immediate site impacts and risks of off-site contamination
Fiji	
Commodity prices	Disruption to established systems of land use and management (particularly for the sugar industry)
Forest management	Risks of erosion, fertility decline, reduced carbon stocks, loss of biodiversity
Urban expansion	Sealing and capping of versatile land

Solomon Islands	
Unsustainable logging	Erosion, fertility decline, reduced carbon stocks, loss of biodiversity
Low Lying Atoll Islands	
Local production of fresh food	Improved management of limited soil resources
Sea-level rise	Salinization, loss of soil resources
Urban expansion	Sealing and capping of land

Table 15.2 | Current population, project population (UNDESA, 2013) and Gross Domestic Product per capita (World Bank, 2014) for countries of the region.

Country	Population 2015 Thousands	Population 2050 Thousands	GDP per capita USD (2009-2013)
Australia	**23 923**	**33 735**	**67 468**
New Zealand	**4 596**	**5 778**	**41 556**
Melanesia			
Fiji	893	918	4 572
New Caledonia	263	364	-
Papua New Guinea	7 632	13 092	2 088
Solomon Islands	584	1010	1 954
Vanuatu	264	473	3 303
Total Melanesia	**9 636**	**15 858**	
Polynesia			
American Samoa	56	62	-
Cook Islands	21	24	-
French Polynesia	283	337	-
Niue	1	1	-
Samoa	193	242	3 647
Tokelau	1	1	-
Tonga	106	140	4 427
Tuvalu	10	12	3 861
Wallis and Futuna Islands	13	13	-
Total Polynesia	**684**	**832**	
Micronesia			
Federate States of Micronesia	104	130	3 235
Kiribati	106	156	1 651
Marshall Islands	53	67	3 325
Nauru	10	11	-
Northern Mariana Islands	55	52	-
Palau	21	28	11 810
Total Micronesia	**519**	**671**	

15.5 | Threats to soils in the region

15.5.1 | Erosion by wind and water

The rates of soil erosion occurring today in Australia and New Zealand are significantly less than in previous decades. The situation in the rest of the region is less clear. Very fast rates of erosion are occurring in countries with uncontrolled land clearing and logging (e.g. Papua New Guinea and the Solomon Islands). Unsustainable rates of erosion are also likely to be occurring in marginal and hilly lands used for agriculture in some Pacific countries, for example Fiji (Liedtke, 1989).

Soil erosion by wind is a significant problem in Australia and an account of trends and current status is presented below. It is not common elsewhere in the region because of the humid climate, although drier areas in New Zealand (primarily on the South Island) are prone to wind erosion (Eyles, 1983).

Australia

Current rates of soil erosion by water in Australia are much less than the peak periods just after land clearing. In many parts of the country, widespread gully erosion occurred during this time and the hydrological regime of many river systems was changed. In southern Australia, gully and river bank erosion are the dominant sources of sediment supplied to streams. Gully erosion in southern Australia has now been largely stabilised, but gullies are still actively forming in northern Queensland and in some agricultural regions of Western Australia (NLWRA, 2001a).

Despite the apparent stabilization, current rates of soil erosion by water across much of Australia now exceed soil formation rates by a factor of at least several hundred and, in some areas, several thousand. As a result, the expected half-life of soils (the time for half the soil to be eroded) in some upland areas used for agriculture ranges from less than a century to several hundred years. The latest assessment concluded that soil erosion by water in Australia is still at unsustainable rates, but there are large uncertainties about the time until soil loss will have a critical impact on agricultural productivity (SOE, 2011; Bui *et al.*, 2010; Bui, Hancock and Wilkinson, 2011). Environmental impacts of excessive sedimentation and nutrient delivery on inland waters, estuaries and coasts are already occurring.

It is estimated that up to 10 million ha of land have less than 500 years until the soil's A-horizon (effectively the more fertile topsoil) will be lost to erosion. Most of this land is in humid subtropical Queensland. Integrated studies of soil formation and erosion using a variety of techniques will be needed to better understand the extent, severity and significance of the problem. However, it is clear that a concerted program of soil conservation is essential to control this chronic form of land degradation across large areas of Australia. The problem is arguably having its greatest environmental impact on the World Heritage listed Great Barrier Reef. The latest scientific consensus is that the decline of marine water quality associated with terrestrial runoff from the adjacent catchments is a major cause of the current poor state of many of the key marine ecosystems of the Great Barrier Reef. The main source of excess nutrients, fine sediments and pesticides from Great Barrier Reef catchments is diffuse source pollution from agriculture (Brodie *et al.*, 2013).

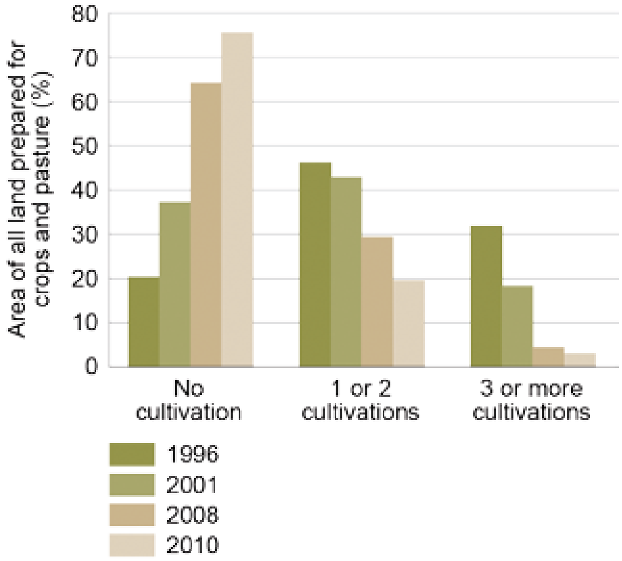


Figure 15.2 | Change in the percentage area of all land prepared for crops and pastures under different tillage practices in Australia, 1996-2010 Source: SOE, 2011.

Land management practices have improved significantly during the past few decades, due to better grazing practices, adoption of conservation tillage, enforcement of forestry codes and soil conservation measures in engineering (e.g. relating to road construction and urban development).

Ground-based monitoring of management practices and land cover along with data on land-management practices (SOE, 2011) reveal a pattern of:

- more careful grazing and maintenance of effective land cover at critical times of the year
- improved adoption of conservation practices, especially across the cropping lands of southern Australia
- an associated large decline in the amount of tillage in farming systems (Figure 15.2)

New Zealand

Soil loss by erosion is a major problem in many areas of New Zealand due to a combination of factors including soil type, topography and climate, as well as the type and associated intensity of land use (especially pastoral agriculture).

Eyles (1983) provided the first systematic inventory of soil erosion in New Zealand and made the following observations and estimates. Surface erosion occurs mainly on the South Island. In 1983 sheet erosion affected 10 million ha of the country, while wind erosion affected 3 million ha (the total area of New Zealand is approximately 27 million ha). Mass movement occurred mainly on the North Island and slip erosion was estimated to affect 7 million ha. Fluvial erosion also occurred mainly in the North Island with rill and gully erosion estimated to affect 2 million ha. The study concluded that a total of 9 million ha of farmed land in New Zealand was at risk of significant erosion, although the level of risk was variable. The large scale afforestation of hill country and steep-land pasture since 1983 (see case study below) will have substantially reduced rates of soil erosion (especially sheet and slip erosion). However, forest harvesting results in significant soil disturbance and this will increase the risk of erosion when the first rotation forest is harvested and the second rotation established.

Dymond (2010) analysed erosion rates in order to refine the national carbon budget because soil erosion in New Zealand results in the large export of sediment and particulate organic carbon (POC) directly to the sea. The North Island of New Zealand was estimated to export 1.9 (–0.5/+1.0) million tonnes of POC per year to the sea, and to sequester 1.25 (–0.3 /+0.6) million tonnes of carbon per year from the atmosphere through regenerating soils. The South Island was estimated to export 2.9 (−0.7/+1.5) million tonnes of POC per year, and to sequester approximately the same amount. Dymond (2010) assumed that exported carbon is buried at sea with an efficiency of 80 percent. This gives New Zealand a net carbon sink of 3.1 (−2.0/+2.5) million tonnes per year (equivalent to about 45 percent of New Zealand's fossil fuel carbon emissions in 1990). There is essentially a 'conveyor-belt' transfer of carbon from the atmosphere to soils regenerating from erosion and to the sea floor where carbon is permanently buried. The large magnitude of the net sink is primarily due to tectonically driven uplift and erosion combined with high biological productivity. The degree to which other elevated and humid islands in the Pacific can operate as a net sink has not been determined.

Pacific Islands

The high rainfall and steep lands in many of the larger Pacific Island countries make them vulnerable to soil erosion. Understanding the significance of this erosion requires information on both rates of soil formation and the baseline rates of sediment movement in different landscapes. The latter can be much larger than in temperate regions and relatively fast rates of erosion around 50 tonnes $ha^{-1} yr^{-1}$ have been recorded in heavily forested landscapes in Fiji (Glatthaar, 1988; Liedtke, 1989) and Samoa (Terry, Garimella and Kostaschuk, 2002; Terry, Kostaschuk and Garimella, 2006). Tropical cyclones, intense rainstorms, steep slopes and landslides are key factors.

The desire for economic development has led to increasing areas of good quality soil being used for cash crops such as coffee and cocoa and for cattle production. These factors in conjunction with population pressures are forcing small-scale agriculture onto steeper less suitable lands. Logging, whether managed or illegal, increases the pressure even further. Condron and Di (2002) indicate that rates of soil loss by erosion in these areas are typically between 10 to 90 tonnes $ha^{-1} yr^{-1}$ (see also Asquith, Kooge and Morrison, 1994; Liedtke, 1989). Accelerated soil erosion has also been associated with plantations, particularly on marginal sloping land.

Although most atolls are away from the cyclone belt in the Southwest Pacific, a few have been severely eroded by large storms. The impacts are devastating as much of the limited soil resource is washed away, and also what is left becomes highly salinized and of limited productive capacity. This was particularly noted on Funafuti, Tuvalu following cyclone Bebe in 1972 (Maragos, Baines and Beveridge, 1973).

Sub-Antarctic Islands

Several sub-Antarctic islands in the Southwest Pacific region continue to have high rates of soil erosion. Macquarie Island, for example, is an isolated island in the Southern Ocean that provides a critical habitat for migratory species. Macquarie Island has been severely eroded and approximately AUD $25 million has been spent in recent years on habitat restoration programmes, mainly for the eradication of rabbits.

15.5.2 | Soil organic carbon change

Most of the region experienced large losses of soil carbon when land was first cleared for agriculture. The major phases of clearing in each country were noted above. In some countries clearing associated with uncontrolled logging continues today, for example in Papua New Guinea and the Solomon Islands (Moorehead, 2011). The initial disruption to soil caused by clearing usually results in a significant loss of nutrients. Organic matter is oxidised and the removal of surface cover (litter and protective vegetation) makes the soil more prone to erosion. Stores and cycles of nutrients adjust under the new land use, but in most cases the net loss of nutrients and leakage are greater than under natural conditions. Most studies indicate that across the region soil carbon typically reduces to 20–70 percent of the pre-clearing amount (e.g. Sanderman and Baldock, 2010). Opportunities for restoring some of this very large stock of carbon have been a focus of soil research in Australia and New Zealand during the last decade.

Australia

In their review of replicated Australian field trials with time-series data, Sanderman and Baldock (2010) concluded that, although the implementation of more conservative land-management practices will lead to a relative gain in soil carbon, absolute soil carbon stocks may still be on a trajectory of slow decline. There are also inevitable trade-offs between agricultural production (e.g. carbon exports in the form of crops, fibre and livestock) and carbon sequestration (capture and storage) in soils.

SOE (2011) provides a district-by-district assessment of trends in soil carbon for Australia. The assessment concluded the following.

- The time since clearing is a key factor determining current trends. For example, large parts of Queensland are still on a declining trend because widespread clearing for agriculture was still occurring in the 1990s.
- Few regions have increasing soil carbon stores.
- Regions with intensifying systems of land use (e.g. northern Tasmania) have decreasing stores.
- Most regions with a projected drying climate have declining trends.
- The savannah landscapes of northern Australia have significant potential for increasing soil carbon stores, but this requires changes in grazing pressures and fire regimes.
- Some of the extensive cropping lands in southern Australia with weathered and naturally infertile soils have not experienced as large a loss of soil carbon since clearing (e.g. generally a 30–70 percent loss and sometimes <30 percent loss) because they had small carbon stores at the time of European occupation and have not changed substantially (although soil biodiversity has undoubtedly changed). Many of these soils have also benefited from the addition of fertiliser and the correction of trace element deficiencies.

A comprehensive study of soil carbon stocks across a wide range of climates, soils and management systems by Baldock, Macdonald and Sanderman, (2013) confirmed that broad patterns of soil carbon variation correlate with first order drivers such as climate and soil type. It also demonstrated that increasing soil stocks in some environments is difficult. For example, results from both trial and commercial grower sites throughout Queensland indicate that no-till systems are not capable of increasing soil organic carbon in either Queensland grain or sugarcane systems. However, no-till may be capable of slowing carbon loss following a period of carbon input from, for example, a pasture ley (Page *et al.*, 2013).

The study also revealed that no individual management practice has the same influence on 0-0.3 m soil carbon stocks across all agricultural regions. Statistically significant differences in 0-0.3 m soil carbon stocks were often not detected despite strong variations in the management practices assessed (e.g. continuous pasture versus continuous cropping). The results suggest that differences in the way individual landowners implement practices in response to personal preferences or business requirements may contribute significantly to the size of stocks. Poor application of a particular management practice which on average has the capability to increase soil carbon may result in a loss of soil carbon. Equally, a very good application of a practice that is found to decrease soil carbon on average may result in increased soil carbon stocks if levels of carbon capture and return to the soil are high enough (Cowie *et al.*, 2013; Page *et al.*, 2013; Cotching *et al.*, 2013; Badgery *et al.*, 2013; Davy and Koen, 2013; Wilson and Lonergan, 2013). These findings have implications for policies and incentive schemes that reward farmers for storing additional organic carbon. Schemes that rely on compliance with a general management practice may not be effective.

New Zealand

The importance of soil erosion to the soil carbon balance of New Zealand was noted earlier. Significant progress has also been made in estimating the carbon balance of different land uses. Trotter *et al.* (2004) estimated that New Zealand's major vegetation types combined to make the total land area a small net carbon source (Table 15.3). A similar conclusion was reported by Tate *et al.* (2005).

As indicated in Table 15.3, 'improved' grassland is New Zealand's most widespread land use but there are large differences in carbon exchange rates within this land type. For example, detailed measurements by Mudge *et al.* (2011) over two years demonstrated that a dairy farm with mineral soil was a net C sink (year 1 = 590 ± 560 kg C $ha^{-1} yr^{-1}$; year 2 = 900 ± 560 kg C $ha^{-1} yr^{-1}$) while Nieeven *et al.* (2005) and Campbell *et al.* (2015), using similar eddy covariance methods, demonstrated that other dairy farms with drained peat soil were net C sources (-1061 ± 500 kg C $ha^{-1} yr^{-1}$ and 2940 C $ha^{-1} yr^{-1}$ respectively).

Table 15.3 | Estimated annual land–atmosphere (net) carbon (C) exchange rate for New Zealand's major vegetation types. Source: Trotter et al., 2004.

Vegetation type	Area (million ha)	Net carbon exchange (T g y^{-1})	Source or sink
Native forest	5.8	-8	Source
Planted forest	1.6	+5	Sink
Scrubland	3.7	-2	Source
Native (tussock) grassland	4.3	+7	Sink
'Improved' grassland (ryegrass and white clover)	6.7	-6	Source
Unimproved grassland (species other than ryegrass and white clover)	3.4	+3	Sink
Total for New Zealand	25.5	-1	Source

Schipper *et al.* (2014) reported the results of repeated sampling of 148 soil profiles across New Zealand over a 20-40 year period under improved pasture grazed by dairy cattle and dry stock (e.g. beef cattle and sheep). For soils on flat land, C stock of the uppermost 0.3 m depth decreased significantly over time (by 5 ± 21 tonnes C ha^{-1}, n = 125 profiles). For silandic Andosols and Gleysols, C stock of the uppermost 0.3 m depth decreased significantly over time (by 14 ± 20 tonnes C ha^{-1} for silandic Andosols (mean ± standard deviation, n = 32), and by 8 ± 14 tonnes C ha^{-1} for Gleysols (n = 25 profiles). Soils of these groups had the highest (initial) C stocks (178 ± 31 tonnes C ha^{-1} for the uppermost 0.3 m depth of silandic Andosols, 101 ± 28 tonnes C ha^{-1} for the Gleysols and 96 ± 28 tonnes C ha^{-1} for the other soils, n = 91 profiles). For soils in hill country, C stock of the uppermost 0.3 m depth increased significantly over time (by 14 ± 22 tonnes C ha^{-1}, n = 23 profiles).

There have been some studies of potential causal factors of temporal change in soil C stock in New Zealand, including fertility and fertiliser application, irrigation and erosion.

- **Fertility and fertiliser application**: Phosphorus (P) fertiliser application and P and nitrogen fertility status have not been found to account for the C stock trends of lowland and hill country soils beneath grazed pasture (Dodd and Mackay, 2011; Schipper *et al.*, 2011, 2013; Parfitt *et al.*, 2014).
- **Irrigation**: The carbon stocks in some irrigated soils used for grazing over many decades have decreased compared to those receiving only rainfall (Kelliher *et al.*, 2012). This may have been caused by greater soil respiration rates in the irrigated system, although other mechanisms are possible (see Kelliher, Curtin and Condron, 2013, Schipper *et al.*, 2013, Condron *et al.*, 2014).

- **Erosion**: As noted earlier, sheet erosion by water and landslides are important factors affecting carbon stocks in New Zealand. In one catchment, these processes each accounted for a loss of 0.5 tonnes C ha^{-1} yr^{-1} (Page *et al.*, 2004). On lower slopes, the soil may be lost or re-distributed or both. One study showed the C stock in erosion scars increased from 10 to 80 tonnes C ha^{-1} within 70 years in the uppermost 0.2 m depth of soil (Parfitt *et al.*, 2013). However, the carbon stocks of soils forming on erosion scars are unlikely to return to more than ~80 percent of the pre-landslide amount because the newly developing soils are relatively shallow and drought-prone in summer (Rosser and Ross, 2011).

Studies of current C stocks and the potential saturation value for the soils of New Zealand (Beare *et al.*, 2014) suggest there is an opportunity to increase the C stock of pastoral soils by increasing the C input rate, including deeper plant roots (e.g. Carter and Gregorich, 2010). However, organic matter stored deeply in soils is poorly understood (Rumpel and Kögel-Knabner, 2011) and determining the contribution of roots to soil C may not be straightforward (e.g. Dodd and Mackay, 2011).

Pacific

Only a few studies of soil carbon dynamics have been undertaken in the countries of the Pacific (e.g. Hartemink, 1998a). General statements are nonetheless regularly made about the decline in soil carbon associated with soil erosion, excessive cultivation and poor soil management. For example, Leslie and Ratukalou (2002) conclude that the small size of farm holdings in Fiji (60 percent are less than 3 ha) forces farmers into intensive cultivation (often mono-cropping) for high output, short-term production without (or with only minimal) fallow periods. Furthermore, competition for land is forcing subsistence gardens onto steeper slopes because of the expansion of cash cropping and grazing on the flatter lands.

Excessive soil erosion in many sugar cane areas along with the burning of cane trash is resulting in serious depletion of fertility and soil loss on poorly managed farms. A 30-year study from Fiji on a series of sugarcane farms showed the expected decrease (15-35 percent) in topsoil organic C at the time of land clearing. However, a redistribution of C occurred, with increases at 30-40 cm depth (below the plough layer) and a very gradual increase in C at greater depth (80 cm). As a result, the overall C content of the fields increased on well-managed farms (Morrison and Gawander in press, Morrison, Gawander and Ram, 2005).

In Papua New Guinea, the expansion of agriculture and uncontrolled logging are most likely causing a reduction in soil carbon stocks but quantitative studies to assess the magnitude of these changes are needed. Even on atolls, land-use change involving replacement of native vegetation by coconuts has led to significant declines in soil carbon, which is critical there because of the key role of organic matter in moisture retention (Morrison and Seru, 1985).

15.5.1 | Soil contamination

The Southern Hemisphere does not have the same history of large scale industrialization as the Northern Hemisphere. However, soil contamination is a significant problem mainly in relation to impurities in phosphate fertilizers, agricultural chemicals, mining, waste disposal, former industrial sites and nuclear testing. Reviews of the history of soil contamination in the region are provided by Naidu *et al.* (1996) and more specifically for Australia (Tiller, 1992; Barzi *et al.*, 1996), New Zealand (Roberts *et al.*, 1996), Papua New Guinea (Singh, Levett and Kumar, 1996) and South Pacific Islands (Morrison, Gangaiya and Koshy, 1996). Throughout the region there are tens of thousands of contaminated sites (SOE, 2011; Ministry for the Environment, 2010) but the scale of the remediation task is not clear. Australia and New Zealand have a long and effective history of working together to coordinate the management and remediation of soil contamination but the waste management problem facing small islands in the Pacific is a serious and escalating problem.

Fertilizers

Impurities in fertilizers and soil amendments such as lime and gypsum can include cadmium, fluorine, lead and mercury. Cadmium and fluorine have been of most concern in the region and the former can move from soil to the edible portions of plants. In Australia and New Zealand, the historically high concentrations of cadmium in phosphate fertilizers resulted from the use of island sources of high cadmium phosphate rock for fertilizer manufacture, primarily from Nauru and Christmas Island (McLaughlin *et al.*, 1996; Loganathan *et al.*, 2003). A large effort has been devoted to determining the magnitude of the problem and to implementing a range of control measures (Warne *et al.*, 2007; MAF, 2011; Cavanagh, 2014). In recent years, levels of cadmium in fertilisers have been reduced, and farming systems have been modified to manage the problem and mitigate future risk. For example, in New Zealand a tiered system for fertilizer management has been established. However, large areas of land that received heavy applications of superphosphate over decades now have elevated levels of cadmium. A recent analysis by de Vries and McLaughlin (2013) concluded that the present cadmium inputs from fertilizer in Australia are in excess of the long-term critical loads in heavy-textured soils for dryland cereals and that all other systems are at low risk. In New Zealand, a recent survey has shown that only isolated soil samples had cadmium concentrations that exceeded the upper-tier threshold value. The evidence to date indicates that cadmium in New Zealand soils poses no immediate concern.

Agricultural chemicals

Many of the more harmful pesticides and herbicides have been banned or tightly controlled in Australia and New Zealand. However, some residues can persist and adversely affect the environment, notably in areas that were, or still are, used for growing potatoes, tomatoes, cotton, bananas and sugar cane. Copper, arsenic and lead are contaminants associated with orchards and market gardens. A widespread problem in both countries has been the thousands of former cattle and sheep-dip sites contaminated with organochlorines such as dichlorodiphenyltrichloroethane (DDT) and other pesticides, including arsenic-based compounds. Urbanisation and the construction of dwellings on or near these sites pose a serious threat to human health. Most of these sites have been investigated and registered. There are far fewer studies on the impact of agricultural chemicals in the smaller island nations of the region.

Mining

The history of mining in the region was outlined above. In Australia, a significant number of current or former mining towns are affected by soil contamination, mostly associated with tailings, mine wastes and pollution from ore processing (e.g. Queenstown in Tasmania, Broken Hill, Captains Flat and Wollongong in New South Wales, Mt Isa in Queensland). At Port Pirie, dispersed heavy metals from smelting (primarily lead, zinc, cadmium and copper) can be detected over thousands of square kilometres, although seriously contaminated areas are restricted to tens of square kilometres (Cartwright, Merry and Tiller, 1976).

In Papua New Guinea riverine disposal of processing residues, waste rock and overburden into rivers (e.g. Ok Tedi and Porgera) is having a large and long-term environmental impact on soils. Since 1984, the 1 000 km long Fly River system has received about 66 million tonnes yr^{-1} of mining waste from the Ok Tedi copper-gold-porphyry mine and this has caused widespread contamination and altered hydrological regimes (Bolton, 2009; Campbell, 2011). Elevated levels of copper, zinc, cadmium and lead occur in the sediments that have been deposited in the alluvial systems of the Fly River but the longer-term impacts on ecosystems and human health are not clear.

Waste disposal

The management of waste is now tightly regulated and managed in Australia and New Zealand. However, waste management is a major contemporary problem for the small island nations of the Pacific. Morrison and Munro (1999) provide an overview of the problem and broader issues of soil management on atoll islands (see case study below, Section 15.6).

Industrial sites

Even though they are less industrialised than many countries in the Northern Hemisphere, Australia and New Zealand have a legacy of contaminated former industrial sites. Most are point-scale but in some cases more widespread contamination has occurred.

The legacy of nuclear testing

Morrison, Gangaiya and Koshy (1996) review the legacy of nuclear testing in the Pacific and the following draws heavily on their account. Nuclear testing occurred throughout the 1940s and 1950s in the Marshall Islands resulting in serious contamination and impacts on resident populations. Sites in the Marshall Islands have been monitored for over 40 years and the detailed investigations include impacts on soils. Some areas will be contaminated forever. In undisturbed areas, most of the radioactivity (>80 percent) is in the upper 0.15 m of the soil profile. The terrestrial food chain is still the most significant potential exposure pathway, but in some locations there may also be health problems from the intake of resuspended radioactive soil particles. Access to several islands is still restricted because of contamination.

Nuclear testing also occurred in the 1950s and 1960s in Kiritimati (Christmas Island). Radioactivity concentrations in soil are consistent with global fall-out levels for a low rainfall equatorial area, and no site on the island presents a risk to the health of the local population or requires any restriction on land use.

Nuclear testing in French Polynesia occurred from the 1960s to the 1990s and it has been a major international environmental issue. Morrison, Gangaiya and Koshy (1996) state that there is no doubt that the atmospheric testing did lead to significant contamination, but the impact of the underground tests is more difficult to assess. A small number of official investigations has been permitted on Mururoa, but these have been limited in extent and some areas have been excluded from investigations. The atoll is under French military control with imported food and restricted use of local resources so the direct impact of contaminants is limited. Morrison, Gangaiya and Koshy (1996) conclude that the impact on surrounding islands which are occupied by Polynesians living a traditional lifestyle is expected to be small, but in the absence of detailed investigations, this cannot be confirmed.

Nuclear testing in Australia occurred during the 1950s and 1960s at Maralinga and Emu Field in South Australia and on the Montebello Islands near the coast of Western Australia. Rehabilitation of the contaminated Maralinga test area has taken decades with a major effort concluding in 2003 (DEST, 2003). Return of the Maralinga test area to its traditional owners was completed in November 2014.

15.5.2 | Soil acidification

Soil acidification is an insidious and widespread problem in many parts of the region. If not corrected, the slow process can continue until the soil is irreparably damaged. The severity and extent of acidification are increasing in many areas due to inadequate treatment, intensification of land management, or both. Soil acidification is of greatest concern in situations where: (i) agricultural practices increase soil acidity (e.g. use of high-analysis nitrogen fertilisers or large rates of product removal); (ii) the soil has a low capacity to buffer the decrease in pH (e.g. infertile, light-textured soils); or (iii) the soil already has a low pH.

The main onsite effects of acidification include: loss or changes in soil biota involved in nitrification, accelerated leaching of plant nutrients, and induced nutrient deficiencies or toxicities. There may also be breakdown and subsequent loss of clay materials from the soil, the development of subsoil acidity, and reduced net primary productivity and carbon sequestration.

The potential offsite effects on waterways include mobilisation of heavy metals, acidification, increased siltation and eutrophication. The process of acidification considered in this assessment is distinct from that associated with acid sulphate soils. Such soils occur primarily in coastal settings or with mines rich in sulphides. In these areas, soils naturally contain metal sulphides that severely acidify when oxidised.

Australia

Acidification 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. More recent studies have confirmed the scale of the problem (e.g. Lockwood *et al.*, 2003, SOE, 2011, DAFWA, 2013).

Soil acidification is widespread in the extensive farming lands of southern Australia and the rates of lime application are well short of those needed to arrest the problem (see below). Acidification is common in intensive systems of land use (tropical horticulture, sugar cane, dairying). In some regions, acidification is limiting biomass production but the degree of restriction is difficult to estimate. Trends in the tropical savannahs are uncertain. If acidification is occurring there, it will be a difficult problem to solve (Noble, Cannon and Muller, 1997; Noble *et al.*, 2002). Carbon losses are most likely occurring across regions in poor condition, and unabated soil acidification will be a major constraint on storing carbon in soils in the future.

Ultimately, soil acidification restricts options for land management because acid-sensitive crops and pastures cannot be grown. It is relatively straightforward to reverse short-term soil acidification through the application of lime. However, it is much harder to reverse the problem if the acidification has advanced deeper into the soil profile, because incorporating lime at depth is prohibitively expensive. Prevention rather than cure is essential. While rates of lime application appear to be increasing (due to active extension programs), they still fall far short of what is needed to arrest the problem. The case study of southwest Western Australia (see below, Section 15.6) provides more details. The rates of lime application are still much lower than what is needed to avoid irreparable damage. In South Australia the average quantity of lime sold annually over the past decade (113 000 tonnes) is only 53 percent of that needed to balance the estimated annual soil acidification rate (SOE, 2011).

New Zealand

The acidification of legume-based pastures as a result of nitrate leaching and nutrient transfer/removal is of particular concern in New Zealand (de Klein, Monaghan and Sinclair, 1997). In many productive lowland pastures soil acidification is overcome through regular liming. However, amelioration of soil acidity by liming is not considered to be feasible in most hill country areas due to the high cost of aerial application (Bolan and Hedley, 2003; Moir and Moot, 2010). These soils are not cultivated and this enhances the risk for subsoil acidification. As a result acidification is affecting a significant portion of hill-country soils in New Zealand, which has the potential to significantly affect the capacity of many soils to sustain plant and animal production.

Papua New Guinea, Fiji and other tropical countries

Soil acidification is a problem in areas used for sugar cane in Papua New Guinea (e.g. Hartemink, 1998b) and in agricultural areas in Fiji (see Section 15.5.6). It may also be significant in tropical and subtropical areas that have been cleared but there have been few investigations.

15.5.3 | Salinization and sodification

Dryland salinity

Dryland salinity is widespread across many parts of southern and eastern Australia. A large proportion of Australia's agriculture is undertaken in areas with a rainfall of 450–800 mm yr^{-1}. In their natural condition, these landscapes had minimal deep drainage (generally less than 20 mm yr^{-1}), and natural stores of salt brought in by rain and dust had accumulated in the soil in many regions. The removal of native vegetation changed the hydrological cycle, because trees and shrubs intercept significant quantities of rain (typically 10–20 percent of rainfall fails to reach the soil surface). When vegetation is removed, more water either infiltrates or runs off the surface. If the original vegetation has been replaced by more shallow-rooted species that use less water (e.g. annual crops and pastures), even more water passes through the soil. This may lead to rising groundwater levels and, in some cases, dryland salinity.

Dryland salinity has been one of Australia's most costly forms of land degradation. The comprehensive assessment by NLWRA (2001b) concluded that, assuming no changes in water imbalance, areas with dryland salinity were expected to increase from 5.7 million ha to 17 million ha by 2050. In many regions, the initial and sometimes primary impact of the change in hydrological regime is an increase in the salinity of streams and rivers (SOE, 2011). However, in low relief landscapes, large areas can be salinized and this dramatically reduces the options for land use.

Between 2001 and 2009, the Millennium Drought (van Dijk *et al.*, 2013) across southern and eastern Australia appears to have slowed the spread of dryland salinity in these regions but the outlook is still problematic. Current projections of climate change are for a drying of southern Australia which should lead to a lessening of the problem. However, the long-term outlook for more recently cleared land in the northern Murray–Darling Basin and central Queensland is unclear. Large areas are yet to reach a new hydrological equilibrium after clearing. However, close surveillance of groundwater systems is essential, particularly in regions that returned to wetter conditions in the last five years. The case study below for southwest Western Australia suggests that the expansion of dryland salinity may be experiencing a temporary lull in many districts. Dryland salinity is not a significant problem in other parts of the Southwest Pacific Region.

Salinity in irrigation areas

Salinity has been a major problem in the irrigation districts of Australia, particularly in the south of the Murray-Darling Basin where most irrigation is concentrated. Salinity developed in the early 1900s soon after the first schemes were completed. The scale of the problem started to be fully recognized in the 1970s and by 2000 it was viewed alongside dryland salinity as the country's highest priority natural resource problem. Major investments in infrastructure, large-scale salt interception schemes, institutional reform, substantial improvements in water-use efficiency and the Millennium Drought all contributed to a mitigation of the problem (e.g. MDBA, 2010). SOE (2011) were equivocal in their assessment of whether salinity was still a major problem. However, the conclusions above for dryland salinity apply equally to irrigation salinity. New irrigation development is occurring in Tasmania where landscapes have stores of salts in some settings. Irrigation developments in northern Australia and the east coast are being explored at present but salinity risks in these areas tend to be less.

Saltwater intrusion

Saltwater intrusion is a critical issue for the management of freshwater groundwater systems on most of the 1 000 or more inhabited atoll islands in the region. This issue is considered in the case study below. Throughout the region, increasing groundwater extraction (e.g. for irrigation), periods of below-average rainfall, urban development and sea-level rise have increased the risk of saltwater intrusion in coastal districts. In Australia, Werner (2010) and Ivkovic *et al*. (2012) provide an initial assessment of the problem. There is a good awareness of salt water intrusion risks in New Zealand and only a small number of actual salt water intrusion problems have occurred. Most have been into shallow unconfined aquifers and the problems have been short-lived and adequately managed by changes to groundwater abstraction (Callander, Lough and Steffens, 2011).

Sodicity

Australia has the largest extent of naturally sodic soils of any continent (FAO, 1988) but they are relatively rare in other parts of the region. There is a large scientific literature on the management and amelioration of sodicity in Australia and reviews are provided by Loveday and Bridge (1983), Summer and Naidu (1998) and Rengasamy (2002, 2006). Sodic soils are difficult to manage because of their poor soil–water and soil–air relations. Swelling and dispersion of sodic aggregates reduces the porosity and permeability of soils and increases soil strength even at low suction (e.g. high water content). Sodic soils have a narrow non-limiting water range (Letey, 1985) and they are typically either too wet immediately after rain or too dry within a few days for optimal plant growth. The dispersive clays cause surface crust and seal formation, poor internal drainage, and in some settings, tunnel erosion.

Rengasamy (2002) estimated that more than 60 percent of the 20 million ha of cropping soils in Australia are sodic and that the actual yield of grains on these soils is often less than half of the potential yield. However, sodicity is often associated with other subsoil constraints to root growth including alkalinity, boron toxicity and salinity so remediation is neither easy nor economic in many cases.

There are few studies on the dynamics of sodicity. Apart from irrigation systems where water supplies contain appreciable sodium, there is limited evidence to indicate that sodicity is increasing or decreasing in either severity or extent or both. In irrigation systems with appreciable sodium, salt loads are generally managed effectively. However, land-based disposal of effluent can be constrained by increasing sodicity and salt loads (Toze, 2006; Balks, Bond and Smith, 1998; Bond, 1998).

Irrigation with waters containing appreciable quantities of potassium is a common occurrence, particularly with waste water. This monovalent cation has a similar effect to sodium, causing clay dispersion and reduced permeability. Smith, Oster and Sposito, (2014) state that the deleterious effect of potassium is estimated to be about one-third of that of sodium.

15.5.4 | Loss of soil biodiversity

Soil biology has been an active field of research in the region for many decades (e.g. CSIRO, 1983; Pankhurst *et al.*, 1994; Abbott and Murphy, 2007). Much of this has had a focus on agriculture and forestry. However, there has also been a concerted effort to understand the evolution, distribution and status of biodiversity. Woodman *et al*. (2008) provide some preliminary assessments for Australia and outline strategies for developing a longer-term system for assessing soil biodiversity. Several significant assessments are underway in the region using molecular biological techniques. The Biome of Australia Soil Environments (BASE) is collecting samples to create a large-scale genomic database of soil biomes across Australia. Until these are completed, it is difficult to provide a definitive assessment of the loss of biodiversity in the region. However, some general inferences can be drawn.

- Above-ground biodiversity is much better characterized than soil biodiversity and there are several megadiverse districts and countries in the region.
- Areas with high above-ground biodiversity are likely to be similarly diverse below ground if their environments were relatively stable throughout the Pleistocene (e.g. the moist forests of southwest Western Australia, rainforests on the eastern escarpment of Australia, intermediate elevations in Papua New Guinea, southern districts in the South Island of New Zealand). Some ancient landscapes were submerged during this period (e.g. New Caledonia) and this most likely had an impact on soil biodiversity present today.
- A range of pressures and stressors directly affect soil biota and some have already been discussed (e.g. loss of soil carbon, acidification, physical disruption through cultivation or other means, intensification of fire regimes). Large areas in the region have experienced these pressures and stressors. As a consequence, they are most likely experiencing a loss of soil biodiversity.

15.5.5 | Waterlogging

No comprehensive survey or monitoring of waterlogging have been undertaken at the district or national level in the region. Waterlogging is a significant constraint on agricultural production and extensive drainage schemes were installed during the twentieth century, particularly in low lying alluvial areas in New Zealand and eastern Australia. Texture-contrast soils with impermeable B-horizons are widespread in the pasture and cropping lands of southern Australia. Waterlogging is a major limiting factor of crop production in south-western Victoria (McDonald and Gardner, 1987). Raised beds are sometimes used to minimize the impact of water logging and enhance crop production.

15.5.6 | Nutrient imbalance

Nutrient imbalances are widespread throughout the more intensively managed landscapes of the region. Some of these have already been outlined in relation to carbon balances and acidification (see below as well). The focus here is on systems of land use where nutrient mining, depletion or accumulation may be occurring.

Nutrient mining refers to situations where there is a large removal of nutrients with minimal additions. In Australia, this has occurred in some extensive, low-input farming systems. For example, Dalal and Mayer (1986) document the decline in soil fertility over 70 years in areas used for dryland cropping in Queensland. This extensive low-input system relied on the natural fertility of its predominantly heavy clay soils (Vertisols) but it now requires fertilizer inputs to offset nutrient exports in harvested products.

Nutrient decline is occurring in other parts of the region although there are few reliable surveys and monitoring systems except in New Zealand. The shortening of rotations in the shifting agricultural systems of Melanesia caused by increased population is most likely causing nutrient decline but minimal evidence is available. Nutrient decline is also likely to be occurring on marginal lands that are degrading due to processes such as acidification and erosion. The magnitude of the nutrient loss has been documented in a few districts, particularly where sediments and nutrients have a major environmental impact.

Nutrient accumulation is a more recent phenomenon in the region and the case study on New Zealand (see below) explores several aspects. In most other Australian farming systems, fertilizer use has increased and most nutrient imbalances are managed because of the economic consequences of over- or under-use. In Australia, the use of nitrogen fertilizers has more than doubled in the last 25 years but application rates are still moderate compared to more intensively managed systems in China, Europe and the United States. Nutrient accumulation has occurred across southern Australia and elsewhere. In a few cases, environmental impacts are significant (e.g. the case of the Great Barrier Reef mentioned earlier and the Peel-Harvey system in Western Australia (e.g. Ruprecht, Vitale and Weaver, 2013).

Phosphorus is naturally deficient across large parts of the region. Despite this, in Australia phosphate fertiliser use is relatively inefficient (McLaughlin, Fillery and Till, 1990; Weaver and Wong, 2011). Most phosphorus is applied in the higher rainfall areas of southern Australia, and around 40 percent is applied to pastures. Nationally, approximately 20 percent of the phosphorus applied as fertiliser is extracted in food and fibre products for export, and about 5 percent is consumed domestically. The remaining 75 percent of the phosphorus applied in Australian agriculture accumulates in the soil, and some of this is lost to the environment, with detrimental impacts on waterways (Simpson *et al.*, 2011; McLaughlin *et al.*, 2011). The accumulation of phosphorus is greatest in soils across southern Australia (e.g. Weaver and Summers, 2013). Apart from the environmental risks caused by this accumulation, the inefficiency has an economic cost that will increase if fertiliser prices rise. In contrast, many grazing systems in northern Australia have pastures and animal production systems that are limited by deficits in phosphorus availability (McIvor, Guppy and Probert, 2011).

Nutrient imbalance has been a persistent issue in soils used for sugar cane in Fiji. Production began in the 1880s and fertiliser use increased after the Second World War. Up to the 1980s, the use of particular fertilizers resulted in excess inputs of N and less than appropriate inputs of P and K (Morrison *et al.*, 1985, 2005). After significant debate and review, blended fertilisers were introduced in the early 1990s and this has led to a more balanced input of N, P and K. However, topsoil Ca and Mg contents have dropped dramatically and this has been attributed to a continuous removal of Ca and Mg in harvested cane and by erosion. In most years, less than replacement quantities of these elements are being added in fertilisers. Another contributing factor has been a dramatic decrease in liming on sugarcane farms.

15.5.7 | Compaction

Until recently, heavy machines such as tractors, harvesters and trucks were driven over most agricultural areas in the region leading to widespread soil compaction (Tullberg, 2010). Damage was greatest when the soil was wet. Some of the compaction can be undone through cultivation, although it is common for plough pans to develop just below the depth of cultivation. The distribution of pressure under a heavy vehicle also results in a zone of compaction halfway between the wheels, usually at a depth of around 0.5 metres. This type of compaction is difficult to remove. Heavy animals can also compact wet soil, leading to a decline in pasture production. Most of the damage occurs in the upper part of the soil profile.

In Australia, the extent of soil compaction due to wheel traffic and its agronomic consequences have not been investigated in detail; however, it is thought to be very widespread (Chan *et al.*, 2006; McGarry, Sharp and Bray, 1999; Tullberg, Yule and McGarry, 2007). A number of studies have documented the extent and severity of compaction under different systems of land use including: irrigated agriculture (McGarry, 1990), cotton (Braunack and Johnston, 2014), dryland cropping (Bridge and Bell, 1994), sugar cane (Braunack, Arvidsson and Hakansson, 2006), and grazing and forestry (Rab, 2004). One study (Geeves *et al.*, 1995) surveyed the physical and chemical properties of 78 soils under crops and grazed pastures judged to be representative for southern New South Wales and northern Victoria. The study concluded that soil-based constraints exist in surface and subsurface soil horizons and that these constraints are in some instances severe. The bulk density of B horizons ranged from 1.25 to 1.95 Mg m^{-3} with a mean of 1.58 Mg m^{-3} indicating that root growth will be restricted in many of these clay-rich horizons (Jones, 1983).

Encouragingly, there has been a rapid uptake of controlled-traffic farming in Australia and New Zealand during the last 15 years. This confines compaction to the smallest possible area and has the potential to alleviate further damage. Tullberg, Yule and McGarry, (2007) provide a detailed account of these systems and their impact on soils. The improved productivity, practicality and economic viability of controlled-traffic systems have led to enthusiastic adoption by farmers in the region. In Western Australia, farm profit has been shown to increase by 50 percent through the adoption of controlled-traffic farming (Kingwell and Fuchsbichler, 2011).

In New Zealand, deterioration of soil physical condition is common in grazed pastoral systems due to animal treading. Sparling and Schipper (2004) in their survey on the condition of soils in New Zealand reported moderate compaction on a large proportion of pasture soils and that about half the sites under dairying were below the critical threshold of 10 percent air-filled porosity. Mixed cropping soils also had low macroporosity. Drewry (2006) indicated that the physical condition of the top 10-15 cm naturally recovers when animals are partially or completely excluded from pasture and proposed the integration of a recovery cycle when managing grazing systems.

15.5.8 | Sealing and capping

Most towns and cities in the region were established during the 19th century and they were nearly all built on, or adjacent to, land highly suited to horticulture and cropping. The encroachment of urban and peri-urban development has seen the capping of this land. For economic reasons, it is highly unlikely that these good quality soils will ever regain their biological function. The loss of these strategically important soils is occurring throughout the region and is particularly significant in Fiji, some of the small Pacific islands, New Zealand and Australia. Population projections (Table 15.2) indicate that the problem will intensify in coming decades. The rate of sealing and capping is not being monitored at present.

15.6 | Case studies

15.6.1 | Case study one: Intensification of land use in New Zealand

Significant changes in land use have occurred across New Zealand in recent decades. The impacts of these changes on soil resources are reasonably well understood because of a substantial commitment to fundamental and applied soil research within the country. This effort is integral to the current and future international success of New Zealand's agriculture sector.

An important development in the evolution of land use in New Zealand that contributed to intensification was the removal of all agricultural subsidies in the mid-1980s (MacLeod and Moller, 2006). New Zealand is unique because it is the only developed country to be largely exposed to international markets. As a consequence, current land management decisions are now driven directly by the response of land managers to market prices for agricultural products.

The removal of subsidies triggered a decline in traditional sheep farming. Sheep numbers dropped from around 69 million in 1980 to 39 million by 2002 and 31 million in 2012. Plantation forests were established on former sheep pastures and forestry expanded from 0.85 million ha in 1980 to 1.8 million ha in 2000, with 95 percent under private ownership. The strong international demand for dairy products caused an increase in the number of dairy farms. Dairy cattle increased from around 3.0 million in 1980 to 5.3 million by 2002 and 6.4 million by 2012. Cattle stocking rates and productivity also increased (Ministry of Agriculture, Fisheries and Food, and Statistics New Zealand). Some of the pastures converted to plantation forests in the 1980s and 1990s are now being converted to dairy pastures following the harvesting of mature trees, and even partly grown trees have been removed and replaced by pasture for dairy farming (Sparling *et al.*, 2014). Horticulture has also expanded. The area of horticultural crops has increased by 40 percent in just over 10 years (HortNZ, 2014).

Most of the intensification of agriculture in New Zealand has occurred on the better class lands. There has also been the expansion of dairy farming onto soils that have previously been less intensively farmed, such as artificially drained Luvisols in the Southland region and irrigated stony Fluvisols and Leptosols of the

Canterbury region. There has been a six-fold increase in N fertilizer use over the past 20 years (compared to a two to three fold increase in Australia) along with substantial increases in use of most other agricultural inputs.

Soil-based primary production is fundamental to the New Zealand economy and agricultural, horticultural and forest products together account for 60–65 percent of the country's total export income. Understanding the pressures and changes occurring in the soils of New Zealand, especially under pasture systems, has been a major task for New Zealand soil scientists. The establishment of a coordinated soil monitoring capability (Sparling and Schipper, 2002) has been important.

Taylor *et al*. (2010) provide an example of the insights gained from the monitoring program for the Waikato region which covers much of New Zealand's central North Island. This monitoring program is linked to systems for setting regional targets for various aspects of the soil resource. About 58 percent of the region is in pastures, 18 percent is in plantation forestry and <1 percent is used for various types of horticulture. Monitoring started in 1995, initially as part of the national '500 soils project' (Hill *et al*., 2003). Four issues that are causing loss of soil resources were identified and quantified in the Waikato region as follows.

1. About a third of monitoring sites under pasture had macroporosity below the lower target. This resulted in reduced infiltration of water which leads to runoff to waterways and attendant problems of contamination, flooding and erosion.
2. Arable and horticulture land-use showed a considerable decline in total carbon since 1990 compared with background levels. Five percent of sites were below the set target and likely to have reduced productive yield.
3. Dairy, horticulture and arable land uses showed considerable increases in fertility compared with non-intensified sites and appear to contribute to the continual increase in N and P detected in regional waterways.
4. Increases were detected in three categories of elements. First were those linked to fertilisers and lime (primarily P, Ca, Cd, F and U); second were elements linked to animal remedies (including Zn and Cu); and third were elements linked to pesticide use in horticultural areas (primarily Cd and Cu).

Agricultural intensification in New Zealand has led to a much greater emphasis on understanding and management of nutrient flows and losses, particularly on grazed pastoral farms. This has included a strong focus on practices and technologies that minimise the leakage of nutrients, especially nitrogen (N) and phosphorus (P), from farms to the wider environment. This has seen farm nutrient management planning shift from a relatively small set of procedures designed to optimise fertiliser application rates for pasture and animal production to a comprehensive whole farm nutrient management approach that considers a range of issues to ensure both farm productivity and environmental outcomes are achieved (Monaghan, Houlbrooke and Smith, 2010).

Agricultural intensification in New Zealand is clearly affecting soil and water quality as measured by a variety of indicators. The effect on soil is being influenced both by legislation directed at the off-site impact of land use on water quality and through a collaborative process that enables farmers and regulators to work together so that land can be managed in the most cost-effective way possible. However, irrespective of these efforts there are still problem areas where the off-site impact may be too much to manage. Even under the best management for water quality issues, other issues such as localised contamination of soil (e.g. with Cd) will still be an issue. These are being highlighted and worked through at a regional and national level – with a requirement that regions formally report on soil quality in addition to existing requirements for water and air quality.

15.6.3 | Case study two: Soil management challenges in southwest Western Australia

The southwest of Western Australia is dominated by ancient landscapes and widespread sandy soils that are strongly weathered. By world standards, the soils are infertile and have a range of physical and chemical constraints to plant growth. During the last one hundred years, large areas have been cleared for agriculture. The original perennial, deep-rooted vegetation has been replaced with shallower-rooted annual crops and pastures. Despite significant soil constraints, extensive cropping and pasture systems have developed, benefitting from the existence of regular winter rains. The farming systems generally operate with low inputs of fertilizer, farm chemicals and soil ameliorants. Despite this, the region generates a large proportion of Australia's agricultural exports.

The serious soil management challenges of the region are reasonably well understood and it is clear that the pressures of climate change and degradation of soil resources are combining to threaten the viability of many agricultural businesses. This case study focuses on just a few of the soil management challenges. The account is based almost exclusively on the comprehensive report card for sustainable natural resource use for agriculture (DAFWA, 2013).

The climate of southwest Western Australia is changing and over recent decades mean temperatures have risen and annual rainfall has declined (Figures 15.3(a) and (b)). The pattern of rainfall is also changing with declines in autumn and winter rainfall and increases in spring and summer rainfall. Predictions indicate that these trends will continue and in the short term, year-to-year climate variability may be more important for agriculture than the longer term trends.

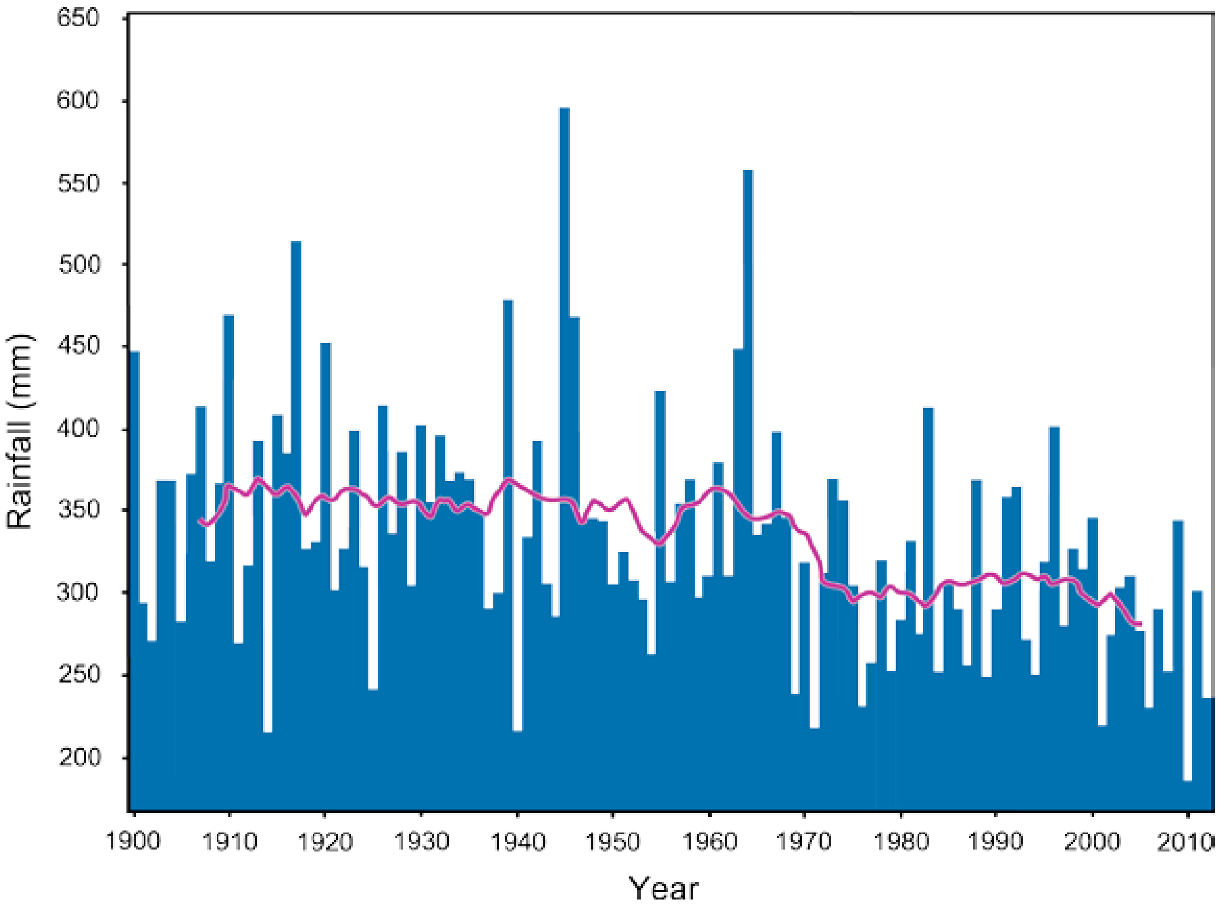


Figure 15.3 (a) Trends in winter rainfall in south-western Australia for the period 1900–2012. Source: Australian Bureau of Meteorology[1].

1 http://www.bom.gov.au/

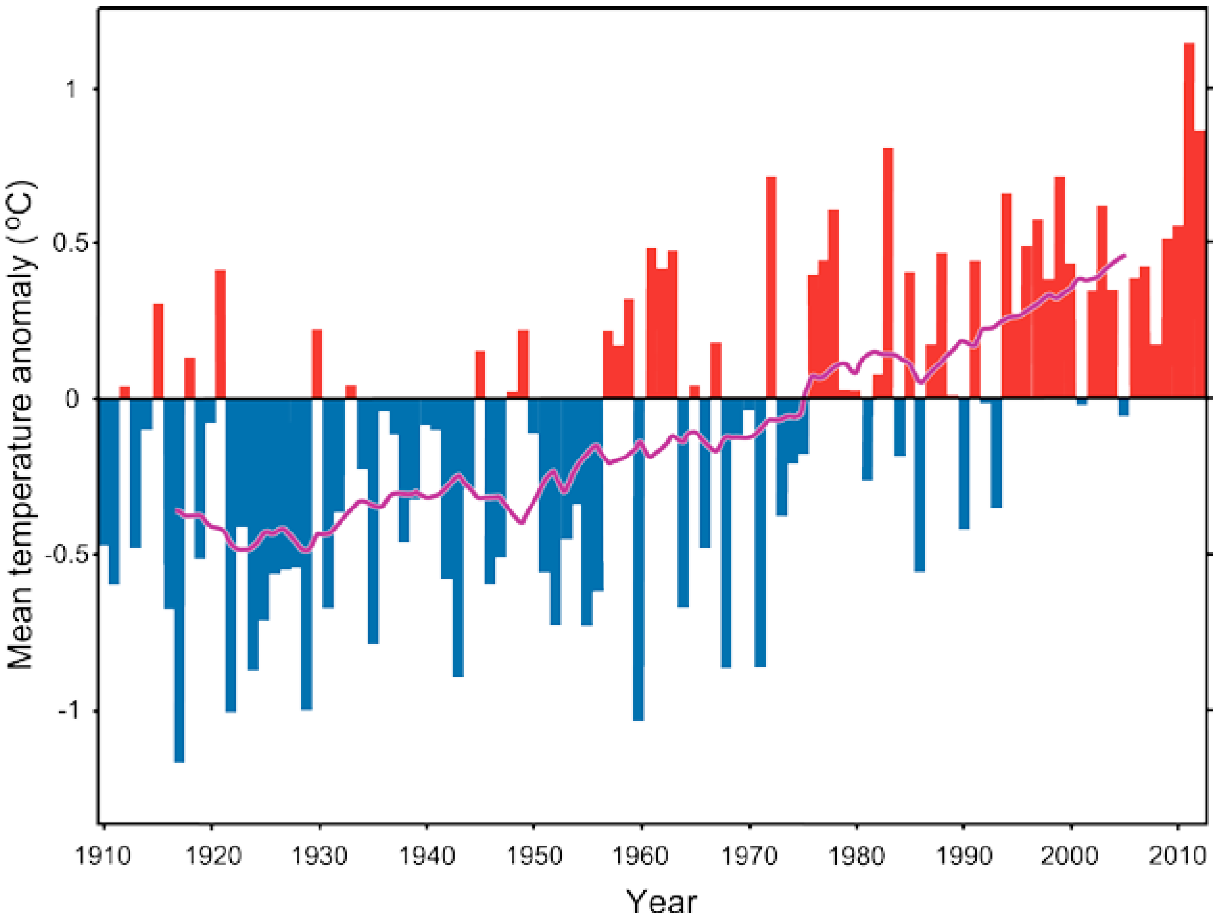


Figure 15.3 | (b) Annual mean temperature anomaly time series map for south-western Australia (1910–2012), using a baseline annual temperature (1961–1990) of 16.3 °C. The 15-year running average is shown by the black line.
Source: Australian Bureau of Meteorology[1].

Soil acidification

Soil acidity in Western Australia is estimated to cost broad-acre agriculture AUD $498 million per year (Herbert, 2009) or about 9 percent of the average annual crop. It is one of the few soil constraints (particularly subsurface constraints) that can be treated with appropriate management.

Between 2005 and 2012 a total of 161 000 samples was collected from over 93 000 sites to determine soil pH (determined in calcium chloride solution - pHCa) status and trend. Figure 15.4 shows the proportion of samples below the nominated targets for Western Australia for the surface layer (0–10 cm) of pHCa 5.5 (desired target) and pHCa 5.0 (critical threshold). Soil acidity is widespread and extreme in many areas of the southwest of Western Australia, particularly in sandy soils. Surface soil pH can be increased to above target (pHCa 5.5) over significant areas with the application of 1 to 3 tonnes ha^{-1} of good quality lime. This will cost from AUD $50 to $150 per ha. If soil surface pH can be raised and maintained above the target this will ensure that management of subsurface acidity will be achieved over time.

There is general recognition that lime use needs to increase and the trend is positive (Figure 15.5). However, current agricultural lime sales are still only 40 percent of the estimated annual amount required to reach the recommended targets over the next 10 years (Gazey *et al.*, 2013).

1 http://www.bom.gov.au/

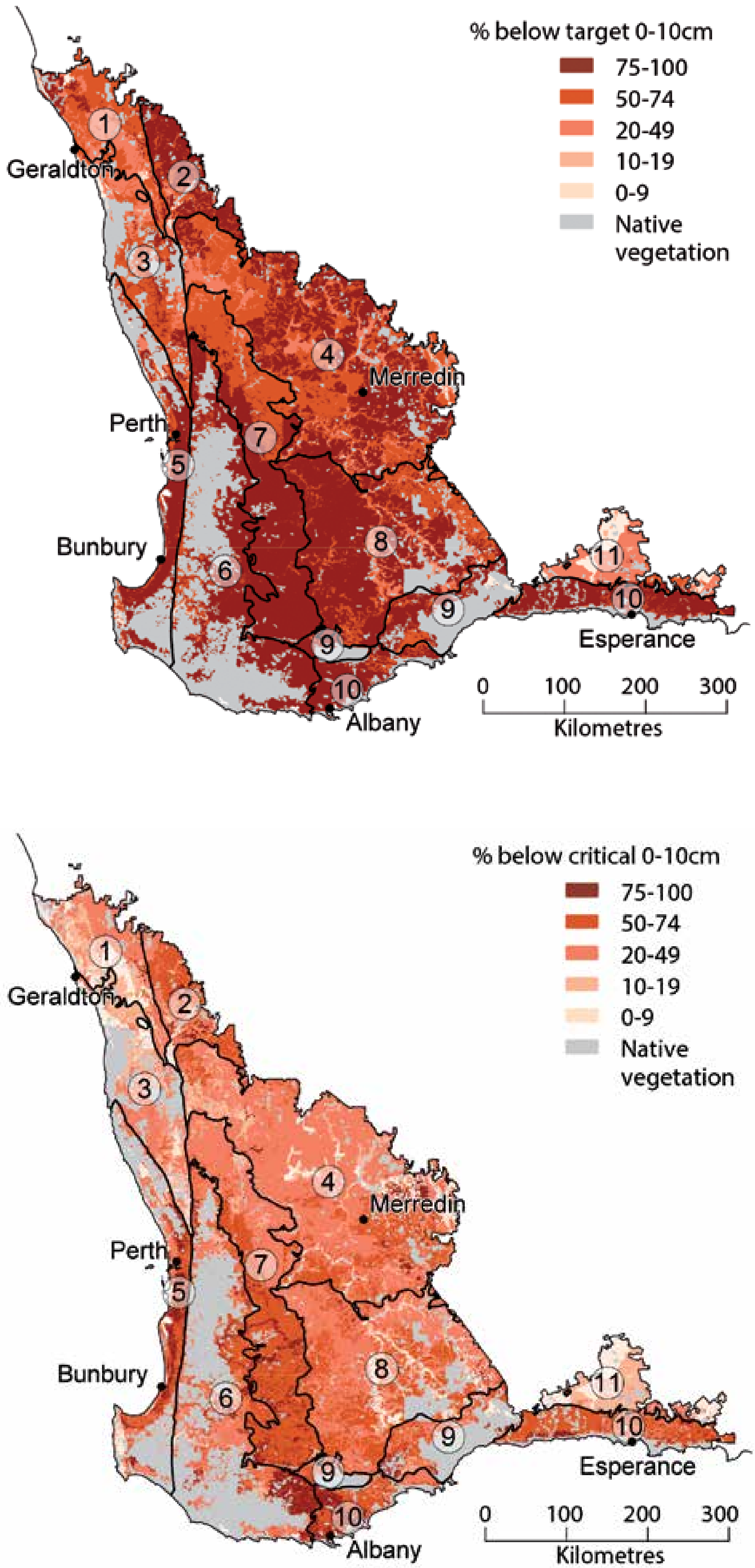


Figure 15.4 | Percentage of sites sampled (2005–12) with soil pH at 0–10 cm depth below the established target of pHCa 5.5 (left) and the critical pHCa 5.0 (right). Grey indicates native vegetation and reserves.
Source: Gazey, Andrew and Griffin, 2013.

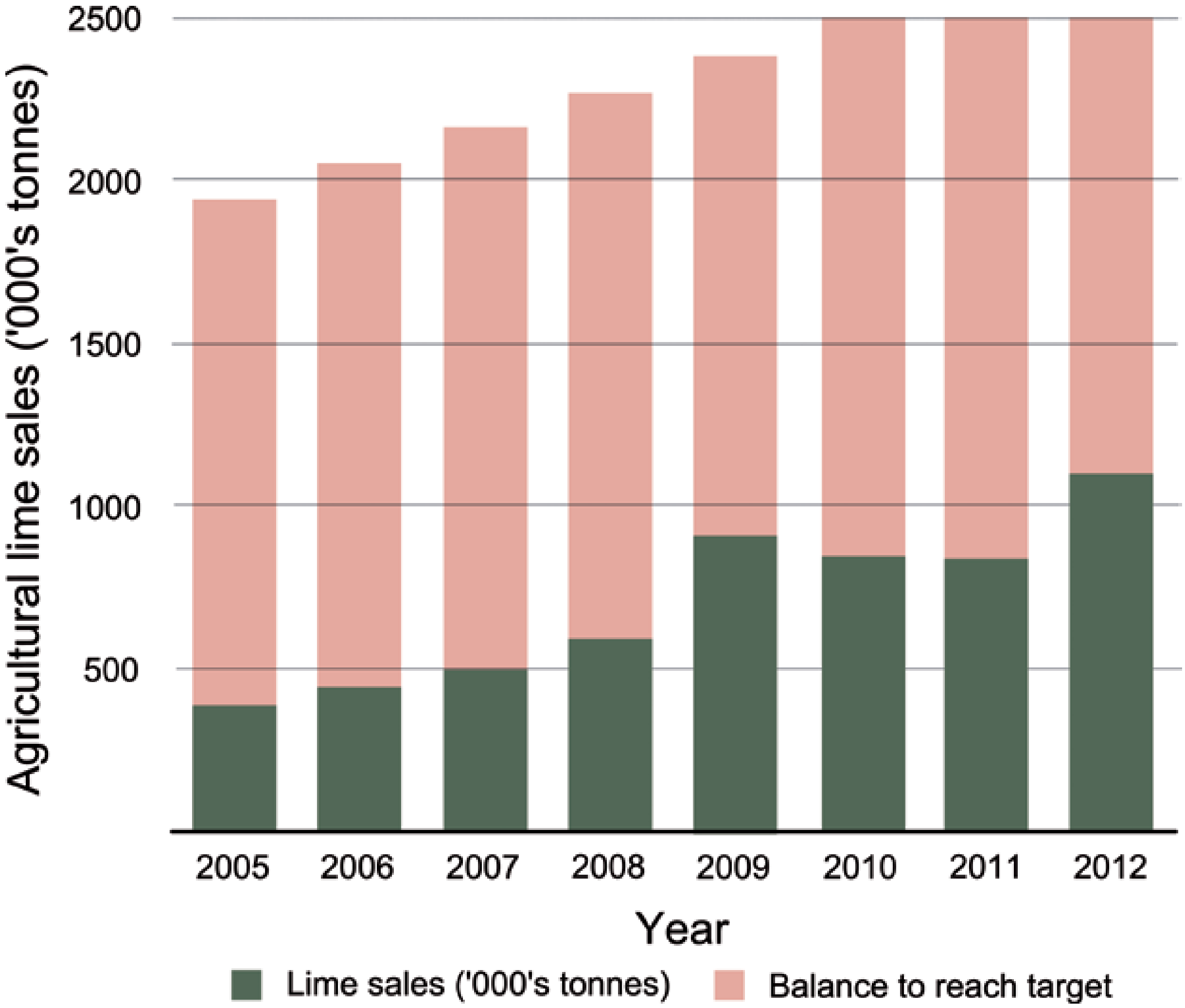


Figure 15.5 | Agricultural lime sales 2005–12 in the south-west of Western Australia based on data for 85–90 percent of the market.

Other soil management challenges

A range of other serious threats to soil function are prevalent in southwest Western Australia. Dryland salinity has been accelerated and enlarged by widespread clearing and land-use change. The problem affects agricultural land, water resources, natural biodiversity and the built environment with an economic cost of hundreds of millions of dollars annually. Currently, more than 1 million ha are severely salt-affected (McFarlane, George and Caccetta, 2004; Furby, Caccetta and Wallace, 2010). Dryland salinity has expanded in most of this region since 1998, especially following episodic rainfall events (George *et al.*, 2008a; Robertson *et al.*, 2010). In areas cleared and developed for agriculture after 1960, most water tables continue to rise, despite a decline in annual rainfall. Dryland salinity is a potential threat to 2.8–4.5 million ha of productive agricultural land (George *et al.*, 2005). The long-term extent of salinity may take decades to centuries to develop (George *et al.*, 2008b). The areas with the highest dryland salinity risk occur mostly in the highly productive, medium to high rainfall dryland agricultural areas. Management to contain or adapt to salinity is technically feasible using plant-based and engineering options, though recovery is economically viable in only a few areas (Simons, George and Raper, 2013).

Wind erosion is a critical issue for the livestock and grain industries. Maintaining sufficient, stable soil cover throughout the year is the challenge and this is difficult in a variable and generally drying climate. Water erosion hazard during the growing season has diminished due to declining winter rains and more sustainable management practices. Water erosion events are mainly caused by intense, localised summer storms, and these are likely to increase with changing climate. Maintaining sufficient soil cover to prevent water erosion is not always possible in grazing systems.

Water repellence is widespread on the sandy soils of the region. The annual average opportunity cost of lost agricultural production is estimated to be AUD $251 million. Water repellence also increases the risk of wind and water erosion, off-site nutrient transport and possibly soil acidification through increased nitrate leaching. The extent and severity of water repellence appears to be increasing as cropping increases together with early sowing, minimum tillage and reduced break of season rainfall.

A warming and generally drying climate associated with lower organic matter inputs could limit future sequestration of organic carbon in soils used for livestock and grains production. Farming systems that increase biomass production and minimise processes such as wind and water erosion are required to maintain and improve soil carbon levels. Soil compaction is suspected to be a major constraint for all agricultural industries in the region but investigations are needed to quantify its extent and severity. Finally, the nutrient status of soils has increased in agricultural systems. On average, pasture soils and arable soils contained 1.3 times and 1.6 times respectively as much phosphorus (P) as is required for optimum production.

15.6.2 | Case study three: Atoll Islands in the Pacific

The fifth assessment report of the IPCC (AR 5) concluded that current and future climate-related drivers of risk for small islands during the 21st century include sea level rise, cyclones, increasing air and sea surface temperatures, and changing rainfall patterns. This confirmed previous IPCC assessments and highlighted the vulnerability of small islands to multiple stressors, both climate and non-climate (Nurse *et al.*, 2014). Limited water and soil resources are additional stressors that contribute to the vulnerability of small islands, and especially atoll islands, in the Southwest Pacific region.

The following is drawn from Morrison's (1990, 2011) reviews of soils on atoll islands. The atoll islands are in essence reefs of variable thickness that have built up by corals resting on a volcanic base. There is a variety of atoll forms but most commonly they occur as a slightly emerged calcareous reef surrounding a lagoon. Low atolls are usually less than 5 m above sea level. Raised atolls (raised coral platforms) can be higher (e.g. Nauru and Niue). Many atolls have deposits of ash or pumice from nearby volcanic activity

The sandy textured soils on most atoll islands are highly calcareous and very permeable. They usually show minimal profile development. The soils are heavily dependent on organic matter to ensure the retention and availability of water and nutrients. Organic carbon can range from 1 to 20 percent in the surface layers depending on the vegetation and position in the landscape. There is minimal organic carbon deeper in the profile. The calcareous mineralogy and associated alkalinity cause a range of plant nutrition problems (e.g. deficiencies of K and micronutrients). The soils are prone to periods of water stress unless irrigated. In summary, the soils are infertile and poorly suited to intensive agriculture.

Population growth (Table 15.2) and increasing urbanization combined with the decline of traditional agriculture has left many of the small island nations largely dependent on imported processed food. This dependency on processed food is a significant factor contributing to poor health associated with the rise of chronic diseases like hypertension and diabetes. Local production of fresh food is therefore a necessity on atoll islands and Deenik and Yost (2006) provide a case study for the Marshall Islands.

The fertility of soils on atoll islands can be improved through the incorporation of composts and organic mulches. This was a central feature of traditional systems. Mechanised mulching and composting systems are being developed on several islands along with various irrigation systems that aim to improve water use efficiency. However, various nutritional and soil management issues have to be resolved, particularly in relation to the suitability of different feed stocks and the unusual soil chemistry of some islands.

The management of water is also fundamental because freshwater reserves on most atoll islands are under significant stress due to excessive groundwater extraction, contamination (e.g. from waste dumps) and saltwater intrusion. Finally, the farming systems on low atoll islands are vulnerable to salinization due to either poor quality groundwater associated with saltwater intrusion or infiltration of seawater after storm surges and flooding.

At present, there is very limited soil science expertise available to help develop more productive and sustainable agricultural systems on atoll islands. White, Falkland and Fatai (2009), in reviewing the management of freshwater lenses on small Pacific islands, conclude that they are some of the most vulnerable aquifer systems in the world. This conclusion applies equally to the soil and food production systems on atoll islands.

15.6.4 | Case study four: DustWatch – an integrated response to wind erosion in Australia

Wind erosion has environmental impacts at the source where soils are eroded (onsite wind erosion), and much greater economic and human health impacts downwind from the source where air quality is reduced (offsite wind erosion). Wind erosion has been a major problem in Australia and it continues to be significant although incidence and severity have declined since the 1940s. The good technical understanding of wind erosion in Australia has relied heavily on the development of DustWatch – a collaborative effort to monitor wind erosion across the country. This case study summarizes the status of wind erosion in Australia and provides a brief introduction to DustWatch.

Wind erosion trends for Australia

Climate is by far the strongest determinant of wind erosion. Land management can either moderate or accelerate wind erosion rates. Unravelling these two influences has been difficult, but between 2001 and 2009 the Millennium Drought – the worst on record (Van Dijk *et al.*, 2013) – provided an opportunity to gauge the effectiveness of improvements in land management that had occurred in recent decades across the country.

Historical accounts indicate that wind erosion was very active during the drought periods of the late 19th and early 20th centuries (e.g. Ratclife, 1938). While these anecdotal reports present dramatic images of huge dust storms engulfing rural towns, and sand drifts burying fence lines and blocking rural roads, until recently it had not been unequivocally established whether the 'dust bowl years' of the 1940s were due to extreme drought, poor land management or both.

McTainsh *et al.* (2011) have analysed wind erosion activity during the 1940s and 2000s. They used archived meteorological data to calculate the dust storm index (DSI) for both periods (McTainsh, 1998, McTainsh *et al.*, 2007). The DSI, which provides a measure of the frequency and intensity of wind erosion activity, is the accepted measure of wind erosion activity in Australia (Bastin, 2008, SOE, 2011). Overall, mean onsite wind erosion in the 1940s was almost six times higher (mean DSI = 11.4) than in the 2000s (mean DSI = 2.0), and the mean maximum DSI for the 1940s was four times that of the 2000s (SOE, 2011).

There were also significant differences between districts. The decrease in wind erosion in the 2000s was much more pronounced in the east and centre of the continent. The 1940s erosion rate in central Australia was large, but the decrease in the 2000s was less than elsewhere. There has also been a large decrease in the number of dust storms reaching the coastal cities.

O'Loingsigh *et al.* (2015) confirmed these findings by using Dust Event Days (DED) and a modified version of a published Dust Storm Index (DSI) to show that wind erosion during the World War II Drought (1937-1945) was up to 4.6 times greater than during the Millennium Drought.

Despite these trends, the Millennium Drought resulted in large dust storms and other wind erosion activity. Two extreme dust storms hit eastern Australian cities on 23rd October 2002 and 23rd September 2009. The 'Red Dawn' dust storm on the 22nd and 23rd September 2009 was the largest to pass over the coastal city of Sydney since reliable records began in 1940 (Figure 15.6). Visibility was reduced to 400 m and the maximum hourly PM_{10} concentration was 15 366 µg m^{-3} – the highest ever recorded for Sydney and possibly for any Australian capital city. The Australian air quality standard of 50 µg m^{-3} per 24 h was massively exceeded in suburban Sydney (1 734 µg m-3) and 150 km to the north in the city of Newcastle (2 426 µg m-3).

Red Dawn was caused by drought and extreme winds. The source of the red dust was primarily the sand plains of western New South Wales, the sand plains, riverine channels and lakes of the lower Lake Eyre Basin and the Channel Country of Queensland. The estimate of total suspended particulate sediment lost off the Australian coast for the 3 000 km long Red Dawn dust storm was 2.54 million tonnes with a plume height of 2 500 m. This is the largest off-continent loss of soil ever reported using measured, as opposed to modelled, dust concentrations for Australia (Leys *et al.*, 2011). This single event resulted in economic costs of between AUD $293–A$313 million with most being associated with household cleaning and associated activities. The dust storm also affected the state of Queensland, but costs there were not included in the study by Tozer and Leys (2013).

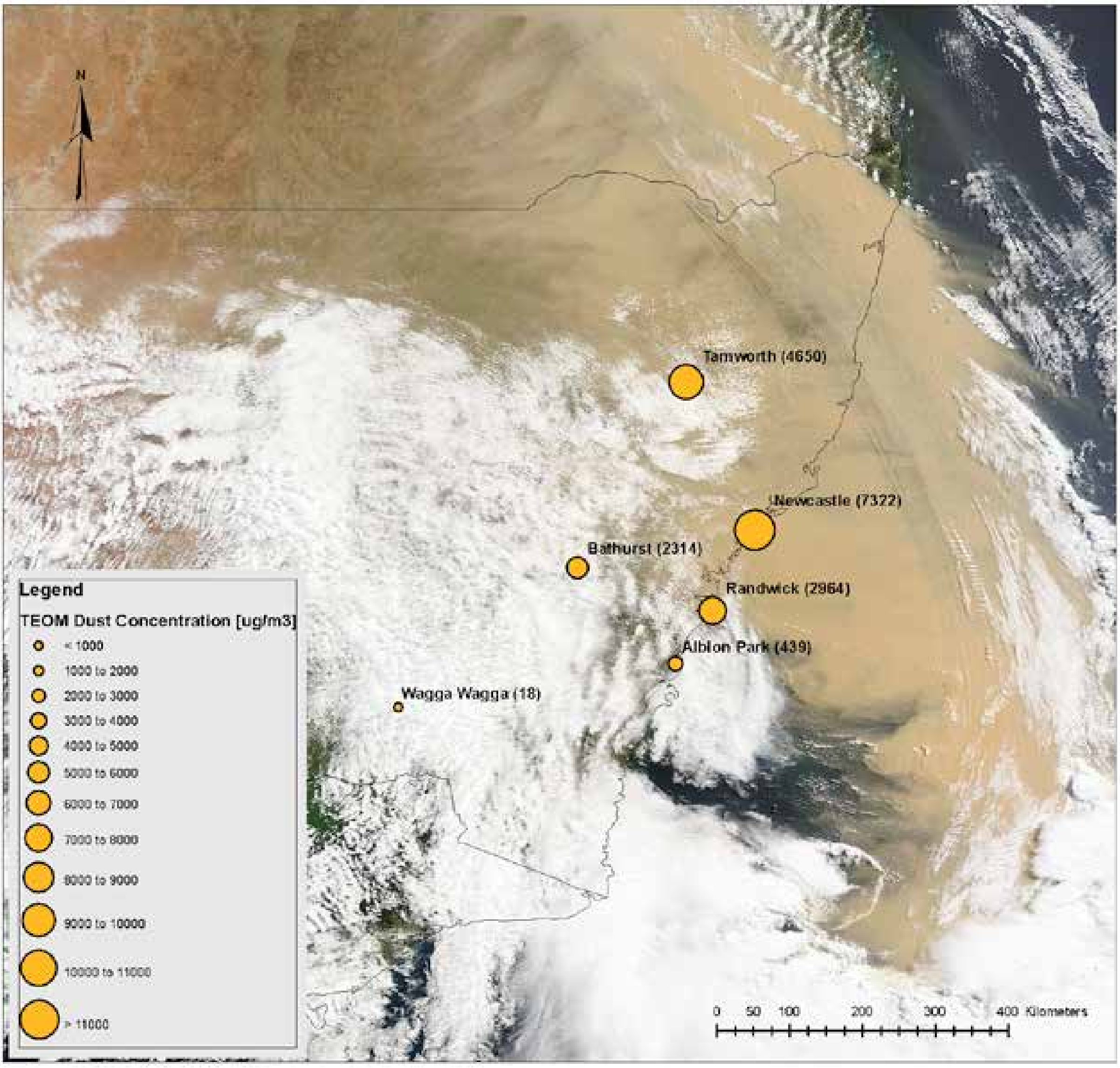


Figure 15.6 | MODIS image for 0000 23 September 2009 showing Red Dawn extending from south of Sydney to the Queensland/ NSW border and the PM 10 concentrations measured using Tapered Element Oscillating Microbalances (TEOM) at the same time at ground stations.

The DustWatch network

Australia has a long tradition of participatory and collaborative approaches to natural resource management. DustWatch (Leys *et al*., 2008, DustWatch, 2014) fits within this broader tradition. It was established to fill some of the gaps in the measurement networks of wind erosion and to supplement land management information provided by community participants.

Measuring and monitoring wind erosion is problematic because of the irregular nature of wind erosion in space and time. DustWatch aimed to build on the long record of observations collected by the Australian Bureau of Meteorology (BoM) by strategically adding a community network of observers and instruments to fill identifiable gaps in the BoM network. DustWatchers use the same basic measurement protocols as the BoM. In addition, satellite imagery is used wherever possible to help define source areas and dust paths. Remote sensing alone has not been successful due to the presence of cloud and the low resolution of geostationary satellites and the low temporal resolution of satellites with better spatial resolution. DustWatch has successfully mixed the formal reporting of wind erosion activity from the BoM with informal community networks, instrumented sites and ad hoc satellite data to provide an understanding of wind erosion in Australia. The major outcomes from DustWatch have been:

- better reporting of the extent and severity of wind erosion across Australia at national to regional scales
- greater dialogue between the scientists and the community
- increased awareness of the impact of wind erosion on the environment
- documentation of local community knowledge on why and where wind erosion is occurring

DustWatch is important because it has provided new data sources when other formal data sources are inadequate. This includes valuable land management information from local landholders. The spatial and temporal data are also provided at the correct scales for testing physical models and remote sensing products. This ultimately has provided the capability for understanding how wind erosion processes are influenced by climate and land management.

15.7 | Conclusions

Based on the above finding, a provisional assessment is made of the status and trend of the 10 soil threats in order of importance for the region. At the same time an indication is given of the reliability of these estimates (Table 15.4).

The status of soil resources in the Southwest Pacific region is mixed. The region is a globally significant exporter of agricultural products and nearly all of the 24 countries rely heavily on soils for wealth generation. The threats to soil function in some countries are serious and require immediate action to avoid large scale economic costs and environmental losses. These threats to soil function combined with other pressures caused by increasing population and climate change are especially challenging in southwest Western Australia and on the atoll islands of the Pacific. It is difficult to assess some threats because of the lack of surveys and monitoring networks. The example of soil monitoring from New Zealand demonstrates the practical value of having the capability to track and respond to soil change. The intensification of land use in New Zealand and to a lesser extent Australia provides an indication of the soil management challenges that will dominate in coming years as countries attempt to substantially increase food production within a resource constrained world. Poor land management practices, and especially uncontrolled logging in the low-income countries in the Southwest Pacific are a significant challenge to national prosperity.

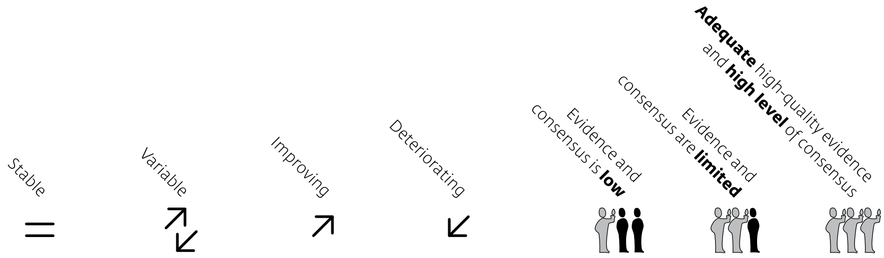


Table 15.4 | Summary of soil threats status, trends and uncertainties in the Southwest Pacific

Threat to soil function	Summary	Condition and Trend					Confidence	
		Very poor	Poor	Fair	Good	Very good	In condition	In trend
Soil acidification	A widespread and serious problem that has the potential to cause irreversible damage to soils particularly in southern Australia, tropical landscapes and areas where product removal and leaching are contributing factors.		↙					
Soil erosion	Improved land management practices in Australia and New Zealand have reduced erosion rates but the problem is still serious in some districts. Unsustainable rates of soil loss are associated with logging and clearing in several Pacific nations			↗				
Organic carbon change	The conversion of land to agricultural uses has generally caused large losses of organic carbon in soils. Improved land management practices have stabilized the situation but there is limited evidence for increasing soil carbon even under these more conservative management systems.		↗↙					
Nutrient imbalance	Rapid intensification of agriculture in New Zealand and more recently Australia is causing significant environmental impact, particularly due to the large increase in fertilizer use and ruminant animals. In other districts, nutrient mining and decline is occurring due to insufficient replacement of nutrients removed through harvest or other loss-pathways.			↙				

Compaction	Limited evidence suggests the problem is constraining plant growth across large areas, particularly in the cropping and pasture lands of Australia and smaller areas in New Zealand. Controlled traffic and other improved management practices may have halted this decline in soil physical fertility.			=				
Soil sealing and capping	Loss of good quality agricultural land due to urban and industrial expansion is an emerging and potentially major problem for all countries in the region.				↙			
Contamination	Most sources of soil contamination are now regulated and controlled although the legacy of past practices is significant (e.g. Cd in fertilizers). Contamination caused by mining and waste disposal is a significant issue for several Pacific nations.				↗			
Salinization and sodification	Salinization is a widespread and expensive problem in Australia and some atoll islands. After a temporary respite due to dry years, the problem may continue to expand and the time to equilibration is likely to be in the order of decades.			=				
Loss of soil biodiversity	Rates of loss were most likely highest during the expansion of agriculture, particularly over the last 100 years, and this may have slowed. However, information on baselines and trends is lacking in nearly all districts and countries.			↗↙				
Waterlogging	Waterlogging is a constraint to agricultural production in some wet years but evidence on the extent and severity is lacking. Large areas were drained to address the problem, particularly in New Zealand and parts of coastal Australia.				=			

References

Abbott, L.K. & Murphy, D.V. 2007. Soil biological fertility: a key to sustainable land use in agriculture. Boston, Dordrecht, Kluwer Academic Publishers.

Asquith, F., Kooge, M. & Morrison, R.J. 1994. Transportation of sediments via rivers to the ocean and the role of sediments as pollutants in the South Pacific. Reports and Studies No 72 Apia. South Pacific Regional Environmental Programme.

Badgery, W.B., Simmons, A.T., Murphy, B.M., Rawson, A., Andersson, K.O., Lonergan, V.E. & van de Ven, R. 2013. Relationship between environmental and land-use variables on soil carbon levels at the regional scale in central New South Wales. Australia. *Soil Research*, 51: 645-656.

Baldock, J., Macdonald, L. & Sanderman, J. 2013. Foreword to Soil Carbon in Australia's Agricultural Lands. *Soil Research*, 51.

Balks, M.R., Bond, W.J. & Smith, C.J. 1998. Effects of sodium accumulation on soil physical properties under and effluent-irrigated plantation. *Aust. J. Soil Res.*, 36: 821–830.

Barzi F., Naidu R. & McLauglin M.J. 1996. Contaminants and the Australian soil environment. In: R. Naidu, R.S. Kookana, D.P. Oliver, S. Rogers & M.J. McLaughlin, eds. *Contaminants and the soil environment in the Australasia-Pacific regions*. pp. 451-484.Springer Netherlands.

Bastin, G. 2008. *ACRIS Management Committee. Rangelands 2007: taking the pulse*. Canberra, National Land & Water Resources Audit.

Beare, M.H., McNeil, S.J., Curtin, D., Parfitt, R.L., Jones, H.S., Dodd, M.B. & Sharp J. 2014. Estimating the organic carbon stabilisation capacity and saturation deficit of soils: a New Zealand case study. *Biogeochemistry*, 120: 71-87.

Bolan, N.S. & Hedley, M.J. 2003. Role of carbon, nitrogen, and sulfur cycles in soil acidification. *In Z. Rengel, ed. Handbook of Soil Acidity*, pp. 29-56. USA, New York, Marcel Dekker, Inc.

Bolton, B. 2009. *The Fly River, Papua New Guinea: environmental studies in an impacted tropical river system. Developments in Earth and Environmental Sciences. Vol.* 9. USA, Burlington, MA, Elsevier. pp 1-620.

Bond, W.J. 1998. Effluent irrigation – an environmental challenge for soil science. Aust. J. Soil Res., 36: 543–555.

Bowler, J.M., Johnston, H., Olley, J.M., Prescott, J.R., Roberts, R.G., Shawcross, W. & Spooner, N.A. 2003. New ages for human occupation and climatic change at Lake Mungo. *Australia. Nature*, 421: 837–840.

Braunack, M.V. & Johnston, D.B. 2014. Changes in soil cone resistance due to cotton picker traffic during harvest on Australian cotton soils. *Soil & Tillage Research*, 140: 29-39.

Braunack, M.V., Arvidsson, J. & Hakansson, I. 2006. Effect of harvest traffic position on soil conditions and sugarcane (Saccharum officinarum) response to environmental conditions in Queensland, Australia. *Soil & Tillage Research*, 89: 103-121.

Bridge, B.J. & Bell, M.J. 1994. Effect of Cropping on the Physical Fertility of Krasnozems. *Australian Journal of Soil Research*, 32: 1253-1273.

Brodie, J., Waterhouse, J., Schaffelke, B., Kroon, F., Thorburn, P., Rolfe, J., Johnson, J., Fabricius, K., Lewis, S., Devlin, M., Warne, M. & McKenzie, L. 2013. 2013 *Scientific consensus statement. Land use impacts on Great Barrier Reef water quality and ecosystem condition*. State Government of Queensland.

Bui, E., Hancock, G., Chappell, A. & Gregory, L. 2010. *Evaluation of tolerable erosion rates and time to critical topsoil loss in Australia*. Canberra, CSIRO Sustainable Agriculture Flagship.

Bui, E.N., Hancock, G.J. & Wilkinson, S.N., 2011. 'Tolerable' hillslope soil erosion rates in Australia: Linking science and policy. *Agriculture Ecosystems & Environment*, 144: 136-149.

Callander, P., Lough, H. & Steffens, C. 2011. *New Zealand Guidelines for the Monitoring and Management of Sea*. Envirolink Project 420-NRLC 50. New Zealand, Christchurch.

Campbell, D.I., Wall, A.A., Nieveen, J.P. & Schipper L.A. 2015. Variations in CO_2 exchange for dairy farms with year-round rotational grazing on peat soils. *Agriculture, Ecosystems and Environment*, 202: 68–78.

Campbell, I.C. 2011. Science, governance and environmental impacts of mines in developing countries: lessons from Ok Tedi in Papua New Guinea. In R.Q. Grafton & K. Hussey, eds. *Water Resources Planning and Management*, pp. 583-598.. Cambridge University Press.

Carter, M.R. & Gregorich, E.G. 2010. Carbon and nitrogen storage by deep-rooted tall fescue (Lolium arundianaceum) in the surface and subsurface soil of a fine sandy loam in eastern Canada. *Agriculture Ecosystems and Environment*, 136: 125-132.

Cartwright, B., Merry, R.H. & Tiller, K.G. 1976. Heavy metal contamination of soils around a lead smelter at Port Pirie, South Australia. *Aust. J. Soil Res.*, 15: 69-81.

Cavanagh, J. 2014. *Status of cadmium in New Zealand soils*: 2014. Lincoln, Landcare Research.

Chan, K.Y., Oates, A., Swan, A.D., Hayes, R.C., Dear, B.S. & Peoples, M.B. 2006. Agronomic consequences of tractor wheel compaction on a clay soil. *Soil & Tillage Research*, 89: 13-21.

Chen, C. & Randall, A. 2013. The economic contest between coal seam gas mining and agriculture on prime farmland: it may be closer than we thought. *Journal of Economic and Social Policy*, 15 (available at http://epubs.scu.edu.au/jesp/vol 15/iss 3/5)

Condron, L.M. & Di, H.J. 2002. Capacity of soils to sustain or extend current crop and animal production: New Zealand and South Pacific Islands Perspective. In R. Lal, ed. *Agricultural Sciences*. Encyclopaedia of Life Support Systems, UNESCO. UK, Oxford, EOLSS Publishers.

Condron, L.M., Hopkins, D.W., Gregorich, E.G., Black, A. & Wakelin, S.A. 2014. Long-term irrigation effects on soil organic matter under temperate grazed pasture. *European Journal of Soil Science*, 65: 741-750.

Cotching, W.E., Oliver, G., Downie, M., Corkrey, R. & Doyle, R.B. 2013. Land use and management influences on surface soil organic carbon in Tasmania. *Soil Research*, 51: 615-630.

Cowie, A.L., Lonergan, V.E., Rabbi, S.M.F., Fornasier, F., Macdonald, C., Harden, S., Kawasaki, A. & Singh, B.K. 2013. Impact of carbon farming practices on soil carbon in northern New South Wales. *Soil Research*, 51: 707-718.

CSIRO. 1983. *Soils: an Australian viewpoint.* Melbourne, CSIRO & London, Academic Press.

DAFWA. 2013. *Report card on sustainable natural resource use in agriculture*. Department of Agriculture and Food, Western Australia.

Dalal, R.C. & Mayer, R.J. 1986. Long term trends in fertility of soils under continuous cultivation and cereal cropping in southern Queensland. I. Overall changes in soil properties and trends in winter cereal yields. *Australian Journal of Soil Research*, 24: 265-279.

Davy, M.C., Koen, T.B. 2013. Variations in soil organic carbon for two soil types and six land uses in the Murray Catchment, New South Wales, Australia. *Soil Research*, 51: 631-644.

de Klein, C.A.M., Monaghan, R.M. & Sinclair, A.G. 1997. Soil acidification: a provisional model for New Zealand pastoral systems. *New Zealand Journal of Agricultural Research*, 40: 541–557.

De Vries, W. & McLaughlin, M.J. 2013. Modeling the Cadmium balance in Australian agricultural systems in view of potential impacts on food and water quality. *Sc. Total Environ*. 461-462: 240-257.

Deenik, J.L. & Yost, R.S. 2006. Chemical properties of atoll soils in the Marshall Islands and constraints to crop production. *Geoderma*, 136: 66-81.

Denham, T.P., Haberle, S.G., Lentfer, C., Fullagar, R., Field J., Therin, M., Porch, N. & Winsborough, B. 2003. Origins of Agriculture at Kuk Swamp in the Highlands of New Guinea. *Science*, 301: 189-193.

DEST. 2003. *Rehabilitation of former nuclear test sites at Emu and Maralinga (Australia*) 2003. Report by the Maralinga Rehabilitation Technical Advisory Committee. Canberra, Australian Government Department of Education, Science and Training.

Dodd, M.B. & Mackay, A.D. 2011. Effects of contrasting soil fertility on root mass, root growth, root decomposition and soil carbon under a New Zealand perennial ryegrass/white clover pasture. *Plant and Soil*, 349: 291-302.

Drewry, J.J. 2006. Natural recovery of soil physical properties from treading damage of pastoral soils in New Zealand and Australia: a review. *Agriculture, Ecosystems and Environment*, 114: 159–169.

DustWatch. 2014. *Official web site*. (available at www.dustwatch.edu.au)

Dymond, J.R. 2010. Soil erosion in New Zealand is a net sink of CO_2. *Earth Surface Processes and Landforms*, 35(15): 1763-1772.

Eyles, G.O. 1983. The distribution and severity of present soil erosion in New Zealand. *New Zealand Geographer*, 39: 12-28.

FAO. 1988. Salt affected soils and their management. FAO Soils Bulletin 39. Rome, FAO.

Friedlaender, J.S., Friedlaender, F.R., Reed, F.A., Kidd, K.K., Kidd, J.R., Chambers, G.K., Lea R.A., Loo, J-H., Koki, G., Hodgson, J.A., Merriwether, D.A. & Weber, J.L. ,2008. The Genetic Structure of Pacific Islanders. *PLoS Genet*, 4(1): e 19. doi:10.1371/journal.pgen.0040019

Furby, S.L., Caccetta, P.A. & Wallace, J.F. 2010. Salinity monitoring in Western Australia using remotely sensed and other spatial data. *J. Environ. Qual.*, 39: 16–25.

Gazey, C., Andrew, J. & Griffin, E. 2013. Soil acidity. *In Report card on sustainable natural resource use in agriculture.* Department of Agriculture and Food, Western Australia.

Geeves, G.W., Cresswell, H.P., Murphy, B.W., Gessler, P.E., Chartres, C.J., Little, I.P. & Bowman, G.M. 1995. *The physical, chemical and morphological properties of soils in the wheat-belt of southern NSW and northern Victoria*. Department of Conservation and Land Management, NSW & Aust. Division of Soils Occasional Report, CSIRO.

George, R.J., Clarke, J.D.A. & English, P. 2008b. Modern and palaeogeographic trends in the salinisation of the Western Australian Wheatbelt. *Australian Journal of Soil Research*, 46: 751-767.

George, R.J., Kingwell, R., Hill-Tonkin, J. & Nulsen, R. 2005. *Salinity investment framework: Agricultural land and infrastructure*. Resource management technical report 270. Department of Agriculture, Western Australia.

George, R.J., Speed, R.J., Simons, J.A., Smith, R.H., Ferdowsian, R., Raper, G.P. & Bennett, D.L. 2008a. *Long-term groundwater trends and their impact on the future extent of dryland salinity in Western Australia in a variable climate.* 2nd International Salinity Forum: Salinity, water and society - Global issues, local action. Adelaide, South Australia 31 March – 3 April.

Glatthaar, D. 1988. The sediment load of the Waimanu River, Southeastern Viti Levu, Fiji. In H. Liedtke & D. Glatthaar, eds. *Report about two research projects in the Republic of Fiji*, pp. 52-75. Sponsored by the German Research Foundation (Deutsche Forschungsgemeinschaft, Bonn, Federal Republic of Germany). Bochum, Ruhr-University Bochum, Geographical Institute.

Hartemink, A.E. 1998a. Soil chemical and physical properties as indicators of sustainable land management under sugar cane in Papua New Guinea. *Geoderma*, 85: 283-306.

Hartemink, A.E. 1998b. Acidification and pH buffering capacity of alluvial soils under sugarcane. *Experimental Agriculture*, 34: 231-243.

Herbert A. 2009. 'Opportunity cost of land degradation hazard in the south-west agricultural region', Resource Management Technical Report 349, Department of Agriculture and Food, Western Australia.

Hill, R.B., Sparling, G., Frampton, C. & Cuff, J. 2003. National Soil Quality Review and Programme Design. Technical Paper 75, Land. Ministry for the Environment, Wellington

HortNZ. 2014. Industry statistics. (available athttp://www.hortnz.co.nz/Overview/About-Us/Industry-statistics.htm, accessed November 2014)

Ivkovic, K.M., Marshall, S.K., Morgan, L.K., Werner, A.D., Carey, H., Cook, S., Sundaram, B., Norman, R., Wallace, L., Caruana, L., Dixon-Jain, P. & Simon, D. 2012. *National-scale vulnerability assessment of seawater intrusion: summary report. Waterlines report*. Canberra, National Water Commission.

Jones, C.A. 1983. Effect of soil texture on critical bulk densities for root growth. *Soil Science Society of America Journa*l, 47: 1208-1211.

Kelliher, F.M., Condron, L.M., Cook, F.J. & Black, A. 2012. Sixty years of seasonal irrigation affects carbon storage in soils beneath pasture grazed by sheep. *Agriculture Ecosystems and Environment*, 148: 29-36.

Kelliher, F.M., Curtin, D., Condron, L.M. 2013. Soil carbon stocks in particle-size fractions under seasonally irrigated, grazed pasture. *New Zealand Journal of Agricultural Research*, 56: 239-244.

Kingwell, R. & Fuchsbichler, A. 2011. The whole-farm benefits of controlled traffic farming: An Australian appraisal. *Agricultural Systems*, 104: 513-521.

Leslie, D.M. & Ratukalou, I. 2002. *Review of rural land use in Fiji: opportunities for the new millennium*. Suva, Secretariat of the Pacific Community.

Letey, J. 1985. Relationship between soil physical properties and crop production. In B.A. Stewart, ed. Advances *in soil science*. Vol. 5, pp. 277-294. Springer.

Leys, J., McTainsh, G., Strong, C., Heidenreich, S. & Biesaga, K. 2008. DustWatch: using community networks to improve wind erosion monitoring in Australia. *Earth Surface Processes and Landforms*, 33: 1912-1926.

Leys, J.F., Heidenreich, S.K., Strong, C.L., McTainsh, G.H. & Quigley, S. 2011. PM 10 concentrations and mass transport during 'Red Dawn' – Sydney, 23 September 2009. *Aeolian Research*, 3: 327-342.

Liedtke, H. 1989. Soil erosion and soil removal in Fiji. *Applied Geography and Development*, 33: 68-92.

Lockwood, P., Wilson, B., Daniel, H. & Jones, M. 2003. *Soil acidification and natural resource management: directions for the future*. NSW, Armidale, University of New England..

Löffler, E. 1977. *Geomorphology of Papua New Guinea*. Canberra, CSIRO & Australian National University Press.

Löffler, E. 1979. *Papua New Guinea*. Australia, Hutchinson.

Loganathan, P., Hedley, M.J., Grace, N.D., Lee, J., Cronin, S.J., Bolan, N.S. & Zanders, J.M. 2003. Fertiliser contaminants in New Zealand grazed pasture with special reference to cadmium and fluorine — a review. *Australian Journal of Soil Research*, 41: 501-532.

Loveday, J. & Bridge, B.J. 1983. Management of salt-affected soils. *In Soils* an Australian viewpoint. pp. 843-856. CSIRO, Australia.

MacLeod, C.J. & Moller, H. 2006. Intensification and diversification of New Zealand agriculture since 1960: an evaluation of current indicators of land use change. *Agriculture, Ecosystems and Environment*, 115: 201–218.

MAF. 2011. *Cadmium and New Zealand agriculture and horticulture: a strategy for long term risk management*. Wellington, New Zealand Ministry of Agriculture and Forestry.

Maragos, J.E., Baines, G.B.K. & Beveridge, P.J. 1973. Tropical cyclone Bebe creates a new land form in Funafuti atoll. *Science*, 1981: 1161-1164.

McDonald, G.K. & Gardner, W.K. 1987. Effect of waterlogging on the grain yield response of wheat to sowing data in south-western Victoria. *Australian Journal of Experimental Agriculture*, 27: 661-670.

McFarlane, D.J., George, R.J. & Caccetta, P.A. 2004. *The extent and potential area of salt-affected land in Western Australia estimated using remote sensing and Digital Terrain Models*. Proc. of Engineering Salinity Solutions, Perth, Western Australia. 9–12 November. Australian Capital Territory, Barton, Institution of Engineers.

McGarry, D. 1990. Soil compaction and cotton growth on a Vertisol. *Aust. J. Soil Research*, 28: 869 877.

McGarry, D., Sharp, G. & Bray, S.G. 1999. *The Current Status of Soil Degradation in Queensland Cropping Soils*. Queensland Department of Natural Resources, DNRQ 990092. 16 pp.

McGlone, M.S. 1989. The Polynesian settlement of New Zealand in relation to environmental and biotic changes. *New Zealand Journal of Ecology*, 12: 115-129.

McIvor, J.G., Guppy, C. & Probert, M.E. 2011. Phosphorus requirements of tropical grazing systems: the northern Australian experience. *Plant and Soil*, 349: 55-67.

McLaughlin, M.J., Tiller, K.G., Naidu, R. & Stevens, D.P. 1996. Review: the behaviour and environmental impact of contaminants in fertilizers. *Australian Journal of Soil Research*, 34: 1-54.

McLaughlin, M.J., Fillery, I.R., Till, A.R. 1990. Operation of the phosphorus, sulphur and nitrogen cycles. In: R.M. Gifford, M.M. Barton, eds. *Australia's renewable resources: sustainability and global change*. IGBP, Australia Planning Workshop.

McLaughlin, M.J., McBeath, T.M., Smernik, R., Stacey, S.P., Ajiboye, B. & Guppy, C. 2011. The chemical nature of P accumulation in agricultural soils-implications for fertiliser management and design: an Australian perspective. *Plant and Soil*, 349: 69-87.

McTainsh, G. 1998. *Dust storm index. In: Sustainable agriculture: assessing Australia's recent performance*. A report to the Standing Committee on Agriculture and Resource Management of the National Collaborative Project on Indicators for Sustainable Agriculture. Collingwood, CSIRO Publishing.

McTainsh, G., Leys, J., O'Loingsigh, T. & Strong, C. 2011. *Wind erosion and land management in Australia during 1940–1949 and 2000–2009*. Report prepared for the Australian Government Department of Sustainability, Environment, Water, Population and Communities on behalf of the State of the Environment 2011 Committee. Canberra, DSEWPaC.

McTainsh, G., Tews, E., Leys, J. & Bastin, G. 2007. *Spatial and temporal trends in wind erosion of Australian rangelands during 1960 to 2005 using the dust storm index* (DSI). Australian Collaborative Rangeland Information System. Canberra, Australian Government Department of the Environment, Water, Heritage and the Arts.

MDBA. 2010. *Murray-Darling Basin Authority*. Report of the Independent Audit Group for Salinity 2009-10. Canberra, MDBA. (available at http://www.mdba.gov.au/sites/default/files/pubs/Report-of-the-Independent-Auditors-for-Salinity-2009-10.pdf)

Ministry for the Environment. 2010. *Proposed national environmental standard for assessing and managing contaminants in soil:* discussion document. Wellington, New Zealand Ministry for the Environment.

Moir, J.L. & Moot, D.J. 2010. Soil pH, exchangeable aluminium and lucerne yield responses to lime in a South Island high country soil. *Proceedings of the New Zealand Grassland Association*, 72: 191-196.

Monaghan, R.M., Houlbrooke, D.J. & Smith, L.C. 2010. The use of low-rate sprinkler application systems for applying farm dairy effluent to land to reduce contaminant transfers. *New Zealand Journal of Agricultural Research*, 53: 389–402

Moorehead, A. (ed.). 2011. *Forests of the Pacific Islands: foundation for a sustainable future*. Suva, Secretariat of the Pacific Community.

Morrison, R.J. & Gawander, J.S. (in press). Changes in the properties of Fijian Oxisols over 30 years of sugarcane cultivation. *Soil Research*.

Morrison, R.J. & Munro, A.J. 1999. Waste management in the small island developing states of the South Pacific: an overview. *Australian Journal of Environmental Management*, 6: 232-246.

Morrison, R.J. & Seru, V.B. 1985. *Soils of Abatao Islet, Tarawa, Kiribati*. INR Environmental Studies Report No. 27. University of the South Pacific. 81 pp.

Morrison, R.J. 1990. Pacific atoll soils: chemistry, mineralogy and classification. *Atoll Research Bulletin*, 339: 1-25.

Morrison, R.J. 2011. Soils of low elevation coral structures. In D. Hopley, ed. *Encyclopedia of Modern Coral Reefs*, pp 1019-1024. New York, Springer.

Morrison, R.J., Gangaiya, P. & Koshy, K. 1996. Contaminated soils in the South Pacific islands. *In Contaminants and the Soil Environment in the Australasia-Pacific Region* , pp. 659-675. Netherlands, Springer.

Morrison, R.J., Gawander, J.S. & Ram, A.N. 2005. Changes in the properties of a Fijian Oxisol over 25 years of sugarcane cultivation. In D. M. Hogarth, ed. *Proceedings International Society of Sugar Cane Technologies XXV Congress*, Vol. 2, pp. 139-146. Guatemala.

Morrison, R.J., Manner, H.I. 2005. The pre-mining pattern of soils on Nauru, Central Pacific. *Pacific Science*, 59: 523-540.

Morrison, R.J., Sivan, P., Krishnamurthi, M., Deo, V., Vula, E. & Jogia, D.L. 1985. *Report of the Cabinet Committee investigating the fertilizer procurement and distribution practices of the Fiji Sugar Corporation*. Report to the Fiji Cabinet. 58 pp.

Mudge, P.L., Wallace, D.F., Rutledge, S., Campbell, D.I., Schipper, L.A. & Hosking, C.L. 2011. Carbon balance of an intensively grazed temperate pasture in two climatically contrasting years. *Agriculture Ecosystems and Environment*, 144: 271-280.

Naidu, R., Kookana, R.S., Oliver, D.P., Rogers, S. & McLaughlin, M.J. 1996. Contaminants and the soil environment in the Australasia-Pacific region. Dordrecht, Kluwer Academic.

Nieeven, J.P., Campbell, D.I., Schipper, L.A. & Blair, I.J. 2005. Carbon exchange of grazed pasture on a drained peat soil. *Global Change Biology*, 11: 607-618.

NLWRA. 2001a. *Australian agricultural assessment 2001*. Volume 1. Canberra, National Land and Water Resources Audit.

NLWRA. 2001b. *Australian dryland salinity assessment 2000: extent, impacts, processes, monitoring and management options*. Canberra, National Land and Water Resources Audit.

Noble, A.D., Cannon, M. & Muller, D. 1997. Evidence of accelerated soil acidification under Stylosanthes-dominated pastures. *Australian Journal of Soil Research*, 35: 1309-1322.

Noble, A.D., Middleton, C., Nelson, P.N. & Rogers, L.G. 2002. Risk mapping of soil acidification under Stylosanthes in northern Australian rangelands. *Australian Journal of Soil Research*, 40: 257-267.

Nurse, L.A., McLean, R.F., Agard, J., Briguglio, L.P., Duvat-Magnan, V., Pelesikoti, N., Tompkins, E. & Webb, A. 2014. Small islands. In V.R. Barros, C.B. Field, D.J. Dokken, M.D. Mastrandrea, K.J. Mach, T.E. Bilir, M. Chatterjee, K.L. Ebi, Y.O. Estrada, R.C. Genova, B. Girma, E.S. Kissel, A.N. Levy, S. MacCracken, P.R. Mastrandrea & L.L.White, eds. *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects*, pp. 1613-1654. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. UK, Cambridge, Cambridge University Press & USA, New York, NY.

O'Connell, J.F. & Allen, J. 2004. Dating the colonization of Sahul (Pleistocene Australia–New Guinea): a review of recent research. *Journal of Archaeological Science*, 31: 835-853.

Ollier, C.D. 1982. The Great Escarpment of eastern Australia: tectonic and geomorphic significance. *Journal of the Geological Society of Australia*, 29: 13–23.

O'Loingsigh, T., McTainsh, G.H., Parsons, K., Strong, C.L., Shinkfield, P. & Tapper, N.J.2015. Using meteorological observer data to compare wind erosion during two great droughts in eastern Australia; the World War II Drought (1937–1946) and the Millennium Drought (2001–2010). *Earth Surface Processes and Landforms*, 40: 123-130.

Page, K.L., Dalal, R.C., Pringle, M.J., Bell, M., Dang, Y.P., Radford, B. & Bailey, K. 2013. Organic carbon stocks in cropping soils of Queensland, Australia, as affected by tillage management, climate, and soil characteristics. *Soil Research*, 51: 584-595.

Page, M.J., Trustrum, N.A., Brackley, H. & Baisden, W.T. 2004. Erosion-related soil carbon fluxes in a pastoral steepland catchment, *New Zealand. Agriculture Ecosystems and Environment*, 103: 561-579.

Pankhurst, C.E., Doube, B.M., Gupta V.V.S.R. & Grace, P.R.1994. Soil biota: management in sustainable farming systems. East Melbourne, CSIRO.

Parfitt, R.L., Baisden, W.T., Ross, C.W., Rosser, B.J., Schipper, L.A. & Barry, B. 2013. Influence of erosion and deposition on carbon and nitrogen accumulation in resampled steepland soils under pasture in New Zealand. *Geoderma*, 192: 154-159.

Parfitt, R.L., Steveson, B.A., Ross, C. & Fraser, S. 2014. Change in pH, bicarbonate-extractable-P, carbon and nitrogen in soils under pasture over 7 to 27 years. *New Zealand Journal of Agricultural Research*, 57: 216-227.

Rab, M.A. 2004. Recovery of soil physical properties from compaction and soil profile disturbance caused by logging of native forest in Victorian Central Highlands, Australia. *Forest Ecology and Management*, 191: 329-340.

Ratclife, F. 1938. *Flying fox and drifting sand: the adventures of a biologist in Australia.* New York, RM McBride & Co.

Reisinger, A., Kitching, R.L., Chiew, F., Hughes, L., Newton, P.C.D., Schuster, S.S., Tait, A. & Whetton, P. 2014. Australasia. In V.R. Barros, C.B. Field, D.J. Dokken, M.D. Mastrandrea, K.J. Mach, T.E. Bilir, M. Chatterjee, K.L. Ebi, Y.O. Estrada, R.C. Genova, B. Girma, E.S. Kissel, A.N. Levy, S. MacCracken, P.R. Mastrandrea & L.L.White, eds. *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects*, pp. 1371-1438. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. UK, Cambridge, Cambridge University Press & USA, New York, NY.

Rengasamy, P. 2002. Transient salinity and subsoil constraints to dryland farming in Australian sodic soils: an overview. *Australian Journal of Experimental Agriculture*, 42: 351–361.

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

Roberts, A.H.C., Cameron, K.C., Bolan, N.S., Ellis, H.K. & Hunt, S. 1996. Contaminants and the soil environment in New Zealand. *In Contaminants and the soil environment in the Australasia-Pacific Region*, pp. 579-628. The Netherlands, Springer.

Roberts, R.G., Jones, R. & Smith, M.A. 1990. Thermoluminscence dating of a 50 000-year-old human occupation site in northern Australia. *Nature*, 345: 153–156.

Roberts, R.G., Jones, R., Spooner, N.A., Head, M.J., Murray, A.S. & Smith, M.A., 1994. The human colonisation of Australia: optical dates of 53 000 and 60 000 years bracket human arrival at Deaf Adder Gorge, Northern Territory. *Quaternary Science Reviews*, 13: 575–583.

Robertson, M.J., George, R.J., O'Connor, M.H., Dawes, W., Oliver, Y.M. & Raper, G.P. 2010. Temporal and spatial patterns of salinity in a catchment of the central Wheatbelt of Western Australia. *Australian Journal of Soil Research*, 48: 326–336.

Rosser, B.J. & Ross, C.W. 2011. Recovery of pasture production and soil properties on soil slips scars in erodible hill siltstone country, Wairarapa, *New Zealand. New Zealand Journal of Agricultural Research*, 54: 23-44.

Rumpel, C. & Kögel-Knabner, I. 2011. Deep soil organic matter – a key but poorly understood component of terrestrial C cycle. *Plant and Soil*, 338: 143-158.

Ruprecht, J., Vitale, S. & Weaver, D. 2013. Nutrient export (phosphorus). *In Report card on sustainable natural resource use in agriculture.* Department of Agriculture and Food, Western Australia.

Sanderman, J. & Baldock, J.A. 2010. Accounting for soil carbon sequestration in national inventories: a soil scientist's perspective. E*nvironmental Research Letters*, 5.

Sanderman, J., Farquharson, R. & Baldock, J. 2010. *Soil carbon sequestration potential: a review for Australian agriculture*. Report prepared for the Australian Government Department of Climate Change and Energy Efficiency. Canberra, Commonwealth Scientific and Industrial Research Organisation.(available at www.csiro.au/files/files/pwiv.pdf).

Schipper, L.A, Parfitt, R.L., Fraser, S., Littler, R.A., Baisden, W.T. & Ross, C. 2014. Soil order and grazing management effects on changes in soil C and N in New Zealand pastures. *Agriculture Ecosystems and Environment*, 184: 67-75.

Schipper, L.A. & Sparling, G.P. 2011. Accumulation of soil organic C and change in C:N ratio after establishment of pastures in New Zealand. *Biogeochemistry*, 104: 49-58.

Schipper, L.A., Dodd, M.B., Fisk, L.M., Power, I.L., Parenzee, J. & Arnold, G. 2011. Trends in soil carbon and nutrients of hill-country pastures receiving different phosphorus fertilizer loadings for 20 years. *Biogeochemistry*, 104: 35-48.

Schipper, L.A., Dodd, M.B., Pronger, J., Mudge, P.L., Upsell, M. & Moss, R.A. 2013. Decadal changes in soil carbon and nitrogen under a range of irrigation and phosphorus fertilizer treatments. *Soil Science Society of America Journal*, 77: 246-256.

Simons, J., George, R. & Raper, P. 2013. Dryland salinity. *In Report card on sustainable natural resource use in agriculture*. Department of Agriculture and Food, Western Australia.

Simpson, R.J., Oberson, A., Culvenor, R.A., Ryan, M.H., Veneklaas, E.J., Lambers, H., Lynch, J.P., Ryan, P.R., Delhaize, E., Smith, F.A., Smith, S.E., Harvey, P.R. & Richardson, A.E. 2011. Strategies and agronomic interventions to improve the phosphorus-use efficiency of farming systems. *Plant and Soil*, 349: 89-120.

Singh, K., Levett, M.P. & Kumar, R. 1996. Contaminants and the soil environment in Papua New Guinea: an overview. In contaminants and the soil environment in the Australasia-Pacific region. R. Naidu, ed. Dordrecht, Kluwer Academic.

Smith, C.J., Oster, J.D. & Sposito, G. 2014. Potassium and magnesium in irrigation water quality assessment. *Agricultural Water Management*. (in press).

SOE. 2011. *Australia state of the environment 2011*. Independent report to the Australian Government Minister for Sustainability, Environment, Water, Population and Communities. Canberra, State of the Environment 2011 Committee, DSEWPaC..

Sparling, G.P. & Schipper, L. 2004. Soil quality monitoring in New Zealand: trends and issues arising from a broad-scale survey. *Agric. Ecosys. Environ.*, 104: 545-552.

Sparling, G.P. & Schipper, L. 2002. Soil quality at a national scale in New Zealand. *J. Environ. Qual.*, 31: 1848-1857.

Sparling, G.P., Lewis, R., Schipper, L.A., Mudge, P.L. & Balks, M.R. 2014. Changes in soil organic C and N contents at three chronosequences after conversion from plantation pine forest to dairy pasture on a New Zealand Pumice soil. *Soil Research*, 52: 38–45.

Summer, M.E. & Naidu, R. 1998. *Sodic soils*. Oxford University Press. New York.

Tate, K.R., Wilde, R.H., Giltrap, D.J., Baisden, W.T., Saggar, S., Trustrum, N.A., Scott, N.A. & Barton, J.P. 2005. Soil organic carbon stock and flows in New Zealand: System development, measurement and modelling. *Canadian Journal of Soil Science*, 85: 481-489.

Taylor, M. D. Kim, N. D., Hill, R. B. & Chapman, R. 2010. A review of soil quality indicators and five key issues after 12 yr soil quality monitoring in the Waikato region. *Soil Use and Management*, 26: 212-224.

Terry, J.P., Garimella, S. & Kostaschuk, R.A. 2002. Rates of floodplain accretion in a tropical island river system impacted by cyclones and large floods. *Geomorphology*, 42: 171-182.

Terry, J.P., Kostaschuk, R.A. & Garimella, S. 2006. Sediment deposition rate in the Falefa River basin, Upolu Island, Samoa. *Journal of Environmental Radioactivity*, 86: 45-63.

Tiller, K.G. 1992. Urban soil contamination in Australia. *Austr. J. Soil Res.*, 30: 937-957.

Toze, S. 2006. Reuse of effluent water - benefits and risks. *Agricultural Water Management*, 80: 147-159

Tozer, P. & Leys, J. 2013. Dust storms–what do they really cost? *The Rangeland Journal*, 35: 131-142.

Trotter, C.M., Tate, K.R., Saggar, S., Scott, N.A. & Sutherland, M.A. 2004. A multi-scale analysis of national terrestrial carbon budget and the effects of land-use change. In M Shiyomi, ed. *Global Environmental Change in the ocean and on land*. TERRAPUB.

Tullberg, J. 2010. Tillage, traffic and sustainability-A challenge for ISTRO. *Soil and Tillage Research*, 111: 26-32.

Tullberg, J.N., Yule, D.F. & McGarry, D. 2007. Controlled traffic farming - From research to adoption in Australia. *Soil and Tillage Research*, 97: 272-281.

Turney, C.S.M., Bird, M.I., Fifield, L.K., Roberts, R.G., Smith, M., Dortch, C.E., Grün, R., Lawson, E., Ayliffe, L.K., Miller, G.H., Dortch, J. & Cresswell, R.G. 2001. Early human occupation at Devil's Lair, Southwestern Australia 50 000 years ago. *Quaternary Research*, 55: 3–13.

UNDESA. 2013. *World Population Prospects: The 2012 Revision. Vol. I. Comprehensive Tables*. ST/ESA/SER.A/336. United Nations, Department of Economic and Social Affairs, Population Division.

van Dijk, A.I.J.M., Beck H.E., Crosbie, R.S., de Jeu, R.A.M., Liu, Y.Y., Podger, G.M., Timbal, B. & Viney, N.R. 2013. The Millennium Drought in southeast Australia (2001–2009): Natural and human causes and implications for water resources, ecosystems, economy, and society. *Water Resources Research*, 49: 1040–1057.

Warne, M., Rayment, G., Brent, P., Drew, N., Klim, E., McLaughlin, M., Milham, P., Shelley, B., Stevens, D. & Sparrow, L. 2007. *Final report of the National Cadmium Management Committee (NCMC) (2000-2006).* (available at http://www.cadmium-management.org.au/documents/NCMC-Final-Report-web.pdf)

Weaver, D.M. & Summers, R.N. 2013. Nutrient status (phosphorus). *In Report card on sustainable natural resource use in agriculture.* Department of Agriculture and Food, Western Australia.

Weaver, D.M. & Wong, M.T.F. 2011. Scope to improve phosphorus (P) management and balance efficiency of crop and pasture soils with contrasting P status and buffering indices. *Plant and Soil*, 349: 37-54.

Werner, A.D. 2010. A review of seawater intrusion and its management in Australia. *Hydrogeol. J.*, 14: 1452-1469.

White, I. Falkland, A. & Fatai, T. 2009. Groundwater evaluation and monitoring assessment, vulnerability of groundwater in Tongatapu, Kingdom of Tonga. SOPAC/EU EDF 8. Reducing the Vulnerability of Pacific APC States. Canberra, Australian National University. 351pp.

Wilson, B.R. & Lonergan, V.E. 2013. Land-use and historical management effects on soil organic carbon in grazing systems on the Northern Tablelands of New South Wales. *Soil Research*, 51: 668-679.

Woodman, J.D., Baker, G.H., Evans, T.A., Colloff, M.J. & Andersen, A.N. 2008. *Soil biodiversity and ecology: emphasising earthworms, termites and ants as key macro-invertebrates*. Final report prepared for the 2008 Collaborative Terrestrial Biodiversity Assessment. Canberra, National Land & Water Resources Audit. 126 pp.

World Bank. 2014. *GDP per capita (current US$), World Bank national accounts data, and OECD National Accounts data files. World Development Indicators*. (availablea at http://data.worldbank.org/indicator/NY.GDP.PCAP.CD, accessed 12th November 2014)

16 | Regional Assessment of Soil Change in Antarctica

Contributing author: Megan Balks

16.1 | Antarctic soils and environment

Antarctica has a total area of 13.9 × 106 km^2, of which 44 890 km^2 (0.32 percent) is ice-free (Fox and Cooper, 1994; British Antarctic Survey, 2005) with potential for soil development. Ice free areas are mainly confined to the Antarctic Peninsula, a few places around the perimeter of the continent and along the Transantarctic Mountains. The largest ice-free area (approximately 5 000 km^2) is the McMurdo Dry Valleys in the Ross Sea Region.

Mean annual temperatures vary from near 0°C in the moister, marine influenced Antarctic Peninsula to about -20°C in dry, higher altitude inland areas (Campbell and Claridge, 1987). Many soils are formed on mixed tills with some directly formed on bedrock. Organic matter is minimal (< 0.1 percent) in drier, colder inland areas. However, in areas where free moisture occurs, mosses and, on the Antarctic Peninsula, higher plants, may grow and accumulate to form peat soil materials. Ornithogenic materials dominate soils at many coastal locations (e.g. Hofstee *et al.*, 2006). Surface ages in coastal and Antarctic Peninsula regions tend to be predominantly Holocene, exposed by retreat of the Last Glacial Maximum ice. At higher elevations in the McMurdo Dry Valleys, surfaces as old as Mid-Miocene (14 Ma) have been reported (Sugden, Bentley and Cofaigh, 2006) indicating low erosion rates under a stable polar desert climate. Soil microclimates, driven by strong topographic variability, also influence soil properties (Balks *et al.*, 2013).

A wide variety of soils occur in the ice-free areas (Campbell and Claridge, 1987). Gelisols (Soil Survey Staff, 2014) or Cryosols (IUSS Working Group WRB, 2014) are the predominant soils in Antarctica. Cryosols contain permafrost at depth and are overlain by an active layer that thaws during the summer and is frozen in winter. In moister coastal areas the permafrost is ice-cemented and thus 'frozen solid'. However in some inland areas of the Trans-Antarctic Mountains, including the McMurdo Dry Valleys, there is not enough moisture to form ice-cement so soils with temperatures well below 0°C are loose and easily excavated (Bockheim, 1978; Campbell and Claridge, 1987). The soils range from Gelisols (Cryosols) in the Ross Sea Region, through Gelisols and Entisols in coastal East Antarctica, to a mixture of Gelisols, Entisols, Spodosols and Inceptisols in the warmer northern Antarctic Peninsula Region where permafrost is not ubiquitous (Balks *et al.*, 2013). Due to limited weathering in the cold climate, many Antarctic soils are dominantly gravelly sands. Where vegetation is absent, a protective desert pavement usually forms at the soil surface. The focus for studies on Antarctic soils is not on their potential for food production, but rather on their genesis, diversity, and vulnerability to impacts of human activity.

16.2 | Pressures/threats for the Antarctic soil environment

Most of the human activities in Antarctica, including historic huts, modern research stations, and tourist visits, are concentrated in the relatively accessible, small, ice-free areas, on the coast, particularly in the Ross Sea region and Antarctic Peninsula (O'Neill et al,. in press).

Antarctica was first recorded by three whaling ships in 1820 leading to regular whaling visits and the Ross expedition of 1839-1940. The 'heroic era' of exploration (1895–1917) included expeditions such as those of Borchgrevink, Scott, Shackleton, Mawson and Amundsen. Since the International Geophysical Year (1957-1958), greatly increased human activity has occurred with over 70 scientific research bases established, mainly around the Antarctic coast. Ship-based Antarctic tourism has become popular with 46 000 tourists reported in the 2007/08 summer and 27 700 in the 2013-2014 season (IAATO, 2014).

Legacies of human occupation are scattered at isolated sites across Antarctica, particularly in areas close to the major research stations and semi-permanent field camps (Campbell, Balks and Claridge, 1993; Kennicutt *et al.*, 2010; Tin *et al.*, 2009). Impacts have included physical disturbance as a result of construction activities, geotechnical studies, and roading (Campbell, Balks and Claridge, 1993; Campbell, Claridge and Balks, 1994; Harris, 1998; Kennicutt *et al.*, 2010; Kiernan and McConnell, 2001); local pollution from hydrocarbon spills (Aislabie *et al.*, 2004; Kim, Kennicutt II and Qian, 2006; Klein *et al.*, 2012) and from waste disposal (Claridge *et al.*, 1995, Snape, Morris and Cole, 2001; Santos *et al.*, 2005; Sheppard, Claridge and Campbell, 2000); introduction of alien species (Frenot *et al.*, 2005; Chown *et al.*, 2012; Cowan *et al.*, 2011); and disturbance to soil biological communities (de Villiers, 2008; Harris, 1998; Naveen, 1996; Tin *et al.* 2009 and references therein). The amount of contaminated soil and waste has been estimated at 1–10 million m^3 (Snape, Morris and Cole, 2001). The presence of persistent organochlorine pollutants in Antarctica has been attributed to long-range atmospheric transport from lower latitudes (Bargagli, 2008).

Antarctic soils are easily disturbed and natural recovery rates are slow due to low temperatures and often a lack of liquid moisture (Campbell, Balks and Claridge, 1993, 1998a; Campbell *et al.*, 1998b; Kiernan and McConnell, 2001; Waterhouse, 2001). Where physical disturbance removes the protective 'active layer' the underlying permafrost will melt with resulting land surface subsidence and, in drier regions, accumulation of salt at the soil surface (Campbell, Claridge and Balks, 1994; Waterhouse, 2001). Campbell and Claridge (1975, 1987) recognized that older, more weathered desert pavements and associated soils were the most vulnerable to physical human disturbance. However disturbances on active surfaces, such as gravel beach deposits, aeolian sand dunes and areas where melt-water flows, have the capacity to recover (visually) relatively quickly (McLeod, 2012; O'Neill, Balks and López-Martínez, 2012b, 2013: O'Neill *et al.*, 2012a).

Fuel spills are the most common source of soil contamination and have the potential to cause the greatest environmental harm in and around the continent (Aislabie *et al.*, 2004). Hydrocarbon fuel spills have been shown to persist in the environment for decades, with fuel perching on top of ice-cemented permafrost (Balks *et al.*, 2002). When spilled on Antarctic soils, possible fates of the hydrocarbons include dispersion, evaporation, and biodegradation. Hydrocarbon degrading microbes are present in the Antarctic environment but within the Ross Sea region their effectiveness is limited by moisture and nutrient (N and P) availability (Aislabie *et al.*, 2004, 2012). Hydrocarbon spills on Antarctic soils can enrich hydrocarbon-degrading bacteria within the indigenous microbial community (Aislabie *et al.*, 2004, 2012; Delille *et al.* 2000).

Elevated levels of metal concentrations have been reported at base sites especially in areas used for waste disposal or affected by emissions from incinerators or fuel spills (Claridge *et al.* 1995; Sheppard, Claridge and Campbell, 2000; Webster *et al.*, 2003; Santos *et al.*, 2005; Stark *et al.* 2008; Guerra *et al.*, 2011). Particularly high metal levels have been reported at Hope Bay on the Antarctic Peninsula (Guerra *et al.*, 2011) and at the Thala Valley landfill at Casey Station, East Antarctica (Stark *et al.*, 2008). Elevated levels of methyl lead have been detected in soil from a former fuel storage site at Scott Base (Aislabie *et al.*, 2004).

Surface trampling has been shown to impact on soil nematode abundances in the McMurdo Dry Valleys (Ayres *et al.*, 2008) and on arthropod abundance on the Antarctic Peninsula (Tejedo *et al.* 2005, 2009). Potential for introduction of invasive plant, insect, and microbial biota is gaining attention (Cowan *et al.*, 2011; Chown *et al.*, 2012; Greenslade and Convey, 2012).

16.3 | Response

All activities in Antarctica are regulated through the national administrative and legal structures of the countries active in the region, underpinned by the international legal obligations resulting from the Antarctic Treaty System. The Protocol on Environmental Protection to the Antarctic Treaty (the Madrid Protocol) was signed in 1991 and designates Antarctica as 'a natural reserve devoted to peace and science'. The Madrid Protocol mandates the protection of Antarctic wilderness and aesthetic values and requires that before any activity is undertaken the possible environmental impacts are assessed. Since the ratification of the Madrid Protocol in 1991 environmental awareness has increased and the standard of prevention of human impacts undertaken by many of the Antarctic programmes, such as those operating in the McMurdo Dry Valleys, is now more stringent than environmental management standards in most, if not all, other regions of the planet (O'Neill *et al.*, in press).

The ice-free areas visited by humans are small, relative to the Antarctic continent as a whole, and impacts occur as isolated pockets amongst largely pristine Antarctic wilderness (O'Neill *et al.*, in press). The most intense and long-lasting visible impacts occur around the current and former research bases, and are often remnants of activities in the 1950s-1970s prior to the Madrid Protocol (Campbell and Claridge, 1987; Webster *et al.*, 2003; Bargagli, 2008; Kennicutt *et al.*, 2010; O'Neill, 2013). Since the 1980s environmental accountability, management and awareness have increased, and the environmental footprints of stations such as Scott Base and McMurdo Station on Ross Island have remained static or decreased (Kennicutt *et al.*, 2010). For example, there are mechanisms in place to prevent spills, remove wastes, phase out incineration, limit soil disturbance, and protect sites of particular cultural or environmental significance. These mechanisms are proving effective at preventing further damage to Antarctic soils.

References

Aislabie, J.M., Balks, M.R., Foght, J.M. & Waterhouse, E.J. 2004. Hydrocarbon spills on Antarctic soils: effects and management. *Environmental Science and Technology*, 38: 1265–1274.

Aislabie, J.M., Ryburn, J, Gutierrez-Zamora, M-L., Rhodes, P., Hunter, D., Sarmah, A.K., Barker, G.M. & Farrell, R.L. 2012. Hexadecane mineralization activity in hydrocarbon-contaminated soils of Ross Sea region Antarctica may require nutrients and inoculation. *Soil Biology & Biochemistry*, 45: 49–60.

Ayres, E., Nkem, J.N., Wall, D.H., Adams, B.J., Barnett, J.E., Broos, E.J, Parsons, A.N., Powers, L.E, Simmons, B.L. & Virginia, R.A. 2008. Effects on human trampling on populations of soil fauna in the McMurdo Dry Valleys, Antarctica. *Conservation Biology*, 22: 1544–1551.

Balks, M.R., López-Martínez, J., Goryachkin, S.V., Mergelov, N.S., Schaefer, C.E.G.R., Simas, F.N.B., Almond, P.C., Claridge, G.G.C., Mcleod, M. & Scarrow, J. 2013. Windows on Antarctic soil-landscape relationships: comparison across selected regions of Antarctica. *Geological Society, London, Special Publications*, 381: 397-410. doi: 10.1144/SP 381.9

Balks, M.R., Paetzold, R.F., Kimble, J.M., Aislabie, J. & Campbell, I.B. 2002: Effects of hydrocarbon spills on the temperature and moisture regimes of Cryosols in the Ross Sea region, *Antarctica*. *Antarctic Science*, 14: 319–326.

Bargagli, R. 2008. Environmental contamination in Antarctic ecosystems. *Science of the Total Environment*, 400: 212–226. DOI: 10.1016/j.scitotenv.2008.06.062

Bockheim, J.G. 1978. Relative age and origin of soils of the eastern Wright Valley, Antarctica. *Soil Science*, 128: 142–152.

British Antarctic Survey. 2005. *Antarctic Factsheet Geographical Statistics*. UK, Cambridge, British Antarctic Survey. 4 pp. (avaialable at http://www.antarctica.ac.uk/about_antarctica/ teacher_resources/ resources/factsheets/ factsheet_geostats_screen.pdf)

Campbell, I.B. & Claridge, G.G.C. 1975. Morphology and age relationships of Antarctic soils. R. *Soc. N. Z. Bull.*, 13: 83–88.

Campbell, I.B. & Claridge, G.G.C. 1987. *Antarctica: soils, weathering processes and environment*. USA, New York, Elsevier Science Publishers B.V. 368 pp.

Campbell, I.B., Balks, M.R. & Claridge, G.G.C. 1993. A simple visual technique for estimating the impact of fieldwork on the terrestrial environment in ice-free areas of Antarctica. *Polar Record*, 29: 321–328.

Campbell, I.B., Claridge, G.G.C. & Balks, M.R. 1994. The effects of human activities on moisture content of soils and underlying permafrost from the McMurdo Sound region, Antarctica. *Antarctic Science*, 6: 307–314.

Campbell, I.B., Claridge, G.G.C. & Balks, M.R. 1998a. Short- and long-term impacts of human disturbances on snow-free surfaces in Antarctica. *Polar Record*, 34(188): 15–24.

Campbell, I.B., Claridge, G.G.C., Campbell, D.I. & Balks, M.R. 1998b. Soil temperature and moisture properties of Cryosols of the Antarctic Cold Desert. *Eurasian Soil Science*, 31: 600–604.

Chown, S.L., Huiskes, A.H.L., Gremmen, N.J.M., Lee, J.E., Terauds, A., Crosbie, K., Frenot, Y., Hughes, K.A., Imura, S., Kiefer, K., Lebouvier, M., Raymond, B., Tsujimoto, M., Ware, C., Van de Vijver, B. & Bergstrom, D.M. 2012. Continent-wide risk assessment for the establishment of nonindigenous species in Antarctica. Proceedings of the National Academy of Sciences, 109(13): 4938–4943.

Claridge, G.G.C., Campbell, I.B., Powell, H.K.J., Amin, Z.H. & Balks, M.R. 1995. Heavy metal contamination in some soils of the McMurdo Sound region, Antarctica. *Antarct. Sci.*, 7: 9–14.

Cowan, D.A., Chown, S.L., Convey, P., Tuffin, M., Hughes, K., Pointing, S. & Vincent, W.F. 2011. Non-indigenous microorganisms in the Antarctic: assessing the risks. *Trends in Microbiology*, 19: 540–548.

De Villiers, M. 2008. Review of recent research into the effects of human disturbance on wildlife in the Antarctic and sub-Antarctic region. *In Human disturbance to wildlife in the broader Antarctic region: a review of findings*. Working Paper 12 for XXXI Antarctic Treaty Consultative Meeting, Kiev, Ukraine, 2–13 June 2008.

Delille, D. 2000. Response of Antarctic soil bacterial assemblages to contamination by diesel fuel and crude oil. *Microbial Ecology*, 40: 159–168.

Fox, A.J. & Cooper, P.R. 1994. Measured properties of the Antarctic Ice Sheet derived from the SCAR digital database. *Polar Record*, 30: 201-206.

Frenot, Y., Chown, S.L., Whinam, J., Selkirk, P.M., Convey, P., Skotnicki, M. & Bergstrom D.M. 2005. Biological invasions in the Antarctic: Extent, impacts and implications. *Biological Reviews*, 80: 45–72.

Greenslade, P. & Convey, P. 2012. Exotic Collembola on subantarctic islands: Pathways, origins and Biology. *Biological Invasions*, 14: 405–417.

Guerra, M.B.B., Schaefer, C.E.G.R., de Freitas Rosa, P., Simas, F.N.B., Pereira, T.T.C. & Pereira-Filho, E.R. 2011. Heavy metal contamination in century-old manmade technosols of Hope Bay, Antarctic Peninsula. *Water, Air and Soil Pollution*, 222: 91–102.

Harris, C.M. 1998. Science and environmental management in the McMurdo Dry Valleys, Antarctica. In J. Priscu,. ed. *Ecosystem Processes in a Polar Desert: the McMurdo Dry Valleys, Antarctica*. Antarctic Research Series 72. Washington, DC, American Geophysical Union.

Hofstee, E.H., Balks, M.R., Petchey, F. & Campbell, D.I. 2006. Soils of Seabee Hook, Cape Hallett, Northern Victoria Land, Antarctica. *Antarctic Science*, 18: 473–486.

IAATO. 2014. *Tourism statistics*. International Association of Antarctic Tour Operators. (available at http://iaato.org/tourism-statistics)

IUSS Working Group WRB. 2014. *World Reference Base for soil resources 2014. International soil classification sstem for naming soils and creating legends for soil maps*. World Soil Resources Reports No. 106. Rome. FAO.

Kennicutt, M.C., Klein, A., Montagna, P., Sweet, S., Wade, T., Palmer, T. & Denoux, G. 2010. Temporal and spatial patterns of anthropogenic disturbance at McMurdo Station, Antarctica. *Environmental Research Letters*, 5: 1–10.

Kiernan, K. & McConnell, A. 2001. Land surface rehabilitation research in Antarctica. *Proceedings of the Linnean Society of New South Wales*, 123: 101–118.

Kim, M., Kennicutt II, M.C. & Qian, Y. 2006. Molecular and stable carbon isotopic characterization of PAH contaminants at McMurdo Station, Antarctica. *Marine Pollution Bulletin*, 52: 1585–1590.

Klein, A.G., Sweet, S.T., Wade, T.L., Sericano, J.L. & Kennicutt II, M.C. 2012. Spatial patterns of total petroleum hydrocarbons in the terrestrial environment at McMurdo Station, Antarctica. *Antarctic Science*, 24: 450–466.

McLeod, M. 2012. *Soil and Permafrost Distribution, Soil Characterisation and Soil Vulnerability to Human Foot Trampling, Wright Valley, Antarctica*. New Zealand, University of Waikato. 219 pp. (Ph.D. Thesis)

Naveen, R. 1996. Human activity and disturbance: building an Antarctic site inventory. In R. Ross, E. Hofman, & L. Quetin, eds. *Foundations for ecosystem research in the Western Antarctic Peninsula region*, pp. 389-400. Washington, DC, American Geophysical Union.

O'Neill, T.A. 2013. *Soil physical impacts and recovery rates following human-induced disturbances in the Ross Sea region of Antarctica*. New Zealand, University of Waikato. 369 pp. (Ph.D. Thesis)

O'Neill, T.A., Aislabie, J. & Balks, M.R. (in press). Human impacts on Antarctic Soils. In J.G. Bockheim, ed. *Soils of Antarctica*. Dordrecht, Springer Publishers.

O'Neill, T.A., Balks, M.R, López-Martínez, J. & McWhirter, J. 2012a. A method for assessing the physical recovery of Antarctic desert pavements following human-induced disturbances: a case study in the Ross Sea region of Antarctica. *Journal of Environmental Management*, 112: 415–428.

O'Neill, T.A., Balks, M.R. & López-Martínez, J. 2012b. The effectiveness of Environmental Impact Assessments on visitor activity in the Ross Sea Region of Antarctica. In L. Lundmark, R. Lemelin, & D. Müller, eds. Issues in *Polar Tourism: Communities, Environments, Politics*. Berlin-Heildelberg-New York, Springer.

O'Neill, T.A., Balks, M.R. & López-Martínez, J. 2013. Soil Surface Recovery from Vehicle and Foot Traffic in the Ross Sea region of Antarctica. *Antarctic Science*, 25: 514–530.

Santos, I.R., Silva, E.V., Schaefer, C.E., Albuquerque, M.R. & Campos, L.S. 2005. Heavy metal contamination in coastal sediments and soils near the Brazilian Antarctic Station, King George Island. *Marine Pollution Bulletin*, 50: 185–194.

Sheppard, D.S., Claridge, G.G.C. & Campbell, I.B. 2000. Metal contamination of soils at Scott Base, Antarctica. *Applied Geochemistry*, 15: 513–530.

Snape, I., Morris, C.E. & Cole, C.M. 2001. The use of permeable reactive barriers to control contaminant dispersal during site remediation in Antarctica. *Cold Regions Science and Technology*, 32: 157-174.

Soil Survey Staff. 2014. *Keys to Soil Taxonomy, 12th ed*. Washington, DC, USDA-Natural Resources Conservation Service.

Stark, S.C., Snape, I., Graham, N.J., Brennan, J.C. & Gore, D.B. 2008. Assessment of metal contamination using X-ray fluorescence spectrometry and the toxicity characteristic leaching procedure (TCLP) during remediation of a waste disposal site in Antarctica. *Journal of Environmental Monitoring*, 10: 60–70.

Sugden, D.E., Bentley, M.J. & Cofaigh, C.O. 2006. Geological and geomorphological insights into Antarctic ice sheet evolution. *Philosophical Transactions of the Royal Society*, 364: 1607–1625. doi:10.1098/rsta.2006.1791

Tejedo, P., Justel, A., Benayas, J., Rico, E., Convey, P. & Quesada, A. 2009. Soil trampling in an Antarctic Specially Protected Area: tools to assess levels of human impact. *Antarctic Science*, 21: 229–236.

Tejedo, P., Justel, A., Rico, E., Benayas, J. & Quesada, A. 2005. Measuring impacts on soils by human activity in an Antarctic Specially Protected Area. *Terra Antarctica*, 12: 57–62.

Tin, T., Fleming, L., Hughes, K.A., Ainsley, D.G., Convey, P., Moreno, C.A., Pfeiffer, S., Scott, J. & Snape, I. 2009. Review: Impacts of local human activities on the Antarctic environment. *Antarctic Science*, 21: 3–33.

Waterhouse, E.J. (ed). 2001. *Ross Sea Region 2001: a State of the Environment Report for the Ross Sea Region of Antarctica*. New Zealand, Christchurch, New Zealand Antarctic Institute.

Webster, J., Webster, K., Nelson, P. & Waterhouse, E. 2003. The behaviour of residual contaminants at a former station site, Antarctica. *Environmental Pollution*, 123: 163–179.

Annex | Soil groups, characteristics, distribution and ecosystem services

Coordinating Lead Authors: Maria Gerasimova and Thomas Reinsch

Peer Reviewer: Neil McKenzie

Contributing authors: L. Anjos, O. Batkhishig, J. Bockheim, R. Brinkman, G. Broll, P. Charzyński , M.R. Coulho, F.O. Nachtergaele, M. Nanzyo, S. Mantel, S.M. Pazos (†), M.H. Stolt, C. Tarnocai , T. Tóth, L.P. Wilding and G. Zhang.

HISTOSOLS[1]

In most Histosols organic materials are deposited in a wetland environment to form peat. The waterlogged conditions, oxygen deficiency and, very often in the north, low temperatures, and acidic conditions, inhibit decomposition and lead to accumulation of organic matter (Kolka *et al.*, 2012). In some Histosols the organic material is derived from upland forest vegetation under cool, wet high rainfall conditions.

The soil profile features of Histosols reflect the origin of the organic material and the degree of decomposition, while the occurrence of permafrost is a common feature in these soils in arctic landscapes (FAO, 2014; Figure A1). The soil materials in Histosols are generally dark brown to almost black reflecting the high organic matter content. These soils support forest, sedge and shrubby-moss types of vegetation and occupy a poorly-drained, level topography. However, some Histosols in the coastal areas are found on slopes or form a continuous cover on the terrain, such as blanket bogs. Most of these soils developed during the Holocene Epoch. The age of the basal peat (the peat layer just above the mineral contact) is usually five to nine thousand years old.

Histosols are common soils in the Boreal and Arctic landscapes of the Northern hemisphere, although they may also occur in temperate and some tropical regions. Globally, peat lands (organic soils developed on peat) cover approximately 4 million km^2 (World Energy Council, 2013). However, most of the Histosols (3.5 million km^2) are found in the Northern Circumpolar Permafrost Zone where 76 percent of these soils are perennially frozen (Tarnocai *et al.*, 2009). Nearly 80 percent of all Histosols occur in Russia, Canada and the United States.

The global significance of Histosols is that they store huge amounts of organic carbon. It has been estimated that they represent a carbon pool of 500 billion tonnes of organic carbon (Strack, 2008). In addition, the present rate of annual carbon sequestration is approximately 100 million tonnes (Strack, 2008), which exceeds the present carbon loss from these soils due to agriculture, peat extraction and other human-made disturbances. Due to climate change, however, the water-saturated Histosols could be a source of greenhouse gases, mainly in the forms of methane (Couwenberg *et al.*, 2010), but they also could be the source of carbon dioxide if these soils dry out and are affected by wildfires.

1 The Reference Soil Group names of the World Reference Base developed by the IUSS Working Group RB (FAO, 2014) are used. Where the approximate equivalent name in the USDA Soil Taxonomy (Soil Survey Staff, 2014) is different, the USDA name is cited in brackets.

Figure A1 | (a) A Histosol profile and (b) a peatbog in East-European tundra.

ANTHROSOLS (Plagganthrepts, Haplanthrepts, some Orthents)

Anthrosols (Figure A2) are soils formed, altered or influenced by intense human agricultural activities. They are associated with long-term agricultural management in many parts of the world, especially where ancient civilizations were present. Technosols (see below) are also human-influenced but are connected with more recent human activities in industrial and urban environments resulting in the presence of artificial and man-made objects in the soil.

The formation of Anthrosols is termed anthropedogenesis. This formation includes various processes induced by human activities in ancient agricultural systems, such as periodic irrigation and drainage, continuous build-up by applying transported manure or other soil materials, and long-term fertilization. These soils are often enriched with phosphorus and carbon, and are characterized by movement and accumulation of clay and clay-organic complexes, reduction and oxidation of iron-manganese oxides and even physical compaction, processes that occur at an accelerated rate compared with that of natural soil changes. This results in a special soil morphology and soil horizon development, such as the formation of a surface horizon with a high organic matter content, the development of compacted plough-pans and the formation of redoximorphic features, all of which represent the outcome of anthropedogenesis.

Anthrosols occur widely across the globe. They appear, for example, in ancient agricultural regions under paddy cultivation, or in semi-arid and arid regions where irrigation and sedimentation have occurred. Anthrosols may also show a long-term build-up of elements from long-term manure application and phosphorus enrichment. Globally, the total extent of Anthrosols is estimated at more than 200 million ha, of which 80 percent are cultivated paddy fields.

Anthrosols make up the most fertile agricultural land in the world and provide food as an essential ecosystem provisioning service. They often are an inherent part of unique agricultural systems and as such have a cultural function as well.

Figure A2 | (a) An Anthrosol (Plaggen) profile and (b) associated landscape in the Netherlands.

TECHNOSOLS (Non soils)

Technosols (Figure A3) are common soils on all continents. They are dominant in urban areas, where there are only remnants of natural soils and where soils radically transformed by different human activities dominate together with 'new soils'. The development of particular horizons and layers in such soils is not reflected in natural conditions of the system (Charzyński *et al.*, 2013). Technogenic activities lead to the construction of artificial soil, soil sealing or extraction due to mining of materials not affected by surface processes in natural landscapes. The largest areas of Technosols in comparison to country total area can be found in countries with an extremely high percentage of urbanization such as Belgium and the United Kingdom. The largest areas dominated by Technosols are located within the largest mega-cities, for example the Yangtze River Delta Megalopolis in China (population of about 90 million); the Taiheiyō Belt in Japan, also known as Tokaido corridor (population of nearly 80 million); and the Great Lakes region in the United States (60 million).

Technosols are soils of urban, industrial, traffic, mining and military areas.

There are four main varieties of Technosols:

- soils sealed by technic hard material (hard material created by humans in industrial processes) e.g. asphalt or concrete.
- soils containing a large amount (more than 20 percent in the upper 1 m of soil) of artefacts. Artefacts are objects in the soil formed or strongly transformed by human activity or excavated from beneath the earth. Examples of artefacts are mine spoils, dredgings, rubbles, organic garbage, cinders, industrial dust, synthetic solids and liquids (e.g. petrol, kerosene)
- soils with geomembranes or synthetic membranes made, for example, of polyvinyl chloride (PVC) laid on the surface or into the soil.
- constructed or naturally developed shallow soils on buildings, without any contact to other soil material.

The soil profile features of Technosols are usually weak. Beneath technogenic deposits occasionally a natural soil profile can be observed. Also original profile development may still be present in contaminated natural soils.

In the urban ecosystem, soils play an essential role with their functions and ecosystem services. However, the ability of Technosols to provide ecosystem services differs from the services secured by natural soils and is often impaired (Morel *et al.*, 2015; Stroganova *et al.*, 1998). Technosols are more likely to contain toxic substances than other types of soils and should be treated with care (FAO, 2014). The benefits of Technosols and other urban soils are nonetheless numerous. They provide groundwater recharge for water supply, plant products for food supply, a medium for alternative storm-water management, sites for recreational activities, and buffering of temperature and humidity. They serve as a medium of retention, decomposition and immobilization of contaminants, and for dust entrapment to reduce dust content in the air. Technosols can be also considered as historical archives (Lehmann and Stahr, 2007).

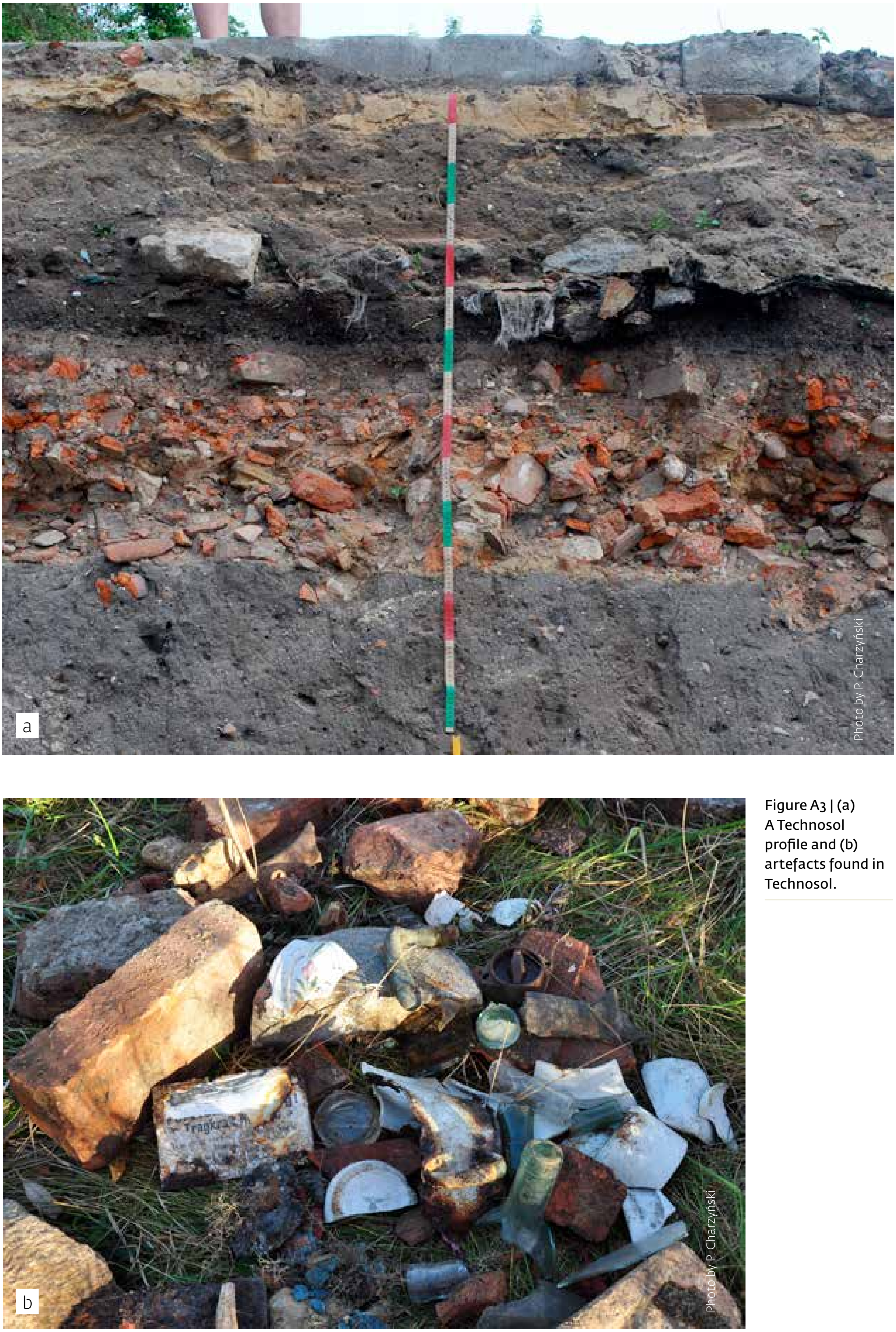


Figure A3 | (a) A Technosol profile and (b) artefacts found in Technosol.

CRYOSOLS (GELISOLS)

Due to the presence of perennially frozen conditions, Cryosols have unique processes and properties different from other soil groups. Cryogenic processes, which dominate the development of these soils, are driven by the mobility of unfrozen soil water as it migrates along the thermal gradient from warm to cold. This unfrozen water then moves into the frozen soil system, and feeds the ice bodies. The increase of ice volume and the volume increase from water to ice lead to differential frost heave. This then results in cryoturbation and the formation of cryogenic macro and micro soil structures (Figure A4).

Cryosols are the major soils in the permafrost areas of the Northern Circumpolar Arctic and Subarctic as well as a large part of the Boreal Region. They also occur in the ice-free areas of Antarctica and in the subalpine and alpine areas of the mountainous regions. They cover approximately 10.2 million km^2 in the Northern Circumpolar Region (Tarnocai *et al.*, 2009) and approximately 46 thousand km^2 in the Antarctic Region (Tarnocai and Campbell, 2002). Globally, most Cryosols occur in Russia and Canada. Cryosols support forest vegetation in the Boreal and Subarctic regions and tundra vegetation in the Arctic and Alpine Regions. Most Cryosols in Antarctica are unvegetated.

Cryosols, especially those affected by cryoturbation, contain large amounts of organic carbon. Cryoturbation moves organic materials from the surface into the subsoil where it is preserved for thousands of years because of the cold soil temperatures. Cryosols in the Northern Circumpolar Permafrost Zone contain approximately 351.5 Gt of carbon in the 0-100 cm depth and 818 Gt in the 0-300 cm depth (Tarnocai *et al.*, 2009). Due to climate change and the resulting thawing of these high carbon content Cryosols, they could be the source of greenhouse gases (carbon dioxide and methane), which would then further increase climate warming.

Figure A4 | (a) A Cryosol profile and (b) associated landscape in West Siberia, Yamal Peninsula.

LEPTOSOLS (lithic sub-groups of the Entisol order)

Leptosols include soils that are very shallow (less than 25 cm) with continuous rock occurring at or near the surface, and soils that are very gravelly (with less than 20 percent soil particles). Bare rock at the surface is included in the concept of Leptosols (FAO, 2014; Figure A5). These are generally young soils with little or no soil profile development. When Leptosols form in calcareous materials, dissolution and removal of carbonates may occur and biological activity may be high.

Leptosols may occur everywhere where rocks are near the surface. They are particularly prevalent in strongly eroding areas in mountainous land at high and medium altitude with a strongly dissected topography. Minor occurrences are also along rivers where gravelly deposits have accumulated without substantial admixture of fine earth material. Leptosols are the most extensive soils in the world with an estimated extent of more than 1 600 million ha. They are associated with mountain ranges, with the Sahara and the Arabian Desert.

In spite of their considerable extent, Leptosols have largely been ignored in soil studies mainly because of their very limited interest for agriculture and the general lack of profile development. This may not be fully justified as more than 12 percent of the world's population lives in a mountainous environment where Leptosols are common (Nachtergaele, 2010).

Leptosols have a potential for seasonal grazing and as forest land. Erosion is the greatest threat in montane Leptosol areas of the temperate zone where population pressure (for example from tourism), over-exploitation and environmental pollution lead to the increasing deterioration of the natural vegetation and to soil erosion.

Figure A5 | (a) A Leptosol profile in the Northern Ural Mountains and (b) associated landscape.

VERTISOLS

Vertisols (Figure A6) are expansive clayey soils that shrink and swell extensively with changes in moisture content. Cracking, gilgai microrelief, and high clay content are common attributes of Vertisols, but these properties are not exclusive to them. Slickensides are the common morphogenetic link to Vertisols and Vertic intergrades (Ahmad, 1983; Coulombe *et al.*, 1996a and 2000; Soil Survey Staff, 1999). The soil mechanics model of shear failure, slickenside formation, and oblique thrusting appear to better fit observed morphological properties, systematic soil property depth functions, and leaching vector transfers than the traditional inversion pedoturbation model (Ahmad, 1983; Coulombe *et al.*, 1996a; Wilding and Tessier, 1988). Shrink-swell in Vertisols is due mainly to inter- and intra-particle pore volume changes that occur under field conditions at high matric potentials (-1/3 to -10 bars). This is in contrast to the commonly held interlayer clay dehydration/rehydration mechanism invoked for shrink/swell dynamics (Wilding and Tessier, 1988). While smectite is a clay mineral component commonly found in Vertisols, many other clays including kaolinite, halloysite, mica and mixed layer assemblages of vermiculite, smectite and chlorite may occur as dominant or co-dominant associates (Coulombe *et al.*, 1996b). Mineralogy controls shrink-swell phenomena by the presence of fine-grained particles which have high external surface areas, high flexibility and a packing geometry that favours micropores a few micrometres in diameter or less (Wilding and Tessier, 1988; Coulombe *et al.*, 1996b).

Vertisols commonly occur in regions of grasslands and savannas, but may also be found under mixed pine and deciduous forests. Parent materials may originate from sedimentary, igneous or metamorphic origins but must provide, either from inheritance or weathering, a high content of clay with high surface area. Soil moisture conditions vary widely from aridic to aquic with the caveat that soil moisture stress, desiccation and cracking must occur at some time in most years. Topography controls gilgai patterns with normal gilgai commonly occurring on slopes < 4 percent, while linear gilgai occur on steeper landforms. Most Vertisols occur on Pleistocene-age or younger geomorphic surfaces that are several thousand to hundreds of thousands of years old (Coulombe *et al.*, 1996a and 2000).

Vertisols occur in over 100 countries and represent about 316 million km^2 or 2.4 percent of the global ice-free land area (Dudal and Eswaran, 1988; Ahmad, 1983; Coulombe *et al.*, 1996a; Wilding, 2000; Coulombe *et al.*, 2000). Over 75 percent of Vertisols are found in India, Australia, Sudan, United States, Chad and China. Forty-seven percent are in tropical regions, 52 percent in temperate zones, and 1 percent in cold boreal climates (Wilding, 2000; Coulombe *et al.*, 2000).

While Vertisols are very productive land resources in many parts of the world, especially in the developed world, they are among the most difficult resources to manage (Coulombe *et al.*, 1996a; McGarity *et al.*, 1984; Ahmad and Mermut, 1996). They require well above average managerial skills for success because of their high energy requirements, limited range of soil-water workability, high physical instability, susceptibility to seasonal flooding, fertility constraints, and susceptibility to wind and water erosion. They are best managed with shallow and infrequent tillage. Irrigation scheduling should be frequent with low application rates. Despite their resilience, Vertisols are subject to structural degradation, loss of macroporosity, loss of biological diversity and formation of tillage pans when used under continuous, long-term mechanical cultivation practices. Rejuvenation of native structural conditions can only be partially achieved after decades of fallow (Coulombe *et al.*, 1996a; McGarity *et al.*, 1984; Puentes and Wilding, 1990). Construction activities are constrained by the propensity of Vertisols for soil failure and high shrink/swell activity (Coulombe *et al.*, 2000; McGarity *et al.*, 1984; Ahmad and Mermut, 1996). High bioremediation and physical/chemical sorption capacities promote favourable habitats for land treatment of waste products because Vertisols when wet are slowly permeable, biochemically reduced, and have long mean residence times for intestinal fluids.

Figure A6 | Vertisol gilgai patterns and associated soils: (a) linear gilgai pattern located on a moderately sloping hillside in western South Dakota. Distance between repeating gilgai cycle is about 4 m. (b) Normal gilgai pattern occurring on a nearly level clayey terrace near College Station, TX. After a rainfall event microlows have been partially filled with runoff water from microhighs - repeating gilgai cycle about 4 m in linear length. (c) Trench exposure of soils excavated across normal gilgai pattern - repeating gilgai cycle about 4 m in linear length. Dark-colored deep soil in microlow (leached A and Bss horizons) with light-colored shallow calcareous soils associated with diaper in microhigh (Bssk and Ck horizons). The diaper has been thrust along oblique slickenside planes towards soils surface. Vertical depth of soil trench in about 2 m. (d) Close up of dark-colored soil associated with microlow and light colored diaper associated with microhigh of the trench in (c).

SOLONETZ (Natric great groups of several different orders)

Solonetz (figure A_7) are formed by salt accumulation and the leaching of the surface horizon. The dominant soil processes involved in Solonetz formation are leaching of the surface horizon combined with argilluviation and sodification manifested in the formation of columnar/prismatic structural elements.

Solonetz generally occur in flat plains that have a source of soluble salts, such as salt-bearing parent material or a shallow saline water table in semi-arid, temperate and subtropical climates that receive less than 500 mm of rainfall per year. The vegetation is commonly dominated by short grasses. The formation of these soils takes generally more than 5 000 years.

The extreme physical characteristics (high water retention, low hydraulic conductivity, strong swelling-shrinking, great plasticity) of Solonetz are linked with the high concentration of sodium in the exchange complex of the soil. Solonetz show strong profile differentiation in terms of colour, texture, structure, sodicity, salinity, alkalinity and calcareousness. Because of the high sodicity, Solonetz have a very short time window between snow melt and the following dry period for optimal ploughing. Plastic wet or dry Solonetz surface horizons cause a number of problems for cultivation. Only Solonetz with a thick surface horizon can be cropped successfully. Other Solonetz may be used for livestock farming. After reclamation, when the adsorbed sodium is replaced by calcium (by applying gypsum) and drainage, these soils can be turned into cropland.

The extent of Solonetz is estimated at about 135 million ha. They are mainly located in North America, Eurasia and Australia.

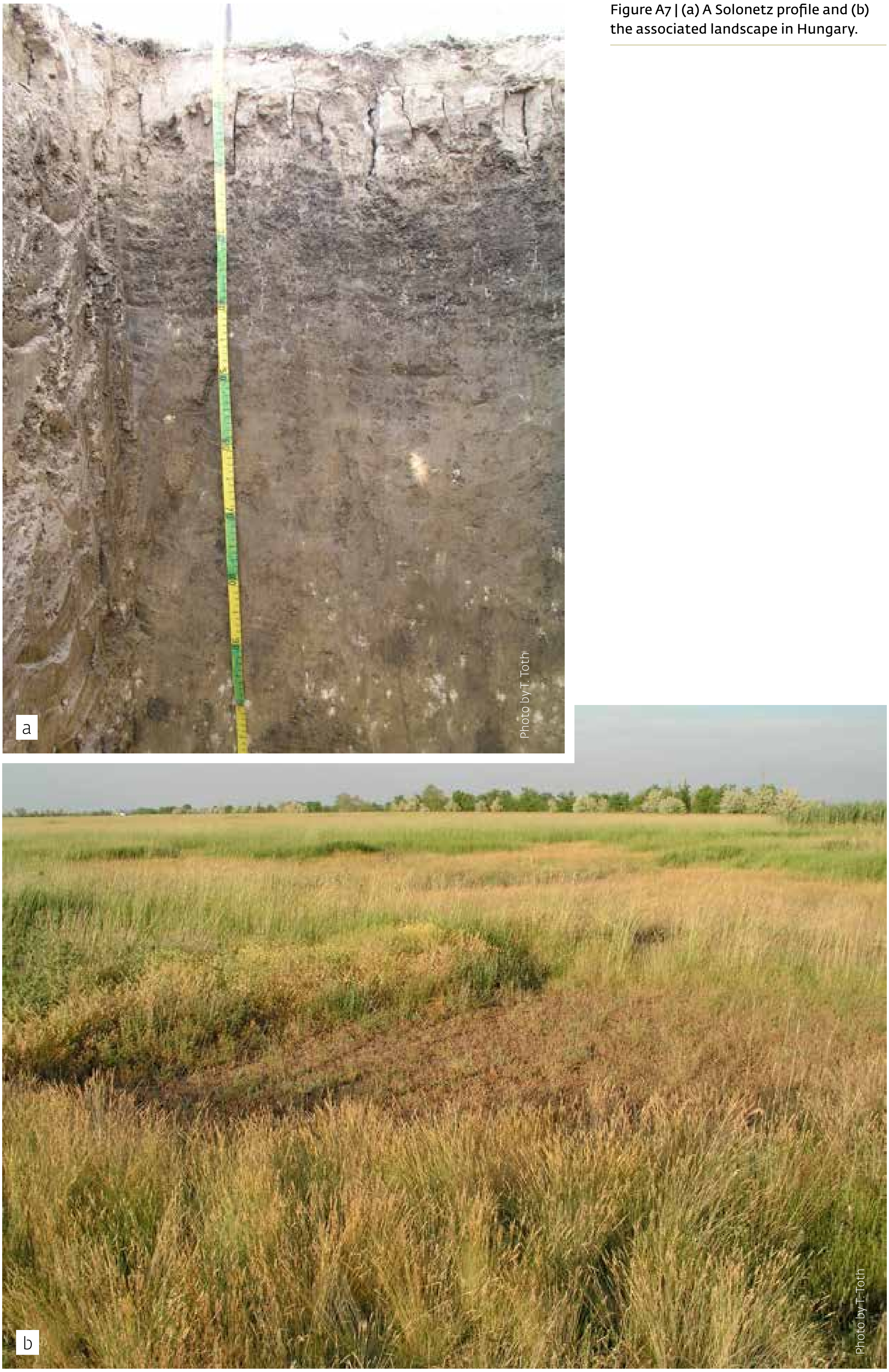


Figure A7 | (a) A Solonetz profile and (b) the associated landscape in Hungary.

SOLONCHAKS (Salids)

Solonchaks (Figure A8) are characterized by their high salt concentration, expressed by the electrical conductivity of the saturation extract (ECe) that exceeds 15 dS m^{-1} (or > 8 dS m^{-1} when the pH is ≥ 8.5). The presence of salt crystals and hydromorphic features are indicators of Solonchaks. The dominant soil processes involved in Solonchak formation are salt accumulation and the development of hydromorphic features.

These soils generally occur in inland river basins and very flat or depressed areas which have a source of soluble salts, such as salt-bearing parent material or a shallow saline water table. They also occur in coastal lowlands. Generally they are formed in arid, semi-arid and sub-humid climates where rainfall is less than 500 mm yr^{-1} and the evaporation exceeds the rainfall. The vegetation consists of salt tolerant grassland, bushes or mangroves.

Globally the extent of Solonchaks has been estimated at 260 million ha; they occur mainly in the drier parts of North America, northern Africa, the Middle East and central Asia, South America and Australia.

Solochaks are typically used for livestock farming or highly adapted irrigation farming. The vegetation on Solonchaks provides ecological services such as coast protection, grazing land and a source of wood. After leaching the salts and with drainage, these soils can be turned into cropland.

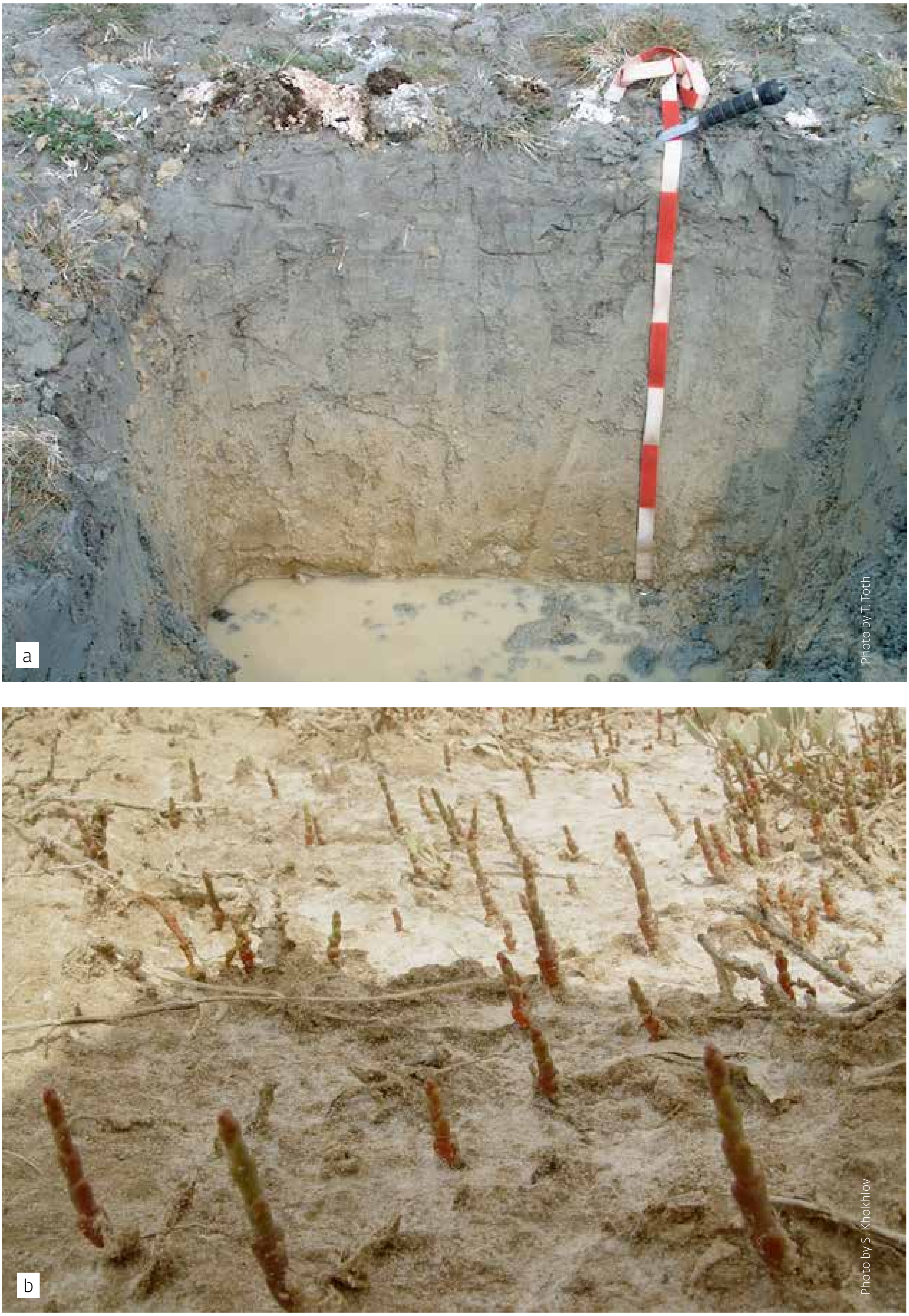


Figure A8 | (a) A Solonchak profile and (b) a salt crust with halophytes.

PODZOLS (SPODOSOLS)

The majority of Podzols (total area 4.8 million km^2) occur in humid boreal and temperate climates on light-textured rocks or quartz sands, on outwash plains and river terraces under pine forests, and on siliceous hard rocks in the mountains (Figure A9). Smaller areas occur under equatorial evergreen forests.

Podzol is a Russian folk name introduced by Dokuchaev (1879) into scientific language. It means either 'similar to ash' or 'under ash', implying that the ash is white or dark, respectively. Hence Podzols initially were mostly identified by their whitish (albic) subsurface layer resulting from the loss of iron-organic compounds. The thickness of the albic material (and of the whole Podzol solum) depends on climate and ranges from 2 m in equatorial 'giant Podzols' (Sombroek, 1966; Figure A10) to 5-10 cm in 'dwarf Podzols' on the Baltic Shield. Equatorial Podzols are confined to areas with annual rainfall from 1 800 to 3 000 mm without a marked dry season, to the weathering products of granites or gneisses, claystone and sandstone in the Amazon basin, and to marine sediments on the coastline of Brazil (Sombroek, 1966: Lucas *et al.*, 2012). A particular combination of environmental factors favours the development of acid hydrolysis and downward migration of its products immobilized at a varying depth to form a spodic horizon. The mechanism of podzolization has been discussed by many researchers. The most recent ideas summarized by Sauer *et al.* (2007).

The properties of spodic horizons and both the regional and local distribution of Podzols are in good agreement with moisture regimes: the spodic horizon is dominated by iron oxide compounds in drier conditions and by dark organic matter in humid ones. This differentiation is distinct at regional and local levels. Giant Podzols have a high organic carbon content in the spodic horizon, indicating that unusually large quantities of dissolved organic carbon were transferred from the topsoil. This is attributed to high volumes of water percolating through the soil, the chemical quality of the organic matter, and the long time for soil evolution (Lucas *et al.*, 2012). The subsoils of Podzols under a continental climate or strong drainage are commonly dominated by iron, while in other more temperate areas or less drained areas they are dominated by organic carbon (Friedland *et al.*, 1988).

Podzols may be young soils, just a few centuries old, or they may have been formed over millennia (Sauer *et al.*, 2007). Podzols buried almost 8 000 years ago were described under Histosols 2-3 m thick in West Siberia (Karavaeva, 1982).

Podzols are unstable soils even without human intervention: tree windfalls or fires induce wind erosion. Their most efficient ecological services are supporting coniferous forests, often of high quality, regulating the water balance in landscapes, and retaining some pollutants. In northern Europe, Podzols on heathlands are poorly preserved if they have been part of a Plaggen ecosystem (Blume and Leinweber, 2004).

Figure A9 | (a) A Podzol profile and (b) an associated landscape, West-Siberian Plain.

Figure A10 | (a) A giant Podzol profile and (b) an associated landscape, Brazil.

FERRALSOLS (OXISOLS)

Ferralsols (Figure A11) form where the weathering conditions are very intense, usually under tropical and subtropical humid conditions, with intense leaching of silica and alkaline and alkaline-earth cations, resulting in the relative accumulation of kaolinite and various amounts of resistant minerals such as (hydr-)oxides of Fe, Al, Mn and Ti. The distribution of clay in the soil profiles is uniform, without marked clay increase with depth. Ferralsols have low-activity clays throughout the lower horizons, and the base saturation is frequently low. Ferralsols have a distinct granular microstructure, due to the strong interaction among kaolinite and (hydr-) oxides of Fe and Al.

These soils show no large variation of clay content or evidence of clay illuviation, and the horizons are marked only by a higher content of organic carbon in the topsoil, which reduces with depth. The subsurface horizons show gradual to diffuse boundaries. The chemical characteristics reflect the leaching of base cations and advanced weathering resulting in low-activity clays. Some Ferralsols formed from basic rocks such as basalt may have better nutrient reserves, although the high iron content will result in strong phosphorus 'fixation'. The structure is usually of granular type, although some Ferralsols may develop a weak subangular block structure. They vary in the colour of their subsurface horizon from red to yellow, mainly according to the iron content in the parent material and to hydrological conditions.

Ferralsols are reported on weathering products of acid and basic rocks, and unconsolidated sediments on old and stable surfaces. They are most common on interior plateaus or slowly undulating topography in humid tropical, humid subtropical and monsoon climates. Because many climatic changes occurred since these soils were formed or the parent materials deposited, they may lack a relationship with the present vegetation, which may vary from Amazon forest to dry savannah.

The most extensive occurrences of Ferralsols are in South America, mainly in Brazil. They cover about 17 percent of Latin America and the Caribbean (Gardi *et al.*, 2014). Ferralsols are also distributed in eastern and central Africa (10 percent of the continent, Jones *et al.*, 2013) and Madagascar, in some areas of Australia and in the United States (Hawaii).

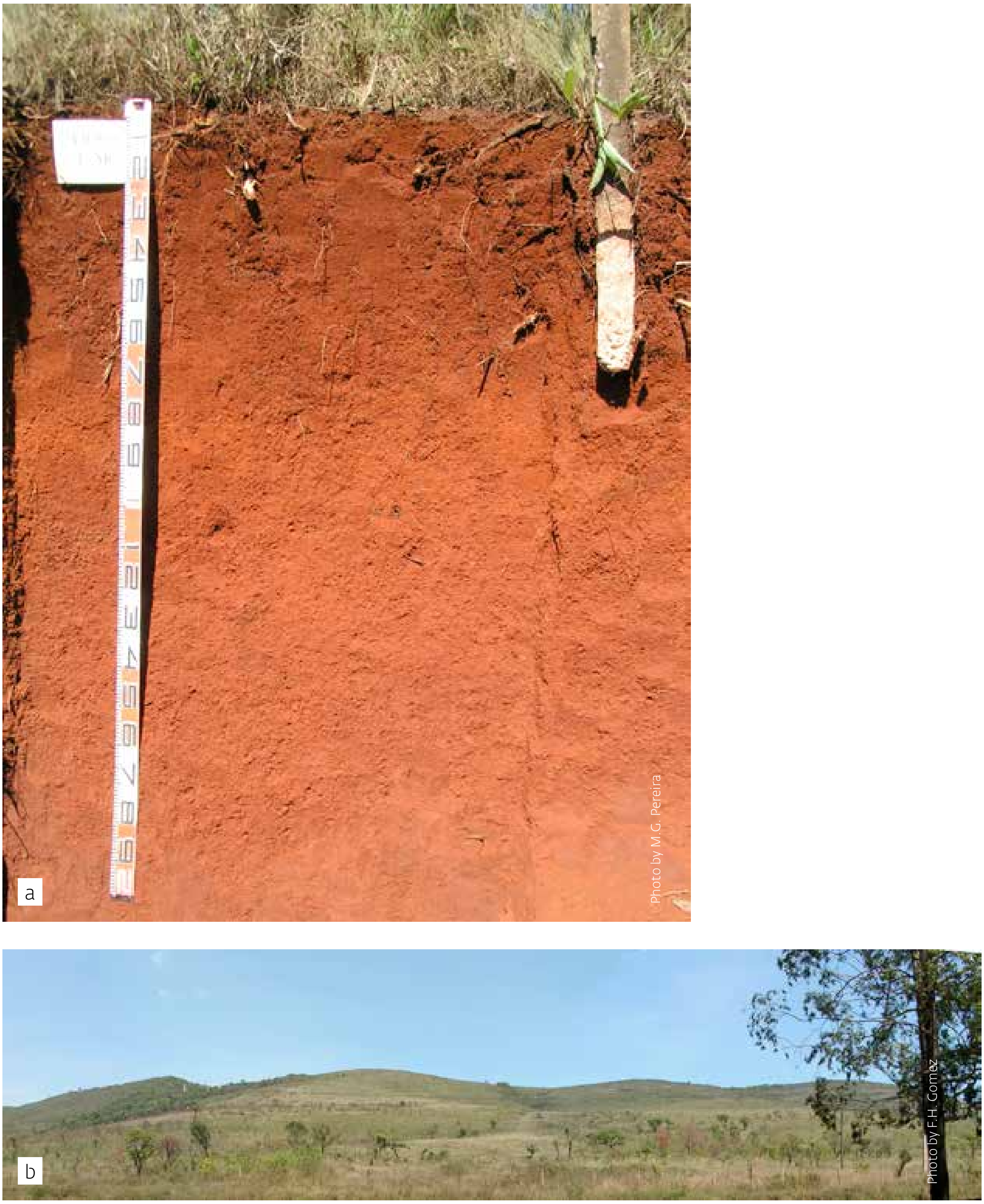


Figure A11 | (a) A Ferralsol profile and (b) an associated landscape, Brazil.

NITISOLS (Alfisols, Ultisols, Inceptisols and Oxisols Great groups)

Nitisols are well drained clayey soils with deep profiles. They are characterized by the strong development of structure, frequently with shiny aggregate faces (Figure A12). They originate from basic and intermediate rocks or sediments derived under relatively intense weathering conditions in tropical and subtropical climates. This leads to the predominance of low activity clays (kaolinite) and (hydr-) oxides of Fe, Al and titanium (Ti). Some Nitisols have high base saturation and high potential for crop production. Others have very high amounts of iron and strong P fixation, or a very low sum of exchangeable bases and high aluminium. Both these latter classes are limiting to crops.

The texture is clay loam or finer, with no large variation of clay content within the soil. The profile development shows intense weathering, with the prevalence of kaolinite and high iron in the nitic horizon, resulting in the strong stability of the aggregates, and the common angular and/or subangular blocks combined in a prismatic structure. The nitic horizon may show clay coatings indicating an illuviation process. The Nitisols are usually red or reddish-brown, and there is no distinct colour variation in the profile, except for the topsoil, due to the higher content of organic carbon. The subsurface horizons show gradual to diffuse boundaries. Nitisol classes vary largely according to base saturation, clay activity (usually low), iron and aluminium content. However, Nitisols formed from basalt may have high base saturation and, due to their good drainage and structure, may have high potential for both intense and low input agriculture.

Nitisols are mainly formed from weathering products of intermediate and basic igneous rocks (basalt and diabase). They may also have originated from clayey sediments in karstic areas (TerraRossa). They occur predominantly on high level plateaus and slightly undulating reliefs, originally under tropical and subtropical forest, or Cerrado (Brazil) and savannah vegetation.

Nitisols occur in eastern Africa and Madagascar (2 percent of the continent, Jones *et al.*, 2013). Although accounting for less than 1 percent of area on the Latin America and Caribbean soil map, Nitisols are prized lands in Southeastern and South regions of Brazil, and in neighbouring Argentina and Uruguay. They are cultivated with crops such as coffee, citrus, soybean, corn and sugarcane; and they play an important role in the agriculture of many tropical countries. Nitisols are also found in Australia (Ferrosol in Australian soil classification, formed from basalt), Europe (the Mediterranean) and the United States.

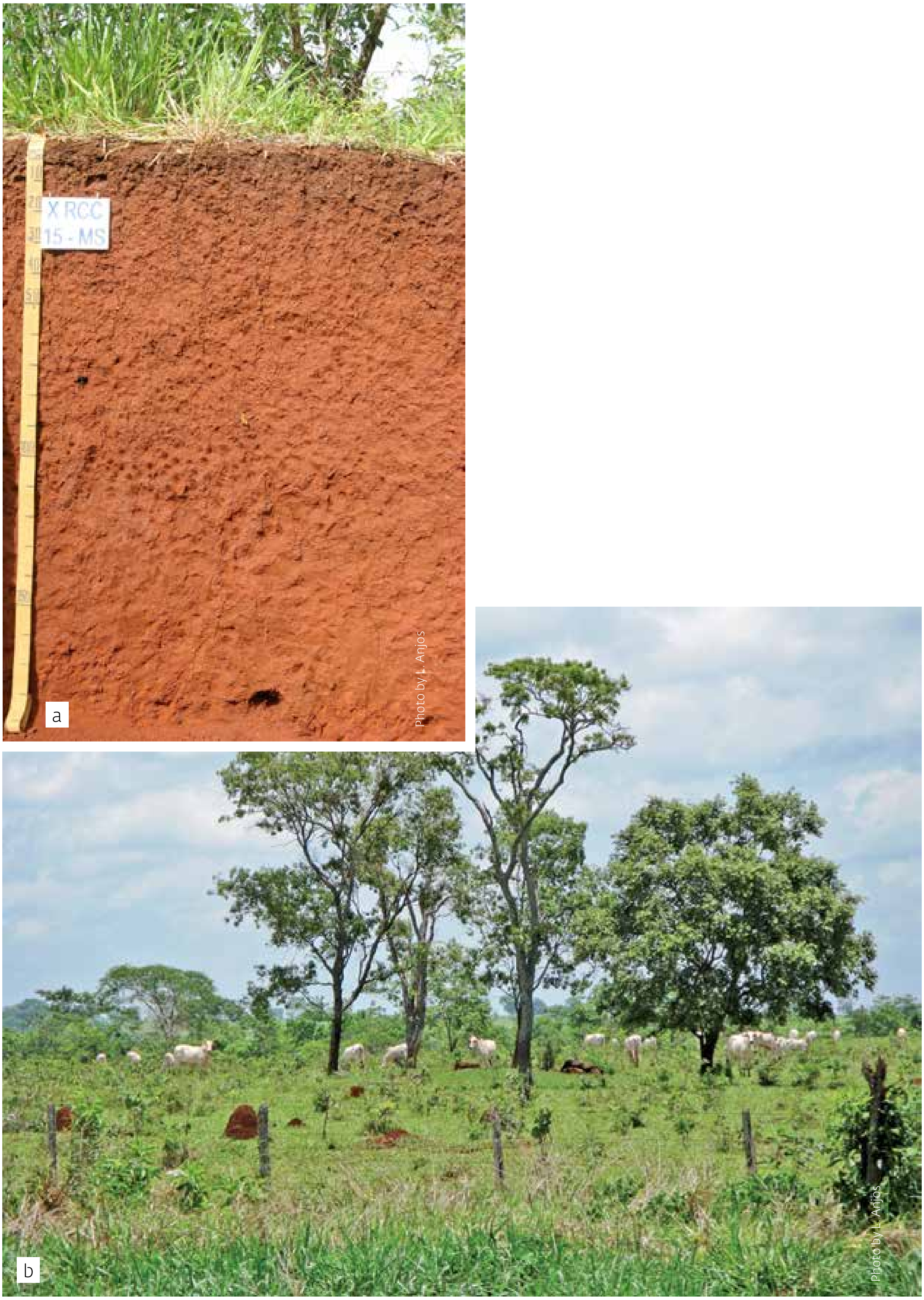


Figure A12 | (a) A Nitisol profile and (b) the associated landscape with termite mounds, Brazil.

PLINTHOSOLS (Plinthic sub-groups)

Plinthosols (Figure A13) are defined by the presence of plinthic, petroplinthic or pisoplinthic horizons, at a certain depth in the soil profile. Their formation is related to accumulation and redistribution of Fe under conditions of alternating wetting and drying cycles over long time periods. The landscape position - low lands with high groundwater or slopes with water seepage conditions - leads to chemical reduction of iron compounds in the parent material, which are redistributed and accumulated in the soil profile. The plinthite may be hard and irreversible (petroplinthite), forming a continuous and highly impermeable layer of ferruginous material (carapace or crust). The reduction, segregation and precipitation of iron (hydr-)oxides in the subsurface horizon forming the plinthite bodies, together with the dominance of kaolinite and other products of strong weathering such as gibbsite, indicate the conditions in which most Plinthosols formed. The profile may develop strongly bleached eluvial horizons and have evidence of clay illuviation; or show morphology associated to Ferralsols or to lesser development in recent sediments where reducing conditions are still present as indicated by gleyic properties. The subsurface horizon has platy, polygonal or reticulate patterns of distinct coloured (red, brown) plinthite bodies that are coherent enough to be separated from the surrounding soil matrix, which is usually of a pale colour. Hardening of the plinthite will form discrete concretions or nodules that characterize the pisoplinthic horizon. Further cementation and interconnecting of the pisoplinthic material will form the petroplinthic horizon, a layer of indurated material which may be continuous, broken or fractured.

Plinthosols are reported as formed from weathering products that have a high amount of Fe or where this element is accumulated due to water seepage or ascension of groundwater. They are most common on level to gently sloping topography, in areas with seasonal fluctuating groundwater in wet climates, humid and tropical, such as in the Brazilian Amazon Basin. However, in the Brazilian Cerrado and the savannahs of Africa, Plinthosols (with petroplinthic or pisoplinthic horizons) are also found on steeper slopes or as hard layers on plateau tops of old erosional surfaces.

Extensive areas of Plinthosols occur in West Africa, where they represent 5 percent of the total 30 million km^2 area of the continent (Jones *et al.*, 2013). Widespread in the Amazon Basin, they cover about 1 percent of Latin America's 22 million km^2, largely in Brazil, Colombia, Venezuela, Guyana and Bolivia, and in the Caribbean region, (Gardi *et al.*, 2014). Plinthosols are also found in Southeast Asia, India, Australia and the United States.

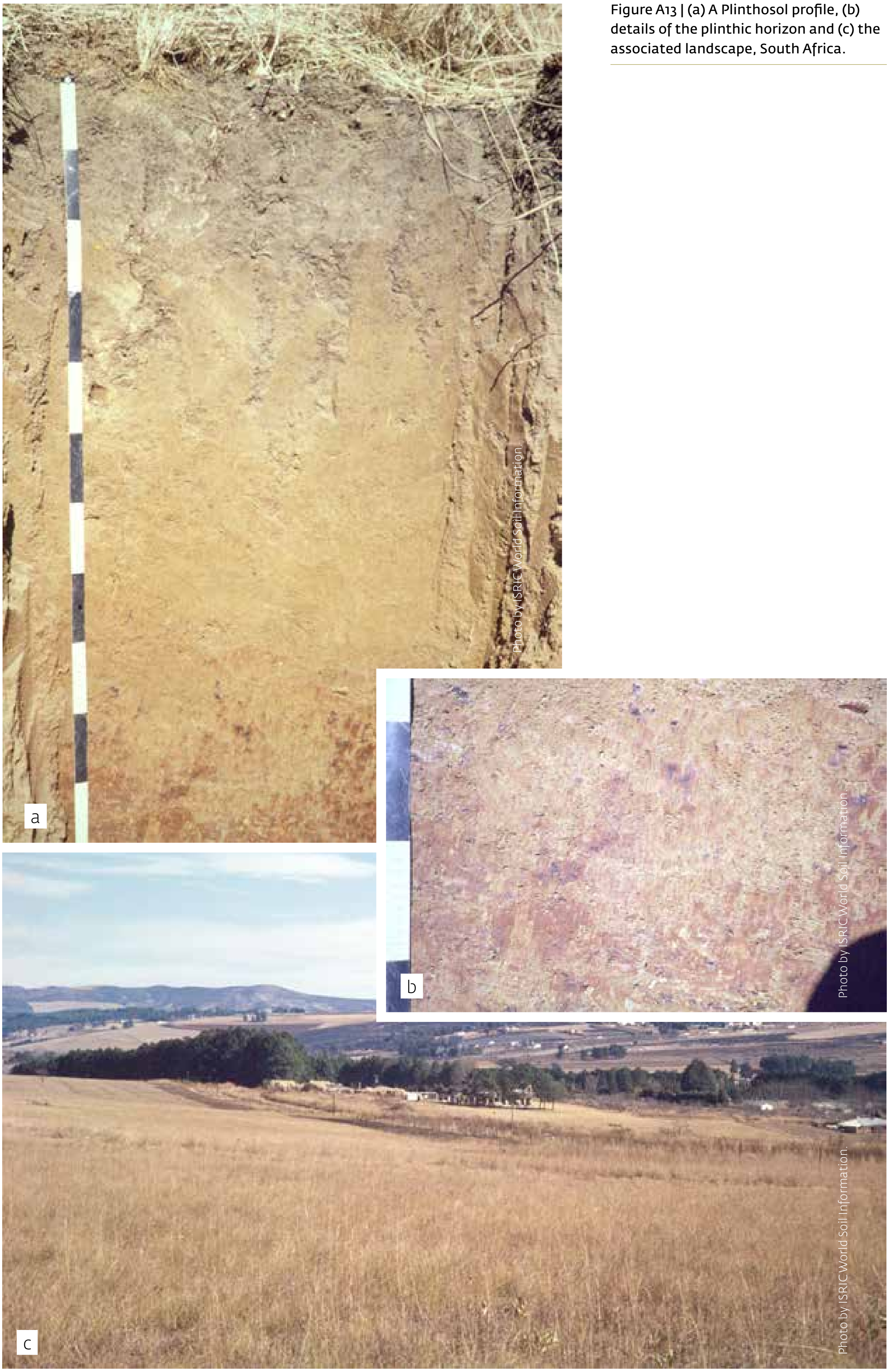


Figure A13 | (a) A Plinthosol profile, (b) details of the plinthic horizon and (c) the associated landscape, South Africa.

PLANOSOLS (Albaqualfs, Albaquults and Argialbolls)

Planosols are seasonally water-saturated or flooded, poor acid soils with bleached, generally silty surface horizons with an abrupt transition to a dense subsoil with significantly more clay (Figure A14). There may be pore infillings of bleached material in the subsoil. Clay destruction and aluminium interlayering driven by periodic iron hydroxide reduction and reoxidation (ferrolysis) has been recognised as a process sometimes involved in the formation of the silty surface horizons (Brinkman, 1979; Van Ranst *et al.*, 2011).

Planosols occur in generally level areas in climates with contrasting wet and dry seasons, mainly in the subtropics but in temperate areas and the tropics as well. Their total extent is estimated at 1.3 million km^2. They are extensive in Latin America (southern Brazil, Paraguay, and Argentina) and Australia, and they also occur in Africa (Sahelian zone, East and southern Africa), the eastern United States, Siberia, China, and Southeast Asia (Bangladesh, Thailand).

Natural vegetation on Planosols is sparse grass with or without shrubs or small trees; extreme Planosols may be barren. They are generally used for grazing or for grain or root crops in temperate areas. In the subtropics and tropics, rainfed paddy (wetland) rice is grown on bunded fields; with irrigation, they can be double cropped with a second paddy rice or dryland crop. Yields are very low without fertilizers and remain sub-optimal even with fertilizers because of the poor physical and chemical soil conditions.

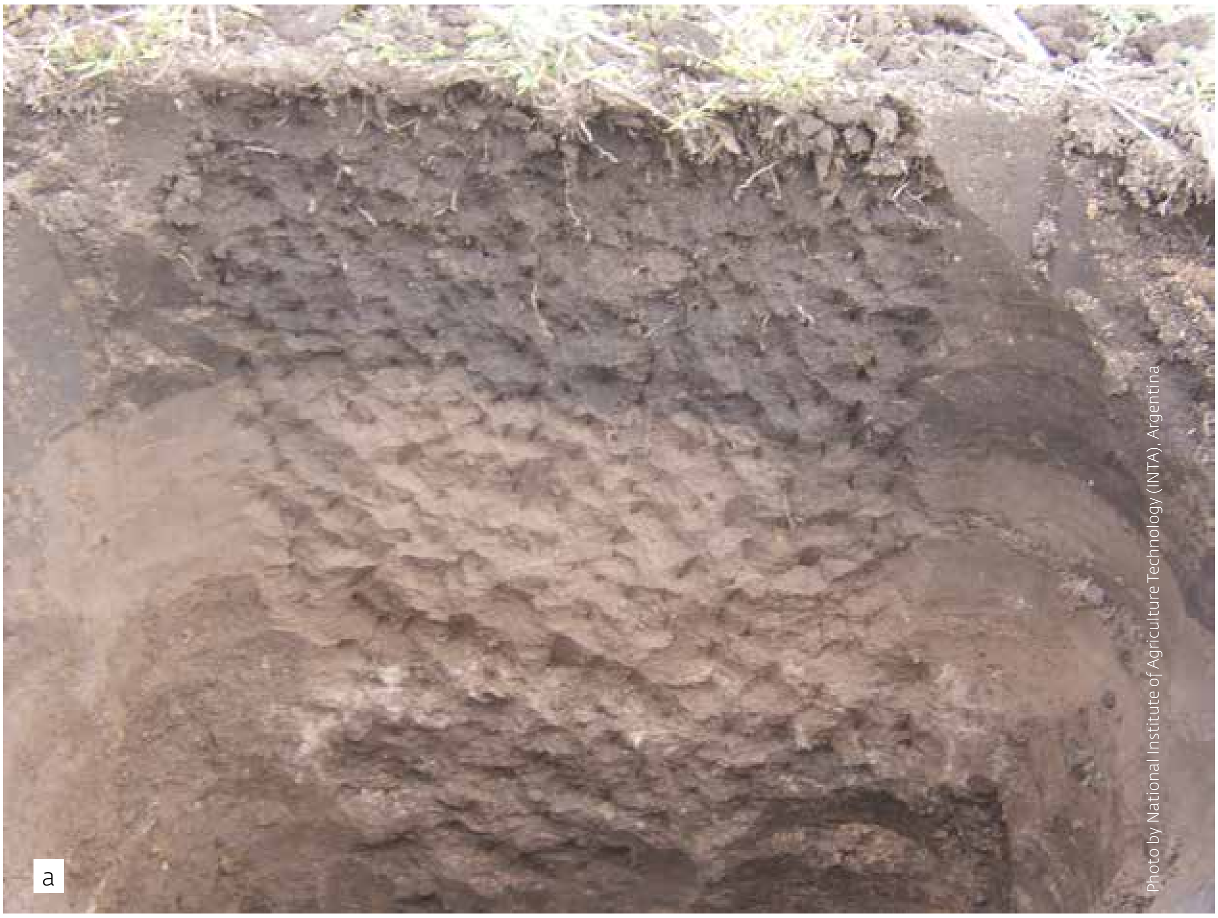


Figure A14 | (a) A Planosol profile and (b) the associated landscape, Argentina.

GLEYSOLS (Aquic suborder and Endoaquic great groups)

Gleysols are easily identified by bluish or greenish grey colours in their mineral horizons that are usually water-saturated, with only a weak or no structure (Figure A15). These horizons are formed under reducing conditions characterized by a low redox potential. In some Gleysols the smell of hydrogen sulphide or methane is noticeable. Iron compounds are easily mobilized in Gleysols, especially in the presence of organic matter and anaerobic microorganisms. These are partially removed or oxidized and may accumulate as iron segregations, nodules, iron pans, bog ores, etc. (Zaidelman, 1994). Above the layer with gleyic characteristics, a topsoil horizon relatively rich in organic matter occurs, that may show rusty root channels. The range in pH values in Gleysols is broad and may vary between 2.5 and 9. In coastal positions Gleysols may show sulphides oxidation resulting in high acidity (Zech *et al.*, 2014).

Globally the extent of Gleysols is estimated at 7.2 million km^2, of which approximately two thirds occur in boreal areas on unconsolidated parent rocks. In humid regions they often occupy depressions, river valleys and deltas, lake kettles and foot slopes. Subaqueous soils of shallow water bodies are also included with Gleysols. Large areas of Gleysols occur in tundra areas, in deltas of great rivers and in lowlands. They occur as associated soils almost everywhere, except in arid lands and on steep slopes.

In tundra regions the melting of the permafrost layer in summer causes excess of water in an environment already enriched in organic matter and induces seasonally reducing conditions and the formation of Gleysols. Water logging is the main prerequisite for the development of gleyic features and is due to high ground water table in depressions; additional water inflow there may contribute to gleying as well as flooding in the valleys and tides in coastal areas. There is no special plant community on Gleysols because they occur all over the world, but everywhere hygrophytes are dominant plants.

The main limitation for Gleysols management is surface water logging and/or shallow ground water hindering the growth of the roots of crops and trees. With artificial drainage the ground water table is lowered and the excessive moisture removed. When drainage is implemented efficiently, as it is in the Netherlands and Germany, Gleysols are productive soils for vegetables, beets and flowers. The main ecosystem threat is related to the Gleysols' low position in the landscape, where they may accumulate pollutants and could turn into 'chemical time bombs' (Stigliani, 1988).

Figure A15 | (a) A Gleysol profile and (b) associated landscape in the East European tundra.

STAGNOSOLS
(Aquic Suborders and Epiaquic Great Groups in Alfisols, Ultisols, Inceptisols, Entisols, and Mollisols)

Stagnosols (Figure A16) have much in common with Gleysols, but are different in their source of waterlogging and in their manifestations in the soil profile. Periodical stagnation of atmospheric water accounts for the name of this soil (originating from the German Pseudogley and Stagnogley). Stagnosols are characterized by the difference in texture between topsoil and subsoil originated due either to illuviation or to initial parent material heterogeneity in areas with humid climate and flat topography. Stagnosols are identified by the colour pattern of the upper 0.5 m of their mineral horizon, where a combination of reductimorphic (bluish grey colours that do not last) and oximorphic colours (rusty, reddish brown mottles inside aggregates and root channels known as Rohrenstein) together with iron-manganic segregations or nodules occur. These pedofeatures may occur within the whole layer, or they may be confined to its lower part, whereas its upper part may be composed of albic material with reductimorphic features. A special case of stagnic properties is the 'marbled' colour pattern described in old German literature as 'Marmorierung' (Muckenhausen, 1963).

Stagnosols are mostly acid to weakly acid and have a low to medium base saturation. Humus accumulation is prominent in these soils with raw or moder humus types; the biological activity in these soils is weak and the physical properties are unfavourable for plant growth: low porosity, reduced water filtration and risks of drying out (Zech *et al.*, 2014). Stagnosols are often localized and do not occur in vast continuous areas. They are mostly associated with other soils - Cambisols, Retisols, Acrisols. They are confined to flat or weakly undulating plains with various unconsolidated parent materials, moderately or heavy-textured. When the textural difference between the top- and subsoil is large, they are replaced by Planosols.

Stagnosols have mostly been described in areas with humid temperate and subtropical climate under hardwood forests. They are most common in Western Europe and the Midwest of the United States. The total area of Stagnosols worldwide is estimated at 1.5^{-2} million km^2 (FAO, 2014). Stagnosols have a low fertility due to their poor physical properties and moisture regime along with the elevated acidity and aluminium toxicity. Applying artificial drainage is less efficient than in Gleysols, unless additionally deep loosening of the subsoil is applied to break the impermeable layer. The same weakly permeable and dense subsoil is a problem for silviculture as it is an obstacle for tree roots and results in a high probability of tree uprooting. Nevertheless, forests of wetness-tolerant tree species and meadows are a preferable land use option.

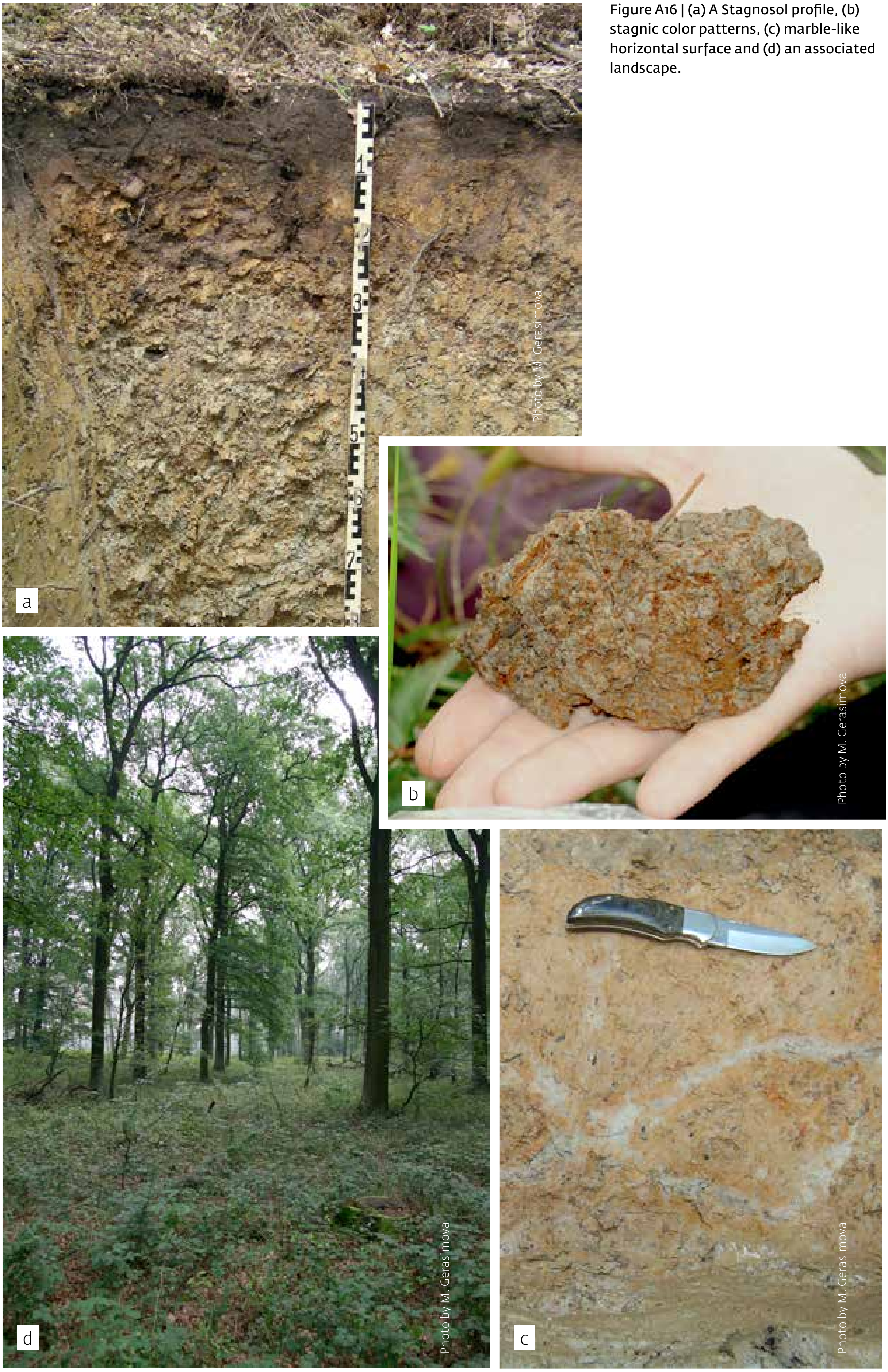


Figure A16 | (a) A Stagnosol profile, (b) stagnic color patterns, (c) marble-like horizontal surface and (d) an associated landscape.

ANDOSOLS (ANDISOLS)

Andosols (Figure A17) form typically from volcanic tephra on uplands. Distinctive properties of matured Andosols are the high content of active Al and Fe materials, and the lowest bulk density among mineral soils. Volcanic glass, the major constituent of tephra, rapidly weathers to form allphane, imogolite, Al-humus complex (non-crystalline active Al materials) and ferrihydrite (poorly crystalline active Fe material), with leaching loss of a large amount of Si, Na, Ca, etc. Under plentiful vegetation a large amount of humus accumulates in the A horizon, forming the Al-humus complex. Halloysite tends to increase under semi-dry climate in addition to non-crystalline active Al materials. Translocation of clays, Al and Fe is minimal in the soil profile. Due to a porous, fluffy and highly aggregated microstructure, Andosols show low solid phase ratio, low bulk density, high water permeability, and high water holding capacity.

A humid climate and uplands are favourable for leaching loss of Si, Na, Ca, etc. and for formation of active Al and Fe materials. Tephra deposits in swamps tend to weather more slowly than those on uplands, and the weathering product is richer in halloysite. Weathering of tephra under arid soil moisture regime appears even slower. The rock type of tephra ranges from rhyolitic to basaltic. The colour of rhyolitic to Andisitic tephra is whitish to greyish and that of basaltic tephra, black.

Andosols cover less than 1 percent of the earth's land surface. The major occurrences of Andosols are in and around the volcanic areas along the circum-Pacific volcanic zone, the Alpine-Himalayan belt, and the great Rift Valleys of Africa. Others are on the Hawaiian Islands, Iceland, etc. Andosols are used as productive farmlands after appropriate improvement of chemical shortcomings.

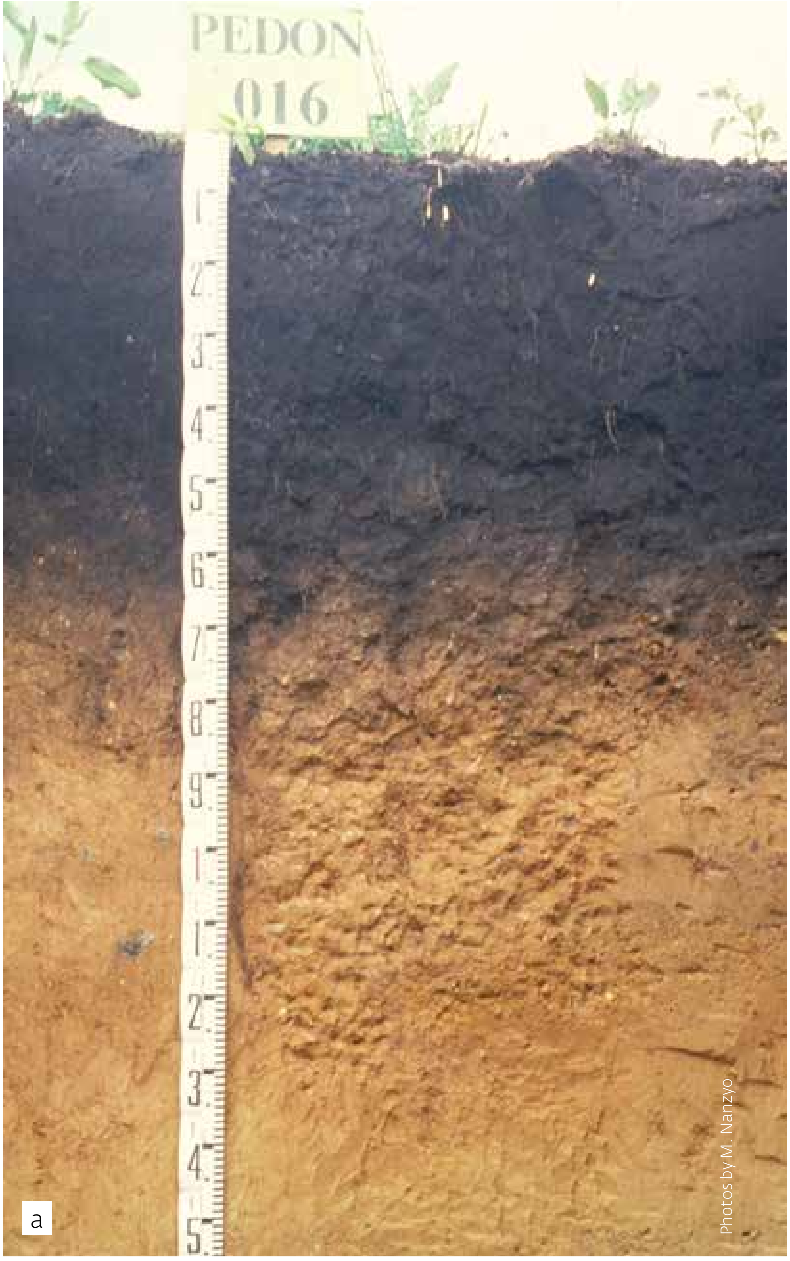


Figure A17 | (a) An Andosol profile and (b) the associated landscape in Japan.

CHERNOZEMS (Udolls)

Chernozems are marked by their deep, dark and well-structured topsoil (FAO, 2014). The soil has an almost black colour (hue < 1, chroma ≤ 2.5), intricate pedality and strong water-stable structure, which is mostly due to the activity of earthworms. The (micro-)structure is granular or crumb-granular in the upper part of the topsoil and spongy in the lower one; density is close to 1 g cm^{-3}; the Corg content ranges between 2.9 and 3.5 percent in the upper 10 cm (with humates as predominant fraction), and exceeds 1.2 percent at the lower boundary of the chernic horizon (Lebedeva, 1974). Earthworms and burrowing mammals (mole rats, marmots, hamsters, ground squirrels) modify the horizons' boundaries, effervescence depth and the pathways of solution flows. They also perform the exchange of material between the top and subsoil, which contributes to the profile stability; dark and brown krotovinas are common. Calcic horizon and/or secondary carbonates are diagnostic for all Chernozems. Secondary carbonates comprise labile forms: pseudomycelium and impregnation mottles corresponding in thin sections to needle-shaped crystals in voids, micritic (quasi) coatings, sometimes with sparite grains. The labile forms of carbonates are in agreement with the data on current hydrothermal soil regimes. Soft segregations – beloglazka – and hard nodules occur in more arid variants of Chernozems transitional to Kastanozems, while micritic pendants are confined to materials with rock fragments. In Russia, Chernozems are differentiated in accordance with secondary carbonate pedofeatures reflecting the current pedoclimate (Figure A18).

Continental climate with summer rains, soil freezing for two to four months, rich forb-grass natural vegetation, mostly loess as parent material, good drainage and level to undulating topography all contribute to the development of most typical profiles. The radiocarbon age of the topsoil ranges within 2^{-3} kA in its upper part, and 5-8 kA in the lower part (Chichagova, 1985). Chernozems first appeared in the Late Miocene under grass ecosystems maintained by grazers (Retallack, 2001). However, most Chernozems have been cropped for at least the last two centuries. They are regarded as very fertile soils.

Chernozems occur as a continuous belt in steppe and forest-steppe landscapes in Russia and the Ukraine, in the Great Plains of the US, in northern Kazakhstan and locally in some countries of Central Europe. They cover approximately 230 million ha.

High fertility of Chernozems is provided by a unique combination of very favourable chemical and physical properties. More than half of their area is cropland - maize, wheat, sugar beet and sunflower are the main crops. In the drier parts of their area, the main limitations to agriculture are droughts with occasional dust storms, whereas in wetter parts both wind and water erosion are the main risks. Climate change along with water conservation measures and irrigation at the background of lithological discontinuity have resulted in the appearance of small wetlands in the steppe landscapes.

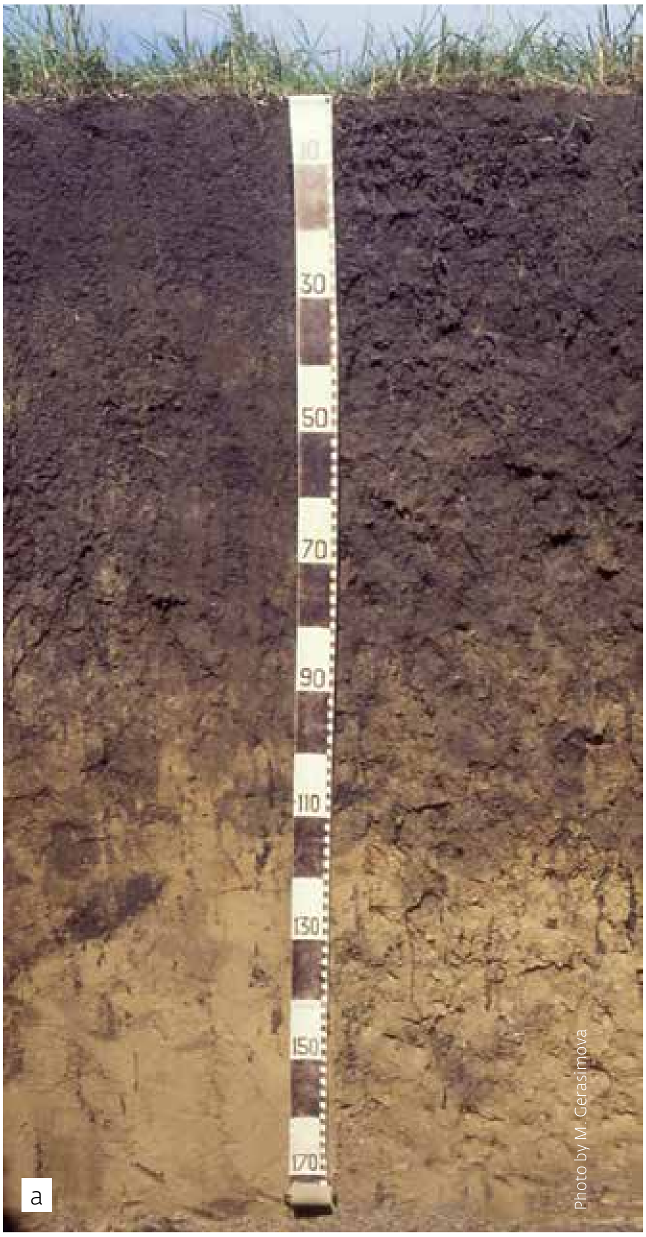


Figure A18 | (a) A Chernozem profile (Photo by J. Deckers) and (b) the associated landscape in the Central Russian Uplands.

KASTANOZEMS (Ustolls and Xerolls)

Kastanozems are humus-rich soils that were originally covered with early-maturing native grassland vegetation which produced a characteristic brown topsoil 20-40 cm thick in which the organic matter content ranges between 2 and 6 percent. Kastanozems have a brown topsoil with a granular or fine blocky structure. The rest of the profile is lighter in colour and is characterized by the secondary accumulation of calcite (Figure A19). Kastanozems are chemically rich soils with a pH slightly above neutral. Near the surface, soil pH may reach a value of 8.0.

These soils are found in relatively dry climatic zones (annual precipitation 200–400 mm). Kastanozems are mostly used for irrigated farming and grazing. Kastanozems have relatively high levels of available calcium ions and other nutrients. Carbonates weakly move down in the soil profile with percolating water to form layers of secondary carbonates; gypsum is also common in these soils. Kastanozems form in semi-arid regions under relatively sparse grasses and shrubs.

The total extent of Kastanozems is estimated to be about 465 million ha. Major areas are in the Eurasian short-grass steppe belt (southern Ukraine, the south of the Russian Federation, Kazakhstan and Mongolia), in the Great Plains of the United States of America, in Mexico, and in the southwestern pampas and Chaco regions of Argentina, in Paraguay and southeastern Bolivia (FAO, 2014).

The main obstacle to the agricultural use of these potentially rich soils is drought (Encyclopaedia of Soil Science, 2008). Irrigation, which brings the threat of secondary salinization, is nearly always necessary to obtain high yields. Another serious problem on Kastanozems is overgrazing (Wang and Batkhishig, 2014), extensive grazing being another important use for these soils. Overgrazing on light-textured soils often produces deflation, destroying the topsoil.

Figure A19 | (a) A Kastanozem profile and (b) the associated landscape in Mongolia.

PHAEOZEMS (Udolls and Albolls)

Phaeozems are soils with a mollic horizon which occur most frequently in the transitional areas between boreal forests and steppes, or in forest-free plains with temperate semi-humid climate (tall-grass prairies) on unconsolidated base-rich sediments, mostly loess or loess-like material. They may also occur locally under sparse herbaceous forests in the mountains (Figure A20). They cover approximately 2 million km^2, and their largest areas are found in the United States (Great Plains) and Canada, in the Argentinian Pampas, and in Manchuria. Mountainous variants were described in Southern Siberia and Northern Mongolia on gentle slopes with colluvium, under larch forests with a rich forb-grass cover (Zech *et al.*, 2014; Vostokova and Gunin, 2005). Phaeozems are formed under milder and more humid climates than Chernozems; the vegetation is mesophytic with less pronounced seasonal rhythms. Typically, natural grassland cover is mostly replaced by high-quality farmland, or may be modified by grazing. Phaeozems are among the most fertile soils owing to their favourable physical and chemical properties, along with moisture and thermic regimes (udic and mesic).

The most conspicuous feature of Phaeozems is their dark, mostly thick, mollic horizon with traces of burrowing mammal activity, weakly acid to neutral, base saturation ranging within 50-100 percent. Phaeozems include some of the traditional forest-steppe Chernozems with deep secondary carbonates, and in this case they have a chernic horizon with its coprogenic structure underlain by a cambic or argic horizon. In the rest of Phaeozems, the subsoil horizons may be diverse: argic, cambic, calcic and petrocalcic; the latter phenomenon is common in Argentinian Phaeozems – a specific hard tosca layer that may occur within 1 m from the soil surface and be a limitation for plant growth (Moscatelli, 1991; Pazos, 2012). Some other properties were described in Phaeozems as well: albic material and uncoated silt grains, clay coatings, stagnic colour pattern, and sodic features (FAO, 2014). This broad array of properties is explained by the occurrence of Phaeozems in different environments providing for the development of additional pedogenic processes, some of them being limitations for farming.

The global significance of Phaeozems is their high agricultural potential, as well as the prominent reserves of organic carbon accumulated in their topsoils. The limitations are not strong: they include wind erosion in dry years, water erosion on uplands, and water stagnation either during short rainy events or in case of high groundwater. In Manchuria, deep freezing and slow thawing are common. Local manifestations of sodicity have been recorded in Argentina and Western Siberia (Gerasimova, 2002).

Figure A20 | (a) A Phaeozem profile and (b) the associated landscape, Argentinian Pampa.

UMBRISOLS (Umbric Great Group in Aquept Suborder, Humic Subgroups in all Suborders of Inceptisols)

Umbrisols are mostly mountainous soils of cool humid climates covered by meadows or sparse forests. They are characterized by a dark, humus-rich and acid topsoil horizon with a low base saturation. A rather weak crumb structure is characteristic for the topsoil horizon (Figure A21).

Umbrisols are formed under dense forb-grass natural vegetation (subalpine meadows) or under deciduous forests with a prominent lower canopy, sometimes with shrubs. This produces a large volume of plant residues, which in part may not be strongly decomposed, and elements of a moder humus form may be identified (Zech *et al.*, 2014). Rather steep slopes and stony parent material provide sufficient drainage in spite of abundant precipitation and high air moisture; the soil is always moist, but stagnic or gleyic properties are absent. Typical examples of landscapes with Umbrisols are (sub-)tropical montane cloud forests in Mexico, Bolivia and Chile (Roman *et al.*, 2010), although Umbrisols also occur at higher altitudes in sub-boreal continental mountain ranges. Igneous and metamorphic rocks are almost always the parent material for Umbrisols. Worldwide, Umbrisols occupy approximately 10 million km^2 (Zech *et al.*, 2014).

The geographical location of Umbrisols poses serious limitations for agricultural activities. Chemical fertility is not low owing to high humus content but is restricted by soil acidity. Liming and mineral fertilizers are required. Another limiting factor is the risk of erosion because of the predominance of steeper slopes in areas of Umbrisols. Most Umbrisols are left under natural forests or forestation activity as hard rock or stony eluvium are not serious obstacles for tree roots. Grazing is less common. Only in New Zealand have high inputs made it possible to practice intensive dairy farming on these soils (FAO, 2014).

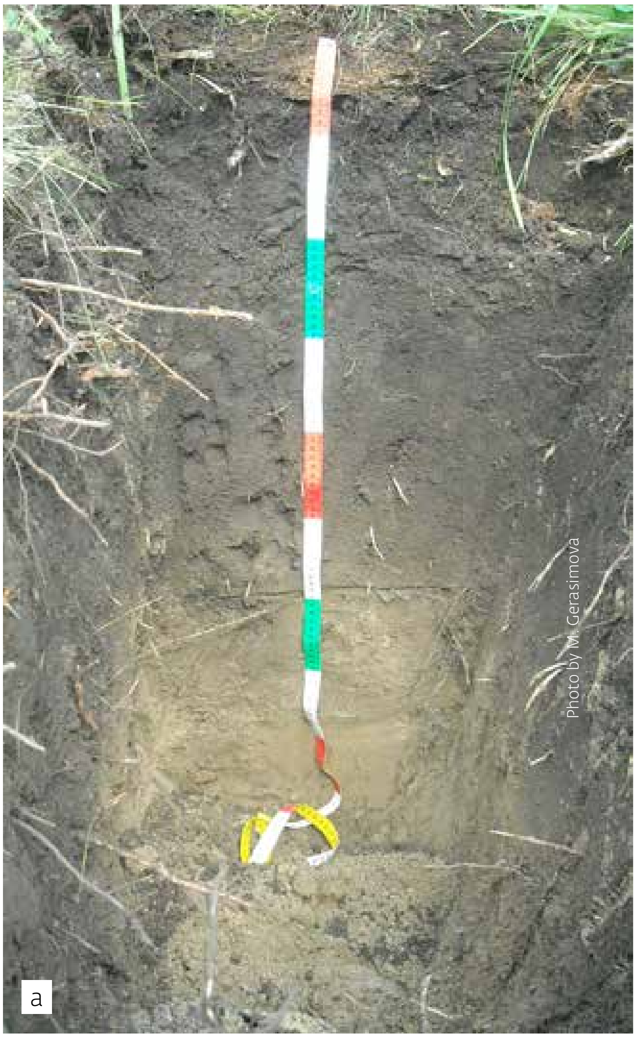


Figure A21 | (a) An Umbrisol profile, (b) an associated landscape.

DURISOLS (Durids)

Durisols may develop in arid and semi-arid conditions when a dissolution and accumulation of silica leads to the formation of a cemented hardpan that restricts the rooting depth of soils. These soils form mainly in alluvial and colluvial deposits in level or slightly sloping alluvial plains, terraces and piedmont plains. Stable landscapes occur where the Durisols have been eroded down to their resistant duripan, the material of which is often used in road construction. Durisols in low-lying areas may suffer from salt accumulation (Figure A22).

The duripan may range in thickness from 10 cm to more than 4 metres. There are two main types of duripans: those which are massive, and those with a platy or laminated structure that are coated with amorphous opal or microcrystalline silica.

Durisols are known to be relatively extensive in Australia, South Africa, Namibia and the drier parts of the southern United States. Minor extents have been observed in South America and Kuwait. No estimate of their global extent is available (FAO, 2014). The agricultural use of Durisols is mostly limited to extensive grazing. Arable cropping is limited to areas where irrigation water is available.

Figure A22 | (a) A Durisol profile and (b) the associated landscape, Ecuador.

CALCISOLS (Calcids, Argids, Cambids, some Cryids)

Calcisols embrace a broad group of soils in arid and semiarid regions. Their name in the FAO-WRB system has been changed, with former Xerosols and sub-categories of Yermosols becoming Calcisols. Most national names for these soils comprise indications of their grey or brown colour and of the (semi-)desertic or aridic landscapes in which they occur (Figure A23). Calcisols are widely spread in Mediterranean countries, central-eastern and southern Africa, the Near East, Mongolia, Australia, and southwestern United States. Calcisols are very widespread and are fifth in importance by surface area of all classified soils - 1000 million ha (FAO, 2014).

Calcisols have light coloured topsoil, poor in humus, sometimes free of carbonates, and a diagnostic calcic horizon. If there is a petrocalcic horizon within the upper 100 cm, the soil is also qualified for Calcisol. Calcic horizon is identified in the profile either 'quantitatively' by an elevated content of calcium carbonate in the fine earth (≥ 15 percent $CaCO_3$), or by an increase relative to the underlying horizon; or 'qualitatively' through the presence of secondary carbonates (FAO, 2014). Both criteria indicate mobilization and accumulation of carbonates in the soil (calcification). Both processes are known to depend on the moisture regime (Boettinger, 2002), which is dry almost all year round but with a short rainy period. Calcic horizon is formed in other soils (salt-affected, Chernozems, Kastanozems) but in Calcisols it is their major characteristic. High content of carbonates may be checked in the field by effervescence with 1M hydrochloric acid: it is quick with abundant foam formed. Secondary carbonates occur as soft nodules (beloglazka), pendants and coatings on stones, impregnation mottles, veins, single or coalescent – pseudomycelium, in the fine earth. Calcisols are always base-saturated, neutral to alkaline, have a narrow C:N ratio, and Corg content below 1-2 percent; the profile curve of $CaCO_3$ usually has a peak in the subsoil (Zech *et al.*, 2014).

Water deficit is a major limitation for using Calcisols, and extensive grazing is common in many lands dominated by Calcisols. Few areas are used for rainfed agriculture. Under irrigation, grain crops, cotton and vegetables are efficiently grown.

Figure A23 | (a) A Calcisol profile, (b) an associated landscape and (c and d) secondary carbonates in Calcisols.

GYPSISOLS (Gypsids)

Gypsisols are characterized by a significant secondary accumulation of calcium sulphate. Accumulation of gypsum takes place initially as crystal aggregates in the voids of the soils. These aggregates grow by accretion, displacing the enclosing soil material. When the gypsic horizon occurs as a cemented impermeable layer, it is recognized as the petrogypsic horizon. These soils occur in the driest part of the arid climatic zone in unconsolidated deposits of base-rich weathering material on level land and in depressions. Natural vegetation on these soils is sparse and limited to xerophytes and ephemeral grasses and herbs (Figure A24).

The worldwide extent of Gypsisols has been estimated at about 100 million ha, exclusively occurring in desert areas. Major occurrences are found in the Near East, Kazakhstan, Turkmenistan, Uzbekistan, the Libyan and Namib deserts, in southern and central Australia and in the southwest of the United States.

Large areas of Gypsisols are used for extensive grazing. When irrigation water is available these soils can be very productive, but the dissolution of gypsum results in the irregular subsidence of the land surface, caving in canal walls, and in the corrosion of concrete structures (FAO, 2014).

Figure A24 | (a) A Gypsisol profile and (b) an associated landscape.

RETISOLS (Glossic great groups of Alfisols and Ultisols)

The clay illuviation within Retisols is typically manifested by an interfingering of bleached coarser-textured soil material into the illuvial horizon, forming a net-like pattern (e.g. a glossic horizon). The dominant soil processes involved in Retisol formation are argilluviation and biological enrichment of base cations. They are often characterized by 'waxy' argillans; a subangular blocky structure; silty or loamy textural classes; active and superactive CEC (cation-exchange capacity) classes; and the occurrence of lithologic discontinuities (Figure A 25).

Retisols occur in climates where winters are cold and summers are short and cool with an annual precipitation between 500 and 1000 mm. They typically carry a temperate needle-leaf evergreen forest/woodland on often steeply sloping land. Their parent material is variable and includes loess, till, lacustrine and alluvium. Retisols are dated from the mid-Holocene or older e.g. > 5 000 years old. These soils generally exist in 'tension zones' (ecotones), reflecting a change in climate and/or vegetation.

Regional distribution of the 320 million ha of Retisols is mainly in Europe and northern and central Asia. There are about 85 000 ha in the United States. Retisols are important for forestry, recreation, and limited livestock farming and they provide ecological services such as watershed protection and ecological sustainability

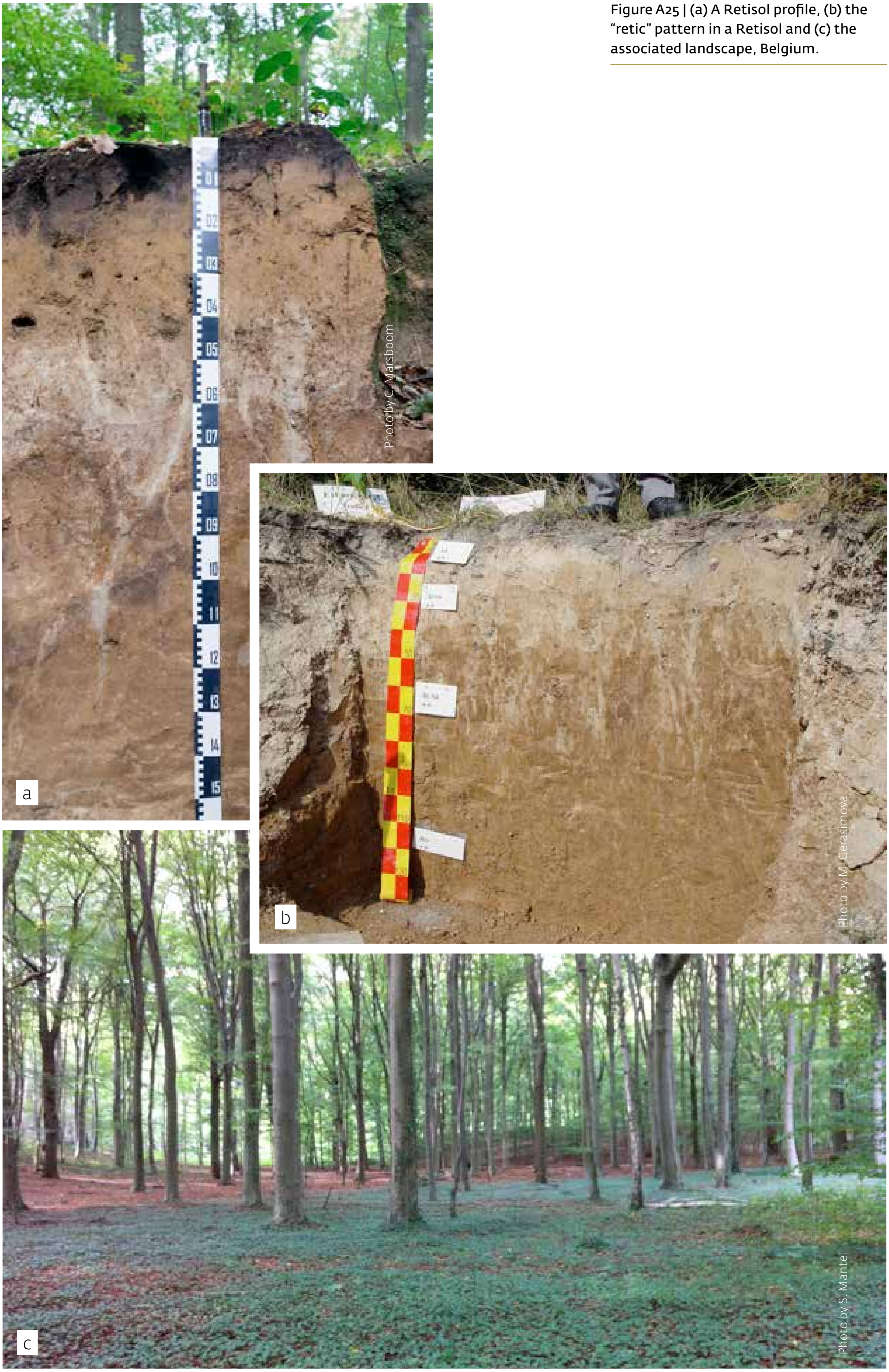


Figure A25 | (a) A Retisol profile, (b) the "retic" pattern in a Retisol and (c) the associated landscape, Belgium.

ACRISOLS (Kan- great groups of Ultisols, e.g. with a kandic horizon)

Acrisols are characterized by movement and accumulation of low-activity clays (cation-exchange capacity < 24 cmolc kg^{-1} clay) and a low base saturation (< 50 percent). The dominant soil processes involved in Acrisol formation include argilluviation and base-cation leaching (Figure A 26).

Acrisols occur under equatorial or warm climates, fully humid or winter-dry with an annual precipitation exceeding 1 200 mm. They typically carry a tropical deciduous or tropical evergreen forest or are under savannah. They occur on old hilly land surfaces where the relief is variable but often steeply sloping. Their parent material is saprolite or colluviums. These soils are commonly more than 200 000 years old.

Acrisols are used for forestry, recreation, agroforestry and shifting cultivation. They provide ecosystem services such as water protection and biotechnology for human health.

Regional distribution is some 1 000 million ha worldwide, mainly in southeast Asia, the southern fringe of the Amazon Basin, southeastern United States, and east and west Africa.

Figure A26 | (a) An Acrisol profile and (b) the associated landform in Kalimantan, Indonesia.

LIXISOLS (Kan - great groups of Alfisols, e.g. with a kandic horizon)

Lixisols are characterized by the movement and accumulation of low-activity clays (cation-exchange capacity < 24 cmolc kg^{-1} clay) and a high base saturation (> 50 percent). The dominant soil processes involved in Lixisol formation include argilluviation and biological enrichment of base cations. These soils are often polygenetic and have strong textural differentiation and advanced weathering but with abundant base cycling (Figure A 27).

Lixisols occur in the drier parts of the tropics and sub-tropics with a precipitation more than 1 200 mm annually. They typically carry a savannah vegetation. They occur on variable reliefs, while their parent material is saprolite or colluviums. These soils are commonly more than 200 000 years old.

Regional distribution is 435 million ha worldwide, mainly in sub-Sahelian and east Africa, Central and South America, the Indian Subcontinent, and southeast Asia and Australia

Lixisols are used for forestry, low-volume grazing and agro-forestry and provide ecological services such as water protection and ecological sustainability.

Figure A27 | (a) A Lixisol profile and (b) the associated landscape, Brazil.

ALISOLS (Ultisols with an argillic horizon)

Alisols are characterized by movement and accumulation of high-activity clays (cation-exchange capacity > 24 cmolc kg^{-1} clay) and a low base saturation (< 50 percent). The dominant soil processes involved in Alisol formation include argilluviation and base-cation leaching. Alisols have high Al and very low plant nutrients (Figure A 28).

Alisols occur under equatorial or warm climates, fully humid or winter-dry with an annual precipitation exceeding 1 200 mm. They typically carry a tropical deciduous forest or tropical evergreen forest. They occur where the topography is variable but often hilly or undulating, while their parent material is strongly weathered basic rocks and unconsolidated sediments. These soils are commonly more than 200 000 years old.

Regional distribution of the approximately 100 million ha of Alisols globally is mainly in Central and South America, the Caribbean, west and east Africa, southeastern Asia, and northern Australia. Alisols are used for forestry, low-volume grazing and, to a limited extent, for agriculture.

Figure A28 | (a) An Alisol profile and (b) the associated landscape, Belgium.

LUVISOLS (Alfisols with an argillic horizon)

Luvisols are characterized by clay movement, accumulation of high-activity clays (CEC > 24 cmolc kg^{-1} clay) and a high base saturation (> 50 percent). The dominant soil processes involved in Luvisol formation include argilluviation and biological enrichment of base cations. They are either derived from base-rich materials or have not been subject to strong weathering (Figure A 29).

Luvisols occur in humid climates with warm summers and snowfall during winter. They typically carry a vegetation of deciduous forest or woodland. They occur on flat or gently sloping topography. Their parent material is till, loess, alluvium or colluvium. These soils are commonly more than 5 000 years old.

There are 500-600 million ha of Luvisols worldwide, mainly in the Eastern European Plain, Western Siberian Plain, north central and northeastern United States, central Europe, and South Australia. Luvisols are used for agriculture, forestry and grazing. They are among the most productive soils worldwide and provide ecological services such as food and energy security, water protection, and ecological sustainability.

Figure A29 | (a) A Luvisol profile and (b) the associated landscape, China.

8 | Soils with little or no profile development

These soils have little or no profile development due to age, parent material, soil depth, transport, or deposition.

CAMBISOLS (Inceptisols)

Cambisols are young soils with beginning subsurface soil development. Characteristics that are more easily modified include structure, colour and bulk density. Structure begins to develop as wetting and drying cycles occur. Colour is modified through additions and removals such as carbonates and silica. Bulk density decreases as elements are weathered and organisms create voids. Typical soil horizonation is A-Bw-C (Figure A 30).

Cambisols are found in a wide range of climates, in all vegetation types, and level to steep reliefs. The typical parent material is medium and fine-textured, derived from a wide range of rocks, mostly in colluvial, alluvial or aeolian deposits. Cambisols form in almost all environments except permafrost.

The spatial distribution of Cambisols is estimated to be 1 500 million ha worldwide. Countries with more than 50 million ha are Russia, China, Canada and India. Cambisols are the dominant soil in San Marino, Saint Lucia, Grenada, Saint Vincent and the Grenadines, Jersey, Fiji, Belize, Italy, Luxembourg, Samoa, Guernsey, Anguilla, Czech Republic, Georgia, Haiti, American Samoa, Solomon Islands, Bosnia and Herzegovina, and New Zealand.

REGOSOLS (Orthents)

Figure A30 | (a) A Cambisol profile and (b) the associated landscape, China.

Regisols are the youngest soils with no pedogenic horizons and no evidence of soil forming processes. They may, nonetheless, support vegetation but do not meet criteria to be classified as another soil. They are located in inert or slowly soluble parent material, recent deposits, or excavation spoils. The profile horizons are usually A-C (Figure A 31).

Regisols exist in all climates and vegetation. They occur on level terrain to steep slopes.

Countries with more than 500 000 km_2 are Canada, Russia and Mexico. These are dominant soils in Curacao, Aruba, Bahamas, Bonaire, Saint Eustatius, Saba, Cayman Islands, Norway and El Salvador.

ARENOSOLS (Psamments)

Figure A31 | (a) A Regosol profile and (b) the associated landscape, China.

Arenosols are sandy soils that may have diagnostic horizons below meter. These soils have low water holding capacity and where there is no plant cover they are subject to wind transport. Profile horizonation is A-C (Figure A 32).

Arenosols occur in any climate except permafrost. Vegetation varies widely with climate. They are found on level to steep slopes. Typical textures are sandy or loamy sand on dunes, beaches, lacustrine deposits or weathered sandstone or coarse granite. The age can be recent to Pliocene or older.

Arenesols occupy approximately 1 300 million ha or about 10 percent of the land surface of the globe. Countries with more than 400 000 km_2 are Australia, Sudan, China, Angola and Botswana. These are the dominant soils in Botswana and Angola.

FLUVISOLS (Fluvents, Fluv-Subroups)

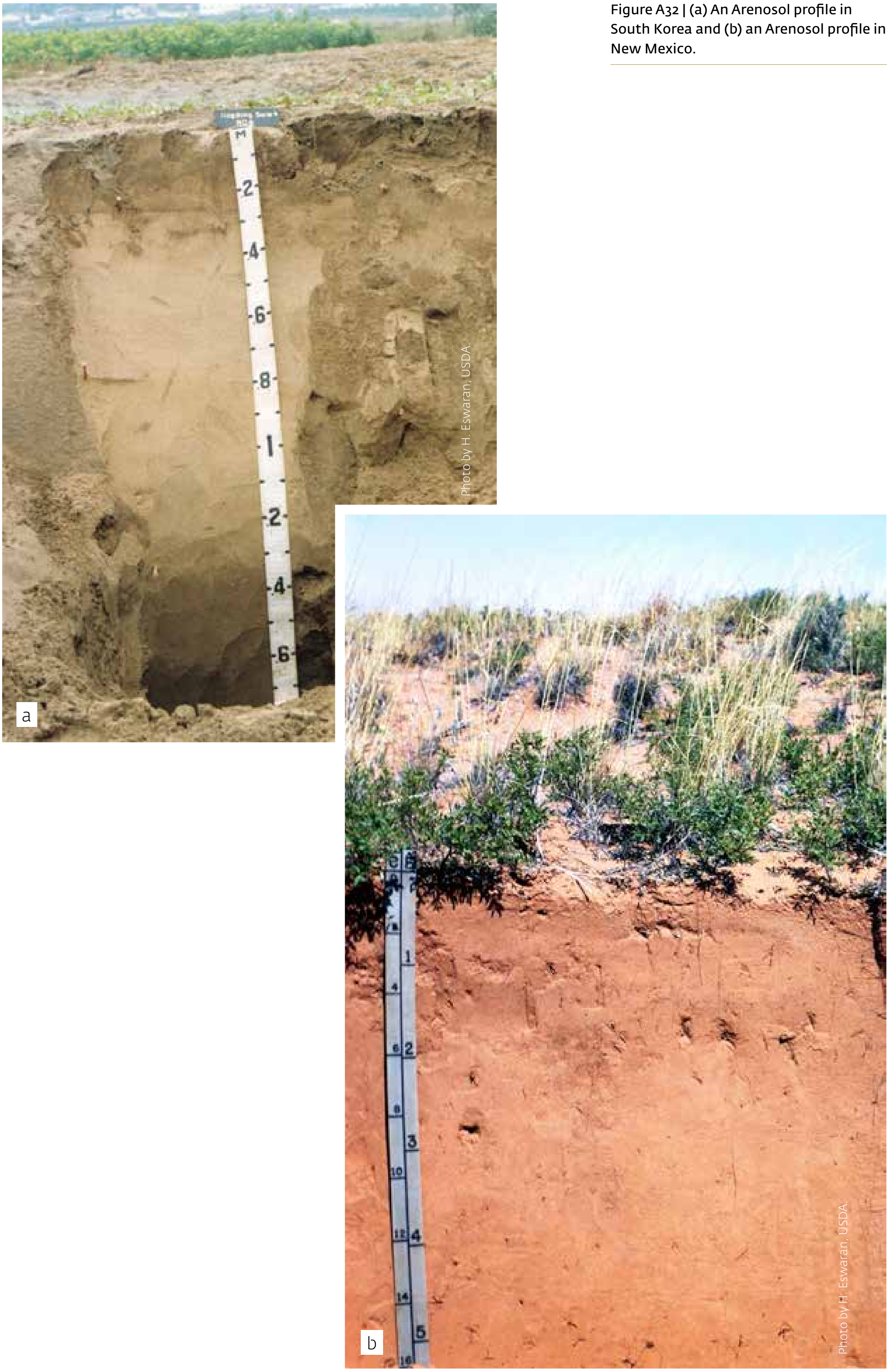


Figure A32 | (a) An Arenosol profile in South Korea and (b) an Arenosol profile in New Mexico.

Fluvisols are soils developedin fluvial, lacustrine or marine deposits A noted characteristic is an irregular decrease in organic carbon. In fact, significant soil organic carbon is buried by depositional events. Typical horizonation is A-C_2-C_3-Ab (Figure A 33).

Fluvisols are found in all climates except permafrost. Vegetation depends on climate and proximity to water. The relief is usually level. Most of the soils are of recent origin.

Fluvisols occupy approximately 350 million ha of the land surface of the globe. Countries with more than 20 million ha are Russia, China and Indonesia.

9 | Permanently flooded soils

Figure A33 | (a) A Fluvisol profile in Wisconsin and (b) a Fluvisol profile in Germany.

WASSENTS, WASSISTS (subaquatic in Histosols and Fluvisols)

Diagnostic horizons are typically absent from subaqueous soils. The exception is horizons formed from the accumulation of organic materials derived from submerged aquatic vegetation. Buried horizons have been observed because of sea level rise. In estuarine subaqueous soils, sulphides typically accumulate in low energy environments (sulphidization). Another important process is pedoturbation (faunal). This is especially the case in estuarine subaqueous soils where benthic organisms such as clams and worms burrow and mix the upper soil materials.

Subaqueous soils are found in shallow areas of lakes, ponds and estuarine systems such as bays and lagoons in any climate. The distribution of the different subaqueous soil types typically follows the submerged landscape which is broken into different units such as submerged beach, bay bottom, wash-over fan, or flood-tidal delta. The parent materials are marine or lake sediments that have been brought in by streams and rivers emptying into the system or through inlets bringing in tidal water and sediment, or sediments brought in during storm events where over-wash events move materials from the barrier island into the lagoon. These are young soils, similar to floodplains in the subaerial system, and having little profile development. Buried horizons are common (figure A 34).

Subaqueous soils provide the structure and habitat for the range of benthic organisms that live in these systems. Submerged aquatic vegetation is rooted in these soils and obtains some nutrients from the soils. Recent studies have shown that subaqueous soils store and sequester equivalent amounts of soil organic carbon as their subaerial counterparts. These soils serve as sinks for heavy metals and under certain conditions are important for water quality, storing N and providing denitrification. Shellfish aquaculture for species such as hard clams and oysters is a common practice on shallow estuarine subaqueous soils.

A range of submerged aquatic vegetation can be found rooted in these soils, depending on the location, climate and water quality. Common species in estuarine systems include eelgrass (Zostera marina), turtle grass (Thalassia sp.), and widgeon grass (Ruppia sp.). In freshwater systems pondweed (Potamogeton sp.), watermilfoil (Myriophyllum sp.), and fanwort (Cabomba sp.) are commonly found.

References

Figure A34 | (a) A Wassent profile and (b) the associated landscape, the Netherlands.

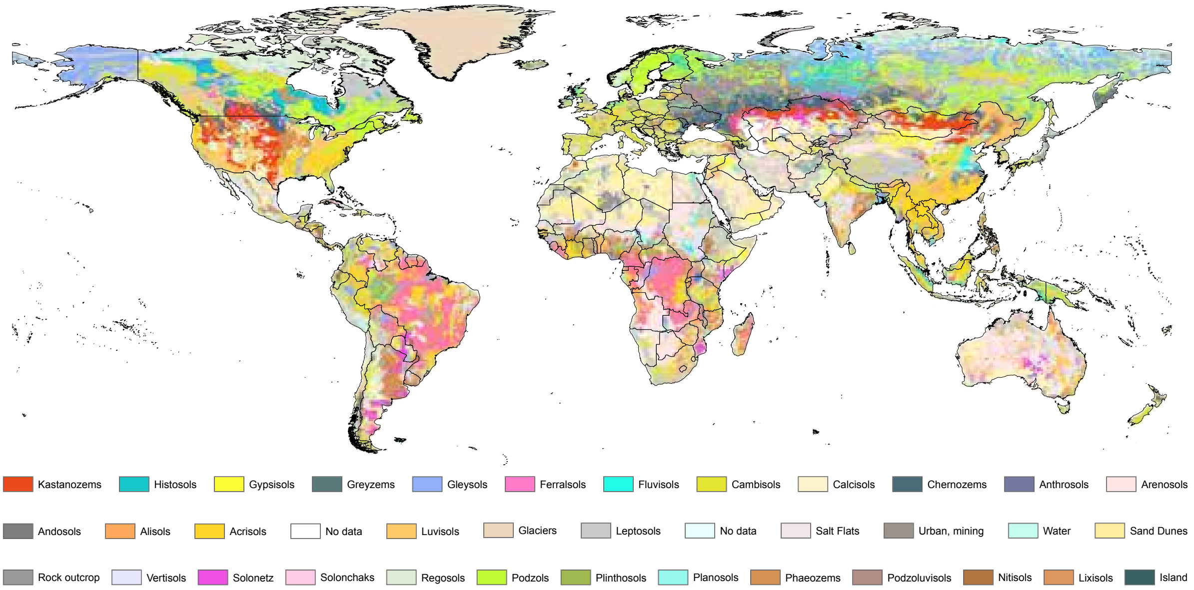


Figure A35 | Global Soil Map of the World based on HWSD and FAO Revised Legend (Nachtergaele and Petri, 2008)

Ahmad, N. & Mermut, A.R., eds. 1996. Vertisols and Technologies for Their Management. *Developments in Soil Science*, Volume 24. Amsterdam, The Netherlands, Elsevier. 548 pp.

Ahmad, N. 1983. Vertisols. In L. P. Wilding., N.E. Smeck & G.F. Hall, eds. *Pedogenesis and Soil Taxonomy II. The Soil Orders. Developments in Soil Science, Volume 11, Part B*, pp. 91-123. Amsterdam, The Netherlands, Elsevier. 410 pp.

Blume, H.P. & Leinweber, P. 2004. Plaggen Soils: landscape history, properties and classification. *J. Plant Nutr*. Soil Sci. 167(3): 319-327.

Boettinger, J. 2002. Calcification. In R. Lal, ed. *Encyclopaedia of Soil Science* , pp. 131-134. New York, Marcel Dekker Inc. 1476 pp.

Brinkman, R. 1979. *Ferrolysis, a soil-forming process in hydromorphic conditions*. Agricututal Research Report 887: vi + 106 pp. Wageningen, PUDOC (PhD thesis)

Charzyński, P., Bednarek, R., Hulisz, P. & Zawadzka, A. 2013. Soils within Toruń urban area. In P. Charzyński, P. Hulisz & R. Bednarek, eds. *Technogenic soils of Poland*, pp. 17–30. Toruń, Polish Society of Soil Science. 357 pp.

Chesworth, W., ed. 2008. *Encyclopedia of soil science*. Springer Science & Business Media

Chichagova, O.A. 1985. *Radiocarbon Dating of Soil Humus*. Moscow, Nauka Publ. 157 pp. [in Russian].

Coulombe, C., Wilding L. & Dixon, J. 1996a. Overview of Vertisols: characteristics and impacts on society. In D.L. Sparks, ed. *Advances in Agronomy, Volume* 57, pp. 289-376. San Diego, Academic Press. Inc. 488 pp.

Coulombe, C.E., Dixon, J.B. & Wilding, L.P. 1996b. Mineralogy and chemistry of Vertisols. *Developments in Soil Science*, 24: 115-200.

Coulombe, C.E., Wilding L.P. & Dixon, J.B. 2000. Vertisols. *In*: M.E. Sumner, ed. *Handbook of Soil Science*, pp. E 269-286. Boca Raton, CRC Press. 1442 pp.

Couwenberg, J., Dommain, R. & Joosten, H. 2010. Greenhouse gas fluxes from tropical peatlands in south-east Asia. *Global Change Biology*, 16(6): 1715-1732.

Dokuchaev, V.V., ed. 1879. *Cartography of Russian Soils*. St.-Petersburg. 123 pp. [in Russian]

Dudal, R. & Eswaran, H. 1988. Distribution, properties and classification of Vertisols. **In** L.P. Wilding & R. Puentes, eds. *Vertisols: Their distribution, properties, classification and management. SMSS Technical Monograph 18*, pp. 1-22. Texas, A&M Printing Center, College Station, TX. 193 pp.

FAO. 2014. *World Reference Base for Soil Resources 2014. International soil classification system for naming soils and creating legends for soil maps*, World Soil Resources Reports No 106, FAO, Rome. 191 pp.

Fridland, M.V., Egorov V.V. & Rudneva, E.N. 1988. *Soil Map of Russian Federation, scale 1:2.5M, 16 sheets. 1988*. Moscow, Dokuchaev Soil Science institute.

Gardi, C., Angelini, M., Barceló, S., Comerma, J., Cruz Gaistardo, C., Encina Rojas, A., Jones, A., Krasilnikov, P., Mendonça Santos Brefin, M.L., Montanarella, L., Muñiz Ugarte, O., Schad, P., Vara Rodríguez, M.I. & Vargas, R., eds. 2014. *Soil Atlas of Latin America and the Caribbean*. Europe Commission – Publications Office of the European Union – L-2995 Luxembourg. 176 pp. [in Spanish]

Gerasimova, M.I. 2002. Genetic and geographic features of the soil cover in Argentinian Pampa. In N.S. Kasimova & M.I. Gerasimova, eds. Landscape geochemistry and soil geography, pp. 324-343. Smolensk, Oecumene. 456 pp. [in Russian]

Jones, A., Breuning-Madsen, H., Brossard, M., Dampha, A., Deckers, J., Dewitte, O., Gallali, T., Hallett,

S., Jones, R., Kilasara, M., Le Roux, P., Micheli, E., Montanarella, L., Spaargaren, O., Thiombiano, L., Van Ranst, E., Yemefack, M. & Zougmoré R., eds. 2013. *Soil Atlas of Africa*. European Commission, Publications Office of the European Union, Luxembourg. 176 pp.

Karavaeva, N.A., ed. 1982. *Bogging and Soil Evolution*. Moscow, Nauka. 296 pp. [in Russian]

Kolka, R.K., Rabenhorst, M.C. & Swanson, D. 2012. Histosols. *In* P.M. Huang, Y. Li & M.E. Sumner, eds. *Handbook of Soil Sciences: Properties and Processes, Second Edition*. CRC Press, pp. 33/8 – 33/29. Taylor & Francis Group, Boca Raton, London, New York.

Lebedeva, I.I. 1974. Modern concepts of chernozems. In V.V. Fridland & I.I. Lebedeva, eds. *Chernozems of the USSR*. Vol. 1, pp. 64-281. Moscow, Kolos. 560 pp. [in Russian]

Lehmann, A. & Stahr, K. 2007. Nature and Significance of Anthropogenic Urban Soils. *J. Soils Sediments*, 7(4): 247–260.

Lucas, Y., Montes, C.R., Mounier, S., Loustau Cazalet, M., Ishida, D., Achard, R., Garnier, C., Coulomb, B. & Melfi, A.J. 2012. Biogeochemistry of an Amazonian podzol-ferralsol soil system with white kaolin. *Geochim. Cosmochim*. Ac., 71: 3211–3222.

McGarity, J.W., Hoult, E.H. & So, H.B., eds. 1984. *The Properties and Utilization of Cracking Clay Soils*. Reviews in Rural Science 5. Armidale, New South Wales, University of New England. 386 pp.

Morel, J.L., Chenu, C. & Lorenz, K. 2015. Ecosystem services provided by soils of urban, industrial, traffic, mining, and military areas (SUITMAs) *J. Soils Sediments*, 15(8): 1659-1666.

Moscatelli, G. 1991. Los suelos de la región Pampeana. *In* O., Barsky, ed. *El desarrollo agropecuario pampeano*. Buenos Aires, Argentina, INDEC-INTA-IICA. 799 pp.

Muckenhausen, E. 1963. Le Pseudogley. *Science du sol*, 1: 21-29.

Nachtergaele, F.O. & Petri, M., eds. 2008. *Mapping land use systems at global and regional scale for land degradation assessment and analysis*. LADA Technical Report #8. FAO, Rome

Nachtergaele, F.O. 2010. *The classification of Leptosols in the World Reference Base for Soil Resources*. 19th World Congress of Soil Science, Soil Solutions for a Changing World 1 – 6 August 2010, Brisbane, Australia. Published on DVD.

Pazos, S.M. 2012. Polygenesis of soils in the central-SE area of Buenos Aires. Trabajo nº 668, Comisión V. Actas XIX Congreso Latinoamericano y XXIII Congreso Argentino de la Ciencia del Suelo. Asociación Argentina de la Ciencia del Suelo y Sociedad Latinoamericana de la Ciencia del Suelo. Mar del Plata. 6 pp. [In Spanish]

Puentes, R. &. Wilding, L.P. 1990. *Effects of long term soil management on infiltration rates and macroporosity of vertisols*. Symposium Session B 1. Trans. 14th Int. Congress Soil Sci., Kyoto, Japan, 5: 244-249.

Retallack, G.V., ed. 2001. *Soils of the Past: an Introduction to Paleopedology*. 2nd Edition. Oxford, Blackwell Science. 600 pp.

Roman, L., Scatena, F.N. & Bruijnzeel, L.A. 2010. Global and local variations in tropical montane cloud forest. In L.A. Bruijnzeel, F.N Scatena & L.S. Hamilton, eds. *Tropical montane cloud forest science for conservation and management*, pp.77-89. Cambridge, UK, Cambridge University Press. 691 pp.

Sauer, D., Sponagel, H., Sommer, M., Giani, L., Jahn, R. & Stahr, K. 2007. Review article – Podzol: Soil of the year 2007 – A review on its genesis, occurrence, and functions. *Journal of Plant Nutrition and Soil Science*, 170(5): 581-597.

Soil Survey Staff, ed. 1999. Chapter 20: Vertisols. *In Soil Taxonomy. A Basic System of Soil Classification for Making and Interpreting Soil Surveys. 2nd Edition. Agriculture Handbook No. 436*. Pp. 783-817. Washington, DC, U.S. Department of Agriculture, Natural Resources Conservation Service. 886 pp.

Soil Survey Staff, ed. 2014. *Keys to Soil taxonomy*. 12th Edition. Washington, DC, U.S. Departmnet of

Agriculture, Natural Resources Conservation Service. 372 pp.

Sombroek, W.G., ed. 1966. *Amazon Soils: A Reconnaissance of the Soils of the Brazilian Amazon Region*. Wageningen, The Netherlands, PUDOC. 300 pp.

Stigliani, W.M. 1988. Changes in valued capacities of soils and sediments as indicators of nonlinear and time-delayed environmental effects. *International Journal of Environmental Monitoring and Assessment*, 10(3): 245-307.

Strack, M., ed. 2008. *Peatlands and climate change*. Jyväskylä, Finland, International Peat Society. 227 pp.

Stroganova, M., Miagkova, A., Prokofieva, T., Skvortsova, I. & Karpachevskiĭ, M.L., eds. 1998. Soils of Moscow and urban environment. Moscow. 177 pp.

Tarnocai C. & Campbell, I. 2002. Soils of the polar regions, In R. Lal, ed. *Encyclopaedia of Soil Science*, pp. 1018-1012. New York-Basel, Marcel Dekker Inc. 1476 pp.

Tarnocai, C., Canadell, J.G., Schuur, E.A.G., Kuhry, P., Mazhitova, G. & Zimov, S. 2009. Soil organic carbon pools in the northern circumpolar permafrost region. *Global Biochemical Cycles* 23(2): GB 2023.

Van Ranst, E., Dumon, M., Tolossa, A.R., Cornelis, J.T., Stoops, G., Vandenberghe, R.E. & Deckers, S. 2011. Revisiting ferolysis processes in the formation of Planosols for rationalizing the soils with stagnic properties in WRB. *Geoderma* 163(3-4): 265-274.

Vostokova, E.A. & Gunin, P.D., eds. 2005. Ecosystems of Mongolia – Atlas (General Scientific Edition). Moscow, Russian Academy of Sciences. 48 pp.

Wang, Q. & Batkhishig, O. 2014. Impact of Overgrazing on Semiarid Ecosystem Soil Properties: A Case Study of the Eastern Hovsogol Lake Area, Mongolia. *Journal of Ecosystem & Ecography*, 4(1): 140.

Wilding, L.P. &Tessier D. 1988. Genesis of Vertisols: Shrink swell phenomena. *In* L.P. Wilding. & R. Puentes, eds. *Vertisols: Their Distribution, Properties, Classification and Management*. pp. 55-81.Texas A&M University Press. 193 pp.

Wilding, L.P. 2000. Classification of Soils.In M.E. Sumner, ed. *Handbook of Soil Science*. pp. E 175-E 183. CRC Press.

World Energy Council, ed. 2013. World Energy insight 2013. London, UK, FIRST. 100 pp.

Zaidelman, F.R. 1994. A concept of gleyization and its role in the pedogenesis. *Archives of agronomy and soil science 38(5):* 323-335.

Zech, W., Schad, P. & Hintermaier-Erhard, G., eds. 2014. Böden der Welt. Ein Bildatlas. Springer Spektrum. Berlin Heidelberg, 164 pp.

Glossary of technical terms

Aerobic: a condition in which molecular oxygen is freely available (ISO, 2013).

Anaerobic: descriptive of a condition in which molecular oxygen is not available (ISO, 2013).

Available water capacity: soil water content useable by plants, based on the effective root penetration depth (ISO, 2013).

Bare Soil: a land cover class that includes any geographic area dominated by natural abiotic surfaces (bare soil, sand, rocks, etc.) where the natural vegetation is absent or almost absent (covers less than 2 percent) (Latham *et al.*, 2014).

Biodegradation: physical and chemical breakdown of a substance by living organisms, mainly bacteria and/or fungi (ISO, 2013).

Contaminant: substance or agent present in the soil as a result of human activity (ISO, 2013).

Cropland: a land cover class that includes all cultivated herbaceous crops, woody crops and multiple and layered crops (Latham *et al.*, 2014).

Decomposition: breakdown of complex organic substances into simpler molecules or ions by physical, chemical and/or biological processes (ISO, 2013).

Desertification: land degradation in arid, semi-arid and dry sub-humid areas resulting from various factors, including climatic variations and human activities (UNCCD, 2011).

Drylands: tropical and temperate areas with an aridity index (annual rainfall/annual potential evaporation) of less than 0.65 (UNEP, 2005).

Grassland: a land cover class that includes any geographic area dominated by natural herbaceous plants (grasslands, prairies, steppes and savannahs) with a cover of 10 percent or more, irrespective of different human and/or animal activities e.g. grazing, selective fire management (Latham *et al.*, 2014).

Habitat ecosystem functions: the ability of soil or soil materials to serve as a habitat for micro-organisms, plants, soil-living animals and their interactions (ISO, 2013).

Humification: decomposition of organic material followed by a synthesis of humic substances (ISO, 2013).

Land: terrestrial bio-productive system that comprises soil, vegetation, other biota, and the ecological and hydrological processes that operate within the system (UNCCD, 2011).

Leaching: the dissolution and movement of dissolved substances by water (ISO, 2013).

Mineralization: final stage of the biodegradation of organic matter or organic substances into carbon dioxide, water and hydrides, oxides or other mineral salts (ISO, 2013).

Mitigation (of land degradation): an intervention intended to reduce ongoing degradation at a stage when degradation has already begun. The main aim here is to halt further degradation and to start improving resources and their functions (FAO, 2015).

Parent material: The unconsolidated and more or less chemically weathered mineral or organic matter from which the solum of soils is developed by pedogenic processes (Soil Science Society of America, 2008).

Particle size distribution: distribution of the soil mineral particles according to predefined classes of size (ISO, 2013).

Pedon: the smallest, three-dimensional unit at the surface of the earth that is considered as a soil. It forms a conceptual foundation for the study of soils as geographic entities (Hole and Campbell, 1985).

pH-value: the negative logarithm (base 10) of the concentration of hydrogen ions, expressed in moles/l in aqueous solution and varying between 0 (extremely acid) to 14 (extremely alkaline) (ISO, 2013).

Rehabilitation: action to restore soil already degraded to such an extent that the original use is no longer possible and the land has become practically unproductive. Generally, long term and often costly investments are needed to show any impact (FAO, 2015).

Shrub-covered area: a land cover class that includes any geographical area dominated by natural shrubs having a cover of 10 percent or more (Latham *et al.*, 2014).

Soil: the upper layer of the Earth's crust transformed by weathering and physical/chemical and biological processes. It is composed of mineral particles, organic matter, water, air and living organisms organized in genetic soil horizons (ISO, 2013).

Soil degradation: the diminishing capacity of the soil to provide ecosystem goods and services as desired by its stakeholders (refined from FAO, 2015).

Soil ecosystem functions: description of the significance of soils to humans and the environment. Examples are: (1) control of substance and energy cycles within ecosystems; (2) basis for the life of plants, animals and man; (3) basis for the stability of buildings and roads; (4) basis for agriculture and forestry; (5) carrier of genetic reservoir; (6) document of natural history; and (7) archaeological and paleo-ecological document (ISO, 2013).

Soil health: the continued capacity of the soil to function as a vital living system, within ecosystem and land-use boundaries, to sustain biological productivity, promote the quality of air and water environments, and maintain plant, animal, and human health (Doran, Stamatiadis and Haberern, 2002).

Soil organic carbon (SOC): a summarizing parameter including all of the carbon forms for dissolved (DOC: Dissolved Organic Carbon) and total organic compounds (TOC: Total Organic Carbon) in soils (ISO, 2013).

Soil organic matter (SOM): matter consisting of plant and/or animal organic materials, and the conversion products of those materials in soils (ISO, 2013).

Soil Processes: physical or reactive geochemical and biological processes which may attenuate, concentrate, immobilize, liberate, degrade or otherwise transform substances in soil (ISO, 2013).

Soil quality: all current positive or negative properties with regard to soil utilization and soil functions (ISO, 2013).

Soil structure: the arrangement of soil particles in a variety of recognized shapes and sizes (ISO, 2013).

Soil threats: see Box 'Soil Threat Definitions'

Box 'Definitions of Soil Threats'

Nutrient imbalance refers to an excess or a lack of nutrients (mainly nitrogen, phosphorus and potassium) in the soil as a consequence of bad land use and management. It may result in soil contamination when nutrients are in excess and in loss of inherent fertility when nutrients are mined.

Soil acidification is defined as the lowering of the soil pH because of the buildup of hydrogen and aluminum ions in the soil and the leaching of base cations such as calcium, magnesium, potassium and sodium. Soil acidification negatively affects soil fertility and compromises the production capacity of most agricultural soils.

Soil biodiversity loss is a decline in the diversity of (micro- and macro-) organisms present in a soil. In turn, this prejudices the ability of soil to provide critical ecosystem services.

Soil compaction is defined as the increase in density and a decline of macro-porosity in a soil that impairs the functions of both the top- and subsoil, and impedes roots penetration and water and gaseous exchanges.

Soil contamination refers to the increase of toxic compounds (heavy metals, pesticides, etc.) in a soil that constitute, directly or indirectly (via the food chain), a hazard for human health and/or for the provision of ecosystem services assured by the soil.

Soil erosion is broadly defined as the removal of (top-) soil from the land surface by running water, wind, ice or gravity. It can be accelerated by human activities (tillage) and animals.

Soil organic carbon loss refers to the decline of organic carbon stock in the soil affecting its fertility status and climate change regulation capacity.

Soil salinization is defined as the increase in water-soluble salts in soil which is responsible for increasing the osmotic pressure of the soil. In turn, this negatively affects plant growth because less water is made available to plants.

Soil sealing refers to the permanent covering of the soil surface with impermeable artificial materials such as asphalt and concrete. This is generally related to urban development and infrastructure construction, which in most cases lead to the absolute loss of the soil resource and of most of its ecosystem services.

Soil sodification is defined as an increase of the exchangeable sodium content of the soil, often accompanied by a loss of soil structure. In turn, it negatively affects soil suitability for crop growth.

Water logging refers to an excess of water on top and/or within the soil, leading to reduced air availability in the soil for long periods.

Solum: comprises the surface layer and subsoil layers that have been altered by soil formation (Soil Survey Staff, 1993).

Sparse vegetation: a land cover class that includes any geographic areas where the cover of natural vegetation is between 2 percent and 10 percent (Latham *et al.*, 2014).

Sustainable land management (SLM): the use of land resources, including soils, water, animals and plants for the production of goods to meet changing human needs while ensuring the long term productive potential of these resources and the maintenance of their environmental functions (UNCED, 1992).

Sustainable soil management (SSM): sets of activities that maintain or enhance the supporting, provisioning, regulating and cultural services provided by soils without significantly impairing either the soil functions that enable those services or biodiversity (adapted from GSP, 2015).

Topsoil: the upper part of a natural soil that is generally dark coloured and has a higher content of organic matter and nutrients when compared to the (mineral) horizons below. It excludes the litter layer (ISO, 2013).

Tree-covered area: a land cover class that includes any geographic area dominated by natural tree plants with a cover of 10 percent or more. Areas planted with trees for afforestation purposes and forest plantations are included in this class (Latham *et al.*, 2014).

References

Doran, J.W., Stamatiadis, S. & Haberern, J. 2002. Soil health as an indicator for sustainable management. Agriculture, Ecosystems and Environment. 88(2002): 107–110.

FAO & ITPS. 2015. Status of the World's Soil Resources (SWSR). Food and Agriculture Organization of the United Nations and Intergovernmental Technical Panel on Soils, Rome, Italy. In press.

FAO. 2015. FAO soils portal. Available at http://www.fao.org/soils-portal/it/

GSP. 2015. Revised World Soil Charter. Available at http://www.fao.org/fileadmin/user_upload/GSP/docs/ITPS_Pillars/annexVII_WSC.pdf

ISO. 2013. Draft international standard ISO/DIS 11074. 9 pp.

Latham, J., Cumani, R., Rosati, I. & Bloise, M. 2014. Global Land Cover SHARE (GLC-SHARE) database Beta-Release Version 1.0 - FAO, Rome.

Soil Science Society of America. 2008. Glossary of soil science terms. Available at https://www.soils.org/publications/soils-glossary

Hole, F.D. & Campbell, J.B. 1985. Soil Landscape Analysis. Rowman and Allanheld, Totowa, NJ. 196 pp.

Soil Survey Staff. 1993. "Soil Survey Manual". Soil Conservation Service. U.S. Department of Agriculture Handbook 18.

UNCCD. 2011. Desertification: a visual synthesis. UN Convention to Combat Dersertification (UNCCD) Secretariat. 50 pp.

UNCED. 1992. United Nations Conference on Environment & Development Rio de Janeiro, Brazil, 3 to 14 June 1992. 351 pp.

UNEP. 2005. Chapter 22. Drylands Systems. In R., Hassan, R., Scholes & N. Ash, eds. Ecosystems and Human Wellbeing: Current State and Trends, Volume 1. pp. 623-662. Millennium Ecosystem Assessment, Island Press.

Authors and affiliations

Adams, Mary Beth	USDA Forest Service, West Viriginia, USA.
Adhya Tapan, Kumar	KIIT School of Biotechnology, Odisha, India.
Agus, Fahmuddin	Indonesian Soil Research Institute, Indonesia.
Al Shankithi, Abdullah	ICBA, United Arab Emirates.
Alavi Panah, Sayed Kazem	University of Teheran, Iran.
Alegre, Julio	National Agrarian University La Molina, Peru.
Aleman, Garcia	Ministry of Agriculture, Cuba.
Alfaro, Marta	INIA, Remehue, Chile.
Alyabina, Irina	Lomonosov Moscow State University, Russian Federation
Anderson, Chris	Massey University, New Zealand.
Anjos, Lucia	Federal Rural University of Rio de Janeiro, Brazil.
Arao, Tomohito	National Institute for Agro-Environmental Science (NIAES), Japan
Arrouays, Dominique	National Institute for Agricultural Research, France.
Asakawa, Susumu	Nagoya University, Japan.
Aulakh, Milkha Singh	Punjab Agricultural University, India.
Ayuke, Frederick	University of Nairobi, Kenya.
Badraoui, Mohamed	National Agronomic Research Institute, Morocco.
Bai, Zhaohai	China Agriculture University, China.
Baldock, Jeff	CSIRO, Australia.
Balks, Megan	University of Waikato, New Zealand.
Balyuk, Svyatoslav	National Institute of Soil Science, Ukraine
Bardgett, Richard	Manchester University, UK.
Basiliko, Nathan	Laurentian University, Canada
Bationo, André	AGRA Kenya. (Burkina Faso).
Batkhishig, Ochirbat	Academy of Sciences, Mongolia
Bedard-Haughn, Angela	University of Saskatchewan, Canada.
Bielders, Charles	Catholic University Louvain, Belgium (USA)
Black, Helaina	The James Hutton Institute, UK
Bock, Michael	Agriculture and AgriFood Canada, Canada
Bockheim, James	Wisconsin University, USA.
Bondeau, Alberte	Mediterranean Institute of Biodiversity and Ecology (IMBE), France
Brinkman, Robert	Soil Science Society The Netherlands.
Bristow, Keith	CSIRO, Australia.

Broll, Gabrielle	Osnabrück University, Germany.
Bruulsma, Tom	Paul Fixen, IPNI. Guelph Ontario, Canada.
Bunning, Sally	FAO, United Nations. (UK)
Bustamante, Mercedes	University of Brasilia, Brazil.
Camps Arbestain, Marta	Massey University, New Zealand.
Caon, Lucrezia	FAO, United Nations (Italy).
Carating, Rodel	Bureau of Soils and Water Management, Philippines
Cerkowniak, Darrel	Agriculture and AgriFood Canada, Canada
Charzynski, Przemyslaw	Nicolaus Copernicus University, Poland.
Chude, Victor	Ministry of Agriculture, Nigeria.
Clark, Joanna	Reading University, England, UK.
Clothier, Brent	Food Ind. Science Centre, New Zealand.
Coelho, Maurício Rizzato	EMBRAPA, Brazil
Colditz, Roland René	Comision Nacional para el Conocimiento y Uso de la Biodiversidad (CONABIO)
Collins, Alison	NLRC, New Zealand.
Comerma, Juan	Soil Science Society Venezuela, Venezuela.
Compton, Jana	EPA, USA
Condron, Leo	Lincoln University, New Zealand.
Corso, Maria Laura	Min. Environment, Argentina.
Cotrufo, Francesca	Colorado State University, USA (Italy)
Critchley, William	VU Amsterdam, The Netherlands (UK)
Cruse, Richard	Iowa state University, USA.
da Silva, Manuela	Joint Research Centre, Italy (Sweden/Brazil)
Dabney, Seth	USDA, USA.
Daniels, Lee	Virginia Tech , USA.
de Souza Dias, Moacir	Universidade Federal de Lavras, Brazil.
Dick, Warren	Ohio State University, USA
Dos Santos Baptista, Isaurinda	INIDA, Cape Verde.
Drury, Craig	Agriculture and AgriFood Canada, Canada
El Mustafa El Sheikh, Ahmed Elsiddig	University of Khartoum, Sudan.
Elder-Ratutokarua, Maria	Secretariat of the Pacific Community
Elliott, Jane	Saskatchwan University, Canada.
Espinosa Victoria, David	Colegio de Postgraduados in Mexico
Fendorf, Scott	Stanford University, USA.
Ferreira, Gustavo	INIA, Urugay
Flanagan, Dennis	Purdue University, USA.
Fraser, Tandra	GSBI, Colorado State University, USA (Can)
Gafurova, Laziza	National University, Tashkent, Uzbekistan.

Gaistardo, Carlos Cruz	INEGI, Mexico
Gardi, Ciro	University of Parma,Italy
Gerasimova, Maria	Moscow State University, Russian Federation.
Govers, Gerard	Catholic University Leuven, Belgium.
Grayson, Sue	University of British Columbia, Canada.
Griffiths, Robert	Centre for Ecology & Hydrology, UK
Grundy, Mike	CSIRO, Australia
Hakki Emrah, Erdogan	Ministry Food, Agriculture and Livestock, Turkey
Hamrouni, Heidi	Ministry of Agriculture, Tunisia
Hanly, James	Massey University, New Zealand.
Harper, Richard	Murdoch University, Perth, Australia
Harrison, Rob	University of Washington
Havlicek, Elena	Université de Neufchatel.(CH)
Hempel, Jon	NRCS, Lincoln, USA.
Henriquez, Carlos Roberto	University of Costa Rica, Costa Rica.
Hewitt, Allan	Landcare Research, New Zealand.
Hiederer, Roland	Joint Research Center - EU (Germany)
Hong, Suk Young	National Academy of Agricultural Science, RDA, South Korea
House, Jo	University of Bristol, England, UK.
Huising, Jeroen	TSBF-CIAT, Kenya (The Netherlands)
Ibánez, Juan José	Spanish National Research Council
Indraratne, Srimathie	University of Peradeniya, Sri Lanka.
Jain, Atul	Department of Atmospheric Sciences, University of Illinois, USA
Jefwa, Joyce	TSBF-CIAT, Kenya.
Jung, Kangho	National Academy of Agricultural Science, RDA, South Korea
Kadono, Atsunobu	Tottori University of Environmental Studies, Japan
Kawahigashi, Masayuki	Tokyo Metropolitan University , Japan
Kelliher, Frank	AgResearch, New Zealand.
Kihara, Job	CIAT, Kenya.
Konyushkova, Maria	Moscow State University, Moscow, Russia
Krasilnikov, Pavel	Moscow State University, Russian Federation.
Kuikman, Peter	Wageningen University, The Netherlands.
Kuziev, Ramazan	National Institute Soil Science, Uzbekistan.
Lai, Shawntine	MWH Americas Inc. Taiwan branch.
Lal, Rattan	Ohio State University, USA
Lamers, John	Bonn University, Germany.
Lee, Dar-Yuan	National Taiwan University, Taiwan
Lee, Seung Heon	Korea Rural Community Corp., Korea
Lehmann, Johannes	Cornell University, USA (Germany)

Leys, John	NSW Office of Environment and Heritage, Australia
Lobb, David	University of Manitoba, Canada
Ma, Lin	Institute of Genetic and Developmental Biology, CAS, China
Macias, Felipe	Universidade de Santiago de Compostela, Spain.
Maina, Fredah	Agricultural Research Institute, Kenya.
Mamo, Tekalign	Ministry of Agriculture, Ethiopia.
Mantel, Stephan	World Soil Information (ISRIC), the Netherlands
McDowell, Richard	AgResearch, New Zealand.
McKenzie, Neil	CSIRO, Australia
Medvedev, Vitaliy	Ukrainan Agricultural Academy
Mendonça-Santos, de Lourdes Maria	EMBRAPA, Brazil.
Miyazaki, Tsuyushi	University of Tokyo, Japan.
Montanarella, Luca	Joint Research Center - EU (Italy)
Moore, John	Colorado State University, USA
Morrison, John	University of Wollongong, Australia
Mubarak, Abdelrahman Abdalla	University of Khartoum, Sudan.
Mung'atu, Joseph	Jomo Kenyatta University, Kenya.
Muniz, Olegario	Soil Institute, Cuba.
Nachtergaele, Freddy	FAO, United Nations (Belgium)
Nanzyo, Masami	Tohoku University, Japan.
Ndiaye, Déthié	CSE, Senegal
Neall, Vince	Massey University, New Zealand.
Norbu, Chencho	Ministry of Agriculture and Forests, Buthan
Noroozi, Ali Akbar	SCWM Institute, Iran.
Obst, Carl	University of Melbourne, Australia
Ogle, Stephen	Colorado State University, USA.
Ogunkunle, Ayoade	University of Ibadan, Nigeria.
Okoth, Peter	TSBF-CIAT, Kenya.
Omutu, Christian	University of Nairobi, Kenya.
Or, Dani	ETH, Switzerland.
Owens, Phil	Purdue University, USA.
Pan, Genxing	Nanjing Agricultural University, China.
Panagos, Panos	Joint Research Center - EU (Greece)
Parikh, Sanjai	University of California-Davis, USA.
Pasos Mabel, Susan (†)	University Buenos Aires, Argentina.
Paterson, Garry	ISCW, Republic of South Africa.
Paustian, Keith	Colorado State University, USA.
Pennock, Dan	University of Saskatchewan, Canada.
Pietragalla, Vanina	Min. Environment, Argentina.

Pla Sentis, Ildefonso	Universitat de Lleida, Spain (Venezuela)
Polizzotto, Matthew	North Carolina State, USA.
Pugh, Thomas	Karlsruhe Institute of Technology, Institute of Meteorology and Climate Research/Atmospheric Environmental Research (IMK-IFU), Germany
Qureshi, Asad	International Centre for Biosaline Agriculture, UAE.
Reddy, Obi	National Bureau of Soil Survey & Land Use Planning, India.
Reid, D. Keith	Ontario Ministry of Agriculture, Canada
Reinsch, Thomas	World Soil Resources, USDA, USA.
Richter, Dan	Duke University, USA.
Rivera-Ferre, Marta	Universita de Vic, Spain.
Robinson, David	Centre for Ecology & Hydrology, UK.
Rodriguez Lado, Luis	Universidade de Santiago de Compostela, Spain.
Roskruge, Rick	Massey University, New Zealand.
Rumpel, Cornelia	INRA, France.
Rys, Gerald	Ministry Primary Industries, New Zealand
Schipper, Louis	The University of Waikato, New Zealand
Schoknecht, Noel	Department of Agriculture and Food, Western Australia
Seneviratne, Sonia	ETH, Switzerland.
Shahid, Shabbir	International Centre for Biosaline Agriculture, UAE.
Sheffield, Justin	Princeton University, USA (UK)
Sheppard, Steve	ECOMatters Inc., Canada
Sidhu, Gurjant	National Bureau of Soil Survey & Land Use Planning, India.
Sigbert, Huber	Umweltbundesamt GmbH, Austria
Smith, Pete	Aberdeen University, Scotland, UK.
Smith, Scott	Agriculture and AgriFood Canada, Canada
Sobocká, Jaroslava	Soil Science and Conservation Institute, Slovakia.
Sönmez, Bülent	Ministry of food agriculture and livestock, Turkey.
Spicer, Anne	Lincoln University, New Zealand
Sposito, Garrison	Berkeley University, USA
Stolt, Mark	University of Rhode Island, USA.
Suarez, Don	USDA Salinity Laboratory, Riverside, USA
Taboada, Miguel	INTA, Argentina.
Takata, Yusuke	National Institute for Agro-Environmental Science (NIAES), Japan
Tarnocai, Charles	Agriculture and Agri.Food, Canada.
Tassinari, Diego	Universidade Federal de Lavras, Brazil.
Tien, Tran Minh	Soils and Fertilizers Research Institute, Vietnam.
Toth, Tibor	RISSAC, Hungary.
Trumbore, Susan	Max Planck Institute, Germany (USA)
Tuller, Markus	University of Arizona, USA.

Urquiaga Caballero, Segundo	Embrapa Agrobiologica, Brazil.
Urquiza Rodrigues, Nery	Ministry of Agriculture, Cuba.
Van Liedekerke, Marc	Joint Research Center, EU. (BEL)
Van Oost, Kristof	Catholic University Louvain, Belgium.
Vargas, Rodrigo	CISECE, Mexico.
Vargas, Ronald	FAO, United Nations (Bolivia)
Vela, Sebastian	La Molina University, Peru.
Vijarnsorn, Pisoot	Chaipattana Foundation, Thailand.
Vitaliy, Medvedev	National Institute of Soil Science Ukraine.
Vrscaj, Boris	Agricultural Institute of Slovenia.
Wall, Diana	Colorado State University, USA
Waswa, Boaz	CIAT, Kenya.
Watanabe, Kazuhiko	Hyogo Agricultural Institute, Japan.
Watmough, Shaun	Trent University, Canada.
Webb, Mike	CSIRO, Australia
Weerahewa, Jeevika	University of Peradeniya, Sri Lanka.
West, Paul	University of Minesota, USA.
Wiese, Liesl	ARC - ISCW, Republic of South Africa.
Wilding, Larry	Texas A&M University, USA.
Xu, Renkou	Institute of Soil Science, CAS, China
Yagi, Kazuyuki	National Institute for Agro-Environmental Science (NIAES), Japan
Yan, Xiaoyuan	Institute of Soil Science, CAS, China
Yemefack, Martin	IRAT/IITA, Cameroon
Yokoyama, Kazunari	National Agriculture and Food Research Organization (NARO), Japan
Zhang, Fusuo	China Agriculture University, China.
Zhang, Gan Lin	Institute of Soil Science, China.
Zhou, Dongmei	Institute of Soil Science, CAS, China.
Zobeck, Ted	USDA, USA.