EEA ETC/ULS Report | 2021

Soil monitoring in Europe

Indicators and thresholds for soil quality assessments

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MORE REVIEWERS WILL BE LISTED AFTER THE EXTERNAL REVIEW!!!!!

Temporary note: the report is being reviewed by the EEA Member countries in the National Reference Centres for Soil under the European Information and Observation Network (EIONET), the European Commission, experts from European research projects (EJP Soils, HoliSoil, BonaRes), the European Soil Partnership, the Common Forum on Contaminated land, and the European Expert Group on Soil Protection.

Abbreviations

Institutions

EEA	European Environment Agency
EC	European Commission
EIONET	European Environment Information and Observation Network
ETC/ULS	European Topic Centre on Urban Land and Soil Systems
EUROSTAT	Statistical office of the European Union.
FAO	Food and Agriculture Organisation of the United Nations
IPCC	Intergovernmental Panel on Climate Change
ITPS	Intergovernmental Technical Panel on Soils
OECD	Organisation for Economic Co-operation and Development
UN	United Nations
UNFCCC	United Nations Framework Convention on Climate Change
UNCCD	The United Nations Convention to Combat Desertification
WHO	World Health Organization

Other abbreviations

1OAO	One out all out principle
ADI	Acceptable daily intake
AEI	Agri Environmental Indicator
ASC	achievable soil organic carbon sequestration
ATSDR	United States Agency for Toxic Substances and Disease Registry
BAU	Business as usual
CAP	Common Agricultural Policy Regulation
Db	Bulk density
DoE	United States Department of Energy
EAP	Environment Action Programme
ENVASSO	Environmental assessment of soil for monitoring (research project)
F2F	Farm to Fork Strategy
GAEC	Good agricultural and environmental condition of land
GLASOD	Global Assessment of Human Induced Soil Degradation
K_{sat}	Saturated water conductivity
IARC	International Agency for Research on Cancer
IRIS	Integrated Risk Information System
LANDMARK	Sustainable management of land and soil in Europe (research project)
LTE	Long-term experiment
LULUCF	Land use, land-use change and forestry
MAES	Mapping and assessment of ecosystems and their services
MAOM	Mineral-Associated Organic Matter
MS	Member States
NDVI	Normalized difference vegetation index
NEC	National Emissions Ceilings Directive
NIR	Near-infrared radiation
n_g	Air capacity (air filled pore volume)
OR	Operating ranges
PD	Packing density
PLFA	Phospholipid-derived fatty acids
POM	Particulate organic matter

PR	Penetrometer resistance
PSI	critical phosphorus saturation index
PTF	Pedo-transfer function
PW	phosphorus in water
RBLM	Risk-based land management
RAR	Risk Assessment Reports
RECARE	Preventing and remediating degradation of soils in Europe (research project)
RED	Visible radiation
RIVM	Dutch National Institute for Public Health and the Environment
RND	Relative Normalised Density or “degree of compaction”
RUSLE	Revised Universal Soil Loss Equation
SDG	Sustainable Development Goal
STE	Short-term experiment
SOC	Soil organic carbon
SOER	State of the Environment and Outlook Report
SOM	Soil organic matter
SSM	Sustainable Soil Management
SV	Screening value
TM	Trace metals
US EPA	United States Environmental Protection Agency
USLE	Universal Soil Loss Equation
VESS	Visual Evaluation of Soil Structure
WFD	Water Framework Directive
WEPP	Water Erosion Prediction Project

About this report

For the last few years, the European Topic Centre on Urban Land and Soil Systems (ETC/ULS) has followed EU-level research and literature on soil indicators, its relation to soil functions and soil threats, its mapping and assessment. The objective of this report is to synthesize the current knowledge about soil indicators in the context of land degradation, ecosystem condition and soil resource use efficiency.

The ETC/ULS is a part of the Environmental Information and Observation Network (Eionet), which supports various EEA activity streams, in close cooperation with the National Reference Centres Soil (NRC Soil). Progress and tasks of the ETC/ULS are regularly discussed, reviewed, and further supported by the NRC Soil. While this report is deeply anchored in this cooperation, additional experts were consulted to complete, and quality assure the various indicators as they cover a broad range of soil threats. In addition, the report has been broadly reviewed by researchers and policy representatives.

Both above-mentioned objectives – soil resource use efficiency and soil degradation assessment - are difficult to achieve, provided the vast number of available publications, and – in contrast - the lack of exhaustive, applicable, repeatable, and agreed definitions, sampling, and analytical methods. This report contains the currently known definitions of priority soil indicators and extends these definitions where needed. It also collects relevant information about available thresholds and evaluation schemes.

This report will not explain how soil properties can be measured, or how a European soil monitoring system can look like[1] - though some specifications are provided. Rather, this report defines prominent indicators related to soil quality, and how they can be evaluated.

1 In contrast to this report about soil indicators and its evaluation, a follow-up report on soil monitoring design in Europe would still be need including agreements on target parameters, sampling and analytical schemes, criteria for spatial-temporal representativity, statistical design specifications to detect uncertainties and trend, and – if appropriate - levels of measurement intensity and integration with other environmental observation activities (climate, air quality, biodiversity, water quality).

Executive Summary

Soil is a finite, non-renewable resource because its regeneration takes longer than a human lifetime. Soil is a fundamental resource of Europe's natural capital, and it contributes to basic human needs by supporting, among others, food provision and water purification, while acting as a major store for organic carbon and a habitat for extremely diverse biological communities.

European soils are under increasing pressure. Key trends are above all:

- Urban sprawl and low land recycling rates condition continued soil loss from sealing and soil replacement (e.g. for construction).
- Intensification of agriculture where use of fertilizers and plant protection products is high,
- Climate change, where it causes weather extremes such as drought and wildfires.

However, land management also positively influences soil quality. Many soil functions can be improved if appropriate practices are in place, particularly to sequester soil carbon and maintain or improve soil biodiversity. Other soil functions can be preserved where certain pressures from intensive land use prevent erosion and compaction.

Resilient, healthy soils are important to help reducing ecological and economic impact from environmental change and extreme conditions. They are an integral element of the European Green Deal, and a target of environmental measures under the Common Agricultural Policy. To support protection targets related to soils, its condition and functioning must be assessed using proper indicator sets and thresholds, which signal to practitioners and policy makers the success of the recommended management practices.

The development of adequate and broadly applicable indicators and thresholds is challenged by the great diversity of European soils and climate, as well as different political, economic, and social conditions which lead to different priority settings for targets and indicators. There are 23 main soil types[2], four prevailing macroclimatic zones[3], and eight recognised soil threats[4], which all together form a complex matrix of basic different environmental growing conditions, whereas each of them requires specific responses to optimize and sustainably use the available resources.

This report describes the rationale for a series of common and broadly accepted soil quality indicators. The indicators were selected in view of their appropriateness to assess the condition of soils, its degradation, its resilience, and its valuable services. In particular, the available state-of-the-art knowledge has been compiled to evaluate each indicator using thresholds for the good condition of soils. In this respect, the report provides a framework for the observation of soils, using a broadly accepted indicators; they are herewith specified with the objective to help achieving the best possible degree of harmonization.

2 JRC (2008): Soil Atlas of Europe; ISBN: ISBN 92-894-8120-X

3 Climate zones according to the Köppen Geiger classification https://en.wikipedia.org/wiki/K%C3%B6ppen_climate_classification#/media/File:Europe_map_of_K%C3%B6ppen_climate_classification.svg

4 COM(2006)/231, Thematic Strategy for Soil Protection (including soil biodiversity)

1. Soil quality assessment

This chapter
- provides definitions for different types of soil degradation,
- explains the importance of thresholds for specific parameters related to soil degradation,
- proposes a methodology for soil quality assessment, and
- provides an overview of current EU policy requirements towards soil protection and gives an outlook to relevant emerging policies and what can be expected from them.

Soil quality can be viewed as a measure of how soil functions (Schjønning et al. 2004); it is the capacity of a soil to sustain its functions, including food and other biomass production, storage, filtering, buffering and transformations of natural and anthropogenic produced substances, a biological habitat and gene reservoir and a sink for carbon. Soil quality is described using **indicators**, which are observed and evaluated soil properties, and which indicate the degree to which degree soils fulfils expected functions as needed for the wellbeing of crops, livestock, and consequently, human health. Important selection criteria for indicators, and the underlying soil properties, are their responsiveness to management and changed environmental conditions; they must correlate with soil functions and the environmental processes affected by disturbances and change.

In the context of sustainability and resource use efficiency, it appeared that many soils do not fulfil their function to the full potential (JRC 2012; EC 2020). Such soils are degraded, its functioning is less than optimal or expected; the ability of soils to provide and regulate ecosystem services is reduced. Soil monitoring, targeted to observe soil properties and its indicators over time, and in relation to management and environmental change, helps to identify the specific region/location where degradation prevails, so that measures to restore soil functions can be applied.

To understand at which level of indicator performance a soil fulfils its functions, thresholds are important, being either critical limits or target values. **Thresholds** are perceived as values above or below which a significant shift or rapid negative change takes place (Van Lynden et al., 2004). Beyond such values, soils would be considered as degraded, with restoring action needed.

1.1 Definitions

Soil health

Soil health is often used as a **synonym for soil quality** to help translate science into practice. Doran et al (2002) refers to soil health as a concept is embedded in environmental sustainability, balancing economic viability with social responsiveness and environment stability. In other words, “soil health” emphasizes the importance of soils to fulfil various societal needs including, and beyond, food production (Bünemann et al 2018).

The concept of soil health is also the foundation of the interim report of the Mission Board on Soil Health and Food. Soil health, or quality, can be broadly defined as “*the capacity of a living soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and promote plant and animal health*. (…) Criteria for indicators of soil quality and health relate mainly to their utility in defining ecosystem processes and in integrating physical, chemical, and biological properties; their sensitivity to management and climatic variations; and their accessibility and utility to agricultural specialists, producers, conservationists, and policy makers.” (cited from Doran et al., 2002; see also Arshad and Martin, 2002). The definition emphasizes the multifunctionality of soils as well as its contribution to ecosystem services (“soil-based ecosystem services”). Bünemann et al. (2018) has recently updated the definition of soil quality as “*the capacity of a soil to function within ecosystems* *and land-use boundaries to sustain biological productivity, maintain environmental quality, and promote plant and animal health*".

Soil threats

Soil threats represent the main structural element for soil protection according to the EU Soil Thematic Strategy 2006. It has also been adopted in the Status of the World's Soil Resources Report (FAO, ITPS, 2015). Soil threats indicate processes and damages towards soils and its functional properties. These damages then reduce the soil's capacity to provide ecosystem services. Spatial data about soil threats indicate focal areas for sensitive management and soil restauration (Huber et al. 2008).

Soil degradation

Soil degradation can be defined as a decline in soil quality, resulting in the reduced functioning of the soil[5]. 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 (FAO, ITPS, 2015). As such, is soil degradation a subset of land degradation, which is itself a subset of environmental degradation (Johnson et al. 1997).

Any assessment of soil degradation needs to address the functions of soil, or more specifically: functions are assessed targeting the good status (health) of **endpoints**, i.e. food quality, human health, water and air quality, and soil biodiversity. The effect of soil degradation on endpoints, thus the reduced performance or loss of soil functions, is observed by specific soil quality indicators, and is characterized by specific **thresholds**.

Thresholds

Thresholds in this context are relevant limits in the environmental media of consideration (water, air, soil, animals, or food quality). For example, water quality standards (*the threshold*) as set by the Nitrate Directive for surface water when used for drinking water, and for groundwater (*the endpoints*), can be used as relevant criteria to evaluate the soil filter function to protect surface and ground water quality. At the same time, various soil properties (as well as land use, climate etc) are relevant parameters of the system that determine the intensity of soil processes related to functions of the soil. Thresholds are needed to inform to which extend soil functions are at risk (or degraded).

1.2 From soil quality to ecosystem services

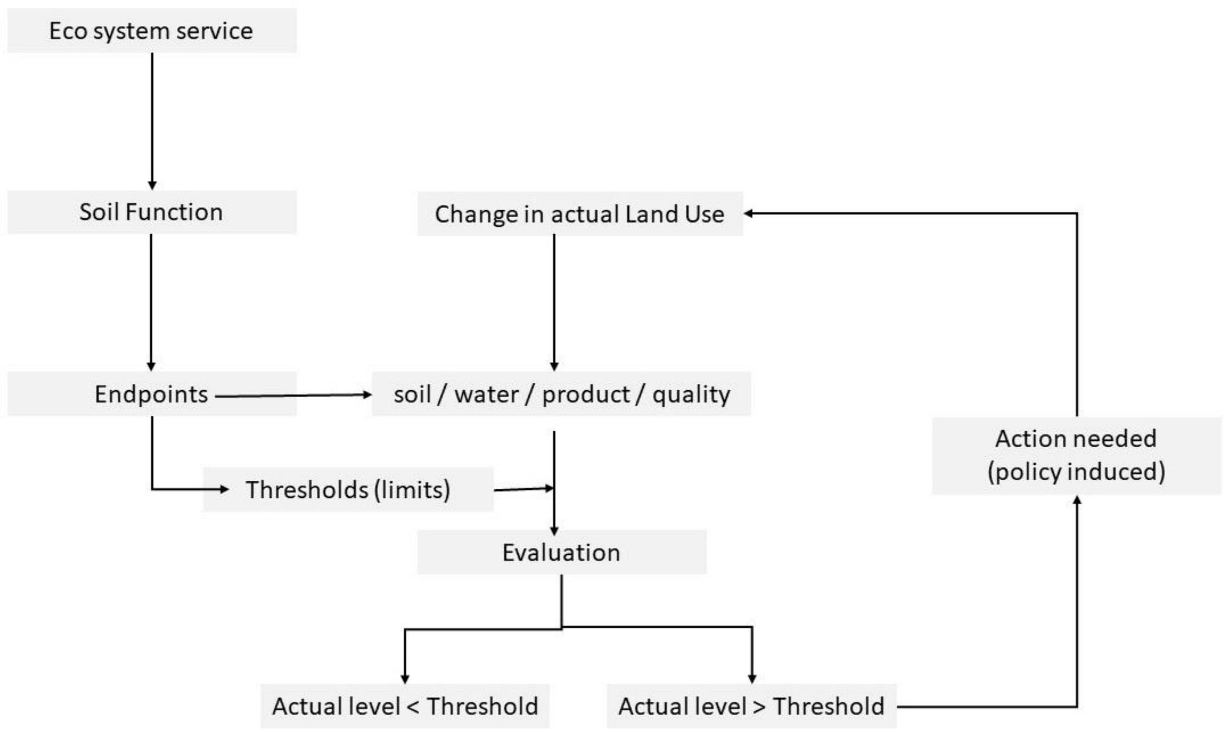
Ecosystem services can be summarized as the goods and benefits people and societies receive from ecosystems. Soil as the belowground compartment of all terrestrial ecosystems plays a key role in the capacity of ecosystems to provide their supporting, regulating, provisioning and cultural services. Paul et al. (2020) considers 29 of 83 ecosystem service classes in the Common International Classification of Ecosystem Services (CICES 5.1, Haines-Young and Potschin 2017) to be related to soil and 40 classes to be affected by agricultural soil management. The potential supply of soil-related ecosystem services is reduced on degraded / eroded soils. To be able to relate soil quality to ecosystem services (or soil functions that make up for a specific service), it is therefore imperative to be able to connect a specific service to a specific soil quality standard or limit in relevant protection targets, such as human health or drinking water quality.

Figure 1-1 presents a conceptual framework for soil degradation assessments. In order to apply this framework for nutrient losses and contamination but also for physical forms of degradation like erosion

5 Soil functions describe the soil's capacity to support ecosystem services essential for human well-being (FAO Soils Portal: http://www.fao.org/soils-portal/soil-degradation-restoration/en)

and compaction, it is important to understand the relationship between soil dynamics (processes which respond to a pressure, indicated by soil properties which can be monitored) and the critical limits for so-called 'endpoints'. This usually requires models which describe the behaviour of a soil under stress, and which help to define thresholds. The key principle underlying this approach is that critical limits (thresholds) in what is called endpoints (e.g. water quality, human health, ecosystem functioning) are converted to equivalent thresholds (or screening values) in soil. If actual levels in soil exceed such threshold levels, further action is required. This can include measures to reduce inputs to soil, clean-up measures or measures to control the impact of the pressure.

Figure 1-1: Conceptual framework for soil degradation assessment



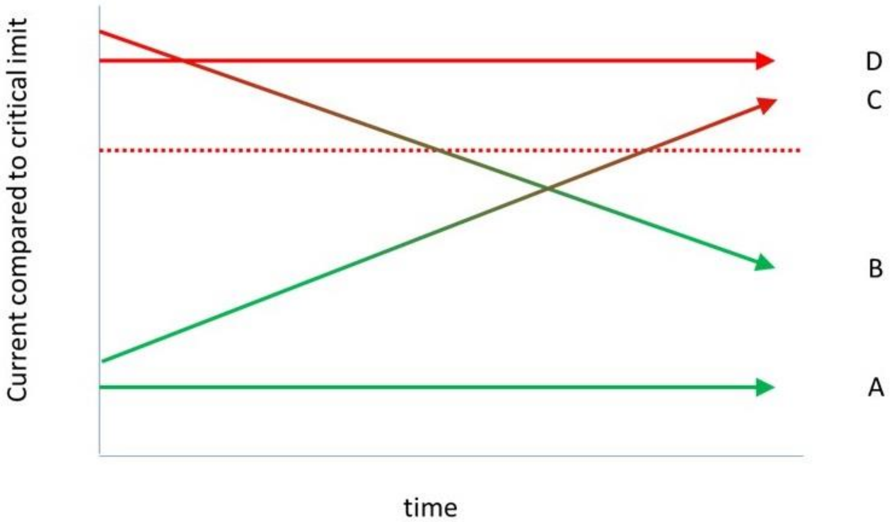
The level of soil degradation can be quantified locally or regionally as the degree to which the *current soil condition* suffers from an exceedance of relevant thresholds in view of specific functions. This approach is also a key element of **Risk-Based Land Management** (**RBLM**; Vegter et al., 2003), in particular regarding contaminated land; it is not necessarily in line with other definitions of soil degradation, where any (undesirable) change in soil properties is seen as degradation. An example of this is the commonly observed accumulation of heavy metals in soil. In most arable cropping systems where animal manure is applied, copper and zinc tend to accumulate in soil. Accumulation may be wanted in case of adding nutrients, like phosphorus (P) in a situation of P limitation, but it is considered unwanted in P saturated soils or when adding toxic pollutants (e.g. Cd). From a risk point of view, accumulation *can* be equivalent to degradation if this leads to exceedance of critical limits in relevant endpoints.

This also adds the second relevant aspect of the risk analysis according to the RBLM principle, which is the *dynamic aspect*. Current conditions (soil properties or concentrations of unwanted substances) in soil and water, can be such that thresholds for relevant endpoints are not yet exceeded. However, depending on e.g. land use[6] (and changes of inputs and/or atmospheric deposition), conditions can change so that thresholds can be exceeded at a certain point in time (Figure 1-2).

6 Land use is dynamic; it includes – among others - inputs to soil, mechanical changes caused by trafficking or different soil preparation (e.g. tillage) and seeding systems

Hence, soil degradation and the impact on soil functions basically can be characterised by four possible scenarios (A to D, as illustrated below). The dashed line represents the relevant threshold in view of the function for a specific soil. Scenario A reflects the best outcome, there is no risk at present and current conditions and land use are such that the risk limit is not exceeded at any point in time or relevant time frame considered. Opposite to this is scenario D where current and future conditions are such that the system is at risk. This would call for either measures to reduce the impact or change land use such that less stringent risk limits can be used. Scenario B (current risk but no risk in the future) and C (currently no risk but thresholds will be exceeded at some point in time) are intermediate outcomes which call for different kind of actions (or acceptance).

Figure 1-2: Dynamic assessment of soil degradation (4 scenarios)



	current	future
Scenario D	risk	risk
Scenario C	no risk	risk
....	threshold	
Scenario B	risk	no risk
Scenario A	no risk	no risk

As far as types of soil degradation are concerned, 4 main types are distinguished (Lal, 2015):

- **Soil physical degradation** is a reduction in structural attributes including pore geometry and continuity, thus aggravating a soil's susceptibility to crusting, compaction, reduced water infiltration, increased surface runoff, wind and water erosion, greater soil temperature fluctuations and an increased propensity for desertification.
- **Soil chemical degradation** can be characterized by changes in soil processes including nutrient depletion, acidification, salinization, and contamination, which in turn leads to a reduced cation exchange capacity, increased aluminium or manganese toxicities, calcium or magnesium deficiencies, leaching of NO_3-N or other essential plant nutrients. For nutrients and contaminants, annual inputs such as those from agricultural management (inputs of N, P, K but also copper, zinc, cadmium and antibiotics via animal manure or mineral fertilisers) or from additional sources including inputs via air or sedimentation are also considered as chemical degradation.
- **Soil biological degradation** refers to reduced soil biological activity, which can be accompanied by loss in soil biodiversity. This leads to lower levels of mineralization and respiration, and an accumulation of incompletely decomposed dead organic matter (necromass). Nutrient availability is reduced, and, in forests, organic matter accumulates in the forest topsoil.
- **Soil ecological degradation**: Even though a clear characterization of the soil ecological condition, and what is to be considered a reference, is largely lacking, ecological degradation reflects a combination of the other three types of degradation. This leads to a disruption in ecosystem functions such as elemental cycling, water infiltration and purification, perturbations of the hydrological cycle and a decline in net biome productivity.

Each of these forms of degradation can be linked to soil threats and the impact thereof as shown in Table 1-1; it combines soil degradation types with soil threats and soil services.

Table 1-1: Soil degradation types, corresponding soil threats and affected soil services

Degradation type	Impact of threats (1)	Affected soil services (2)
Soil physical degradation	Subsoil compaction Soil erosion Landslides	Growth of crops Wood & fibre production Water storage Substance filtering Storage of geological material Carbon storage Habitat for plants, insects, microbes, etc. Support for buildings or transport network
Soil chemical degradation	Accumulation of contaminants and nutrients in soil Salinisation Acidification	Growth of crops Wood & fibre production Water storage Substance filtering Carbon storage Habitat for plants, insects, microbes, etc.
Soil biological degradation	Accumulation of contaminants and nutrients in soil Reduced humus formation and reduced metabolization of contaminants SOM/SOC decline	Habitat for plants, insects, microbes, etc. Water storage Substance filtering Carbon storage
Soil ecological degradation	Combination of above	Combination of above

Note: (1) The listed soil threats are a combination of those mentioned in the Soil Thematic Strategy and the RECARE project according to Stolte et al. (2018)
(2) According to Adhikari and Hartemink (2016)

1.3 Assessing soil degradation by soil quality indicators

The fact that different soil functions have different endpoints implies that soil degradation and the assessment thereof cannot be performed based on one or few soil parameters, valid for all circumstances. Each type of connection between a specific endpoint, be it a critical limit in water or a critical erosion rate, requires a specific approach. Indicators, or soil properties such as pH, organic matter or texture, can be used in risk-based models (e.g. fate of substances) in order to connect the endpoint to the current status of the system.

At present, many risk assessment models are still being developed, while some models are already in use, e.g. to assess the soil status regarding pollution (e.g the Dutch risk assessment model SansCrit[7] and the Risk Assessment Toolbox (www.rivm.nl); the CLEA model[8] used in the UK). This means that it is not possible to assess soil degradation at large with one single indicator. At present, soil degradation assessment according to the current state of research, can only be carried out for specific soil services. For these, it is imperative to consider, in addition to general soil properties (or indicators) used in these models, specific regional conditions like for example climate, crop type etc.

7 SansCrit: https://www.risicotoolboxbodem.nl/sanscrit/
8 CLEA model: https://www.gov.uk/government/publications/contaminated-land-exposure-assessment-clea-tool

In this report, 8 soil threats and 12 soil quality indicators were selected (see Table 10-2), in view of their appropriateness to assess soil degradation related to various important soil functions or ecosystem services. These are described and discussed in chapters 2 to 8. For most cases, the selected indicators are well established, data availability is, at the European level, at least acceptable and they are appropriate to describe key soil degradation types and impairment of key soil services. As stated, several indicators, like for example soil organic carbon, have multiple functions and are used to assess several forms of soil degradation related to different soil services. Table 1-2 illustrates where the proposed indicators are needed to assess the degree of soil degradation, as related to different soil services.

Table 1-2: Soil threats and their linkage to soil services and key societal needs

	Societal needs				
	Biomass	Water	Climate	Biodiversity	Infrastructure
Soil services	Wood & fibre production	Filtering of contaminants	Carbon storage	Habitat for plants, insects, microbes, funghi	Platform for infrastructure
	Growth of crops	Water storage			Storage of geological material
Soil Quality Indicators					
Soil Organic Carbon	+	+	+	+	(2)
Soil nutrient status	+	indiff.	indiff.	+	indiff.
Soil acidification	-	-	(1)	-	indiff.
Soil heavy metal contamination	-	-	indiff.	-	indiff.
Soil biodiversity	+	+	+	+	indiff.
Soil erosion	-	-	-	-	indiff.
Soil compaction	-	-	-	-	indiff.
Soil sealing	-	-	-	-	+

(1) Soil acidification / carbon storage: fulvic acid (from acidified forest floors) enhances bleaching and nutrient loss, as well as loss of dissolved organic carbon; acidic soils favor reduced decomposition
(2) Soil organic carbon / infrastructure: organic soils are instable as platform for infrastructre

LEGEND	
+	positive impact on soil service
-	negative impact on soil service
indiff.	neutral or unknown impact

1.4 Existing indicator systems including soil quality

1.4.1 Global and European soil indicator systems

Table 1-3 provides an overview of commonly discussed European and global soil indicators. The sorting element for these indicators is the Soil Thematic Strategy[9] of the European Commission. It shall be mentioned that Eurostat, FAOSTAT and OECD also maintain indicator systems, which contain soil-related indicators as an element of agri-environmental indicator sets. Soil indicators are also included in EEA's indicator system, which is populated by the members of the Environmental Information and Observation Network (EIONET), and which is – among others - used for the regular Status and Outlook of the Environment reporting. EEA's system also includes indicators under different EU legislation, for which EEA acts the knowledge centre and data hub (reporting in the context of soil for LULUCF and NEC– see Table 1-3).

9 Directive (COM (2006) 232 establishing a framework for the protection of soil and amending Directive 2004/35/EC

Table 1-3: Overview of broadly discussed soil indicators

Degradation types/soil threats	GLASOD1)	EU2)	ENVASSO indicators, modified3)	Indicators mentioned in Status of the World Soil Resources Report4)	SEEA5) and FDES6) 7)
Water erosion	X	X	- Soil loss [t/ha] - Observed erosion features [type/amount per area]	Soil loss	Area affected by soil erosion**
Wind erosion	X				
Overblowing	X		- deposited soil [to/ha]		
Loss of organic matter	X	X	- Topsoil organic matter (SOM) or carbon (SOC) content - SOC stock [t/ha] - Peat stock [t/ha]	C pool: Organic C stocks	Soil carbon*
Salinization	X	X	- Salinity state: total salt content [% EC] - Exchangeable sodium [pH unit ESP %]	Spatial distribution of salt-affected soils	Area affected by salinization**
Acidification	X	(X)	- Top soil pH - % exchangeable acid cations (Mn, Al, Fe)	pH acid neutralization capacity	Area affected by acidification**
Loss of nutrients	X	(X)	- % exchangeable basic cations - % trace elements (includes micronutrients)	**Soil fertility:** % nutrients, pH **Nutrient balances:** applied and excess N, P (fertilizer consumption, nutrient loss from soil) **SOM concentration**	Nutrient concentrations: N, P, Ca, Mg, K, Zn, other**
Pollution	X	X	- Heavy metal content [mg/ha] - Critical load exceedance (S, N)	Contaminated land area	
			- Progress in the management of contaminated sites		Area/number of potentially contaminated and remediated sites**
Compaction and physical degradation	X	X	- Soil density - Air capacity - Vulnerability to soil compaction		Area affected by compaction**
Waterlogging	X		-		Area affected by waterlogging**
Subsidence of organic soils	X		-		
Loss of soil biodiversity		X	- Macrofauna (Earthworms) - Mesofauna (Collembola) - Microbial respiration		
Landslides		X	- Occurrence of landslides		
Soil Sealing		X	- Sealed area		
Desertification		X	- Vulnerability to desertification - Wildfires - SOC on desertified land		Area affected by desertification**
Water cycle			-	Soil moisture	

Source: FAO 2019, draft SoilSTAT

Note: [(1)] GLASOD: GLobal Assessment of human induced SOil Degradation: 12 types of human-induced soil degradation recognized
[(2)] Soil Thematic Strategy of the European Commission (Directive (COM (2006) 232)
[(3)] ENVASSO: **ENV**ironmental **AS**sessment of **SO**il for monitoring. 6th EU Framework Research Programme. Volume 1 identifies 290 potential indicators related to 188 key issues for 9 soil threats (Huber et al. 2008)
[(4)] ITPS 2015
[(5)] SEEA: System of Environmental-Economic Accounting: internationally agreed standard to produce comparable statistics and accounts (stocks and changes in stocks of environmental assets); it follows the accounting structure as the System of National Accounts (SNA). The SEEA is a guide to integrating economic, environmental and social data into a single, coherent framework for holistic decision-making.
[(6)] FDES: Framework for the Development of Environment Statistics (FDES 2013); it uses SEEA definitions and classifications
[(7)] by soil type, by nutrient, national, subnational (in the case of pollution: by location, subnational, by type of pollutant, by source

Under the 7th EU Environment Action Programme, the initiative “Mapping and assessment of ecosystems and their services” (MAES) has been launched. There, a set of soil-related indicators has been proposed (EC et al. 2017) (Table 1-4).

Table 1-4: Proposal for MAES soil indicators (modified)

<table>
<tr><th rowspan="2">ECO-SYSTEM TYPE</th><th colspan="7">SOIL ECOSYSTEM</th></tr>
<tr><th>Urban</th><th>Cropland</th><th>Grassland</th><th>Woodland and forest</th><th>Heathland and shrub</th><th>Wetland</th><th>Sparsely veget. land</th></tr>
<tr><td rowspan="8">Soil pressures</td><td rowspan="3"></td><td colspan="6">Intensive management (e.g. tillage)</td></tr>
<tr><td colspan="2">Loss of organic matter (% per year)
Gross nutrient balance</td><td colspan="2">Landslides (number)
Acidification (kmol H^+ ha^{-1})</td><td>Climate change</td><td></td></tr>
<tr><td colspan="2">Salinity</td><td colspan="3">Salinity</td><td></td></tr>
<tr><td colspan="3">Compaction (kg/m^3)</td><td colspan="3" rowspan="2"></td><td rowspan="2"></td></tr>
<tr><td colspan="3">Imperviousness (%)</td></tr>
<tr><td></td><td colspan="6">Soil erosion (kg/ha/year)</td></tr>
<tr><td colspan="7">Soil sealing (% area)
Soil contamination (from point or diffuse sources, nutrient deposition)</td></tr>
<tr><td colspan="7">Land use change (i.e. land use intensification)</td></tr>
<tr><td rowspan="5">Soil state</td><td rowspan="5">Vegetation coverage</td><td colspan="4" rowspan="2"></td><td>Soil moisture</td><td>Available water capacity</td></tr>
<tr><td>Bulk density</td><td>Soil nutrient availability</td></tr>
<tr><td colspan="2">Soil erosion susceptibility
Soil productivity</td><td colspan="2"></td><td colspan="2" rowspan="2"></td></tr>
<tr><td colspan="4">Available water capacity
Soil nutrient availability</td></tr>
<tr><td colspan="6">Soil carbon stock (%)</td></tr>
<tr><td rowspan="3">Soil bio-diversity</td><td rowspan="2">Earthworms div./abun.</td><td colspan="6">Microbial biodiversity (fungi and bacteria)</td></tr>
<tr><td colspan="6">Soil pH and carbon</td></tr>
<tr><td colspan="7">Soil biodiversity potential (this index measures possible not real biodiversity)</td></tr>
</table>

A subset of these indicators was considered essential for covering the role of soils for the condition and functioning of most ecosystem types, namely

- Soil erosion (kg/ha/year)
- Soil sealing (% area)
- Soil contamination or pollution (from point or diffuse sources)
- Available water capacity
- Soil nutrient availability
- Soil carbon stock (%)
- Soil biodiversity potential

The 2020 MAES assessment then presents an updated collection of soil indicators by the Joint Research Centre (JRC) (Maes et al., 2020).

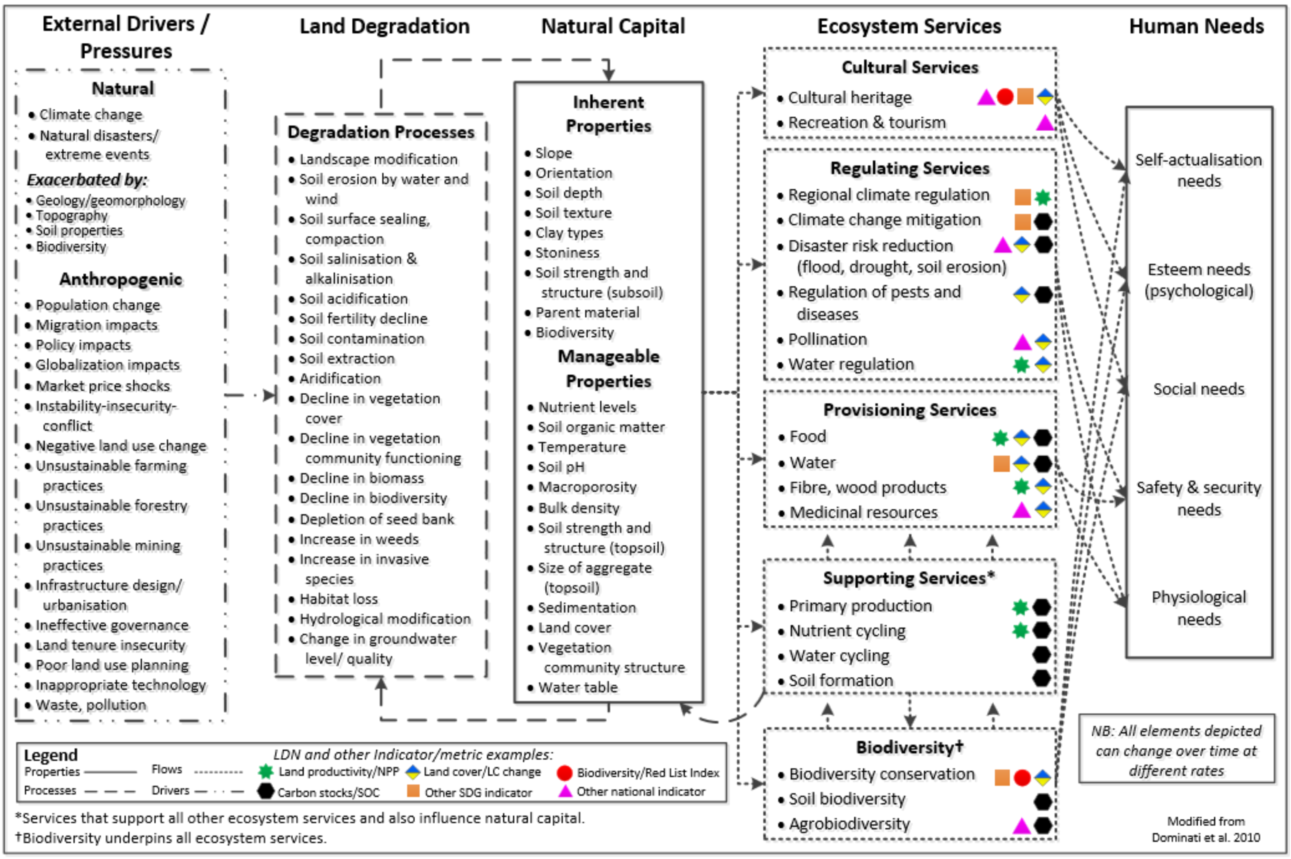
- based on modelling: erosion by water, agricultural area under severe erosion, soil erosion rates per land cover, wind erosion rate
- based on Eurostat indicators: gross nutrient balance
- based on LUCAS Soil measurements: topsoil nitrogen and phosphorus concentrations, trends of soil organic carbon (SOC) stocks in croplands
- based on EU research: area extend of organic soils, susceptibility to compaction
- based on EEA indicators: soil sealing, contaminated sites

Additional results from the literature – without specific reference to indicators - were presented on diffuse pollution, salinization and desertification.

1.4.2 UNCCD land degradation and SDG 15.3.1

The context-specific nature of land degradation requires the combination of several indicators to fully describe the condition of land. Figure 1-3 presents an overview of processes leading to degradation, and how the current subindicators for SDG 15.3.1 (land productivity, cover change, carbon) relate to ecosystem services as affected from land degradation. Countries are encouraged to use additional indicators. Important land degradation processes are in fact the soil threats as mentioned above. Indicators in this context represent "key processes which underpin land-based natural capital" (Orr et al., 2017).

Figure 1-3: Operational definition of land degradation and linkage with sub-indicators



Source: United Nations Statistics (2018)

1.5 Soil indicators for EU policy targets

Despite the fact that the proposal for a EU Soil Protection Directive was withdrawn in 2014, various aspects of soil protection have been incorporated in sectoral policies or other non-soil-related policies. Most relevant developments regarding restoration and protection of soil functions are described below in chronological order, and their targets are summarised in Table 1-6.

- The 2000 **Water Framework Directive**[10] aims at preventing and reducing pollution from agricultural and industrial sources to water bodies by prescribing specific measures. The Directive indirectly regulates diffuse soil contamination because soil pollution is in numerous cases responsible for surface or groundwater pollution. The Directive requires that Member States, "*produce River basin Management Plans*" and establish "*programmes of measure and implement 'basic' measures*". This includes the identification of point sources as well diffuse sources of pollution, a quantitative estimation of their impact and measures to reduce their impact. The Directive is backed up by a clear implementation schedule, including monitoring and evaluation.

- Three EU policy documents of non-binding nature with a clear focus on soil protection were released between 2006 and 2013. Firstly, the **Soil Thematic Strategy**[11] which for the first time identifies and presents soil threats for Europe, secondly the **Road Map for a Resource Efficient Europe**[12] prescribing non-binding targets for land take, soil erosion and local soil contamination, and thirdly the **7th Environment Action Programme (EAP) to 2020**[13] with objectives towards land take reduction, management of local soil contamination, prevention of soil erosion and increase of soil organic matter. All three strategies demand further action: targets, incentives for implementation, and monitoring. While the EU is now moving towards the 2030 agenda, the achievements of these strategies are summarized in the European Environment Agencies "State and Outlook of the Environment" report (EEA 2019). Regarding soil, action is not on track regarding the above-mentioned targets. However, while some soil threats are not covered by any target (e.g. compaction), there is a lack of evidence for many soil threats. In the context of the proposal for the **8th Environment Action Programme** (2020/0300 (COD)), the EU Commission is consulting on a **monitoring framework with headline indicators**[14], among them soil organic carbon, and a placeholder for healthy soils, and soil sealing.

- The **Common Agricultural Policy Regulation (CAP)**[15] introduced standards for Good Agricultural and Environmental Condition of land (GAEC) which are linked to agricultural subsidies. Also important are the Rural Development measures as set out in Regulation (EU) No 1305/2013. Three GAEC standards refer directly to soil, namely GAEC 4 "Minimum soil cover", GAEC 5 "Minimum land management reflecting site specific conditions to limit erosion", and GAEC 6 "Maintain soil organic matter through appropriate practices". GAEC 4 requires that a cover of growing plants or other organic residues should remain on the soils to reduce erosion by water and wind. Member States (MS) set quantitative targets and report through annual implementation reports. Both the implementation of GAECS as well as rural development measures in support of soil quality have been poor across the EU (EC 2020). The 2013 CAP regulation will be replaced by a new regulation for the CAP period 2021-2027.

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10 (2000/60/EC) Directive establishing a framework for Community action in the field of water policy
11 COM(2006)/231, Thematic Strategy for Soil Protection
12 COM(2011) 571, Roadmap to a Resource Efficient Europe
13 Decision No 1386/2013/EU of the European Parliament and of the Council, EU Environment Action Programme to 2020 'Living well, within the limits of our planet'
14 Consultation on a Monitoring Framework for the 8th EAP; https://ec.europa.eu/environment/system/files/2021-07/Explanatory%20Note%208EAP%20Indicators.pdf (Sept. 2021)
15 Regulation (EU) No 1306/2013 on the financing, management and monitoring of the common agricultural policy. The regulation covers the period 2014-2020.

- The 2016 **National Emissions Ceilings Directive (NEC)**[16] contributes to avoiding diffuse contamination, particularly from acidifying pollutants, from industry as it sets limits to air emissions for defined substances. For each Member State and each pollutant group "*annual ceilings*" or maximum amounts are defined, and their exceedance is assessed in the National Emission Ceilings reporting status (published by EEA[17]). The five main air pollutants NO_x, NMVOCs, SO_2, NH_3 and PM2.5 as well as carbon monoxide (CO) are monitored and reported every four years for selected monitoring sites. Since 2018, Member States must follow Air Pollution Control Programmes by establishing monitoring sites to assess the impacts of air pollutants to sensitive receiving environments (freshwater, non-forest natural and semi-natural habitats, and forest ecosystem types). The Directive defines a list of soil indicators for this assessment[18].

- With the adoption of the 2018 **LULUCF regulation**[19] greenhouse gas emissions and carbon dioxide removals from the LULUCF sector (land use, land use change and forestry) have become part of the 2030 Climate and Energy targets: "to ensure the contribution of the LULUCF sector to the achievement of the Union's emission reduction target of at least 40 % and to the long-term goal of the Paris Agreement in the period 2021 to 2030". The Regulation sets a binding commitment for each Member State to ensure that accounted emissions from land use are entirely compensated by an equivalent accounted removal of CO_2 from the atmosphere ("no debit" rule). Although Member States already partly undertook this commitment individually under the Kyoto Protocol until 2020, the Regulation establishes this commitment in EU law for the period 2021-2030. Moreover, the scope is extended from only forests today to all land uses (including wetlands by 2026). The "no net debit" obligation will be assessed for the periods 2021-2025 and 2026-2030. The LULUCF regulation hence encourages land management practices that increase soil organic carbon stocks, as for example restoration of forests and wetlands, and avoiding conversion of grassland to cropland.
  The EU has meanwhile updated the 2030 GHG emissions net reduction target of 55% below 1990 levels. This target has been set in the **2030 Climate Target Plan**[20], and inserted into **European Climate Law[21]**, and is needed for the process towards a climate neutral Europe by 2050. This includes the recognition of the need to enhance the EU's carbon sink through a more ambitious LULUCF regulation – therefore: the provisions under the Green Deal also include the reforming of the 2018 LULUCF Regulation[22].

  To achieve climate-neutrality in 2050, the capacity of land to capture CO_2 will have to increase; this includes soils. Two mechanisms are foreseen:
  - Carbon farming (COWI et al. 2021)
  - Carbon removals certification mechanism (CRCM)[23]: study by UBA, Ecologic, Rambøll and Carbon Counts

16 Directive (EU) 2016/2284 on the reduction of national emissions of certain atmospheric pollutants

17 Progress made by the European Union (EU) and its Member States is published annually by the European Environment Agency; i.e. NEC Directive reporting status 2019 https://www.eea.europa.eu/themes/air/air-pollution-sources-1/national-emission-ceilings/nec-directive-reporting-status-2019

18 For terrestrial ecosystems the assessment of soil acidity, soil nutrients loss, nitrogen status and balance as well as biodiversity loss is required based on the following indicators: soil acidity (every 10 years); soil nitrate leaching (annual); carbon-nitrogen ratio (C/N) (every 10 years)

19 Article 1, Regulation (EU) no 2018/841 on the inclusion of greenhouse gas emissions and removals from land use, land use change and forestry in the 2030 climate and energy framework

20 COM(2020) 562 final: Communication "Stepping up Europe's 2030 climate ambition. Investing in a climate-neutral future for the benefit of our people", Sept 2020.

21 COM(2020) 80 final: proposal European Climate Law; amending Regulation (EU) 2018/1999 (on the on the Governance of the Energy Union and Climate Action)

22 COM(2021) 554 final: Proposal for a regulation amending Regulations (EU) 2018/841 as regards the scope, simplifying the compliance rules, setting out the targets of the Member States for 2030 and committing to the collective achievement of climate neutrality by 2035 in the LULUCF sector, and (EU) 2018/1999 as regards improvement in monitoring, reporting, tracking of progress and review.

23 Study in preparation: https://ec.europa.eu/clima/tenders/2020/305336_de

The uptake of carbon removals and increased circularity of carbon is incentivized by the Circular Economy Action Plan, while Farm to Fork Strategy may enable payments to farmers and foresters for the carbon sequestration they provide. **These policy demands will require improvements of SOC monitoring with regard to reliability (uncertainties) as well as spatial and temporal resolution.**

Table 1-5: Objectives, targets and recommended indicators of the EU's Soil Health and Food Mission Board

Objective	Target	Presence of soil pollutants, excess nutrients and salts.	Soil organic carbon	Soil structure & bulk density & absence soil sealing /erosion	Soil biodiversity.	Soil nutrients and pH	Vegetation cover	Landscape heterogeneity	Area of forest and other wooded lands
1. Reduce land degradation, including desertification and salinization.	1.1: 50% of degraded land is restored moving beyond land degradation neutrality.	●	●	●	●	●	●	●	●
2. Conserve (e.g. in forests, permanent pastures, wetlands) and increase soil organic carbon stocks.	2.1: current carbon concentration losses on cultivated land (0.5% per year) are reversed to an increase by 0.1-0.4% per year;		●				●		
	2.2: the area of managed peatlands losing carbon is reduced by 30-50%.		●				●		
3. No net soil sealing and increase the re-use of urban soils for urban development.	3.1: switch from 2.4% to no net soil sealing;			●			●		
	3.2: the current rate of soil re-use is increased from current 13% to 50% to help meet the EU target of no net land take by 2050.			●			●		
4. Reduce soil pollution and enhance restoration	4.1: 25% of land under organic farming	●							
	4.2: A further 5-25% additional land (i.e. over and above the 25% in full organic) with reduced risk from eutrophication, pesticides, anti-microbials and other contaminants	●							
	4.3: Doubling of the rate of restoration of polluted sites	●							
5.Prevent erosion	5.1: Stop erosion on 30-50% of land with unsustainable erosion risk			●			●	●	●
6.Improve soil structure to enhance habitat quality for soil biota and crops	6.1: Reduction by 30-50% of soil with high density subsoils			●			●	●	
7.Reduce the EU global footprint on soils	7.1: The impact of EU's food, timber and biomass imports on land degradation are reduced by 20-40%	Food, feed and fibre imports							
8.Increase soil literacy in society across Member States	8.1: soil health is firmly embedded in schools and educational curricula	●	●	●	●	●	●	●	●
	8.2: uptake of soil health training by land managers is increased	●	●	●	●	●	●	●	●
	8.3: understanding of impact of consumer choices on soil health is increased	●	●	●	●	●	●	●	●

- The 2019 **European Green Deal communication**[24] contains a roadmap for making the EU's economy sustainable, by turning climate and environmental challenges into opportunities across all policy areas. Under the umbrella of the Green Deal several policy documents are directly linked to soil protection, thus the importance of soil health is broadly addressed. The following key objectives of the Green Deal Communication include policy developments, which are highly relevant for future soil protection:
  - Preserving and restoring ecosystems and biodiversity. In May 2020 the European Commission released the new **Biodiversity Strategy to 2030**[25]. Overall objective is to reverse biodiversity loss

24 COM/2019/640 final, The European Green Deal

25 COM(2020) 380 final, EU Biodiversity Strategy for 2030

in the EU, and to increase resilience towards natural threats such as climate change impacts, forest fires, food insecurity or disease outbreaks. The strategy includes quantitative targets, three of which are directly linked to soil protection (see Table 1-6). Furthermore, the Strategy calls for an **EU Nature Restoration Plan**: Member States are requested to ensure that no deterioration in conservation trends and status of all protected habitats and species by 2030. In addition, Member States will have to ensure that at least 30% of species and habitats not currently in favourable status are in that category or show a strong positive trend". Currently, legally binding EU nature restoration targets are being prepared, which are likely addressing protected as well as unprotected habitat areas, including soil-related indicators. There is a need to developing an EU-wide methodology to map, assess and achieve good condition of ecosystems; and this must be based on indicators for all ecosystems including soils.[26]

- From 'Farm to Fork': a fair, healthy and environmentally friendly food system. Also in May 2020, the **Farm to Fork strategy (F2F)** was published, which focuses on fair and environmentally friendly food production. The strategy repeats the target for organic farmed land as mentioned in the Biodiversity Strategy and besides that, defines two targets related to soil pollution, firstly "to reduce the overall use and risk of chemical pesticides by 50% and the use of more hazardous pesticides by 50% by 2030" and secondly "to reduce nutrient losses by at least 50%, while ensuring that there is no deterioration in soil fertility and reduce the use of fertilisers by 50%". While these targets apply at EU level, Member States will be asked to define their own targets in the CAP Strategic Plans.
- A **zero pollution ambition for a toxic-free environment** has the overall objective to avoid harmful levels of pollution to air, soil and water, as it is one of the main reasons for the loss of biodiversity and ecosystem services, as well as economic losses (e.g. yield loss, health-related cost, remediation cost). It includes two actions relevant for soil protection.
    - The **Chemicals Strategy for Sustainability (CSS)**[27], published in October 2020, sets out concrete actions to make chemicals safe and sustainable and to ensure that chemicals can deliver all their benefits without harming people and the environment. SWD(2020) 249 final[28] addresses the hazard from PFAS contamination of soils; SWD(2020) 250 final[29] raises the concern about mixtures of chemicals in environmental media. Currently, a **framework of indicators** is being developed to monitor drivers and impacts of chemicals pollution and to measure the effectiveness of chemicals legislation. Likely, CSS will deliver a list of substances which soil monitoring needs to address.
    - **The Zero Pollution Action Plan for Air, Soil and Water** has been published[30]. The plan has the ambition to improve the governance framework of the Member States regarding pollution prevention. SWD(2021) 141 final[31] outlines monitoring and outlook framework for the zero pollution ambition; it foresees regular reporting on a) monitoring (relying on indicators on diffuse and local soil pollution) and b) outlook, including a Clean Soil Outlook

In addition to the above listed policies, the State and Outlook of the Environment report 2020 (SOER2020) published by the European Environment Agency (EEA 2019) mentions several other policies with indirect effects on soil:

- Nitrates Directive (Directive 91/676/EEC),
- Sustainable Use of Pesticides Directive (Directive 2009/128/EC),

26 Roadmap to develop a regulation on ecosystem restauration (https://ec.europa.eu/info/law/better-regulation/have-your-say/initiatives/12596-Protecting-biodiversity-nature-restoration-targets-under-EU-biodiversity-strategy_en

27 COM(2020) 667 final. Chemicals Strategy for Sustainability Towards a Toxic-Free Environment

28 SWD(2020) 249 final https://ec.europa.eu/environment/pdf/chemicals/2020/10/SWD_PFAS.pdf

29 SWD(2020) 250 final https://ec.europa.eu/environment/pdf/chemicals/2020/10/SWD_mixtures.pdf

30 COM(2021) 400 final: EU Action Plan: 'Towards Zero Pollution for Air, Water and Soil' https://ec.europa.eu/environment/pdf/zero-pollution-action-plan/communication_en.pdf

31 SWD(2021) 141 final https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:52021SC0141&from=EN

- Sewage Sludge Directive (Directive 86/278/EEC),
- Fertilisers Regulation (Regulation EU 2019/1009),
- Mercury Regulation (Regulation EU 2017/852), and
- Plant Protection Products (PPP) (Regulation EU 1107/2009).

Several additional initiatives shall be mentioned here:

- The EU **Mission Board for Soil Health and Food**[32] has the ambition to support the Green Deal and in particular the Farm to Fork Strategy. Most noteworthy is the overarching goal: "*By 2030, at least 75% of all soils in each EU Member State are healthy, i.e. are able to provide essential ecosystem services*". In order to achieve this goal, the Mission Board sets out eight objectives which are complemented with quantitative targets. For each quantitative target, indicators for monitoring are specified (see Table 1-5).

- The **'4 per mille' initiative**[33]. The importance of SOC sequestration in arable soils on climate change mitigation and food security has been recognized by the initiative '4 per mille - Soils for Food Security and Climate'. It was initiated by the French Ministry of Agriculture, and launched during the UNFCCC Conference of the Parties in 2015 (COP 21). Many European countries are partners of the initiative. The rationale is that an annual growth rate of 4‰ in SOC stocks in the top 40 cm of all soils over a time frame of 20 years would equal the annual anthropogenic C emissions of 8.9Gt, and therefore halt the annual atmospheric CO_2 increase from the agricultural sector. Despite comments on the overestimated CO_2 sequestration potential, in view of the need for sequestering N and P in stored SOC (e.g. Van Groenigen et al., 2017, De Vries, 2018), it is an important initiative that stimulates climate smart agriculture, focusing on SOC sequestration.

- The **UN Sustainable Development Goals (SDG)** were agreed in 2015 as part of the UN 2030 Agenda for Sustainable Development. The European Union committed to implement the SDG. Based on COM(2016) 739 final "Next steps for a sustainable European future", the European Commission has developed in 2017 a reference indicator framework[34] to monitor the SDGs in the EU, and since then, reports annually on the progress of SDG implementation in the context of EU policies. Of the 100 selected indicators (which do not cover all aspects of the global SDGs), 36 are multi-purpose (are used for more than one goal). In the following, the methodologies of the most important global SDG indicators with regard to soils are described:
    - **Indicator 2.4.1 "Proportion of agricultural area under productive and sustainable agriculture"** of **SDG 2.4**, focussing on sustainable food production systems and resilient agricultural practices. Its scope includes ecosystem maintenance and soils, among others. At global level, it is doubted whether this indicator can be monitored with remote sensing, soil and water sampling. Therefore, FAO (2018) recommends farm surveys. In Europe, farmers may be capable to assess the environmental impact of their practices. One of the 11 **sub-indicators** refers to soil health: **"Prevalence of soil degradation".** This sub-indicator follows FAO and ITPS (2015), and proposes the observation of 10 soil threats. The soil threats are then reduced to <u>**4 indicators**</u>**:**
        - soil erosion
        - reduction in soil fertility
        - salinization of irrigated land,
        - waterlogging

      All indicators can be monitored with farm surveys. For the combined agricultural farm area affected by any of the four threats the targets are as follows:

32 European Commission Directorate General for Research and Innovation (2020) Caring for soil is caring for life https://op.europa.eu/en/web/eu-law-and-publications/publication-detail/-/publication/32d5d312-b689-11ea-bb7a-01aa75ed71a1

33 4 pour 1000 initiative: https://www.4p1000.org/

34 EU SDG Indicator set: https://ec.europa.eu/eurostat/documents/276524/10369740/SDG_indicator_2020.pdf

- desirable target: less than 10% of area affected
- acceptable target: 10 - 50% of area affected
- unsustainable target: more than 50% of are affected

– **Indicator 15.2.1 "Progress towards sustainable forest management"**: currently, none of the UN sub-indicators includes soil.

– **Indicator 15.3.1: "Proportion of land that is degraded over total land area"**. The following sub-indicators are used:
  - Trends in land cover
  - Land productivity
  - Carbon stocks (above and below ground: currently only SOC stocks).

Positive, stable, or negative trends are monitored. It is noteworthy to mention that the One-out-all-out (1OAO) principle[35] is applied, meaning that an area is considered as degraded if only one indicator shows a negative trend.

All EU Member States and the EU Commission committed themselves to "achieve a land degradation neutral world by 2030". The indicator on "Land Degradation Neutrality" includes three sub-indicators, being land carbon stocks (above- and belowground), land productivity, and land cover change. EUROSTAT reports on land degradation in response to SDG 15.3 by using two soil-related indicators, being "Soil sealing index" and "Estimated soil erosion by water".

Table 1-6: Soil-related policy objectives and targets at EU- and global level (binding as well as incentive-based non-binding policies and measures)

Policy Document	Relevant policy objectives or targets
Water Framework Directive (WFD) (2000/60/EC)	- Member States to produce River Basin Management Plans, requiring the identification of point sources and their impacts. - Member States to establish programme of measures and implement 'basic' measures, including among others adapted agricultural production schemes to reduce nitrogen input on agricultural soils and as a consequence connected water bodies.
Road Map for a Resource Efficient Europe (COM(2011) 571)	- Soil erosion is reduced by 2050. - Increase of soil organic matter between 2011 and 2050. - By 2020 remedial work on contaminated sites well underway. - Achieve no net land take by 2050.
National Emissions Ceilings Directive (EU) 2016/2284[36]	- Air pollution and its impacts on ecosystems and biodiversity are further reduced with the long-term aim of not exceeding critical loads and levels (based on 7th EAP). - To reduce the ecosystem area subjected to eutrophication by 35% by 2030, compared with 2005 (Clean Air Programme for Europe[37]). - To achieve national emission reduction targets for anthropogenic emissions. - Member States to assess the impacts of air pollutants to sensitive receiving environments (natural and semi-natural habitats and forest ecosystem).

35 One Out, All Out (1OAO) principle considers changes in the sub-indicators: (i) positive or improving, (ii) negative or declining, or (iii) stable or unchanging. A location is considered degraded if at least one of the three land-based indicators shows a negative change (Cowie et al., 2018).

36 Ecosystem monitoring under Article 9 and Annex V of Directive 2016/2284 (NECD)
Note: The extent of ecosystem impacts of air pollution in the EU is based on the exceedance of critical loads and levels for sulphur, nitrogen, and ozone. The definition of thresholds is largely based on the WG on Effects under the Gothenburg Protocol to the Convention on Long-Range Transboundary Air Pollution (CLRTAP) (including – among others - International Cooperative Programmes (ICPs) on Forests, Vegetation, and Integrated Monitoring.

37 COM(2013)918 final

Policy Document	Relevant policy objectives or targets
LULUCF Regulation (EU) 2018/841	- To ensure the contribution of the LULUCF sector to the achievement of the Union's emission reduction target of at least 40 % and to the long-term goal of the Paris Agreement in the period 2021 to 2030. - Member States have binding commitments to compensate CO_2 emissions from the land use sector; land management practices which increase soil organic carbon stocks are accountable compensation measures.
Common Agricultural Programme (2021-2027) Regulation (EU) No 1306/2013	- CAP post-2020: continues to promote practices beneficial for the climate and the environment; introduces eco-schemes for additional measures. Impact indicators indicate the increase in soil carbon, reduction in soil erosion and nutrient (N) loss; a result indicator covers practices targeted to improve soils. - GAEC as conditional standards remain valid in the future CAP: GAEC in support of soil protection and quality: GAEC 6: Tillage management; GAEC 7 No bare soil; GAEC 8: Crop rotation.
8th Environment Action Programme 2030 (COM(2020) 652 final)	- Protecting, preserving and restoring biodiversity and enhancing natural capital, notably air, water, soil, and forest, freshwater, wetland and marine ecosystems. - Umbrella programme[38] against biodiversity loss and ecosystem services degradation, climate change and its impacts, and unsustainable use of resources, pollution, and associated risks to human health.
Biodiversity Strategy to 2030 COM(2020) 380 final	- Legally protect a minimum of 30% of the EU's land area. - At least 25% of the EU's agricultural land must be organically farmed by 2030. - At least 10% of agricultural area is under high-diversity landscape features. - The risk and use of chemical pesticides is reduced by 50% and the use of more hazardous pesticides is reduced by 50%.
Farm to Fork strategy COM(2020) 381 final	- To reduce the overall use and risk of chemical pesticides by 50% and the use of more hazardous pesticides by 50% by 2030. - To reduce the use of fertilisers by 2030 by at least 20%. - At least 25% of the EU's agricultural land must be organically farmed by 2030.
Zero Pollution Action Plan for Air, Soil and Water (in preparation)	- A zero-pollution ambition for a toxic free-environment, including for air, water and soil. - To better monitor, report, prevent and remedy pollution from air, water, soil, and consumer products to levels that are no longer harmful to human health and the environment. - To propose new legislation covering significant pollution sources, which are not yet addressed by other policies, strategies, and protocols. - To facilitate remediation of soil pollution via i) a monitoring framework on the state of pollution and ii) an outlook report including a specific assessment of the evolution of human health and environmental impacts.
Revised Soil Thematic Strategy (in preparation 2021)	The revised Soil Thematic Strategy is expected to be published as an EC communication by the end of Q2 2021; it is likely to refer to or contain: - Targets and objectives of the Horizon Europe-Mission Board 'Soil health and Food': ensure that 75% of soils are healthy by 2030. - Restoration targets of the Biodiversity Strategy 2030 (see above).

Considering the objectives and targets listed in Table 1-7, it becomes evident that monitoring of soil quality indicators and the respective evaluation schemes are needed. Assessments of the European soil condition

38 Including: (1) COM (2018)773 "A Clean Planet for All", followed by the "Long-term low greenhouse gas emission development strategy (2020), (2) Circular Economy Action Plan for a clean and competitive Europe, and (3) new strategies under the Green Deal (Biodiversity Strategy 2030, Farm2Fork Strategy, Zero Pollution Action Plan)

(EEA 2020, JRC 2012, EC 2020) have been lacking a systematic and complete indicator set, and information on trend and reliable statistics are still limited.

1.6 Conclusions

Soil degradation finds its expression as a reduction or elimination of soil functions, thus the loss of the ability of soils to support ecosystem services (FAO and ITPS, 2015). The loss of soil functions can affect the health and survival of organisms living in and from the soil, including humans, which feed from the soil, and which are in daily contact with soil. The degree to which functions are reduced, depends on the degree to which critical limits are exceeded, as was shown in Figure 1-3. The loss of functions may be visible as a reduction of plant production (e.g. yield reductions), reduced soil biodiversity or loss of soil stability (soil losses through erosion and landslides).

As such, the approach to quantify the degree of soil degradation via linkage between critical thresholds and current soil (functional) condition is a big step forward compared to risk assessment schemes from the past. However, this method inherently has several drawbacks related to terminology, methodology and local conditions:

- While various indicators related to soil threats have been proposed over the recent past, specifications for monitoring and evaluation are missing.
- There is no consensus yet between countries regarding valid regionalized critical limits used as thresholds for specific soil functions.
- The methodology to link a specific threshold (via models) to the current condition in soil, or water, differs between countries or group of countries.

The result is that risk assessment approaches used to define the degree of soil degradation and the outcome thereof can vary widely between countries.

A more general limitation is that for some forms of soil degradation, the actual linkage between current soil condition and the specific threshold is not established yet due to either lack of process-based knowledge (e.g. related to the biogeochemical behaviour in soil for ‘new’ contaminants), or the fact that for a regional or national assessment, soil data needed to feed the models are simply not available or interactions are highly conditional and/or complex.

However, in order to evaluate the current soil status or the impact of relevant soil threats on the environment including human health, risk-based limit values are essential. At present, there is no consensus at EU level on unified critical limits in view of the listed soil threats. Even though progress has been made, for example, in the development of effect-based critical limits for metals such as Cd, Pb, Cu and Zn, national standards for soil protection within the EU still vary widely. One reason for not having such a harmonized set of standards is the complicated interconnection between soil functions and site conditions (e.g. climate, soil fertility level), management measures (e.g. fertilizer management) and corresponding soil threats. In addition, views from Member States on targets or endpoints (e.g. water quality, food quality, ecosystem health) to protect, and at what level, widely differ. This further complicates the quest for a harmonized concept to derive such standards or critical limits. For soil contamination, this has resulted in multiple soil quality standards within the EU, ranging from non-effect-based target values (largely related to natural background levels in soils at MS level) to effect-based critical limits (targeting e.g. human health as endpoint). And even though both types of standards are of use, there is a need to describe the background of policy-based and effect-based approaches more clearly, to identify the common grounds on which risk assessment at EU level is to be built.

In Chapters 2 to 9, the current findings regarding thresholds for soil threats are summarized, while chapter 10 provides recommendations how this information can be used, e.g. for soil degradation assessments in Europe.

2 Soil organic carbon

Soil organic carbon (SOC) is a central soil indicator for several soil functions. Although the indicator is already abundantly applied, it is challenging to define thresholds for optimal or critical content, below which soil functions are hampered. This is because of the complex biochemical processes involved with decomposition, including mineralization and stabilization. Also, soil and environmental conditions vary profoundly across Europe. SOC dynamics is a result of the interplay between vegetation, climate, and soil; depending on its chemical composition and nature of its binding with the reactive soil mineral environment, SOC can be very responsive to climatic changes or changes induced by land management. While natural equilibrium under undisturbed conditions can hardly be observed anymore, also the threshold of lowest necessary SOC content to fulfil SOC-dependent soil functions, is hardly known. This makes it difficult to determine the level at which soils are degraded from SOC loss. In this chapter, several approaches to define thresholds are summarized. They mostly rely on the relationship between SOC content and yield response for agricultural soils, but also include the role of SOC for structural stability of soils.

Conservation or increase of soil organic carbon has positive impacts on almost all key societal needs related to soil and almost all soil functions. The need for infrastructure is the only exception. Table 2-1 below provides the relevance of the given indicator to the soil health objectives.

Table 2-1: Relationship of soil organic carbon (SOC) to key societal needs and soil functions

Soil organic carbon		
Societal need	**Soil function**	**impact**
Biomass	Wood & fibre production	+
	Growth of crops	+
Water	Filtering of contaminants	+
	Water storage	+
Climate	Carbon storage	+
Biodiversity	Habitat for plants, insects, microbes, funghi	+
Infrastructure	Platform for infrastructure	indifferent (1)
	Storage of geological material	indifferent

Note: (1)Soil organic carbon / infrastructure: organic soils are instable as platform for infrastructure

2.1 Rationale: role of soil organic carbon for soil productivity, filter and storage of water, nutrients and pollutants

Soil organic carbon (SOC), and hence soil organic matter (SOM), play a key role in carbon sequestration in agricultural and forest ecosystems and hence in climate change mitigation, removing CO_2 from the atmosphere. It also has direct impact on storage and filtering capacity of water, nutrients, and pollutants in soils and thereby on the environmental quality of water, air, soil and, ultimately, in water security. While there is a close relationship between soil nutrient status and SOC, it is not surprising that soil productivity is closely related to SOM levels (Korschens et al. 2005; Feller et al., 2012). SOM (as much as SOC) is today recognized as critical to preserve food security, and SOM decline leads to soil degradation because its loss is often followed by decreases in soil fertility and stability (Stolte et al., 2016). SOC can be considered a “universal keystone indicator” (Loveland and Webb 2003).

SOC has been widely used as an indicator to evaluate soil quality in response to management impacts under various environmental conditions (Bünemann et al., 2018). While under stable environmental conditions, the SOC stock develops towards a long-term equilibrium of mineralization and stabilization, changes in management and natural disturbances affect this equilibrium (Wiesmeier, 2019), causing the depletion of the SOC stock. If the SOC content falls below a certain threshold (or critical limit), all major soil functions are affected, causing soil degradation.

SOC dynamics is closely related to nitrogen dynamics (Van Groenigen et al. 2015). N affects the composition of microbial communities (e.g. proportion of fungi and bacteria), root turnover and chemical composition of SOM. SOM (research is nowadays focused on SOC) is thus an important indicator to regulate N application (use of fertilizers) while it contributes to minimizing environmental pollution (Musinguzi et al., 2013). It seems that anthropogenic N deposition increases soil C storage in terrestrial ecosystems, while increasing SOC improves the N use efficiency i.e. less mineral N is needed from fertilizer to obtain a potential crop yield (Schjønning et al. 2018). Critical SOM (or SOC) levels or thresholds may provide orientation to restore SOC- (and N-) depleted soils. Also, the connection between SOC and soil fertility and crop yield (where yield gaps are diagnosed) has now become the basis to identify SOC sequestration potentials (Amelung et al., 2020).

2.2 Indicator specification “Loss of SOC below critical levels”

Soil organic matter (SOM) is the sum of all dead organic components of different decompositional stages in a soil that are made from basic elements including carbon, nitrogen, oxygen, hydrogen and an array of cations and ions attached to it. Some definitions also include undecayed plant and animal residues as well as microbial biomass. Since SOM is difficult to measure directly, it is common practice to measure and report soil organic carbon (SOC). Historically, for the conversion of SOC to SOM a factor of 1.724 is used, based on the assumption that organic matter is 58% carbon. However, a review by Pribyl (2010) shows that a factor of 2 would often be more accurate especially in the case of soil layers rich in organic matter, such as in forest floors; this is because of differences in the degree and kind of humification (a process which generates humus) related to different stages of decomposition and mineralization.

While the loss of total SOC concentration over a monitoring period is often suggested as an indicator, the change of the bulk SOC concentration in a given soil may not be a good indicator for assessing how well a particular soil function is likely to perform. This is mainly because **labile (active) and stable pools** of soil organic matter vary considerably in their physical and chemical properties, resulting in a wide range of turnover (Gobin et al., 2011; overview of SOM fractions: Table 2-2). Monitoring these different fractions is important because it helps to understand how carbon dynamics in soil is affected by disturbances and how it can be effectively restored (Lehmann et al., 2008, Poeplau et al. 2018).

Recent findings indicate that it is not the “humic substances” as partly decomposed plant compounds, which form the basis for the carbon sequestration potential in soil, and soil fertility, but rather microbial necromass (microbial residues and their biomolecular coating of dead fungi and bacteria). The contribution of microbe-derived carbon to SOC could reach up to 82% (47% to 80%) (Liang et al. 2019). On that basis, Cotrufo et al. (2019) distinguish a mineral-associated organic matter (MAOM) pool, and a particulate organic matter pool (POM). It must be additionally considered that SOC storage and accumulation requires a specific amount of N (**N efficiency of C sequestration**). According Cotrufo et al. (2019), this amount of N depends on the share MAOM and POM, and their respective C/N ratios. POM consists of partly decomposed plant origin (with low N content), while MAOM seems mostly of microbial origin and is chemically bounded to minerals thus physically protected in small aggregates. Cotrufo et al. (2019) hypothesize that any additional C storage in soil is only realized through POM accrual. This research indicates that the determination of carbon sequestration potential and monitoring of SOC in response to management actions would require a different SOC analysis compared to what is currently still common practice.

Table 2-2: SOM pools in soils

Organic matter fractions/pools	Approximate share of total SOC	Characteristics of pool
Microbial biomass-C (bacteria, fungi)	5%	labile, active
Fresh organic material (freshly added plant and animal residues)	10%	labile, active
Active organic matter (partially decomposed residues)	33%	labile, active
Humified[(1)] organic matter (i.e. the well-decomposed and highly stable organic material also known as humus)	33%	stable, activity level depends on degree of organo-mineral complexation
Inert organic matter	20%	constant, not active

Source: Gobin et al. (2011)

Note: [(1)] Recently, the role humic of substances (uncharacterized structural composition, persistent, large-molecular-size constituents) has been questioned; rather, humus is perceived as a continuum of progressively decomposing organic compounds (Lehmann and Kleber 2015); see Liang et al. (2019) about the role of microbial necromass in the mineral-associated organic matter fraction; it can make up more than 50% of SOC.

Despite of the importance to look at the different components of SOC (labile and stable fractions), and because of the challenges to routinely determine them, measurement of **total SOC** is common practice in soil monitoring networks, e.g. from dry combustion (for which many laboratories have routine operations in place).[39] In addition, in order to avoid overestimations of SOC, the amount of mineral soil carbon must be determined and removed from any SOC estimate (with calcareous soils, or after liming).

SOC as an indicator is commonly expressed as concentration or stock (*syn*. pool size, density), and its quantification refers to a specific soil depth. In the following, key references are selected from the vast literature base about **SOC measurement and monitoring**:

- Reference literature about SOC to address policy needs including greenhouse gas inventories: Bispo et al. (2017), FAO (2019), IPCC (2019)
- Reference literature about SOC analysis: Nelson and Sommers (1996), Standard Operating Procedures (SOP) of the Global Soil Laboratory Network (GLOSOLAN)[40]
- Reference literature about SOC monitoring: Goidts et al. (2009), Schrumpf et al. (2011), Poeplau et al. (2017), Arrouays et al. (2018)

In the context of the assessment of soil degradation, the following functional soil carbon indicator is proposed, based on the spatial quantification of soil carbon concentrations or soil carbon stocks in a given depth:

$$SOC\ area_{\ degraded} = soil\ area_{\ SOC\ concentration\ (or\ stock)\ <\ threshold}$$

$SOC_{concentration}$ is expressed as the concentration of organic carbon in fine soil (fractions < 2 mm) on a mass basis (e.g. in units such as g C kg^{-1} soil), from a sample representing a certain soil layer or soil horizon of a specific depth.

39 For SOC analysis, additional reference literature (not cited here) includes alternative methods such as hot water extractable carbon (with seasonality aspects), and Near-Infrared Spectroscopy (NIR) and Pyrolysis, methods which will allow more accurate analysis of labile and stable humus fractions.

40 GLOSOLAN Best Practice Manual, Ch. 2, Volume 2.2. Soil Carbon http://www.fao.org/global-soil-partnership/glosolan/soil-analysis/sops/volume-2-2/en/

SOC_{stock} is the total amount of $SOC_{concentration}$, per area and layer/horizon thickness (e.g. in units such as ton C ha^{-1}); it depends on the bulk density and the stone content of the soil.

SOC_{stock} is the reporting unit in greenhouse gas inventories (Goulding et al., 2013). Because bulk density is not always measured, pedo-transfer functions are available (see e.g. Hollis et al., 2012), although this approach is error-prone (Wiesmeier et al. 2012). Farm advisors usually build their recommendations on $SOC_{concentration}$. Regarding the stone content, it is often neglected in agricultural topsoils; however, following recommendations by IPCC (2006), it is good practice to quantify carbon stocks in subsoils in order to quantify, which amount of carbon is vertically redistributed, and which part is lost (or gained) in the local soil under investigation. For this reason, the weight, or the volume of fine gravel (inside the sampling cylinders for determining bulk density) as well as coarse gravel and stones need to be estimated.

2.3 Critical limits for soil organic carbon

2.3.1 Overview of approaches to determine degradation by SOC

About the effects of soil management and the derivation of SOC thresholds for sustainable soil management, most available studies about thresholds have focused on the effect of SOC decline on crop yield. Crop yield is the result of the interaction of many factors in particular soil fertility, for which SOM and nutrient availability is important, as much as sufficient water. Hence, crop yield could be considered as a parameter for the definition of SOC threshold.

Recently, Oldfield et al. (2019) developed a quantitative model exploring how SOC relates to crop yield potential of maize and wheat, considering co-varying factors of management, soil type and climate; SOC is found to have an impact on yield also with zero input of N. Yields of these two crops are on average greater with higher concentrations of SOC, with yield increases levelling off at 2% SOC (Oldfield et al., 2019). Significant correlations between SOM and soil productivity have been found also for cereals, even under fertilization regimes (Pan et al., 2009); for rice, SOM positively correlated with the yield under no fertilization, contributing to 70% of the rice yield when under fertilization (Zhao et al., 2016). While these studies are often local, it seems difficult to drive globally valid thresholds for SOC-yield relationships.

An often-mentioned SOC threshold is 2% (~3.4% SOM) (Kemper and Koch, 1966; Greenland et al., 1975, both cited from Huber et al., 2008), below which potentially serious degradation of soil would occur. Loveland and Webb (2003) summarised what is known about critical thresholds of SOC respectively SOM, mainly in soils of temperate regions (for tropical soils, see Musinguzi et al., 2013). They concluded that the quantitative evidence for thresholds is limited; a SOC threshold of 1% seems more appropriate than 2%. Below that level, "and without addition of exogenous soil organic matter and fertilizers, a disequilibrium in N-supply to plants might occur, leading to a decrease of both SOM and consequently biomass production" (Körschens et al. 1998). Wessolek et al. (2008) also question the 2% threshold for SOC, because it cannot be achieved for various soils with naturally low SOC levels – not even through practices with optimal supply of organic matter (e.g. sandy cropland soils in north-eastern Germany).

Loveland and Webb (2003) conclude that quantitative evidence for single thresholds in relation to crop yields is difficult to broadly apply. Rather, any typical SOC content can only be determined if specific soil, management, and climatic conditions are considered. Given the diversity of soils and growing factors, one universal value for a critical minimum SOC level may not be appropriate (Goulding et al., 2013). Table 2-3 presents an overview of thresholds for SOC as discussed in this report.

Table 2-3: Overview of Chapter 2.3 on SOC thresholds

Chapter	Definition	Comments on the practicability of existing thresholds
2.3.2	**Reference values**	
	Site-specific, typical SOC or SOM values under current management	Can be easily derived from existing monitoring systems (e.g. as baseline)
	Benchmark SOC values from	Requires extensive monitoring evaluations (forest soil and natural grassland incl. organic soils) Requires validation
	Natural soils (forest soils with low historic disturbance)	
	25 quartile of the SOC median for permanent grassland	
	Optimal SOC content for soil functioning (based on the role of SOC in soil functional PTF, combined with data long term field experiments)	Reference values for central European soil and climate conditions are available Needs to be validated for clay-rich soils and climate regions outside central Europe
2.3.3	Soil **vulnerability** index based on the SOC/clay ratio	Optimum SOC content as 10% of the clay content (piloted in Switzerland, England, and Wales)
2.3.4	Reciprocal SOC **sequestration** potential	Optimum SOC content for CO_2-mitigation function of soils; target values represent SOC equilibrium under long-term sustainable soil management
2.3.5	Thresholds from **long-term field experiments**	Minimum SOC levels for sustainable crop production (values for central Europe)
2.3.6	**Farmers perspective** on deficient SOC	Degraded SOC levels according to farmer's perception (values for Europe)

Note: *Benchmark sites* reflect environmental and management conditions that are representative for a larger area (Van Lynden et al. 2004). Each site represents a very specific set of local conditions which are distinct from other environments. Benchmark sites are particularly important to validate simulation models of indicators.

2.3.2 Optimal or site-specific SOC reference values

There has been much discussion in the soil science community about whether there is a common optimal or critical minimum SOM or SOC level (Goulding et al., 2013), below which soil fertility, water retention (drought resistance), soil structure and other soil properties become insufficient, such that crop yields are affected even at optimal nutrient fertilization rates. This concept derives from the fact that SOM provides and represents key properties to soils, while depending on, and regulating, various biologically mediated soil processes and soil functions.

A simplified approach to thresholds, Arshad und Martin (2002) suggest deriving site-specific SOM values as reference for monitoring and as proxy for optimal SOM levels (see also benchmark SOC stocks, as proposed by De Vos et al. 2015, for forests soils). Such values can be taken from more or less undisturbed soils under natural vegetation (e.g. forests) or modelled, which thus would theoretically represent the highest SOC stock that can be reached by a given soil ("reference SOC stocks" according to Batjes 2011). Barré et al. (2017) suggest that the "highest reachable SOC stock for a given pedoclimatic condition under a given land-use could correspond to the mean of the top 10% of the measured SOC stocks for these conditions". Sparling et al. (2003), for New Zealand, has proposed as target value the SOC median for permanent grassland, and its 25 quartile as minimum value. This is a pragmatic solution and can be easily determined. The 25 quartile represents a conservative orientation, which seems quite realistic. An example to derive modelled reference SOC stocks is Lugato et al. (2015), who have produced a spatially explicit estimation of soil C storage potential in European arable soils by 2050, applying different management scenarios.

Wessolek et al. (2008; in German) have reviewed a great variety of soil models which were developed to predict soil functions and potential threats (e.g. soil water storage, cation exchange capacity), and which contain SOM as a driver. Models are mostly pedotransfer functions, and most of them entail the quantified relationship between SOM and soil properties, because soil organic matter is one of the key covariates. Despite the long tradition and vast research invested in SOM dynamics, the derivation of site-specific SOM content in relation to soil functions is still difficult. This is largely determined by the limited availability of representative and long-term SOC monitoring data. Based on a set of 16 German long-term field trials, the authors have developed a matrix of soil organic matter concentrations, depending on soil texture, climatic water balance and management intensity (type of fertilization).

Table 2-4 and Table 2-5 present matrixes of site- and management-specific reference values as derived by Wessolek et al. (2008). The approach seems promising (a) to serve as a proxy for **minimum SOC values in soils** , and (b) to be validated/extended for all of Europe (especially for loamy and clayey soils, following the conclusion by Wessolek et al. 2008).

Table 2-4: Matrix for optimal SOM values for different soils, management type and climate

Soil texture class	Management intensity (fertilizer)	Climatic water balance [mm] during summer[1]		
		< -1002)	-100 to 0	> 0
Sand	max. both(2)	1.74	2.60	3.47
	org. & mineral	1.64	2.50	3.36
	organic	1.43	2.29	3.15
	mineral	1.26	2.12	2.98
	Null	1.21	2.07	2.93
Silt	max. both	4.09	3.31	2.48
	org. & mineral	3.78	2.97	2.14
	organic	3.57	2.78	2.03
	mineral	3.26	2.59	1.91
	Null	2.95	2.14	1.33
Loam and clay	max. both	1.71	2.83	4.83
	org. & mineral	1.64	2.07	4.60
	organic	1.57	1.93	4.53
	mineral	1.50	1.84	4.47
	Null	1.41	2.00	4.24

Note: To be consistent with the metric in the following sections, initial SOC values been converted to SOM applying the factor 1.724; original source: Wessolek et al. (2008).
(1) negative water balance: potential evapotranspiration larger than precipitation during summer; positive values indicate climate-induced surplus in the water budget; related to April-Sept.
(2) maximal application of organic and mineral fertilizer; Null=no fertilizer

Table 2-5: Approximation for a matrix for lower SOM limits for extensive management

Soil texture class	Climatic water balance [mm] during summer[1)]		
	< -100[2)]	-100 to 0	> 0
Sand	0.9	1.6	2.1
Silt	2.6	1.7	1.4
Loam and clay	1.0	1.6	3.3

Note: Calculated as as 50% standard deviation of Table 2-4.
SOM calculated from the original SOC values applying 1.724 as conversion factor.
Source: Wessolek et al., 2008.

2.3.3 SOC/clay ratio

As mentioned above, SOC content is highly correlated to various other soil properties, in particular, the clay content. In a recent study, Johannes et al. (2017) have reviewed and investigated the role of soil structural parameters (soil aggregate stability, soil porosity, mechanical properties, penetration resistance), soil texture, and its relation to soil organic matter. Their study was inspired by the work of Dexter et al. (2008), who studied the relation between soil texture, in particular the clay content, and SOC. They propose an **optimum SOC content** as 10% of the clay content (later specified by others as the dispersible clay rather than total clay). This threshold was refined by Johannes et al. (2017), based on 161 samples representing a major part of the Swiss agricultural land (Cambic Luvisols) (Table 2-6). The threshold translates to a so-called **vulnerability limit** %SOC = 0.1 * %Clay. Prout et al. (2020) suggest < 1/13 as the threshold to indicate degradation because hardly any grassland and woodland sites fall into that category.

Table 2-6: SOC/clay ratio as an index for good soil structure

<table>
<tr><th>SOC:clay ratio</th><th>Explanation</th><th>Soil structure[(1)]</th><th>Explanation</th></tr>
<tr><td>> 0.125 (1:8)</td><td>Field-level optimum for good structure quality[(2)]</td><td rowspan="2">> 1:10 (VESS < 3)</td><td rowspan="2">Acceptable or good structure</td></tr>
<tr><td>0.1 (1:10)
(1:8-1:13)</td><td>Goal for farmers as minimum desired SOC level</td></tr>
<tr><td>< 0.07 (1:13)</td><td>Structural soil quality is most likely unacceptable[(3)]</td><td>< 1:10 (VESS > 3)</td><td>Degraded structure</td></tr>
</table>

Note: [(1)] VESS: Visual Evaluation of Soil Structure[41] (Ball et al., 2017; see also Chapter 8 – Compaction), the score ranges from 1 (good structure) to 5 (poor structure)
[(2)] enriched in SOC relative to the clay content
[(3)] depleted in SOC relative to the clay content
Source: Johannes et al., 2017

It can be expected that SOC increase has a positive effect on the recovery of soils from degradation of SOC loss and soil compaction, both processes inducing the degradation of soil structure. The current level of SOC, and its sequestration potential, may be a good indicator of the resilience potential of soils (Fell et al. 2018).

41 A simple method description plus video are available at https://www.sruc.ac.uk/info/120625/visual_evaluation_of_soil_structure/1553/visual_evaluation_of_soil_structure_-_method_description

Prout et al. (2020) applied this index for England and Wales. They demonstrated that the SOC/clay ratio as an index for good soil structure applies for a wide range of soils and land uses (arable land, grassland and woodland), and conclude that it can be used to monitor and understand the state of soils at larger scales. The ratio seems valid for at least all western and central European conditions.

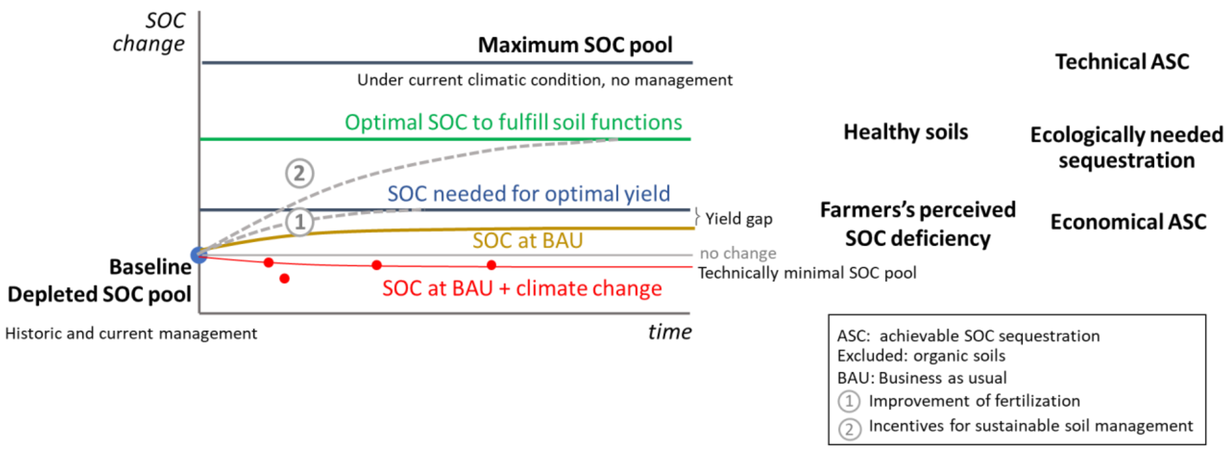
2.3.4 SOC critical limits as a reciprocal of the SOC sequestration potential

Carbon sequestration is "*the process of transferring CO_2 from the atmosphere into the soil of a land unit, through plants, plant residues and other organic solids which are stored or retained in the unit as part of the soil organic matter with a long residence time*" (Olson et al., 2014). The potential of soils to sequester carbon is because historic management has depleted the carbon pool of many soils. And even nowadays, certain soils still loose carbon under current management in some areas of Europe, especially on cultivated carbon-rich soils (drained organic soils) and for forest and grassland conversions.

Among the many benefits of SOC, Amelung et al. (2020) emphasize the potential contribution of soil to mitigate CO_2 increases in the atmosphere, thus its role stabilizes the climate. The major potential for carbon sequestration is in cropland soils, especially, where large yield gaps still exist, and/or where large historic SOC losses have occurred (Lessmann et al., 2020; Amelung et al., 2020). A map of yield gap could then serve to indirectly provide reference values of carbon sequestration potential, hence SOC limit values. It would basically correspond to a SOC level achieved with optimal fertilizer management Table 2-6.

Figure 2-1 presents the theoretical concept of the meaning of different SOC thresholds, and how it relates to soil degradation and carbon sequestration. Many aspects of the SOC dynamics under different management regimes and climate change, are still uncertain, for example, how **optimal SOC** increases the **resilience of soils to climate change**, thus could help mitigating a likely negative SOC balance under business-as-usual + climate change scenarios in the future. Climate change may offset all management efforts to sequester carbon (Meersmans et al., 2016). Research has focussed primarily on the productivity-related function of SOC (yield), which is not sufficient to prepare soils for future aridic conditions in many parts of Europe during the summer months. In this context, the current, extremely low SOC baseline in many European soils needs to be considered. These aspects influence under which conditions which SOC sequestration is realistically possible (see also Amundson and Biardeau 2018).

Figure 2-1: Conceptual overview of SOC thresholds and carbon sequestration



It becomes clear from Figure 2-1, that some of the kinds of thresholds identified in this report correspond to the SOC sequestration potential of soils. In particular the optimal SOC content, which corresponds to healthy soils considering all its functions, seems to be an achievable target for policy incentives. This has also been proposed by De Vos et al. (2015), who suggest that the SOC sequestration potential could serve as orientation for target values for optimal SOC content, to be derived from modelling or sampling undisturbed locations.

According to Sandermann et al. (2017), the identification of historically degraded land indicates where new carbon can be stored now and in the future. Currently, the degree of degradation and its economic impact can be best measured with a yield decrease and even its loss. However, ecosystem services of SOC (including its stabilization and protection from climate change and including the improved water dynamics of SOC-enriched soils) are still hardly accounted for among practitioners. This is probably the reason why soil degradation maps are not considered reliable (Amelung et al., 2020; Gibbs and Salmon, 2015). According to Lugato et al. (2015), the ratio between the actual and the potential SOC stock can be called the **SOC saturation capacity**. Depending on the SOC management scenario chosen for the modelling of the potential future SOC stock and depending on the level of degradation of the current SOC levels for cropland, this storage capacity is likely to be higher than the “optimum” for soils (compare to Ch. 2.3.2).

Current methods to calculate the sequestration potential apply models to compare baselines with assumed management scenarios: business-as-usual (BAU), and sustainable soil management methods (SSM) (examples: Lugato et al., 2015; FAO, 2020). The loss of SOC below a critical level relates to the potential for soil carbon sequestration: the more severe the SOC loss, the higher the carbon sequestration potential.

Based on Angers et al. (2011), the soil C saturation deficit can be defined as the difference between the maximum content of organic carbon in the mineral (fine) fraction (<20 µm; clay + fine silt particles) of soil. This is based on the observation that stable SOC compounds are particularly adsorbed onto the reactive mineral surface of smallest soil particles (see also Kleber et al., 2015). However, the mineral matrix is believed to have a finite storage capacity for organic carbon. These findings were taken further by Wiesmeier et al. (2014), who reported significant amounts of carbon stored in the >20 µm fraction (>20% in the analysed literature, 40-60% in this study). Considering that a large proportion of SOC increase (for example, through carbon farming) would occur in coarse soil mineral fractions, and considering that this SOC is less stabilized, then any such newly stored SOC may be susceptible to rapid loss if the changed farming practices would not be sustained.

2.3.5 Thresholds from long-term field experiments

Körschens et al. (1998) evaluated long-term field experiments (begun in 1902), which included treatments to ‘discern the influence of highly different SOM contents on yield and C and N dynamics’. The authors propose an upper limit to SOM, above which there is an increased risk of N and CO_2 loss; lower limits represent the SOM level to maintain optimum crop production. These limit values increase with increasing clay content, i.e. from 1% SOM at 4% clay, up to 3.5% SOM at 38% clay (Table 2-7). This means that even with the addition of fertilizer, below 1 % SOM, mineralizable N is too low so that potential optimal yields cannot be reached any more.

Table 2-7: Guideline ranges for the SOM content of sandy and loamy soils without groundwater influence (% SOM in ploughing layer) depending on fine silt (< 6.3 µm) and clay

Clay + fine silt [%]	% SOM in sandy soils		% SOM in loamy soils	
	upper value	lower value	upper value	lower value
4	1.5	1.0		
5	1.5	1.0		
6	1.5	1.0		
7	1.5	1.0		
8	1.6	1.1		

Clay + fine silt [%]	% SOM in sandy soils		% SOM in loamy soils	
	upper value	lower value	upper value	lower value
9	1.7	1.2		
10	1.7	1.2	2.0	1.3
11	1.8	1.3	2.1	1.4
12	1.9	1.4	2.2	1.4
13	1.9	1.4	2.2	1.5
14	2.0	1.5	2.3	1.6
15	2.1	1.6	2.4	1.7
16	2.1	1.6	2.5	1.8
17	2.2	1.7	2.6	1.8
18	2.3	1.8	2.7	1.9
19	2.3	1.8	2.8	2.0
20	2.4	1.9	2.8	2.1
21	2.5	2.0	2.9	2.1
22	2.5	2.0	3.0	2.2
23	2.6	2.1	3.1	2.3
24	2.7	2.2	3.2	2.4
25	2.8	2.2	3.3	2.5
26			3.4	2.5
27			3.4	2.6
28			3.5	2.7
29			3.6	2.8
30			3.7	2.8
31			3.8	2.9
32			3.9	3.0
33			4.0	3.1
34			4.1	3.2
35			4.1	3.2
36			4.2	3.3
37			4.3	3.4
38			4.4	3.5

Source: Körschens et al. 1998

The thresholds by Koerschens et al. (1998) demonstrate how the existing variability of soil properties affects the development of thresholds. As soon as values become simplified or grouped for easier application, ranges of values apply (see Table 2-8). Value ranges are always more difficult to handle because policy recommendations usually require exact values. For this reason, the aggregation of thresholds proposed by BMLFUW (2017), which reflects the scheme proposed by Koerschens et al. (1998), could be more easily implemented to provide guidance for optimal application of fertilizers for arable land and pastures (Table 2-8).

Table 2-8: Aggregation of SOM thresholds for soil groups

Soil groups	% clay/fine silt	Koerschens et al., 1998		BMLFUW, 2017
		% SOM in sandy soils	% SOM in loamy soils	Minimum SOM threshold*)
		Upper threshold (range)	Upper threshold (range)	
Light (<15% clay)	4-7	1.0-1.5		> 2.0
	8-14	1.5-2.0	1.6-2.3	
Medium (15-25% clay)	15-22	2.0-2.5	2.2-3.0	> 2.5
	23-25	2.2-2.8	2.5-3.3	
Heavy (>25% clay)	25-32		3.0-4.0	> 3.0
	> 32		3.5-4.4	

Note: *plot specific optimal humus contents
Source: Trombetti et al. 2019

At this stage, for a Europe-wide application, it is currently difficult to apply the thresholds defined by Körschens et al. (1998) with the available Europe-wide texture data: the “fine silt” class (6.3-2 µm) cannot be isolated. Beside this, no definition is given for “Sandy soils” or “Loamy soils”. Similar to the results of Table 2-4 and Table 2-5, larger data sets are needed to improve representativity of the values for larger areas outside Germany, and to validate whether soil functions are limited below these thresholds. However, it can be concluded that thresholds must consider textural class, and that the values presented provide orientation to evaluate the SOC measurements from monitoring.

2.3.6 SOM thresholds from a farmer survey

Farmers´ perceptions of minimal carbon concentration needed for maintaining agricultural production levels in a sustainable manner were investigated by Hijbeek et al. (2016). Besides a literature review, an extensive farm survey was conducted involving 1452 arable farmers in five European countries (Austria, Belgium. Germany, Italy and Spain). Thresholds were derived based on a subset pool of 635 farmers (out of 1452) which also reported an average field-level SOM content. Frequency distributions were stratified by soil texture and macro-climatic region. Due to the ‘fuzziness’ of responses to the questionnaire (some farmers perceive SOM deficiency at the same level of SOM compared to farmers who don’t), and the corresponding statistical weakness, two thresholds were derived in order to have a conservative approach (see Table 2-9 and Table 2-10).

Table 2-9: Definition of SOM Thresholds for values derived from a farmer’s questionnaire

Threshold 1	Threshold 2
positive judgment of the SOM content (low or very low deficiency)	perceive a high or very high deficiency of SOM
lowest SOM percentage (10th percentile)	highest SOM content (90th percentile)
below this threshold value, no farmers are expected to be satisfied with their SOM content	above this threshold, no farmers are expected to be dissatisfied with their SOM content

Source: According to Hijbeek et al. (2017), extended for this study

Table 2-10: SOM thresholds for cropland based on a farmer's questionnaire

Climate	Texture	Threshold 1	Threshold 2
Atlantic	Coarse	2.1	3.5
	Medium	1.7	2.6
	Medium fine	2.8[(1)]	n.a.
Continental	Coarse	1.8	2.1
	Medium	2.3	3.2*)
	Medium fine	2.0	2.4*)
Mediterranean	Coarse	1.7[(1) (2)]	1.1[(1) (2)]
	Medium	1.0	2.0
	Medium fine	1.3	1.4[(3)]

Note: [(1)] high uncertainty
[(2)] effect of different groups of farmers in the questionnaire and their perceptions about the SOC level on their farms

The suggested thresholds (Table 2-10) were tested and compared in Trombetti et al. (2020). The shares of cropland which fall below the threshold 1 are 0.1%. 0.5% and 11.1% in the Mediterranean, Atlantic and Continental climatic zones, respectively. When applying the vulnerability index for soil structure degradation from SOC loss, Prout et al. (2020) found that 38.2%, 6.6%, and 5.6% of arable, grassland and woodland sites in England and Wales, respectively, were degraded.

Applying the farmer's perception of SOC depletion as threshold, there is very little "degraded" cropland area in most of the intensively managed agricultural areas in Europe. This threshold largely considers SOC important for yield dynamics. It is obvious that the initial SOC content of uncultivated soils under permanent vegetation cover has been much higher (see Table 2-10). Soils – through cultivation – have lost more than 50% of their initial carbon stock. This indicates that cropland soils have already lost a significant amount of their potential to deliver soil functions so that in theory, many cropland soils could be perceived as SOC-degraded. On the other hand, this loss of soil functions, and the re-gain of soil functioning with increasing SOC, is yet difficult to quantify (see also Wiesmeier et al. 2019).

When determining and applying SOC thresholds, not only the production function counts. The role of SOC in relation to the various soil functions needs to be considered, for which SOC is an important controlling and enabling factor (Wessolek et al. 2008).

3 Soil nutrient status – N and P

Soil nutrient status affects biomass production in natural soils and crop yields in agricultural soils, although the impact is less in fertilized soils. An appropriate nutrient status has positive impacts on biomass production and crop yield. it is defined by appropriate levels of available macronutrients i.e. nitrogen (N), phosphorous (P), calcium (Ca), magnesium (Mg), potassium (K) and sulphur (S) and micronutrients, i.e. boron (B), zinc (Zn), manganese (Mn), iron (Fe), copper (Cu), molybdenum (Mo). In addition, it affects the diversity of soil microorganisms, soil animals and plant species, as well as water quality.

Table 3-1: Relationship of soil nutrient status to key societal needs and soil functions

Soil nutrient status		
Societal need	**Soil service**	**impact**
Biomass	Wood & fibre production	+
	Growth of crops	+
Water	Filtering of contaminants	indiff
	Water storage	indiff
Climate	Carbon storage	+/-
Biodiversity	Habitat for plants, insects, microbes, fungi	+
Infrastructure	Platform for infrastructure	indiff
	Storage of geological material	indiff

The most important soil nutrients are N and P. Together with soil pH, which is strongly related to the availability of the base cations Ca, Mg and K but also to the availability of micronutrients (especially Fe, Zn and Mn) and of toxic aluminium (Al), they are the main determinants of soil fertility. These soil parameters, i.e. N, P and pH with related impact on other elements, are affected in agriculture by inputs from fertiliser and manure application and atmospheric deposition - the latter being the main source of input in non-agricultural soil.

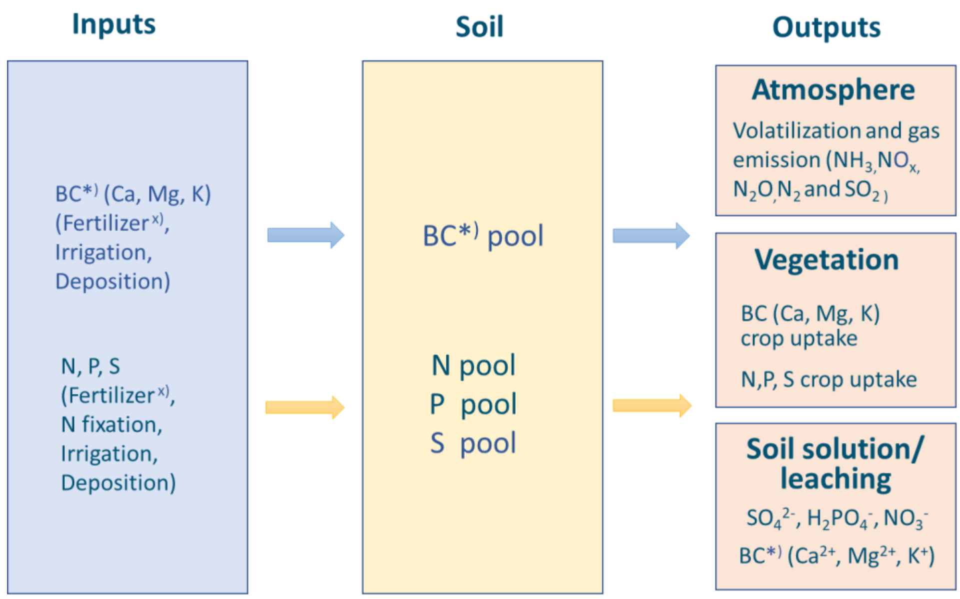
This section focuses on the Importance of N and P monitoring. First, the impact of soil N and P input is described, considering biomass production and crop growth, soil and plant biodiversity and water quality. This is followed by an overview of indicators for those impacts. Finally, thresholds are described, below (target levels) or above (critical levels) which the nutrient status should preferably not come. In Chapter 4, the same is done for soil acidity.

3.1 Rationale: impacts of soil N and P levels on biomass production and crop growth, soil and plant diversity and water quality

3.1.1 The fate of N and P in soil, in response to N and P inputs

Nutrient inputs to soils affect the soil nutrient status (content sand pools) which in turn affects the output of nutrients in vegetation (nutrient uptake due to biomass production, such as tree growth and crop growth) to air and water (Figure 3-1). Nitrogen (N) and phosphorus (P) are essential macronutrients, both of which widely limit primary productivity across terrestrial ecosystems (Elser et al., 2007; Vitousek et al., 2010). In non-agricultural systems (i.e. forestry), biomass growth is thus enhanced by additional inputs. At a certain level, however this production increase will also adversely affect plant species diversity and it may cause losses of N and P to water where it may cause eutrophication (see 3.1.3) and it even may reduce growth at very high levels (see 3.1.2). In agricultural sols, crop growth is strongly stimulated by N and P fertilizer and manure inputs, since plant species diversity is aimed for in agricultural production, but here the risk for N and P losses to water and related eutrophication impacts is even much higher. The losses of N and P to air (N only) and water (N and P) in response to N and P inputs, however strongly differ in view of differences in N and P dynamics in soils as explained below.

Figure 3-1: Link between nutrient inputs and soil nutrient pools and nutrient outputs to air, vegetation and water



*) BC: base cations x) incl. Sewage sludge, waste waters, composts, digestates

Nitrogen

Unlike other nutrients, such as P, Ca, Mg, K and S, there are no N minerals in (agricultural) soils that can be formed via precipitation or secondary mineral formation; there is also no N sorption to either clay or organic matter. While there is usually a chemical equilibrium between available and reactive ions and those in the soil solution (a prerequisite for deriving thresholds based on simple nutrient concentration analysis), no such equilibrium exists for N. Instead, biologically mediated processes affect the N availability to crops and the N surplus in agricultural soil. This N surplus is either emitted to air, as ammonia (NH_3), nitrogen oxides (NO_x), nitrous oxide (N_2O) or dinitrogen (N_2), or to ground- and surface-water, mainly as nitrate (NO_3), or it accumulates in soil, but the long-term change in organic soil N stock is very limited. Due to this behaviour, changes in N inputs by fertilizer and manure directly affect crop yields, even in soils with a high soil N status

Phosphorous

Unlike N, the concentration of P in soil solution is buffered by the stock of reactive or readily available P. Consequently, plant P uptake is strongly governed by the soil P status and this holds also for losses of P to ground water and surface water. Due to this behaviour, changes in P inputs by fertilizer and manure has smaller impacts on crop yields in soils with a high soil P content. Jungk et al. (1993) found that 14 years of different fertilizer P application rates varying between 0 and 180 kg P_2O_5 yr^{-1} at 4 equal intervals hardly affected the yield and plant P concentrations of winter wheat and sugar beet planted in rotation. They concluded that plant P demand was fully satisfied by uptake from soil P reserves that was accumulated during previous soil P applications in excess of plant demand.

3.1.2 Impact of N and P on biomass production

Enhanced N deposition often stimulates forest growth and hence carbon sequestration (Högberg, 2007; De Vries et al., 2009; Thomas et al., 2010), but the expected growth acceleration might be diminished when the accompanied P supply is deficient (Braun et al., 2010; Li et al., 2015; Lang et al., 2016). Soil P availability in terrestrial ecosystems is primarily driven by mineral weathering and atmospheric deposition (Vitousek et al., 2010). P input from atmospheric deposition is low and this also holds for weathering is also generally low. Newman (1995) reviewed P deposition and weathering in global terrestrial ecosystems and estimated a range of 0.07 – 1.7 kg P ha^{-1} yr^{-1} for P deposition and 0.01–1.0 kg P ha^{-1} yr^{-1} for P weathering, indicating that both fluxes are in the same order of magnitude.

The unbalanced atmospheric deposition of N and P (Peñuelas et al., 2013, Du et al., 2016) implies an increase in the area with P-limited ecosystems. Using the leaf N:P ratio of 15 dominant tree species as an indicator, through which the spatial variation of plot-level shift towards N or P limitation across 163 European forest plots during 1995-2017 has been demonstrated. In total, 38% of the studied plots showed a shift towards P limitation, while only 6% of the plots showed a shift towards N limitation, as indicated by a significant increase and decrease of leaf N:P ratio, respectively. Forests are thus increasingly suffering from P deficiency in forest nutrition (Talkner et al. 2020). Especially beech forests are affected by P deficiency (Lang et al., 2019). The increasing use of wood will lead to further nutrient deprivation. Liming also increases this effect.

3.1.3 Impact of N and P on biodiversity and water quality

Soil nutrient (N and P) status affects crop yields, but in case of N it also influences soil biodiversity and in case of P it affects the P accumulation and P losses to surface water, as discussed below. Important soil parameters affecting the availability of nutrients soil nutrients status include the content of aluminium and iron oxides (for P) and the combination of pH, organic matter, and clay content (N and P). Nitrogen in soil also affects the biodiversity of soil organisms, whereas the soil P content has an impact on surface water quality, while both N and P affects biodiversity, especially the plant species diversity, in non-agricultural ecosystems, as discussed below.

Nitrogen and soil biodiversity.

Fertilization and affluent available N reduce the abundance, activity and composition of soil fungi, saprotrophic decomposers as well as mycorrhizal fungi, and N fixing bacteria (Streeter, 1988; Johansson et al., 2004; De Vries et al., 2006). Both plant litter and microorganisms form the basis for detritivores in the food web of soils: thus, excess N leads to bottom-up effects on the whole belowground food web, on plants and eventually also the aboveground food web (Wardle et al., 2004). Impacts of N on soil biodiversity mostly come from studies where organic farming systems were compared with conventional intensive farming systems, but interpretation is partly hampered due to other differences, such as the avoidance of pesticides in organic farming systems (see Velthof et al., 2011). An overview of impacts of N on soil biodiversity indicators such as soil microbial biomass, activity, N mineralization, and microbial diversity (genetic diversity, number of genotypes or species of bacteria) is given in Velthof et al. (2011) (see also Chapter 6).

Nitrogen (and phosphorus) and plant species diversity.

Both N and, to a lesser extent P, affect the biodiversity, especially the plant species diversity, in non-agricultural ecosystems. In this context, forest ecosystems is the largest non-agricultural form of land use. In forests, N is generally a limiting nutrient and deposition may first enhance growth and productivity through enhanced N availability, but in a later stage it may cause eutrophication and acidification, negatively affecting nutrient balances and leading to an increased susceptibility to drought, diseases and pests (for a review of impacts, see Erisman and De Vries, 2000 and for an overview of effects in European forests, see de Vries et al., 2014a). In other ecosystems, growth enhancement is not a benefit but a threat, as it causes a decrease in plant species diversity, also being the trade-off in forest ecosystems. Atmospheric N deposition thus affects various ecosystem services, by its impacts on the fertility (quality) of forest soils and thereby being the forests' capacity to provide services such as wood production (provision service), carbon sequestration (climate regulation service), buffer capacity (water quality regulation service) and pest/disease regulation (see Erisman et al. 2014 and De Vries et al. 2014b for overviews). In this context, the soil P status is also important as it affects the impacts of N in situations where P is limiting growth and thereby also impacts on plant species diversity.

Nitrogen and phosphorus and water quality.

Elevated N and P concentrations in surface waters and coastal/marine waters contribute to the phenomenon of eutrophication with related impacts on the biocoenosis of freshwater ecosystems and coastal/ marine ecosystems. The enrichment of N and P in freshwater is largely due to surface runoff from (agricultural land). Specifically, in marine ecosystems, where N is considered to be the most important element in limiting phytoplankton growth, effects can be considerable and include many negative effects.

3.2 Indicators for N and P status of soils

Indicators for the soil N and P status can be given in terms of total values (total N and P content) and (plant) available N and P contents. These indicators are specifically used to gain insight in the soil fertility status and the need for N and P fertilization in view of crop growth.

- For **N in agricultural soils, target levels or critical levels** are <u>not</u> defined, but there are indicators for **forest soils** (see also section 3.3).
- For P, target and critical levels can be defined, however, a distinction needs to be made between the two: below the **target level,** the soil P status should be increased in view of P limitation for crop growth; above the **critical level,** there is an enhanced risk for negative effects on water quality due to enhanced P loss to surface waters. Details on indicators are described below.

3.2.1 Indicators about the N status in soils

Mineral N in agricultural soils.

In agricultural soils, the total concentration of mineral N (N min), being the sum of available NH_4 and NO_3 in the soil profile, determined by for example an extraction of 1N KCl, is the most relevant indicator for the N status of an agricultural soil in relation to crop yield. It is an indicator of potentially available N, due to its relationship with N mineralization, which increases the sum of dissolved NH_4 and NO_3 in the soil solution. Only this fraction of N is directly available to plants. The concentration of N min is assessed each year for farmers, since N min is highly variable, depending on soil and crop properties and climate and it is used to give advice for N fertilizer applications.

Advice about N fertilization follows the objective of so-called balanced N fertilization. Based on a target crop yield and the N content in harvested crop, the required N uptake in plants is derived. Then the effective N input by other sources than fertilizer is derived. This includes N mineralisation related to mineral soil N, as mentioned above, and external N inputs by manure, crop residues, N fixation and atmospheric N deposition. Effective N inputs are used since there are unavoidable losses of N to air and water, for example, due to the fact that N that is mineralized or comes in by manure and deposition comes partly available outside the growing season. The gap between the required N uptake and the effective N input by other sources than fertilizer is then used to recommend a certain N fertilizer application, accounting again for unavoidable losses of N to air and water.

Carbon to nitrogen ratio in non-agricultural (forest) soils)

Aber et al. (1998) launched the theory on ecosystem N saturation: with a focus on forest ecosystems, they distinguished different stages in view of:

- impacts on soil chemical processes such as mineralization, immobilization, nitrification, affecting N leaching, acidification
- plant nutrition and forest growth, and
- plant species diversity.

Below a specific threshold, terrestrial ecosystems will react to additional N inputs by an increased biomass production. Above a physiological optimum, production remains constant, or even decreases. When the ecosystem approaches "N saturation", N leaching will increase above (nearly negligible) background levels, associated with soil acidification in terms of elevated leaching of base cations as well as increased levels of aluminium at low pH (see chapter 4). At this level, a decrease in plant species diversity and changes towards more nitrophilic species are also observed (e.g. Bobbink and Hettelingh 2011).

One possible indicator for N eutrophication impact in forests is the C/N ratio for either the highly humified organic layer (H horizon: for moderate to nutrient poor forest soils) or for the top few centimetres of mineral soils (nutrient-rich forest soils with absent H horizons). There are indications that N retention is reduced with a decreasing soil C/N ratio, especially in the organic layer, as shown first by Dise et al. (1998, 2009) and Gundersen et al. (1998). This allows to derive a critical C/N ratio in these soils as discussed below in section 3.3.

Output indicator: N concentrations on air and water:

At high target crop yields and/or in soils with limited possibilities for denitrification (e.g. well drained sandy soils), the current N inputs may cause an exceedance of critical limits for nitrate (NO_3) in ground water or total N in surface water (see 3.3 for critical limits). Similarly, at high manure N inputs, typical for areas with intensive livestock husbandry, the emission of ammonia (NH_3) may be such that it exceeds critical levels for NH_3 in air or critical loads of N to ecosystems. It is thus the air and water quality that limits N management rather than soil quality (apart from soil acidification, as discussed in chapter 4 but that effect is in agricultural soils and is generally counteracted by liming). In this context, critical N inputs to soils are calculated, being the inputs that cause concentration of NO_3 in ground water or total N in surface water or NH3 in air which are equal to the critical levels of those N compounds (De Vries and Schulte-Uebbing, 2020). In Europe, there are many regions where current input exceeds those critical N inputs, but this cannot be monitored by a soil N indicator by air and water quality indicators.

3.2.2 Indicators about the P status in soils

Crop yields and available soil P contents

The P indicator used in agricultural soil is the **available P concentration**. The P concentration in the rooted topsoil is derived from soil-P tests (extractants); it indicates the availability of P, and is used for P fertilizer recommendations, based on their linkage with crop yields (Jordan-Meille et al., 2012). There are many extractants that are used to assess the available soil P level, and all extract a different soil P pool. Examples of available soil P parameter are P- Bray (Bray and Kurtz,1945), P-Olsen (Olsen et al., 1954), P-ammonium oxalate (Joret and Hebert, 1955), P-ammonium lactate (Egner et al., 1960) and P-Mehlich (Mehlich, 1984). Each extraction method yields a varied amount of a given nutrient in a soil sample due to differences in extraction mechanism. For instance, fourteen extraction methods in Europe were evaluated by Jordan-Meille et al. (2012), and they concluded that the tested extraction methods yielded different amounts of P from different P pools. Considering the above-mentioned methods, they showed that P extracted increased in the order of the following P extraction methods:

Olsen < Ammonium lactate < Mehlich3 < Bray II < Oxalate < Total P

Regarding thresholds, it would be highly advisable to apply a harmonized extraction, but unfortunately this is not the case, since many countries have related P in a given soil extraction method to crop yields and do prefer to stay with their approach. Actually, it would be even more preferable if a harmonized approach was taken for both a reactive (long term available) soil P pool (e.g. ammonium lactate and a dissolved P concentration like P in water or 0.01 M $CaCl_2$) as this gives information on the soil P buffer capacity, i.e. the speed with which P in solution is replenished form the available pool after P uptake or P leaching.

Water quality and soil P saturation index

One indicator for the effect of P applications on water quality is **P-$CaCl_2$, or P water (Pw)**. Both values represent the dissolved P concentration in the soil solution. Both variables can, however, be highly variable. The **P status in relation to leaching** is expressed by the so-called **P saturation index**; it is defined as the ratio $P_{ox}/(Fe + Al)_{ox}$, where P_{ox} and $(Fe + Al)_{ox}$ stands for the P, Al and Fe extracted in ammonium oxalate.

3.3 Critical limits or target values

3.3.1 Critical limits for N status indicators

N min in agricultural soils

In agricultural soils, critical limits for total N or available N (the mineral N content) in soil, related to specific soil functions are difficult to define. They do affect crop growth in unfertilized soils by affecting N mineralization (see above), but there is no critical limit for it, because N fertilization ensures affluent N supply. Furthermore, excess N does not limit crop growth when related soil acidification is properly counteracted by liming (see chapter 4). High N min content may negatively affect soil biodiversity, but limit values cannot be defined since impacts are not related to differences in soil N status but to effects of N addition to the soil (via fertilizer), as described in section 3.1. Finally, losses of N to air and water, negatively affecting air and water quality, are more related to N inputs and the soil properties affecting denitrification and thus N leaching (especially clay content and ground water level) and N emissions than the actual soil N status.

Critical limits for the C/N ratio in the organic layer of forest soils (H horizon)[42]

The N retention capacity of forest soils is strongly affected by N transformation (mineralisation and immobilisation) processes in the organic layer (LF and H horizon) and to a lesser extent in the mineral topsoil. At high ratios of carbon to nitrogen (C/N ratio) in the soil, most incoming N is retained by microbial immobilisation and limited N is plant available. When more N is stored, the C/N ratio declines, and more N becomes available by mineralisation for plant uptake and leaching. Based on relationship between N leaching and the C/N ratio in the organic layer of forests, C/N ratios varying around 25 (between 20 and

42 H horizon, part of the forest floor: commonly understood to be dominated by humified organic matter and mineral compounds < 30 %

30) are considered critical, with lower C/N ratios indicating an increasing risk for N leaching from forest soils, as shown in the Table below.

Table 3-2: C/N ratio in the organic layer of forest soils

Indication	C/N range in organic layers
High N retention and thus low N leaching potential	>30
Moderate to high N retention and thus low to moderate N leaching potential	25-30
Low to moderate N retention and thus moderate to high N leaching potential	20-25
Low N retention and thus high N leaching potential	<20

Table 3-2 illustrates that above 30 there is very limited leaching risk while it is high below 20 and in between there is strong variation. In more detail, De Vries et al. (2006) derived a N retention fraction based on the NH_4-fraction in the N input and the C/N ratio of the organic layer.

However, a C/N ratio below a value of 25 is often suggested as a threshold value for enhanced leaching. For example, Gundersen et al. (1998) presented a very limited C/N range in organic layers (30-25) to distinguish sites with high N retention and thus low N leaching potential (>30), from those with low N retention and thus high N leaching potential (<25). Using a dataset of published N budgets and C/N ratios of the organic layer, MacDonald et al. (2002) found the strongest relationships between N output and N input when the data were divided for 'N-rich' sites (C/N ≤ 25) and 'C-rich' sites (C/N > 25). This was confirmed by Dise et al. (2009), however, they introduced a threshold of C/N = 23 in the organic layer. Using a subset of the ICP Forests level II database, Van der Salm et al. (2007) found that N leaching was best explained if the C/N would be further refined based on annual average temperature and N throughfall.

Critical limits for N concentrations air and water

As mentioned above, N inputs in excess of N uptake, called the **N surplus**, cause emissions of ammonia (NH_3) to air, leaching of nitrate (NO_3) to ground water, and runoff of total N to surface water and there are critical limits for these concentrations in air and water in view of impacts on ecosystems and health).

- NH_3 in air: 1 – 3 mg NH_3 m^{-3}
  Evidence shows that ammonia in air can have significant toxic impacts on plants by direct uptake through the foliage above a threshold level (for an overview of direct effects of atmospheric ammonia on terrestrial vegetation, see Krupa, 2003 and Cape et al., 2009). The sensitivity of (plant) species to NH_3 increases going from lichens < native vegetation < forests < agricultural crops. Cape et al. (2009) reviewed methods to set a critical level for NH_3 and collated the available evidence to propose an updated NH_3 critical level for different types of vegetation. Based on the evidence a long-term (several year) average critical limit for NH_3 in air of 1 mg NH_3 m^{-3} is now proposed for lichens and bryophytes and of 3 mg NH_3 m^{-3} for higher plants, including forests.

- N in soil solution: leakage from forests: 1 mg N l^{-1}
  De Vries et al. (2007c) suggest an upper limit of 1 mg N l^{-1} as differentiation between undisturbed and 'leaky' N saturated forest sites, based on Gundersen et al. (2006). These authors gave an overview of current water quality in forests by compiling a list of studies from the 1990s on nitrate concentration in seepage water from temperate forests, including >500 sites of seepage water from Europe. From the survey data it is difficult to conclude exactly at which level a forest ecosystem can

be considered ‘leaky’, but they suggest an annual average N concentration level of 1 mg N l^{-1} for seepage water and 0.5 mg N l^{-1} for streams/catchments. Stoddard (1994) characterised four progressive stages of N saturation based on changes in seasonality and levels of nitrate leaching in streams and a value of 1 mg N l^{-1} coincides with the limit for the near final stage.

- N in soil solution: impacts on forests: 1 to 5 mg N l^{-1}
Empirical data suggest that critical dissolved N concentrations in view of adverse impacts on fine root biomass/root length and an increased sensitivity to frost and fungal diseases vary between 1–3 mg N l^{-1} and 3–5 mg N l^{-1}, respectively (De Vries et al. 2007c). The critical values for impacts on fine root biomass and root length are based on Matzner and Murach (1995), who found that total fine root biomass of Norway spruce saplings decreased significantly when the dissolved N (NO_3 + NH_4) concentration was >2 mg N l^{-1}. Critical dissolved N concentrations in view of an increased sensitivity to frost and fungal diseases has been derived from a critical N concentration in the needles of 18 g kg^{-1}, above which this sensitivity increases. De Vries et al. (2007c) derived a relationship between foliar N contents and dissolved annual average N concentrations on the basis of the results for 120 Intensive Monitoring plots in Europe. Below 3 mg N l^{-1}, the N contents in foliage were always below 18 g kg^{-1}, while above 5 mg N l^{-1} values were nearly always above this value. In this rang, adverse vegetation changes are also found (De Vries et al. 2007c).

- NO_3 in ground water: 50 mg NO_3 l^{-1}
The critical NO3- concentration in groundwater is generally set to the WHO drinking water limit of 50 mg NO_3 l^{-1} or 11.3 mg NO_3-N l^{-1}. This limit is based on epidemiological evidence for methemoglobinemia in infants (WHO, 2011).

- N in surface water: 1.0 to 2.5 mg N l^{-1}
Critical limits for dissolved total N in surface water, as indicator for eutrophication of aquatic ecosystems, vary mostly in the range of 1.0 to 2.5 mg N l^{-1}. This range is based on (i) an extensive study on the ecological and toxicological effects of inorganic N pollution (Camargo and Alonso, 2006) and (ii) an overview of maximum allowable N concentrations in surface waters in national surface water quality standards (Liu et al., 2011).

Note: Critical loads, being the critical deposition level from the atmosphere, can be calculated with models that make use of critical limits of N in soil solution (see de Vries et al., 2015, where this is explained in detail). In addition, critical limits for N in air, ground water and surface water are used to assess critical N inputs in agricultural soils (see e.g. De Vries and Schulte-Uebbing, 2020).

3.3.2 Target levels and critical limits for P status indicators

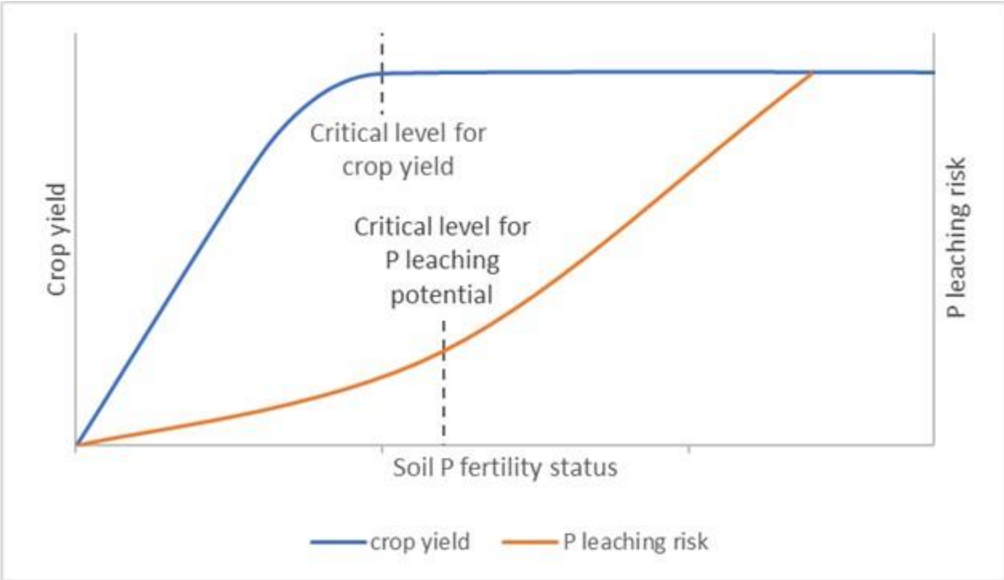
Agricultural soils

In order to avoid losses in crop production as well as negative environmental impacts, available soil P levels should ideally stay:

- above a critical level below which crop yield is limited, defined as the ‘soil P status above which crop yield does not respond to P application’ (Mallarino and Blackmer 1992), and
- below a critical level above which P leaching and runoff is significantly enhanced (e.g. Li et al., 2011).

This principle is illustrated in Figure 3-2. Both plant uptake and leaching are predicted by using both the buffer of available P bound in soil (in the figure defined as Olsen-P) and available P in solution (in the figure defined as $CaCl_2$-P). The figure shows a critical available P level above which the crop yield does not further respond (critical level for crop yield in Figure 3-2), and a critical available P level above which the risk for P leaching increases (critical level for P leaching potential in Figure 3-2). Bai et al (2013) used the change-point between available soil P and $CaCl_2$-P, where $CaCl_2$-P is indicative for dissolved P that is leached out of the system, as an indicator for risk (see also Heckrath et al., 1995 and Hesketh and Brookes, 2000) and in their approach this level is higher than the critical level (or target level) for crop yield. This implies that there is a safe range for available soil P levels, being enough for crop growth and not causing a risk for water quality, but this approach does not account for the potential risk of elevated P below the above mentioned change point (see section 3.3). Below critical limits for crop yields and water quality are given based on this principle

Figure 3-2: Relationships between crop yield (left y-axis) and P leaching risk (right y-axis) and available soil P fertility status



Source: Bai et al. (2013)

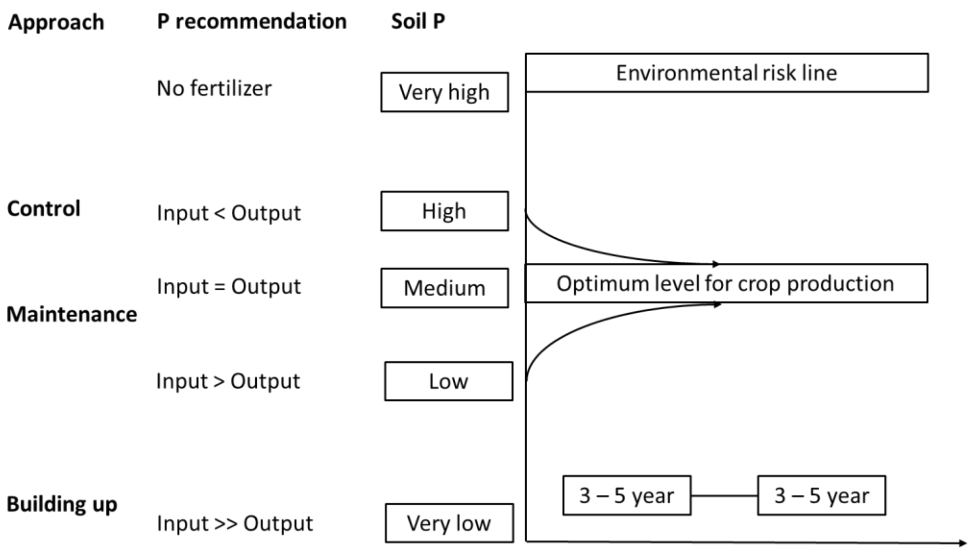
Target levels for available P in view of crop yields

The concept of thresholds for P has been commonly developed and applied in fertilizer recommendations: of common practice is the “Build-up and Maintenance” approach. The principle of this approach is that P application should

- not occur in soils with available soil P levels above the change-point (threshold) for P leaching,
- equal the P withdrawal in harvested crops*) if:
  - available soil P > critical level for crop yield
  - available soil P < critical level for P leaching
- equal the P withdrawal in harvested crop plus an additional amount of P fertilizer, to build up available soil P to the required agronomic level, if:
  - available soil P < critical level for crop yield (Li et al. 2011).

This approach is illustrated in Figure 3-3. The objective is to move from the environmental risk level (very high P-status) or P deficient level (very low P-status) to the level of ensuring stable crop yield (medium P-status).

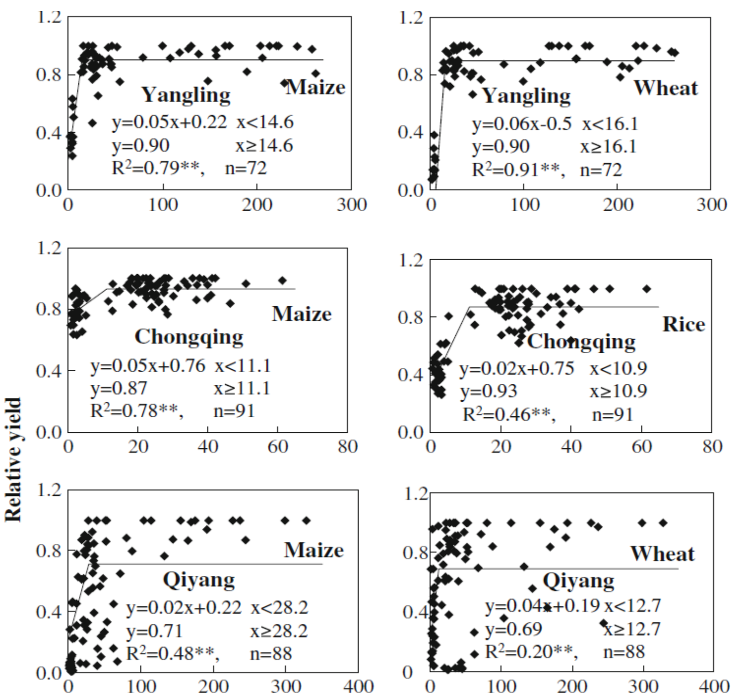
Figure 3-3: Principle of building-up and maintenance approach



Source: Li et al. (2011)

The critical P level for crop yield can be derived by short term (months) pot experiments (in a laboratory r greenhouse) and in long-term (years) field experiments in which P is added to soil by fertilizer and crop yields are recorded while accounting for differences in soil P status. The advantage of short term (months) pot experiments is that soil P level is the only varying element while all other circumstances are equal; the disadvantage is the different environment conditions between laboratory and field responses. Inverse, the advantage of long-term field experiments is that the impacts are derived field conditions, but the disadvantage is that other factors affecting crop yield, such as climatic variables may change in time.

Figure 3-4: Left: relationships between relative crop yield and soil Olsen-P levels. Right: soil fertility class and extraction method



Fertility class	Extractable P concentration (ppm)		
	Mehlich 3-ICP	Bray P1	Olsen P
Very low	<15	0-8	0-5
Low	15-25	9-15	6-9
Medium (Optimum)	25-35	16-20	10-13
High	35-45	21-30	14-18
Very high	>45	>30	>18

Source: left: Bai et al. (2013), right: literature review Siatwiinda et al. (in prep)

A critical level is defined as the soil test level below which crop yield response to additional nutrients is expected, and above which crop yield do not respond anymore to nutrient additions (Voss, 1998). For example, (Bai et al., 2013 using long-term experiments) assessed critical Olsen-P value for maize, wheat, and rice, of 18, 14 and 11 mg kg^{-1}, respectively, based on relationships between crop yield and soil Olsen-P (Figure 3-4 Left). Overall, critical Olsen-P values ranged mostly from 7 mg to 18 mg kg^{-1}. Critical limits significantly vary, depending on the type of analysis (extraction method), crop type and soil properties, and thus must be experimentally derived. The impact of extraction method is illustrated in Figure 3-4 right.

Critical limits for dissolved P and soil P saturation index in view of water quality

Thresholds for P in the soil are also important to protect surface- and groundwaters from eutrophication. For that purpose, the indicator “P saturation index” has been introduced above, i.e. the ratio:

$P_{ox}/(Fe + Al)_{ox}$

The critical **P saturation index (PSI)** ranges mostly around 0.15, i.e. 15% (12.5-17.5%) of the concentration of $(Al+Fe)_{ox}$, based on data for the Netherlands (Schoumans and Chardon, 2015) and Canada (Beauchemin and Simard, 1999). Commonly, the critical value is mostly expressed as 25%-35% of a P sorption capacity, which in turn is calculated as 0.5 x $(Al+Fe)_{ox}$ for sandy soils and non-calcareous clay soils. The critical PSI can be related to a **critical value for P in soil water (Pw)** according to (Chardon, 1994):

$Pw = 481 * PSI^{1.433}$

with $PSI = P_{ox}/(Fe_{ox}+Al_{ox})$

Using a PSI of 0.15 would lead to a critical Pw level near 20 mg P l^{-1}, which is slightly higher than the agronomic optimum Pw level for crop yield found by Jungk et al. (1993) (near 10 mg P l^{-1}), it is lower than an agronomic optimum Pw near 35, as suggested by Ehlert et al. (2004) in the Netherlands. For some aquatic systems, it has also been found that relatively low soil P levels may lead to P runoff that exceed a critical threshold for P in surface water (Hart and Quin, 2004). There is thus a potential overlap between optimal P levels for crop yield and critical p levels for water quality, and this should be kept in mind when using agronomic optimum P levels, as threshold for P fertilization.

Critical limits for N/P ratio in the organic layer of forest soils

In principle, there are no critical limits for soil P status indicators or for N/P ratios with respect to impacts of growth or nutritional quality of forests. Instead, data are given on the P concentration and N/P ratios in foliage (needles and leaves) that are indicative for P limitation (P concentration) or imbalanced growth (N/P ratios). Critical N:P ratios vary strongly between coniferous and deciduous tree species, i.e. for conifers, an N:P < 12:1 indicates N limitation and a N:P >18:1 P limitation, while for deciduous trees an N:P < 17:1 indicates N limitation and N:P >25:1 P limitation (Mellert and Gottlein, 2012). One could use these values as indictors for the N/P in the organic layer since that layer rather reflects the N/P ratio in foliage, so:

N/P ratio in organic layer > 18 (coniferous forests)
N/P ratio in organic layer > 25 (deciduous forests)

Most likely the values should be lower since N in foliage is retained before litterfall. Data of 150 Dutch sandy soils show for example that 95% of the sites had a N/P ratio above 18, indicating P limitation in almost all coniferous sites, whereas more than 60% had a N/P ratio above 25, indicating P limitation for at least 60% of those sites (De Vries and Leeters. 2001).

4 Soil acidification

Soil acidification occurs when pH decreases. This can be caused by acidic precipitation of sulphur dioxide, ammonia, and nitric acid, and has historically affected both forest and agricultural soils. Nowadays, the most important effect is observed on agricultural land through the application of ammonium-based fertilizers and urea, especially on naturally acidic soils such as sandy soils. This is because ammonium nitrogen is readily converted to nitrate and hydrogen ions, and its presence decreases the availability of plant nutrients, such as phosphorus and molybdenum, but also base cations; it increases the availability of elements such as aluminium and manganese, sometimes even to toxic levels. As a consequence, crop yields decline, and in severe cases, clay minerals become dissolved and the soil's cation exchange capacity is reduced, which then leads to structural deterioration. Soil acidification is counteracted by liming.

Table 4-1: Relationship of soil acidification to key societal needs and soil functions

Soil nutrient status

Societal need	Soil service	impact
Biomass	Wood & fibre production	-
	Growth of crops	-
Water	Filtering of contaminants	-
	Water storage	-
Climate	Carbon storage	+/-
Biodiversity	Habitat for plants, insects, microbes, fungi	-
Infrastructure	Platform for infrastructure	indiff
	Storage of geological material	indiff

4.1 Rationale: impacts of soil acidification on soil fertility and crop growth

Nitrogen generally has a positive effect on the quality of agricultural soils because it enhances soil fertility and conditions for crop growth. However, the overuse of N fertilizer can also lead to significant cropland acidification, reflected by pH decline (Guo et al., 2010), unless soils are properly managed (e.g. limed). In slightly acidic soils (4.5<pH<7.0), base cation nutrients, i.e. calcium (Ca), magnesium (Mg) and potassium (K), adsorbed on soil organic matter and clay, are crucial in buffering produced protons by elevated nitrogen inputs (De Vries et al., 2015). During acidification, these base cations are replaced by protons and subsequently leached from the rooted zone, accompanied with nitrate (De Vries et al., 1989; Lucas et al., 2011), which decrease their availability. This is an adverse effect since this loss of base cations implies a loss of the acid neutralization capacity and it may affect plant growth at low base saturation (being the ratio of adsorbed base cations on clay and organic matter as compared to the so-called cation exchange capacity).

In forest soils, the link between acid deposition and changes in soil and soil solution chemistry is well documented. In calcareous soils, the input of acidifying compounds (N and S) will not change soil pH until almost all the calcium carbonate has been depleted. In these soils protons (H+) are buffered by the dissolution of bicarbonate (HCO3-) and calcium (Ca2+) from calcium carbonate, with HCO3- and Ca2+ ions leaching from the system, while the pH remains the same. In non-calcareous soils, buffering is taken over by weathering of silicate minerals and by cation exchange processes of the soil adsorption complexes. In these soils, protons are exchanged for calcium (Ca2+), magnesium (Mg2+) and potassium (K+) and these cations are leached from the soil together with anions (mostly nitrate or sulphate). Subsequent leaching of Ca2+, Mg2+ and K+ leads to loss of the soil's buffering capacity by base cations and to nutrient imbalances for plant growth. Because of the restricted capacity of this buffering system, soil pH will decrease. In many forested catchments, it has been shown that acid deposition has caused prolonged export of base cations, such as Ca2+ and Mg2+, from forest soils, resulting in base cation nutrient depletion (Akselsson et al., 2007; Sverdrup et al., 2006; Watmough et al., 2005). When the soil pH drops below 4.5, the acid input is also buffered by aluminium (Al) release, causing Al toxicity. Significant correlations between S and N deposition and enhanced concentrations of Al3+ in soil solutions have been demonstrated in acidic forest soils in Europe (De Vries et al., 2003). This has led to liming campaigns of forest soils, which has – in combinations with decreasing acid deposition - then significantly increased soil pH.

With decreasing pH, there is thus generally a more limited availability of base cation nutrients (from leaching, together with nitrate and sulphate), such as calcium, magnesium, potassium, and elevated concentrations of toxic elements, such as aluminium, manganese and heavy metals, which can restrict plant and soil biota growth due to nutrient deficiency and metal toxicity (Kochian et al., 2004; Rengel, 1992; Wang et al., 2007). In addition, pH can also affect the availability of phosphorus (P). There are indications that the soil can be limiting crop growth in acidic soils (Baquy et al., 2017; Lucas and Davis, 1961). Al toxicity is a major constraint for crop production in highly acidic soils (pH<4.5) by damaging and stunting root systems (Delhaize and Ryan, 1995; Kochian et al., 2015) and potentially decreasing the availability of phosphate by formation of Al-P precipitates (Hinsinger, 2001).

4.2 Indicators for acidity status of soils

There are various indicators for soil acidification, including pH, base saturation, Al concentration and the ratio of Al to base cations (De Vries et al., 2015). In agricultural soils, pH, and related base saturation, is the indicator that is used to assess the soil acidity status and the related need for liming. Dissolved Al concentrations or the ratio of Al to base cations are never used as indicators since Al release happens at pH values below 4.5 and a base saturation below 25% considered as (far) too low for agricultural soils, since crop yield is clearly affected below such values (see 4.3).

Ulrich and co-workers (e.g. Ulrich & Matzner, 1983) were among the first who postulated that increased Al concentrations, specifically inorganic Al, and elevated Al/Ca ratios in soil solution are a major cause of forest dieback, by damaging the root system of tree species. Effects of high concentrations of Al on trees were tested with seedlings, either grown in water cultures, pot trials or in a greenhouse, mainly carried out in 1980s (for overviews, see Rengel (1992) and Kinraide (2003). Hypothesized mechanisms of Al toxicity include hampered root growth and inhibition of uptake of nutrients (Matzner & Murach, 1995; Schulze, 1989; Sverdrup et al., 1990 and 1992; Sverdrup & Warfvinge, 1993; Warfvinge et al., 1993). Furthermore, several authors (e.g. Roelofs et al., 1985) showed that release of Al by soil acidification and imbalances of ammonium to base cations, due to excessive N inputs and reduced nitrification, may cause nutrient deficiencies, which may be aggravated by a loss of mycorrhiza or root damage. This coincided temporally with field observations and foliage analyses where deficiencies of Mg and K caused yellowing of needles of Norway spruce (Zöttl & Mies, 1983). In the eighties, several authors (for example Hutchinson et al., 1986; Ulrich & Pankrath, 1983) considered soil acidification, especially the increase of the concentration of Al^{3+} in soil solution, responsible for forest decline, since Al^{3+} is very likely to be toxic to plant roots (Cronan & Grigal, 1995; Marschner, 1990; Mengel, 1991; Sverdrup & Warfvinge, 1992). The risk of Al^{3+} for forest health in the field is considered lower but the adverse impact of Al^{3+} on root functioning is an established fact, at least under laboratory conditions.

In forest soils, critical levels for Al concentrations and for the Al/BC ratio have been derived and an overview of these levels is given in e.g. De Vries et al (2015). However, the standard indicator for soil acidity is the pH level, being the indicator used in this study.

4.3 Critical limits

Critical pH levels for agricultural (crop) land

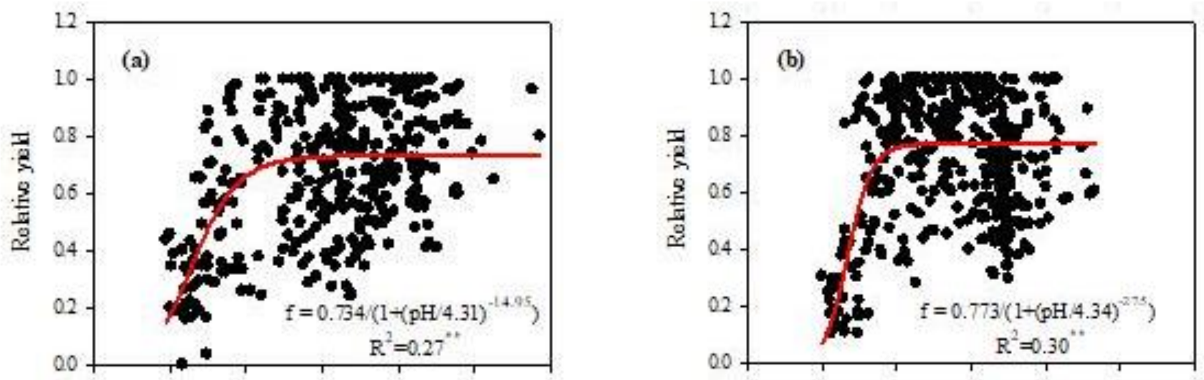
To avoid losses in crop production and environmental impacts, in terms of enhanced metal uptake and metal leaching, soil pH should stay above a critical level below which crop yield is limited, being generally a level, in which also soil metal availability is limited. As with P, the critical pH level can be derived by:

- Short-term manipulation experiments in which the soil is manipulated to pH values by adding H+ or OH- and then a crop is grown on it (pot experiments in a laboratory or greenhouse). Advantage is that soil pH being the only variance while all other circumstances, such as soil type, temperature, water availability, nutrient availability, etc. were kept equal. Disadvantage is the different environment conditions between laboratory and field responses. Also, an adjusted soil pH by add acid or alkali can strongly affect the microorganism community, as well as the nutrients availability and the biomass accumulation.

- Long-term field experiments on the impacts of declining soil pH on plant growth/crop yield. Advantage is that the impacts are derived field conditions, but the disadvantage is that other (confounding) factors may change in time, including changes in climatic variables and the occurrence of pests and diseases, requiring careful consideration of the data.

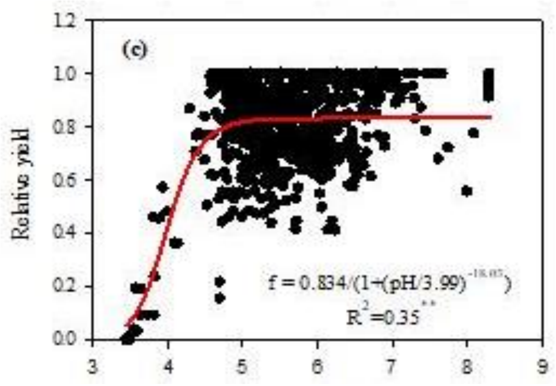
An example of results thus obtained is given in Figure 4-1. A significant non-linear relationship was found between soil pH and relative crop yield, defined as a fraction of the maximum crop yield without acidification impacts, both in short-term manipulated experiments (STE) and long-term experiments (LTE) for wheat, maize and rice. In STE, the critical pH values, related to an expected yield loss of 5%, i.e. a crop yield that equals 95% of the maximum yield, were comparable (4.5 - 4.7) for all three cereal crops (Table 4-2), being close to the pH value of 4.5 at which aluminium release starts to occur.

Figure 4-1: S-functional relationships of soil pH impacts on the relative yield of wheat (a), maize (b) and rice (c) using combined short-term pH manipulation experiments and long-term observations



Soil pH

soil pH



Soil pH

Note: ** denotes P<0.01, indicating a highly significant non-linear relationship.
Source: Zhu et al.; 2020

Table 4-2: Summarized critical pH values of wheat, maize and rice derived from short-term manipulation experiments (STE) and long-term observations (LTE)

Crops	pH at 95% yield		
	STE	LTE	STE+LTE
Wheat	4.5	5.9	5.3
Maize	4.6	5.1	4.8
Rice	4.7	5.0	4.7

Note: Based on Zhu et al.(2020)

Various studies indicate that crop production is already restrained at pH values below 5.5-6.0 due to limited availability of Ca, Mg, K and P (Holland et al., 2019; Walker et al., 2011). The results at least indicate that a pH value below 5 should for sure be avoided while 4.5 is really critical in view of Al toxicity. Preferably, the pH stay above 5.5 or even 6.

4.4 Critical levels of dissolved free aluminium and the molar base cation/aluminium ratio in forest soils

Free aluminium concentration of 2 mg l^{-1}

The sensitivity of a tree to Al varies as a function of solution pH, Al speciation, Ca concentration, overall ionic strength, the form of inorganic N (NH4 or NO3), mycorrhiza interactions, soil moisture etc. Consequently, a wide range of Al toxicity thresholds for various tree species has been reported in the literature, varying between less than 1.5 and more than 30 mg l–1 (e.g. Cronan et al., 1989; Joslin & Wolfe, 1988, 1989; Keltjens & van Loenen, 1989; McCormick & Steiner, 1978; Ryan et al., 1986a, b; Smit et al., 1987; Steiner et al., 1980; Thornton et al., 1987). The sensitivity increases from red spruce, with significant biomass reductions starting to occur near 2 mg l–1 of inorganic Al, to Douglas fir, spruce and European beech, whereas Scots pine, oak and birch are relatively insensitive to Al (Cronan et al., 1989).

Molar base cation/aluminium ratio of 1 (0.5-2.0)

Results in a variety of laboratory experiments described above showed that the Ca/Al ratio was a better indicator for root impacts than inorganic Al (Cronan & Grigal, 1995; Sverdrup & Warfvinge, 1993; Sverdrup et al., 1992). As with Al, a wide range in toxicity thresholds for the Al/Ca ratio has been reported. Sverdrup and Warfvinge (1993) carried out a systematic review of impacts of Al on the growth of tree seedlings and plants in laboratory experiments, based on approximately 200 studies. The response in acid soils, as expressed by root growth, stem growth or plant growth in experiments, has been determined for different species of coniferous and deciduous trees. Studies showed that the plant response can be described better as a function of the base cation and Al concentration in soil solution than just as a function of Al alone or a Ca/Al ratio. The critical limit was most conveniently expressed as a molar Bc/Al_{tot} ratio, with Altot being the total (inorganic and organically complexed) Al concentration and Bc denoting Ca+Mg+K. In many calculations of critical loads of acid deposition to forest ecosystems, either a general limit value of 1 is used for Bc/Al, or a tree species specific value, ranging mostly between 0.5-2.0.

The relevance of laboratory experiments addressing Al toxicity under field conditions has been disputed (Binkley & Hogberg, 1997; De Wit et al., 2001; Kreutzer, 1995; Løkke et al., 1996). Indeed, healthy trees have been found at sites where high soil solution Al concentrations were measured (Huber et al., 2004), while nutrient deficiency symptoms in trees have been found at other sites with similar conditions (Alewell et al., 2000). In addition, whole-ecosystem experiments, designed to test effects of acid deposition on forests (Abrahamsen et al., 1993; Beier et al., 1998; Huber et al., 2004; Kreutzer & Weiss, 1998), have been inconclusive with respect to Al-toxicity effects on root growth and nutrient uptake. Despite this criticism, the above-mentioned critical values are still often used in risk assessment (De Vries et al., 2015).

5 Contaminants in soil

Contamination of soil is one of the pressing concerns about human health and food quality, as well as ecosystem condition and biodiversity. Sustainable land management needs to address the corresponding risks, and this requires reliable information about the accumulation of contaminants to soil to a level, at which risk-based thresholds may be exceeded; beyond that level, soils, and the ecosystem they are part of, do not properly function anymore. To derive meaningful thresholds for contaminants in soil, the entire chain from loads to soil and losses from soil as well as biogeochemical filter and transformation processes need to be considered. This chapter provides an overview of some of the existing approaches for setting such thresholds at European level, highlighting the remaining challenges, such as the lack of harmonized terminology as well as monitoring methods and risk levels. The latter calls for an accurate and updated compilation of current approaches and values set at the national and regional level in the EU.

The aim of this chapter is to provide a more consistent view about soil contamination in view of relevant environmental risks and risks to human health. This includes, but is not limited to, the impact of pollution by metals. At present, the concept of the risk-based land Management has been elaborated predominantly for metals and metalloids. Depending on the balance between inputs and outputs of contaminants to soil, accumulation can occur. Depending on the rate of accumulation, contaminant concentrations in soil may, or may not, exceed risk-based thresholds (which can be specific for soil type, crop type, climatic conditions, and parent material). A key aspect in this respect is the degree to which pollutants in soil are bioavailable for plants or organisms, that is, whether or not contaminants can be taken up by roots, permeate cell membranes or are translocated to deeper soil layers and/or ground- and surface water systems.

Table 5-1: Relation of soil pollution with other threats and main soil services affected by pollution

Soil erosion		
Societal need	**Soil service**	**impact**
Biomass	Wood & fibre production	-
	Growth of crops	-
Water	Filtering of contaminants	-
	Water storage	-
Climate	Carbon storage	+/-
Biodiversity	Habitat for plants. insects. microbes. funghi	-
Infrastructure	Platform for infrastructure	indifferent
	Storage of relocated material or artefacts (excavated geological material, sediments, cables and pipelines, archaeological material)	indifferent

5.1 Methodical aspects to detect and treat soil pollution

5.1.1 Protection targets

Soil pollution can be defined as the presence of contaminants in the upper (unsaturated) soil layer, including the root zone, and in groundwater in excess of levels deemed acceptable in view of risks. It affects human health and degrades the functionality of soils by affecting some of its major functions, such as hosting biological diversity, production of food, and the filter function to protect water bodies. Soil pollution as such is one of the relevant threats known to affect soil quality.

Clearly several relevant links or interactions exist between soil threats, in particular with soil biodiversity and carbon storage. For example, the decline in soil organic matter, acknowledged as a serious threat notably in agricultural soil, can aggravate the impact of soil pollution since effects the active micro-surface and substrate for biochemical soil processes affecting storage, breakdown and release of contaminants. Nevertheless, this chapter will focus largely on how risks of contaminants in soil can be quantified in relation to soil functions. Here, especially relevant soil functions are the production of sufficient (quantity) and safe (quality) food and fodder crops as well as animal products, filter function to preserve water quality, carbon storage and habitat function for soil life (including the health of animal feeding from soil such (i.e., cattle, horses, sheep etc) are prime functions to be considered.

If present in excess levels, soil pollution can lead to health effects and decrease the activity of soil organisms and affect its functional composition. Crop performance, micro-organisms and enzyme activity in the soil are proven to be negatively affected in contaminated soil, on the long-term leading to a decline of aggregate stability and affecting the decomposition of organic matter in soils (Stolte et al., 2016). This in turn can aggravate erosion. However, even more importantly, substances entering the root zone may accumulate in food chains. Moreover, mobile substances leaching into the unsaturated groundwater zone can threat drinking water resources, often after years, decades or even centuries.

The goal of soil protection regulations and procedures is to protect human health, the environment, agricultural production, and groundwater resources. In Table 5-3, three main groups of environmental compartments and endpoints that need to be protected are summarized: arable cropping systems (targeting food and feed quality as well as animal health), the soil ecosystem (targeting life support functions and biodiversity), and protection of water bodies, including both groundwater and surface water bodies (targeting human health and aquatic ecosystem functioning).

Table 5-2: Overview of relevant entities to be protected and critical limits in relevant endpoints

Relevant compartment	Protection target/ endpoint	Assessment criterion	Regulation addressing the issue
Arable, pasture and allotment soils	Food quality for human consumption	Food quality standards and toxicological limit values	EU, WHO, FAO, national regulations
	Fodder quality for animal feed	Feed quality standards	EU
	Animal health	Toxicological limit values	Recommended levels
	Animal products	Food quality standards	EU
(Urban) soils	Human health	Tolerable daily TDI) or access cancer risk	National regulations
Soil Ecosystem	Ecosystem health	PAF[43]	National regulations
Adjacent ground- and surface water Systems	Ecosystem health	PAF	EU, national
	Drinking Water quality	Drinking water standard	EU/national

5.1.2 Terminology important for soil pollution

As long as thresholds are defined differently, applications in different risk assessment systems and planning instruments are not comparable across Europe. This section provides the definitions and explanations of some key terms related to thresholds:

43 Potentially Affected Fraction (PAF) of species and ecological processes

Table 5-3: Terminology important for soil pollution

Term	Definition
Background level	Level of contaminants in soil that can be found without human interference. Heavy metals for example are present in almost all soils as part of the soil matrix composed of clay minerals, oxides and/or organic matter. Clearly, the level at which metals occur can vary and depends among others on the rock type from which the soil developed. For a large number of man-made organic contaminants, background values are zero since they are not part of any soil forming mineral (e.g. microplastics, PFAS, most PAHs or dioxins).
Protection target (endpoint)	Here we refer to endpoints as the entity to be protected. This can refer to a water body to be used by human beings, or human beings themselves when considering exposure to for example polluted soils in an urban setting. Common endpoints considered here include arable (food or fodder) crops, animals, water bodies, terrestrial ecosystems as represented by a number of key species, or humans themselves.
Critical limit	These refer to limit values of specific contaminants in endpoints not to be exceeded. Examples include water quality guidelines in place for drinking water or ecological thresholds to protect aquatic organisms in surface water bodies. Usually, such limits or thresholds are set at EU level or, in case of WHO standards, world-scale. Such critical limits or thresholds therefore do not refer to contaminant levels in soil. To convert critical limits in endpoints to corresponding screening levels in soil, transfer models are required (see below)
Risk limit	A critical concentration in soil or groundwater, related to a specific protection target, *without a formal position in legislation*. Risk limits are often derived as basis for thresholds (the latter may refer to, or be a part of, a legal framework)
Screening value (SV)	Screening values are levels of contaminants *in soil* at which the critical limit or threshold in endpoints would be exceeded. These screening values therefore depend on the function considered and furthermore depend on the soil type that is considered if the pathway between critical limit in the endpoint and the corresponding concentration in soil is affected by one or more soil properties (e.g., soil pH that affects the transfer of most metals from soil to crops). Depending on the desired degree of protection, screening levels can be defined at different levels ranging from low to medium (acceptable risk, no immediate action required) to high levels (beyond which the risk is deemed unacceptable and further research or soil remediation would be required). Screening values are, in contrast to risk limits, part of a legislative framework (however, there may be differences among member states)
Transfer models	In order to convert critical limits in endpoints to corresponding risk limits or screening values in soil, transfer models are needed. Examples include soil to crop models that are able to predict concentrations in crops based on the corresponding level in the soil; this transfer depends on relevant soil properties such as acidity (pH) and or organic matter. Other relevant pathway models are those used to predict the solution concentration of chemicals (nutrients, organic contaminants, and metals alike), influenced by specific soil properties. In case of human exposure, all relevant transfer pathways towards the human endpoint are to be considered (inhalation, intake via water and food which in turn requires the aforementioned soil-to-crop models or soil-to-solution prediction models).

5.1.3 Characteristics of diffuse pollution and point source pollution

Land affected by diffuse or point source pollution suffers from the wide-spread application and distribution of contaminants (see Figure 5-1). **Diffuse pollution** originates from a range of sources including atmospheric deposition (from industry and traffic mostly) and **agricultural soil management.** It is usually affecting larger areas and is characterized by a relatively homogeneous contamination pattern. In some cases, the link between the source of pollution and its destination is not clear e.g. in case of **atmospheric**

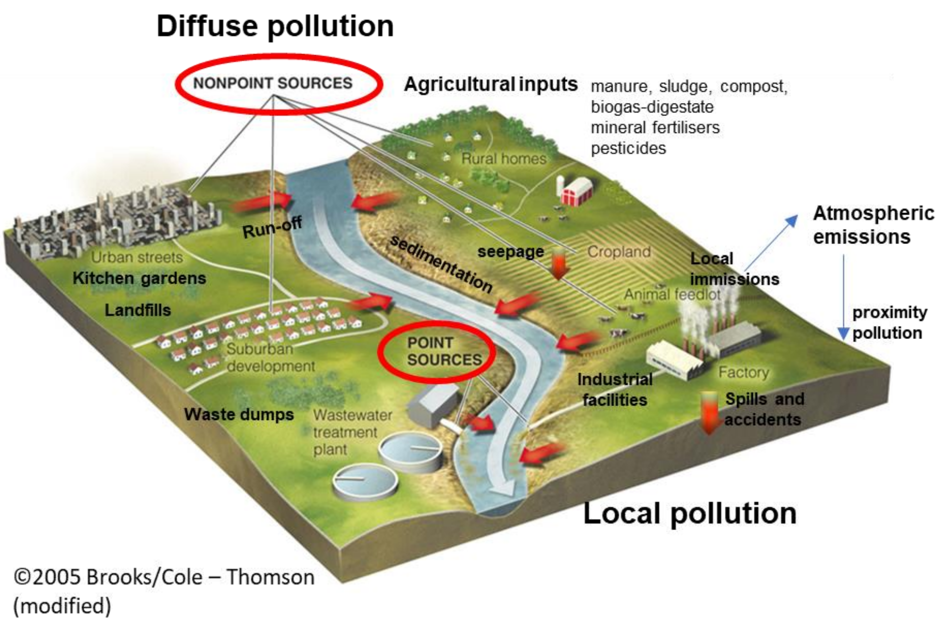
deposition which represents the sum of multiple sources. Also, former or ongoing, deposition of polluted sediments in river **floodplain soils** is a form of diffuse pollution.

A specific type of diffuse pollution is called **proximity pollution** which is a wide-spread form of diffuse pollution originating from a single industrial source, outside the property boundaries of the industry (Van Camp et al. 2004). A typical example of proximity pollution is the regional impact of smelters of non-ferro metals in areas like the Belgian-Dutch border zone de Kempen. Typically, soils are characterized by elevated levels of, in case of the Kempen, cadmium and zinc in areas up to 30 or 40 km away from the smelter.

In case of arable soils, **diffuse pollution** is, for contaminants like cadmium, copper and zinc, but also emerging contaminants like animal medicinal products or microplastics, most often the prime type of contamination. Dominant sources of polluting substances are atmospheric deposition as well as management-related inputs of exogenous organic matter (manure, sludge, compost and biogas-digestates), mineral fertilisers and plant protection products. Usually, input rates of these products are higher in arable cropping systems compared to extensively managed forms of land use like forests or pasture land used for extensive grazing.

In contrast to diffuse pollution, **point source pollution** (also called: local pollution) is at a smaller scale and is characterized by a heterogeneous contaminant pattern. Point source pollution is often caused by anthropogenic activities, e.g., industrial activities, storage an application of waste materials, leaking reservoirs, spills, or calamities. Since in all EU Member States pollution prevention is a key issue and in most EU Member States the precaution principle is leading, a large part of the cases of point source solution is from a historic nature.

Figure 5-1: Forms of pollution and its impact on the environment



Source: Brooks Cole Publishing 2005, modified

In Table 5-4, the major characteristics for the assessment of soil quality for diffuse and point source pollution is listed.

Table 5-4: Main characteristics to assess diffuse and point source soil pollution

	Diffuse pollution*	Point source pollution
Procedure framework	risk limits and screening values for soil, risk limits in groundwater and agricultural products	Tiers approaches, combining thresholds for soil and groundwater and site-specific risk assessment
Availability thresholds	For pesticides, nutrients and several metals	for metals/metalloids, PAH, aromatic compounds (including BTEX), VOClc, mineral oil, asbestos, and others
Source	active (e.g agriculture/air in case of (pesticides, nutrients, metals) or historic (e.g. in case of floodplains soils, PAHs, metals)	in most countries source from the past ('historic pollution')
Policy	source regulation via reduction of inputs and, in specific cases, accumulation or the lack thereof, e.g. in case of application of sludge	remediation/ soil quality management to contain or reduce risk (including restriction of land use)
	EU-wide limit values in place (e.g. in fertilisers, sludge). No EU-wide corresponding soil screening values or framework to derive those	No common European regulation (except for WFD – see Table 1-6)
Assessment criteria	background concentrations and risk-based values (risk limits or sometimes screening values)	primarily risk-based values, sometimes also background concentrations
Protection targets	crops, cattle, humans, ground- and surface water ecosystem (after leaching)	primarily human health, soil ecosystem and drinking water resources (after long-term leaching); moreover crops, surface water, wildlife

5.1.4 Relevant groups of contaminants found in soil

Several soil contaminants such as most metals are naturally found in soils but are found at increased levels due to anthropogenic activities. Other contaminants are synthesized and brought into soils by a range of human activities. The types of contaminants found in soil and groundwater is described hereunder.

Metals and metalloids

For some metals and metalloids, notably lead, mercury and cadmium, policy measures have been or are being enforced, and in consequence, inputs to arable systems seem to have decreased. For lead, emissions via air originating from fuel burning in Europe decreased by about 85% in the last 20 years of the last century (Lorenz et al., 2010). Due to the immobile nature of lead in soil and former applications of lead containing waste materials for soil elevation purposes, lead is often found in urban soils, sometimes at high levels. Children are especially sensitive to lead exposure due to hand – mouth contact; high exposure can impact their neurological development and lower the intelligence quotient (Lanphear et al., 2005; JECFA, 2011). Potential inputs via other agriculture-related sources such as sludges, manure or fertilisers, are at least partially legally regulated; there, limits are set for the allowed content of metals in fertilising products or soil improvers (a.o. EU fertilizer regulation 2019/1009; Sewage Sludge Directive 86/278/EEC).

Cadmium is often a problem in the soils of vegetable gardens located in or near urban areas, in particular in combination with fast growing green crops such as spinach and endive. It is grouped as a priority hazardous substance in the Environmental Quality Standards Directive (EU, 2008), and thus belongs to one of the most toxic environmental chemicals. Moreover, arsenic is often found at high levels in commercially available crops (EFSA 2009; WHO 2011; EFSA 2014). The following metals are regarded as essential for human health: iron, zinc, copper, chromium, cobalt, molybdenum, manganese, and selenium (WHO, 1996; Becking et al. 2007). However, at elevated concentrations these metals might become toxic for humans.

Organic contaminants in soils

- Plant protection chemicals
  Plant protection products (PPPs) are largely introduced from agriculture practices, where they are directly applied during the growing season. Consequently, a series of PPS, mainly herbicides, can be found find at high levels in soil and groundwater. Although in most European countries, the application of PPPs is regulated (type of chemical to be used as well as application thereof, Regulation (EC) No 1107/2009), screening values are still exceeded at a substantial scale, notably in surface water bodies as well as shallow groundwater bodies that are in close contact to the receiving soils. Especially in countries with shallow groundwater tables like the Netherlands, a large number of PPPs is found in most groundwater abstraction wells.

- Other organic contaminants
  Except for plant protection chemicals, a range of organic contaminants is found in soil and groundwater. In contrast to plant protection chemicals, these substances are usually not introduced actively by farmers but end up in soil due to emissions elsewhere. An exception is the application of sewage sludge, which introduces a series of organic compounds into soils, including PFAS.
  Chemicals most commonly found in soils include persistent organic pollutants (POPs), which are chemical substances, persistent in the environment, and which can bioaccumulate in the food chain. They can be naturally occurring (e.g. polycyclic aromatic hydrocarbons (PAHs), or be derived from industrial processes (e.g. polychlorinated biphenyls (PCBs), or organochlorine pesticides such as DDT, dieldrin, and hexachlorobenzene HCB). Polycyclic Aromatic Hydrocarbons (PAHs) form another important, wide-spread group of soil contaminants.

  PAHs are primarily formed by incomplete combustion of carbon-containing fuels such as wood, coal, diesel, fat, tobacco, or incense and are concentrated in oil, tar and coal. Common PAHs in soil are naphthalene, phenanthrene, anthracene, fluoranthene, benzo[a]anthracene, benzo[k]fluoranthene indeno[1,2,3-cd]pyrene, benzo[g,h,i]perylene and benzo[a]pyrene. PAHs in soils might show point source or diffuse (due to atmospheric deposition) contamination patterns. Some PAH representatives are known or suspected to be carcinogenic, mutagenic, or teratogenic. Also, **monocyclic aromatic hydrocarbons** are frequently found in urban soils and groundwater. The representatives most often found are usually categorized as **BTEX** (benzene, toluene, ethylbenzene and xylenes). They were, or are, used on a large scale in cleaning applications such as degreasing. Another important group of organic contaminants often found in urban soils are **volatile organic compounds** (VOC), including trichloroethylene, tetrachloroethylene, 1,1,1- trichloroethane and vinyl chloride. They enter soils through several industrial activities, including dry cleaning. Most VOCs are readily soluble in fat. VOC compounds are generally volatile and mobile.

Emerging substances

Currently, there is concern about emerging chemical substances in soils. These are substances not previously considered or known to be significant in the environment and may have no regulatory standard. The Norman network currently lists 860 substances, of which some are prioritized, forming the basis for the first EU watch list on emerging contaminants, most of them organic (Commission Implementing Decision (EU)2015/49). Emerging substances include Per- and Polyfluoroalkyl substances (PFAS), nanoparticles, antibiotics, and other medicinal products like anthelmintics. PFAS include more than 4,700 different substances (OECD, 2013), which are of very high concern (SVHC) because of their high environmental persistence and toxicity. Soil and groundwater contamination with PFAS has become evident in Europe. Among others, contaminated sewage sludges used as organic fertiliser have caused

PFAS pollution of soil (Ghisi et al. 2019). As a first step to monitor its accumulation, background concentrations for mobile forms of PFAS have been determined for perfluorooctanoic acid (PFOA) or perfluorooctane sulphonic acid (PFOS). A limit value of 0.1 µg/L for each individual PFAS in drinking water has been introduced in the EU (2020).

At present, research is still ongoing about the toxicity of many of these emerging compounds. However, the derivation of toxic limits is hampered by the lack of analytical techniques and other new emerging contaminants at environmentally relevant levels (e.g., in case of microplastics or nanoparticles once present in soil). Also, the conversion of medicinal products to secondary products with different properties and corresponding toxicity and their interactions with the soil matrix (mixing effects and interactions with co-contaminants) pose challenges to address.

Contaminant mixtures

Soil and groundwater quality assessment is generally approached from a single contaminant perspective. However, in most cases different contaminants are found in soil and groundwater. As a consequence, humans and organisms generally are exposed to more than one contaminant at the same time. For contaminants with the same toxicological endpoint (e.g., target organ) that act through a common mode of action, dose addition is appropriate when assessing human health risks. If contaminants have the same endpoint, but act through a different mode of action, **response addition** applies (Swartjes and Cornelis, 2011). The effect of combined exposure of organisms can be assessed using the multiple PAF procedure accounting for the **multi-substance Potentially Affected Fraction** (msPAF) (Posthuma and Suter, 2011). Moreover, multiple contaminants may interact and alter their bioavailability, depending on soil properties and ageing (degradation products and metabolites).

5.1.5 Mechanisms to trigger action for local and diffuse contamination

When risk-based screening values are exceeded, site specific risk assessment recommends soil remediation (or restoring and rehabilitative land use and management practises, aiming to enable further human use of the soils). This concept has predominantly been developed and established for heavily contaminated sites including brownfields, city soils used for playgrounds, or allotments where contact between soil and user is intensive and hence risk levels need to be reduced.

A risk-based approach does not a priori differentiate between soils affected by diffuse or point source pollution. When action is required, that is, when **screening values are exceeded**, management of pollution sources becomes relevant. But for the risk assessment as such (i.e. the evaluation of the current situation in a given location or area), it is not relevant whether a site was affected by diffuse or point sources pollution. However, contaminated sites that are in need of remediation or other action, are largely those who have been affected by point source pollution due to the higher impact of such sources on the quality of soils.

Clearly there is a distinction between soils affected by point source pollution (PSP) versus those that are affected by diffuse pollution (DP) which explains why, until now, most soil remediation actions are confined to PSP:

- Contamination levels observed in sites affected by PSP usually are such that action is imminent. Often, effects are obvious such as degraded soil surfaces, visual impact on vegetation (or the lack thereof) as is the case in for example many former mining areas.
- Contamination levels in PSP-affected soil often pose a direct threat to human health resulting from contamination of drinking water, heavily polluted dust particles blowing into nearby housing areas or transfer into the food chain if such soils are used for local crop production as is the case in or near city areas.

- Diffuse pollution on the other hand has not yet reached levels at which effects become immediately obvious. The rate of accumulation is - in most cases - far less compared to that of PSP and as of now there are few examples of areas where DP required action. Exceptions include for example areas affected by high industrial emission, formerly introduced as proximity pollution.
- By nature, DP affects large areas which would imply that possible measures (remediation, monitoring) affect large areas and, by definition, will be very costly. Examples from areas affected by proximity pollution such as the Belgian-Dutch border area of Kempen show that the development of a regional approach to deal with this can take decades and requires, in this specific case, harmonization of risk assessment approaches between member states.
- So far, DP has not created urgent or visible issues with for example food safety or animal health. This can be misleading though since the slow build-up of contaminants in soil, like for cadmium or lead, can result in a slow but steady increase in exposure to such contaminants. This however is often difficult to quantify since in most industrialised countries in the EU, food usually comes from a vast array of sources and few people depend on food grown in one place. Nevertheless, it was documented (Rietra et al., 2017) that there is a relationship between the average cadmium concentration in soils in the EU and the exposure to cadmium via food.
- Monitoring soil contamination prone to DP is very difficult or requires long (decades) monitoring intervals. This is mostly because of the low accumulation rate of metals in soil (see e.g. Römkens et al., 2018 for cadmium at EU scale), but also because of the high spatial variability (within a monitoring site). Very small changes in concentration levels over long time spans (5 or 10 years) need to be detected. At present, the assessment of trend for most contaminants is largely model-based.
- For many of the recently introduced contaminants of concern (e.g. medicines, PFAS, microplastics), DP can be a relevant source to large areas. At present, however, regional data and risk-based limits in soils are largely absent or are in need for validation; for some contaminants of concern, robust analytical techniques to measure actual levels in soil are still being developed, like those for nanoparticles or microplastics. Nevertheless, there is growing concern that if DP is to continue, issues with emerging contaminants can become critical within decades to come such as in case of microplastics (EU, 2018) or PFAS (EU, 2020b).

Also at EU level, the impact of diffuse pollution has been recognized as a potential issue. The new fertilizer regulation (EU2019/1009) and other policy proposals consider at least partially a risk-based approach with the aim to minimize long-term deterioration of soil quality. Examples for such proposals are end-of-waste criteria for materials like compost and digestate (Saveyn et al., 2014) and, more recently, also for upcoming materials like biochar, struvite, or ash (Huygens et al., 2019). These proposals, however, largely target the quality of inputs to soil rather than to the evaluation of soil quality with screening values.

Aside from the assessment of the status of an agricultural soil based on the actual concentration of contaminants, expected, likely effects on soil quality are also sometimes used as criterion for required actions. Basically, two policy driven approaches can be distinguished:

- *Future concentrations in soil should not exceed the defined screening value at any given point in time* (or a predefined time window like 100 years from now). Usually, risk-based limits are used to derive meaningful acceptable inputs to soil (e.g. in case of the Waste Directive referring to the use of sludge in agriculture (86/278/EEC). In some cases, also background concentrations can be used (except for lithogenic anomalies), even though this would inevitably lead to very strict acceptable loads to soil.
- *Avoidance of any accumulation of contaminants in soil* is an alternative approach currently under discussion (see also: **"stand-still" scenario**). Inputs to soil shall not exceed outputs including crop uptake and leaching, so as to maintain the current concentration of contaminants (or nutrients, like P). This approach is not risk-based in that the current level is considered the relevant criterion and not so much a level at which effects become unacceptable.

Maintaining soil metal levels at levels that pose no risk to human health and the ecosystem is to be preferred from an environmental point of view. Due to intensive land management including both industrial and agricultural activities from the early 1900's levels for metals like lead, cadmium or copper in arable soils frequently exceed background levels. This however does not imply that such soils are at risk despite the observed increase in the total contaminant level.

To avoid further loading of contaminant levels in soils it is necessary to strive towards a balance between inputs and outputs such that the net accumulation becomes zero. This approach, also called the stand-still principle thus aims to maintain current levels of contaminants in soil.

However, when considering inputs to intensively managed arable cropping systems for example, where inputs of metals like copper, zinc and cadmium exceed average inputs at regional or even national levels, a stand-still principle seems hard to achieve. An input-output balance would require serious reductions of the allowed amount of fertiliser or manure applied. This was shown already in 2004 for zinc (de Vries et al. 2004) and for copper (Groenenberg et al. 2006) at a national scale for the Netherlands. Recently, this spatially explicit approach was applied also at EU level for cadmium (Römkens et al., 2018).

For copper, zinc and cadmium, areas where accumulation occurs are related to soil properties with accumulation prevailing in near neutral, mostly clayey soils, whereas zinc and cadmium are largely lost from soil via leaching in acidic sandy soils. These studies reveal the importance of not only considering current concentrations in soil, but to combine this with a dynamic assessment on inputs to and outputs from soil. Such a dynamic assessment will reveal if, and if so at what time scale, screening values are to be exceeded.

Need to incorporate dynamic models to predict changes with time

At present, concentrations of most contaminants in arable, grassland and forest soils are such that relevant critical limits in food or (ground)water are not exceeded (with known exceptions of course like regional issues with cadmium in soils). At the same time there is a growing concern about long term changes and the impact on the ecosystem, water quality and food quality. To evaluate such potential changes, dynamic models such as the one already operational for cadmium (Römkens et al., 2019) are needed. This requires among others information on inputs to the system (atmospheric deposition, inputs related to agricultural use), and outputs from it (crop uptake, leaching). At present, the quality of integrated models to predict such changes over decades is still limited, even for cadmium (note the related high uncertainties to predict leaching of cadmium from soil), and especially for emerging contaminants: robust models as well as data about inputs to soil, and about the fate of substances in the soil-water continuum are needed.

5.2 Indicators for soil pollution

5.2.1 The indicator paradigm

The objective of an indicator on soil pollution is to make the soil and groundwater pollution status of a contaminated site, region, or country visible, either in numbers or as maps. In Table 5-5, examples are given of indicators for diffuse and point source contamination.

Table 5-5: Indicators for soil pollution

Diffuse contamination		Covered in this report
Inorganic contaminants	Critical heavy metal contents in excess of national thresholds	Chapter 5 (here)
	Critical load exceedance by heavy metals	

Nutrients and biocides	Area under organic farming	Land use (not dealt with here)
	Gross nutrient balance	Critical N and P limits (Chapter 3)
Persistent organic contaminants	Concentration of persistent organic pollutants (POPs)	Chapter 5 (here)
Soil acidifying substances	Topsoil pH	Chapter 4
	Critical load exceedance by sulphur and nitrogen	Critical N limits (Chapter 3)
Point source pollution		
Contamination by point sources	Progress in management of contaminated sites	Payá Pérez and Rodríguez Eugenio (2018)
	New settlement area established on previously developed land	*Land use (not dealt with here)*
	Status of site identification number	Payá Pérez and Rodríguez Eugenio (2018)

Source: ENVASSO project; Huber et al., 2008

- Indicators on point-source pollution
  Freudenschuss et al. (2001) have distinguished several subindicators (better: statistical parameters), including: soil polluting activity, number of contaminated sites, progress in the management od contaminated sites, expenditures on remediation, and groundwater incidents. Since then, the work has been taken further but the formerly EINOET ad-hoc Working Group Contaminated Sites (now: WG Soil Contamination) in the form of questionnaires related to the EEA Indicator LSI003 "Progress in the management of Contaminated sites", applying 6 site statuses representing some of the statistics mentioned above (for details see also Payá Pérez and Rodríguez Eugenio 2018). The indicator is now being updated based on the last questionnaire in 2016. Future updates may include polluting activities, dominant contaminants, and spatial reference to regional administrative borders (number of sites per polluting activity and site status per NUTS 3); the proposal is currently in discussion and will address issues raised by Van-Camp et al. (2004). There, the establishment of a European Point Source Assessment System (EPSAS) has been suggested. The development of such a register must be closely aligned with existing data collections about current industrial installations, reported to EEA's European Pollutant Release and Transfer Register (E-PRTR), and data collections under the Mercury Regulation.

- Indicators for soil contamination from diffuse sources
  The following indicators were suggested during several EIONET workshops (Freudenschuss et al. 2001):
  - Average pesticide consumption per unit area of agricultural land
  - Sewage sludge application per unit area of agricultural land
  - Exceedance of critical loads of heavy metal contents in soils related to different land uses
  - Heavy metal balance for agricultural soil
  - Important are also the SOC content and the presence of key soil fauna and organisms.

  A more extend rationale on these parameters and indicators related to diffuse pollution are found in Van-Camp et al. (2004). The suggest that the following metals and nutrients could be realistically monitored, recommending 5-10 intervals:
  - Heavy metals (Cadmium, Copper, Lead, Zinc, Mercury, Arsenic, Nickel and Chromium);
  - Nutrients (nitrogen and phosphates).

These recommendations were then evaluated and synthesized by Huber et al. (2008), as a suggestion for a European soil monitoring system (Table 5-5). Due to the lack of soil data, and particularly due to the lack of a European political incentive, the definition of an indicator on diffuse soil pollution was until now never realized in the EEA or any other soil indicator system. The lack of soil data about heavy metals seems to prevail in many countries (Bünemann et al. 2018), while progress has been achieved with the LUCAS Soil Survey (starting with the 2009/2012 samplings) (Toth et al. 2013; Ballabio et al. 2018).

Due to the continued discussion in the EIONET WG Soil Contamination and other networks, a **soil pollution indicator on diffuse pollution on metals (exceedance of (national) screening values for heavy metals)** seems realistic. However, **important is agreement about common criteria for the definition of thresholds.** Any such an agreement must be based on a common terminology and definitions. The indicator would require broadly valid thresholds, possibly stratified at European level for different soil characteristics and land uses. It must be recognized that because of large differences in soil and climate across EU Member States, the validity of such thresholds would be still limited. For example, soil thresholds aimed at the protection of crop quality for e.g. cadmium, are far more strict in acid soils low in organic matter common in NW parts of the EU compared to those valid for calcareous clay soils in the Mediterranean areas. Agreement on or harmonization of the *approach* and underlying assumptions on how to derive meaningful critical limits in soil therefore seems the relevant issue to accompany this indicator with the objective to derive EU wide generic critical limits in soil.

5.2.2 Methodical references

Different stages are recognised in soil and groundwater sampling:

- analyses of pollution pattern;
- development of a sampling protocol;
- sample conservation;
- sample analyses in the laboratory;
- data interpretation.

The analyses of pollution pattern and the development of a sampling protocol are different for diffuse and point source polluted sites. Since diffuse soil pollution is characterized by a homogeneous contamination pattern, a limited number of samples and analyses of composite samples is appropriate. For point source polluted sites, several options are available for sampling, depending on the contaminant pattern. In the Netherlands, as an example, a ppreliminary, exploratory and main investigation are used (Lamé, 2011). The preliminary Investigation is a desk study combined with a site visit. A preliminary investigation can be performed both for sites where contamination is expected and for sites that are probably uncontaminated. The main objective of the exploratory investigation is to proof that the assumptions made in the preliminary investigation are indeed correct. The goal of the main investigation is to provide the necessary information to deal with the contamination on a cost-efficient basis. The main Investigation is an iterative process, where after each step the question has to be answered if the available information is 'fit for purpose'.

Sampling of soil and groundwater has also been described in international protocols. i.e., for the sampling of soil (ISO, 2018) and for the sampling of groundwater (ISO, 2009).

5.3 Thresholds: screening values for soil contamination

5.3.1 The principle of soil pollution thresholds: from screening value to risk prevention

To identify whether or not a soil is at risk, that is, whether specific functions attributed to the soil are affected by the contaminant present, it is imperative to connect the quality of the soil under investigation to a specific critical limit or threshold, i.e. screening value (SV), which is related to a specific endpoint or protection target (e.g. quality guidelines for drinking water, tolerable daily intake levels via food and other exposure pathways). SV are linked to critical levels in soil via transfer models. Such models depend on representative actually measured concentrations of soil contaminants, or values that are predicted to occur within a specific timeframe (de Vries et al., 2007, for Cd: see Römkens et al., 2019). It must be noted that end-points differ depending on the type of receptors. In Table 5-2, an overview of relevant end-points and related critical limits are listed. In essence, the endpoint refers to the target to be protected.

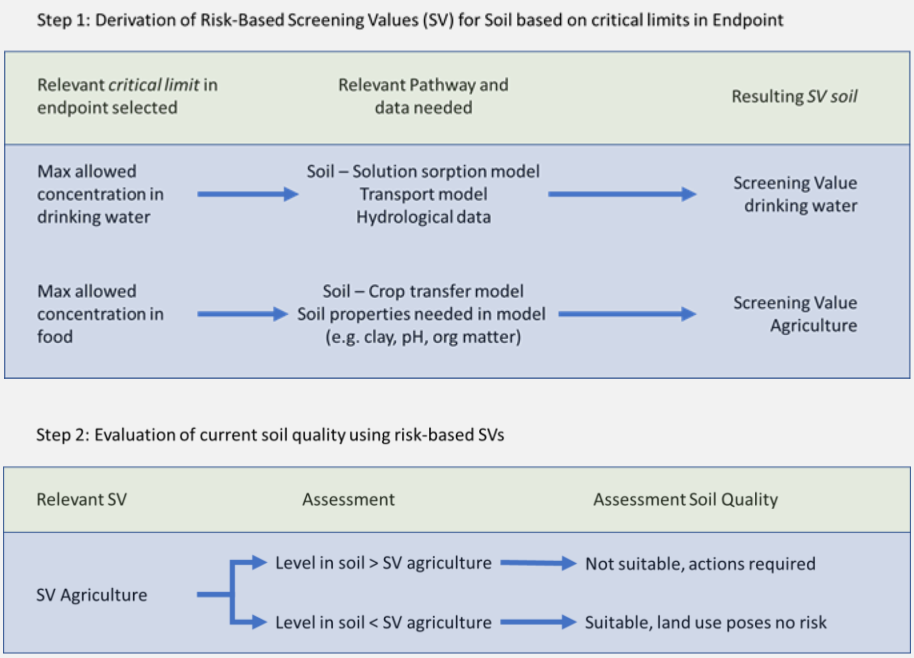
A large variety of screening values (SVs) for different levels of risk have been developed, differing by methodical/scientific and political choices in different countries (see also Carlon et al., 2007). Through parameters such as pH, organic matter or clay, the SVs become soil-specific. In order not to let soil variability limit the application of SV's, the Dutch system has developed a generic SV using a fixed set of soil properties. Such **SVs for a 'standard soil'** (a soil with 25% clay and 10% organic matter) are to be used as generic, first-tier national standard. Based on local conditions, **local SVs** can be derived based on an adopted set of correction formulas to convert the generic SV to the local or regional conditions (Wezenbeek et al., 2008).

Box 5-1 Key principles of RBLM and related screening values

In order to avoid risk from pollution towards the consumer, the ecosystem or livestock **risk based land management (RBLM)** has been developed as restorative or remedial action triggered by the exposure of endpoints. Several methodical steps can be identified how screening values for soils are developed so that the proper management response can be triggered:

Step 1: Relevant critical limits for each form or land use are identified. This can be a single critical limit for example if the main function of an area is to protect drinking water quality but can include multiple criteria if the land use includes multiple relevant endpoints. For agriculture for example it can include both, critical limits in food products, critical limits related to animal health, and critical limits in nearby surface waters (for N and P for example). Each of the critical limits is then converted to a screening value for soil. This screening value represents the acceptable quality of soil below which the function is not affected by the level of contaminants in soil.

Step 2 includes the actual assessment, as shown in Figure 5-2. It involves the comparison of the actual quality in soil with relevant SVs. If the actual soil quality exceeds the relevant SV (or minimum of SVs in case of multiple protection goals), site-specific risk assessment follows (for a more comprehensive elaboration of this, see Ehlert et al., 2013).



As stated earlier, risk limits or screening values derived based on RBLM-principles need to be consistent in both, endpoints addressed (human, ecosystem) ánd protection level, in order to be comparable. And even then, the resulting national screening value or risk limit can depend on for example soil properties. Uptake of metals by crops for example depends (for metals) on pH and, in some cases, organic matter. This means that given a critical limit in food (for example as defined for cadmium and lead in Regulation (EC) No 2006/1881) the resulting screening level in soil derived from this varies according to soil pH and or organics matter content. This can even be the case on a regional level and a first step to correct for this is to use default soil properties (e.g., soil pH, organic matter or clay) to derive a 'standard' screening value (as is done in the Netherlands using a fixed content for organic matter and clay content). When applied locally or regionally, local or regional screening values are derived using regional (or local in case of point source pollution) soil data.

5.3.2 Knowledge base regarding thresholds for soil contamination

A vast variety of thresholds in particular for heavy metals have been developed in many countries, for both point source and diffuse soil pollution. As shown below, screening values as a specific kind of threshold, have been derived from, and for, risk assessment methods, of which many different approaches exist (Swartjes et al., 2009).

At present, most thresholds consider a critical endpoint to be protected, usually human health and the (soil)ecosystem. Other end points often used are groundwater, drinking water and surface water (Carlon and Swartjes, 2007). In some countries, wildlife, animal products, or crops are considered as end points. Current UK (Soil Guideline Values), German (BMU 2020) or Dutch intervention values, for example, are based on effects to humans and the ecosystem. They also share a common approach in that risk assessment is at the core of the system to derive screening values that depend on the actual land use. By relating exposure to acceptable exposure (e.g., TDI or access cancer risk) of human beings[44], a human health-based threshold results. By selection of a Potentially Affected fraction (PAF) of organisms, an ecology-based threshold can be derived from a species sensitivity distribution (Posthuma and Suter, 2011). In case of the Dutch approach, the minimum of the human health-based and ecology-based value serves as the final threshold in soil. Similar approaches have been adopted by other Member States, but the assumptions underlying the models considering the variation in soil across the EU, has resulted in a wide range of screening values (Swartjes et al., 2007).

Despite the inherent range in thresholds across member states, there is a common principle underlying most risk assessment schemes developed so far. The following main groups of thresholds are commonly used (see also Figure 5-1):

Threshold as an overall term can be specified into background and screening values as schematised in Figure 5-2. Not all kinds of thresholds are used in every country, but all existing thresholds in Europe fit in this schematic presentation.

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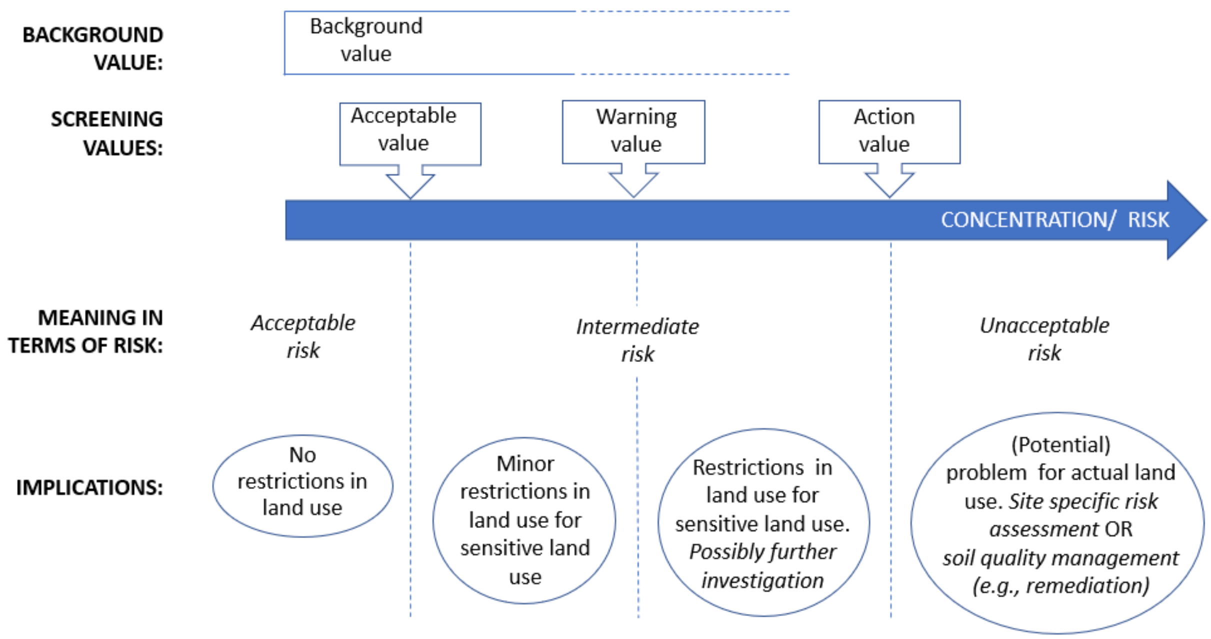
44 Tolerable Daily Intake (TDI): the amount of a potentially harmful substance (e.g., contaminant) in food or drinking water that can be ingested daily over a lifetime without appreciable health risks (Becking et al. 2007).

Figure 5-2: Schematic representation of thresholds



The specification of the thresholds is as follows:

- **Background values**. According to Reimann et al. (2005), the "background value" is often used as a base value to evaluate whether or not a specific soil has been under the influence of anthropogenic pollution in soils. It is commonly expressed "in terms of average, typical, median, mode, a range of values or a background value" [ISO 19258 (2018)]. In some countries background values are assumed to pose no risk or negligible risk and are considered suitable for any type of land use. Background concentrations, however, **do not have any relation with risks**. In the Dutch system, for example, background values are determined as the 95 percentile of values taken from 100 sites that are considered as the most strict thresholds for practical reasons. There is much variability among countries in the definition of the percentile, the population of measuring points, and the level of stratification. Due to the variability of most contaminants in the parent material from which soils are derived, differences in background values related to soil type or geographic distribution can be quite large.
- **Acceptable value.** Several countries use acceptable values in their procedures. The acceptable value generally relates to the **negligible risk level**. The basic idea of an acceptable value is that there are no restrictions in land use, as long as the acceptable value is not exceeded. Acceptable values are sometimes also used a generic remediation target.
- **Action value**. Action values mark the **unacceptable risk level**. Exceedance of the first generation of action values often meant 'polluted soil' and required some kind of intervention (such as remediation). Currently, most countries have more advanced procedures based on *frameworks*, in which the thresholds generally act as a trigger for further, more detailed site-specific investigations in one or two additional assessment steps (Swartjes, 2019).
- **Warning value**. **Intermediate risk** falls in-between the acceptable value and the action value. These values can indicate when (minor) restrictions is soil use for sensitive land uses are appropriate (e.g., no cropping recommended). In other procedures the value is used as a trigger for further soil or groundwater sampling, with the purpose to increase reliability of the judgement whether or not the action value is exceeded.

5.3.3 Currently known screening values

Heavy metals

In view of the assessment of risks from metals in soil, the concept of the derivation of background, warning and action values has been applied in a large number of EU member states. The screening values presented in Table 5-6 reveal a large ranges, mostly related to differences in land uses, the underlying risk limits in endpoints, protection targets, as well as different methodologies to convert those to screening values in soil. In addition, variation also results from the diverse soil properties (e.g. acid soil having more strict SV) or land use. Table 5-6 thus represents the current knowledge. Until now, for the four heavy metals (Cu, Cd, Pb, Zn) as presented in Table 5-6, a total of 444 screening values were found, roughly 50 – 60 per metal and risk level. The values, definition criteria and sources are documents in a data base by the European Topic Centre on Urban, Land and Soil Systems (ETC/ULS). The data base is currently expanded for As, Hg, Ni and Cr, before is handed back to the Eionet Working Group Soil Contamination for review and updating. In parallel, supplementary information is collected in order to understand the differences about how the SV are defined and derived.

For example, for cadmium, screening values have been retrieved for 17 European countries and 3 regions. In accordance with the underlying principle of risk assessment, these are specific for a certain land-use and specific texture classes or parent material categories. Incidentally, other soil properties are included like saturated hydraulic conductivity and soil depth, as in case of Poland. This relates not only to differences between countries but equally so for limits inside one country.

At the European level, thresholds for cadmium range between 0.4 mg kg^{-1} (agricultural soils, Czech Republic) and 1400 mg kg^{-1} (industrial land-use, United Kingdom). The variation for intermediate values is equally high with a factor of 1000 between the lowest and highest values. For Copper, the range of variation is less extreme for both critical risk thresholds (60-1500 mg kg^{-1} within the same country, Czech Republic) and intermediate values (factor 100). A similar degree of variation is reported for thresholds for lead (the critical values range between 50 mg kg^{-1}, Poland, and 2500 mg kg^{-1} , Brussels and Flanders).

Table 5-6: Current screening values for intermediate and intervention values of selected heavy metals in soil

When the values are stratified nationally, e.g. by another parameter such as soil type or parent material, a range of values is provided. All SVs provided in [mg/kg].

Geographical region	Cadmium (Cd)				Copper (Cu)			
	Intermediate risk		Critical risk		Intermediate risk		Critical risk	
	Stratification	SV	Stratification	SV	Stratification	SV	Stratification	SV
Albania_Tirana		0,7				36,3		
Austria	*Land use*	1 - 40		10	*Land use*	100-1500		600
Belgium_Brus		1	*Land use*	2-30		40	*Land use*	145-800
Belgium_Fland			*Land use*	2-30			*Land use*	200-800
Brussels_Wall.	*Land use*	1-10	*Land use*	10-50	*Land use*	40-120	*Land use*	80-500
Bulgaria	*pH*	0.04 - 3			*pH*	15-280		
Czech Republic		10	*Land use and Texture*	0.4-30		500	*Land use and Texture*	60-1500
Denmark		5		5		500		1000
Finland		1	*Land use*	10-20		100	*Land use*	150-200
Germany		20						
Hungary		1		10		75		1000
Ireland				1				
Italy			*Land use*	1.5-15			*Land use*	100-600
Lithuania				3				100
Netherlands	*Land use and Texture*	1-10	*Land use*	12-13	*Land use and Texture*	40-200		190
Poland			*Land use, Saturated hydraulic conductivity and Soil depth*	1-20			*Land use, Saturated hydraulic conductivity and Soil depth*	30-1000
Slovakia	*Land use*	0.1 - 5		20	*Land use*	1-100		500
Slovenia		2		12		100		300
Sweden	*Land use*	0.4 - 12		4	*Land use*	100-300		1000
United Kingdom			*Land use*	2-1400				500

Geographical region	Lead (Pb)				Zinc (Zn)			
	Intermediate risk		Critical risk		Intermediate risk		Critical risk	
	Stratification	SV	Stratification	SV	Stratification	SV	Stratification	SV
Albania_Tirana		85,5				151		
Austria	*Land use*	100-300		500		300		
Belgium_Brus		120	*Land use*	200-2500		120	*Land use*	300-3000
Belgium_Fland			*Land use*	200-2500			*Land use*	600-3000
Brussels_Wall.	*Land use*	80-385	*Land use*	170-360	*Land use*	120-320	*Land use*	215-1300
Bulgaria	*pH*	20-80			*pH*	20-370		
Czech Republic		250	*Land use and Texture*	100-800		1500	*Land use and Texture*	130-5000
Denmark		40		400		500		1000
Finland		60	*Land use*	200-750		200	*Land use*	250-400
Germany		400						
Hungary		100		750		200		2500
Ireland								
Italy			*Land use*	100-1000			*Land use*	150-1500
Lithuania				100				300
Netherlands	*Land use and Texture*	15-590		530	*Land use and Texture*	150-720		720
Poland			*Land use, Saturated hydraulic conductivity and Soil depth*	50-1000			*Land use, Saturated hydraulic conductivity and Soil depth*	100-3000
Slovakia		150		600	*Land use*	2-500		3000
Slovenia		100		530		300		720
Sweden	*Land use*	80-300		800	*Land use*	350-1050		3500
United Kingdom			*Land use*	450-750				

The ranges given in this table represent different soil (texture class)/land use conditions. This table only provides an overview.

5.4 Development of an EU-wide harmonized approach

Within the EU, there is a wide diversity of risk assessment tools for the same purpose (Carlon et al., 2007). This diversity is partly due to different geographical and cultural conditions in the EU Member States. However, also the lack of scientific consensus explains part of the differences (Swartjes, 2011). Additionally, different policy positions in different countries contribute to the differences in screening values in the EU. Due to differences in geography and culture, there is no need for identical screening values in all EU Member States. The derivation of screening values, however, would benefit from a more harmonized approach. For the sake of scientific integrity, a stronger convergence of risk assessment tools that do not include geographical, cultural, or policy elements would be favorable (standardized risk assessment tools). Risk assessment tools that do include geographical, cultural or social elements must be applied with a certain level of flexibility so as to account for these geographical, cultural or social elements (flexible risk assessment tools). Within the Heracles network, the development of a toolbox was therefore proposed, including standardized and flexible risk assessment tools (Swartjes et al., 2009).

In the case of the Water Framework Directive (2000/60/EC), critical limits in endpoints are agreed upon at EU-level; the Regulation (EC) No 2006/1881 on setting maximum levels for certain contaminants (cadmium, lead and arsenic) in foodstuffs; and also, the Drinking Water Directive 98/83/EC sets quality standards of drinking water.

Development of critical limits in endpoints and transfer models for emerging contaminants

Currently, thresholds can only be compiled for a limited number of contaminants. For an array of relevant emerging contaminants, no such critical thresholds in food or water exists due to the lack of TDIs for human or animal health. The difficulty there is the huge variation in biogeochemical properties even within one group of compounds like nanoparticles. Aside from the lack of relevant thresholds in endpoints, also the models to link such endpoints to corresponding levels in soil is only at its infant stage. For PFAS for example, experimental data on the transfer of PFOS, PFOA or other PFAS-compounds are still scarce or have been derived at concentrations unlikely to be found under normal conditions in soils.

Revision of current thresholds in view of actual risk to humans or the environment

Even for an intensively studied element like lead or arsenic, there are ongoing discussions on how to define an ‘acceptable’ level in human bodies related to effects. For example, the lowering of the TDI for lead would immediately lead to even more strict acceptable levels in soils in for example city soils.

In Table 5-7 a summary is given of the current status of several groups of relevant contaminants, their sources, and relevant limits in both endpoints as well as, if available, in soil.

Table 5-7: Status of knowledge about different groups of contaminants in soil

Substance group	Represen-tative chemicals	Relevant diffuse sources related to soil	Source control (quality)	Key endpoints considered	Critical limits in endpoints	Transfer models available	Critical limits soil in place
Heavy metals	Cadmium, Lead,	Compost, mineral fertiliser, sludge, aerial deposition	Mineral fertilisers (EU), sludge (EU), compost (EU)	Food quality (crops/animal products)	Yes (As, Cd, Pb, Hg), EU/WHO standards	Soil specific for a limited number of metals (Cd, Pb)	National risk-based limits, no EU-wide limits
				Fodder quality	Yes (Cd, Pb), EU standards	For limited number of metals (Cd, Pb)	
				Soil ecosystem and animal health	Yes (most metals. PNEC + TDI)	For most metals, national approaches	
Micronutrients	Copper, Zinc	Animal manure, compost, sludge	Compost (EU + national), manure usually not regulated	Soil ecosystem	Yes, national	Yes (biotic ligand models based)	National, risk-based
				Aquatic ecosystem	Yes, EU-wide (WFD)	Yes (biotic ligand model based)	National, risk-based
Persistent Organic micropollutants	PAH's	Sludge, aerial deposition	Sludge	Soil ecosystem	No	Few, poor quality in relation to predicted levels in endpoints	National, partly risk-based
Plant protection chemicals (pesticides)		Aerial deposition (spraying)	management control	Aquatic ecosystem	Yes	Yes	National, risk-based
				Soil ecosystem	Yes	Yes	National, risk-based
Persistent emerging pollutants	Medicinal products	Animal manure, sludge	No, management control	Soil ecosystem	No?	Few, in development	No soil criteria
				Aquatic ecosystem	Yes (Water quality)		
	PFAS	Sludge, water (fire fighting)	Not yet in place	Food quality	No (TDI has been derived by EFSA)	Few, based on initial experimental studies	No risk-based soil criteria, few countries are developing background values.
				Drinking water quality	Yes (drinking water)	Limited, based on documented cases, large discrepancy with laboratory tests	
Other	Micro-plastics	Compost, sludge	Compost (limited to particles > 1-2 mm).	Soil ecosystem Food quality (?)	No	No, experimental case studies only.	No soil criteria

6 Soil biodiversity

Soil organisms are the "biological engine of the earth" and are crucial for the functioning of soils. An active microbiome and below-ground food web controls energy transformations and nutrient turnover of ecosystems. The aim of this chapter is to compile the current approaches for defining the loss of soil biodiversity at European level. Species-diversity may seem, by definition, a robust indicator of healthy soil communities. However, there is a general lack of knowledge about the good status of soil biodiversity and its baselines. While an enormous number of soil-dwelling species is still undescribed, experimental evidence is lacking about the critical role of functionally relevant (flagship) species, and the effects of its loss. Rather than net species numbers, research focus is currently steered on the interactions between functional groups of organisms and their abiotic environment. Although it is currently impossible to quantitatively and accurately measure soil biodiversity as a whole, and to assess its health or level of degradation, it can be approximated using combinations of subindicators.

Increase of soil biodiversity has positive impacts on almost all soil related societal needs and soil functions, The need for infrastructure is not dependant on soil biodiversity.

Table 6-1: Relationship of Soil biodiversity to key societal needs and soil functions

Soil biodiversity

Societal need	Soil service	Impact
Biomass	Wood & fibre production	+
	Growth of crops	+
Water	Filtering of contaminants	+
	Water storage	+
Climate	Carbon storage	+
Biodiversity	Habitat for plants. insects. microbes. fungi	+
Infrastructure	Platform for infrastructure	indiff.
	Storage of geological material	indiff.

6.1 Rationale "Loss of Soil Biodiversity"

The majority of soil processes are driven by the soil biota (i.e. communities of many different microbial and invertebrate species) - thus its important role for many soil functions (Ritz et al. 2009). Loss of soil biodiversity directly affects soil ecosystem services (Breure 2004). Field research has shown that altered levels of soil biodiversity impact ecosystem services. For instance, De Vries et al. (2013) demonstrated that adequate C and N cycling processes require a certain level of biodiversity, i.e. a minimum number of specific feeding groups (e.g. microbes and invertebrates), total biomass of the soil food web, and biomass of the fungal, bacterial, and root energy channel. According to Orgiazzi at al. (2016), in 14 out of the 27 investigated EU countries covering more than 40% of the EU's soils, moderate-high to high potential risks for soil biodiversity do exist.

Soil biodiversity commonly includes all organisms living in the soil (including the soil surface, e.g. the litter layer): macro, meso and microfauna, and microorganisms (bacteria, fungi, protists, archaea and algae). According to Bloem et al. (2006), the main functional groups of the soil food web are:

- Earthworms consume plant residues and soil including (micro)organisms. Often they form the major part of the soil fauna biomass with maximally 1000 individuals per m2, 3000 kg fresh biomass per hectare, or a few hundred kg of carbon (C) per hectare.
- Enchytraeids (potworms) are relatives of earthworms with a much smaller size and a similar diet. Their densities are between 102 and 106 per m2, with a biomass up to 1 kg C ha^{-1}.

- Mites (fungivores, bacterivores, predators) have a size of about 1 mm, densities of 104-105 per m2, and a biomass up to 0.1 kg C ha^{-1}.
- Springtails or Collembola (fungivores, omnivores) also have a size of about 1 mm. They reach densities of 103-105 per m2 and a biomass up to 1 kg C ha^{-1}.
- Nematodes (bacterivores, herbivores, fungivores, predators/omnivores) have a size of about 500 µm, densities of 10-50 per g soil, and a biomass up to 1 kg C ha^{-1}.
- Protists (amoebae, flagellates, ciliates) are unicellular animals with a size of 2-200 µm, densities of about 106 cells per g soil, and a biomass of about 10 kg C ha^{-1}.
- Bacteria are usually smaller than 2 µm, with densities of about 109 cells per g soil, and a biomass of 50-500 kg C ha^{-1}.
- Fungi grow as networks of threads (hyphae) which usually have diameters from 2 to 10 µm, and reach total lengths of 10 to 1000 meter g-1 soil, and a biomass of 1 to 500 kg C ha^{-1}.

Without anthropogenic impacts the occurrence and diversity of these groups is mainly determined by the site-specific soil properties; Rutgers et al. (2009) found that biomass or numbers (abundance) of major groups of soil organisms varied among groups of land use and soil type. In order to assess the level of biodiversity below which soil functioning would be hampered, FAO and ITPS (2015) suggest the development of thresholds (see also Van der Heijden et al., 1998; Liiri et al., 2002; Setälä and McLean, 2004). However, at first, a clear relationship between biodiversity parameters and indicators for specific soil functions must be established (see Van Leuween et al. 2017). This begins with clear objectives for quality criteria of individual ecosystem services for specific ecosystems, and ends with the selection of the most appropriate indicator organism group(s)/species.

Although there have been many initiatives approaching soil biodiversity mapping across Europe in the past, currently there is a lack of knowledge for establishing (site or biotope specific) soil biodiversity baselines (EEA, 2019). Recently. Rutgers et al. (2019) highlighted two main constraints: the lack of consensus on the way to quantify the soil biodiversity provisioning function, and the scarcity of data necessary to map it at the European scale, but recently progress has been made by mapping nematode community composition on a global scale (van der Hoogen et al. 2019)

Soil biota is primarily impacted by land use (which determines the degree of physical disturbance, input of chemicals, and amount and quality of organic material such as litter). Agricultural intensification not only changes the diversity of individual groups of soil biota, it also reduces the complexity in the soil food webs, as well as the community-related mass of soil fauna (Tsiafouli et al. 2015). Furthermore, soil faunal communities had fewer and taxonomically more closely related species. Bloem et al. (2006) found that microbial biomass, microbial activity (respiration) and soil fauna functional groups tend to be more abundant at organic and extensively managed farms. Similarly, the number and diversity of species of the soil food web components, e.g. nematodes, in general decreases with increased land use and management intensity, in some cases intentional for example by applying an intensive rotation will decrease the abundance of potentially harmful organisms such as phytophagous nematoden. But on overall intensive land use is thought to make soil food webs less diverse and composed of smaller bodied organisms (Tsiafouli et al., 2015). In the medium term (up to 4 years), earthworms under organic management are two or three times the level of those found in conventionally managed fields (Blakemore, 2018). Because the findings about the relationship between management regimes and soil biota is fairly stable across regions, agricultural policies may be steered to halt and/or reverse this loss of soil biodiversity (Tsiafouli et al. 2015).

6.2 Soil biological indicators: state of the art

"Loss of soil biodiversity" means that species richness (presence and abundance) as well as its activity level is reduced so that soil processes (e.g. organic matter decomposition) and consequently soil functions (e.g. nutrient provision) are hampered. This requires an indicator which would monitor the **presence (diversity)**

and amount (abundance) of key species and/or functional groups in the soil (based on Rutgers et al. 2009, Bispo et al. 2009). Accordingly, high species diversity combined with high species abundance within functional groups would then provide a greater contribution of organisms to ecosystem services, in a spatially diverse habitat.

Over the recent years, several proposals on possible soil biodiversity indicators have been presented. The following section summarizes these efforts in order to draw a clear picture about the feasibility of current solutions. The overview also helps to identify gaps for further research and the steering of further monitoring efforts.

6.2.1 Concepts for identifying soil biodiversity indicators

Huber et al. (2008; based on Bispo et al. (2007); EU FP6 ENVASSO project) proposed the use of three key indicators to assess the threat of potential loss of soil biodiversity and associated ecosystem functions: i) diversity of earthworms. ii) diversity of collembolans and iii) soil microbial respiration. The results are presented in Bispo et al. (2009; see below 6.2.2)

Breure (2004) proposed i) microbial biomass (bacteria and fungi, which represent the highest amount of ecological soil capital); ii) nematodes (family level and feeding types), the relative and absolute abundance of which provides good information of the diversity and stability of the ecosystem; iii) earthworms: due to their influence on soil properties and since they are (relatively) easy to determine taxonomically and to characterize ecologically.

Ritz et al. (2009) reviewed 183 biological so-called candidate indicators, of which they selected 21 genotypic-, phenotypic- and functional-based indicators for different trophic groups; of that, 13 indicators would be currently fully deployable in monitoring activities (see also Black et al.2011). The indicator selection process has been quite complex because the authors ranked biological indicators against ecological processes and soil properties associated with its functions. In addition, they considered the applicability in large-scale monitoring schemes. The following list presents the most commonly discussed indicators:

- Indicators based on genotypic methods (most common among the selected indicators due to recent advances in molecular (sequencing) techniques): presence and amount of actinomycetes; ammonia oxidisers; archaea; de-nitrifiers; eubacteria; fungi. Indicators related to the structure of the microbial community, and are determined using DNA yield.
- Indicators based on phenotypic methods, such as extractions, visual recordings or catchings (pitfall traps): total abundance and functional groups involved with N cycling; presence and amount of microarthropods and nematodes; all soil fauna and flora, in particular ground-dwelling organisms as well as macro soil invertebrates.
- Indicators based on "*functional*" methods: substrate-induced respiration; potential enzyme activity (microbial biomass and total community activity).

These methods are often not species-based (in particular almost never for microbes) because of the extremely high species diversity and the lack of simple relationships between taxonomic status and functional traits in most soil microbial communities. Nevertheless, the authors stress the importance of knowledge about observation methods in combination with indicators in order to ensure comparability of results from different monitoring networks. Ritz et al. (2009) also note that a substantial amount of research and testing is still needed, in order to understand the sensitivity of these indicators for soil management, and how they correlate with soil functions, and their variability across spatial (landscape) scales (soil types, etc.) as well as seasonality effects.

The approaches by Ritz et al. (2009) were further developed by Stone et al. (2016): genetics-based indicators related to microbial and nematode diversity ranked highest, considering that indicators must be practical and sensitive to soil and management types. Griffiths et al. (2016) selected 18 soil biological indicators from literature and tested them at 6 experimental sites in different European regions. Besides methods, which address the diversity of individual groups (invertebrates and microbes), functional methods were identified which relate to different ecosystem services. However, further development and standardization of methods (sampling as well as analysis) as well as inter-laboratory comparisons are necessary to accompany the indicator measurements in monitoring. Actually, so far there was only one Europe-wide sampling program covering almost 100 sites in which at the same time and place both structural as well as functional endpoints have been measured (Stone et al. 2016b). Starting 2018, a subsample of LUCAS-Soil plots will be analysed for soil biodiversity using DNA-based methods (see Box 1, see also Orgiazzi et al. 2018).

Recently, Guerra et al. (2021) have proposed essential biodiversity variables (EBVs), which they closely relate to policy needs (UNCBD, SDG, Paris Agreement). The authors represent the global Soil Biodiversity Observation Network (SoilBON; https://geobon.org/bons/thematic-bon/soil-bon), which operates under the Group on Earth Observations Biodiversity Observation Network (GEOBON), and which invites researchers globally to systematically collect and sample observational data on the condition of soil biodiversity and functions. While the suggested EBVs have been discussed in previous frameworks including the ones cited above, Guerra at al. (2021) recommend specific analytical methodologies for each indicator:

- Intraspecific genetic diversity (DNA extraction)
- Abundance of species populations (DBA-based bacteria and fungi, nematode extraction)
- Community traits of roots (fine root weight, length and diameter distributions)
- Taxonomic community composition (DNA-based soil archaea, bacteria, fungi, protists, and invertebrates)
- Functional diversity (microscopic analysis of functional groups nematodes; functional diversity of bacteria, Archaea and fungi)
- Soil biomass (substrate-induced respiration method)
- Litter decomposition (litter bags, followed by incubation)
- Soil respiration (O_2 consumption from microbial respiration)
- Enzymatic activity (incubation followed by fluorescence measurements)
- Soil aggregation (soil aggregate resistance index)
- Nutrient cycling (amount and availability of nitrogen, carbon and phosphorus)
- Habitat extend (bulk density and soil structure)

Following the vision of one or several holistic indicators for soil biodiversity, it becomes clear from Guerra et al. (2021) and its predecessors, that a large amount of observation and research is still necessary in order to properly build and interpret monitoring of soil biological diversity and its functioning.

Box 6-1 DNA-based methods

Currently, several institutions are developing DNA-based methods to investigate soil-living communities. The recording and evaluation of the diversity of soil organism communities through DNA was hampered considerably by the lack of trained taxonomists and by using morphological features alone. In this context, considerable progress has been made within the last decade, but only recently some of these methods were standardized through ISO Standard 11063 (see also Plessard et al. 2012).

It is expected that in the near future efficient, cost-effective and routinely applicable DNA-based methods for soil biodiversity monitoring system will be available in order to monitor and evaluate soil biodiversity. Thus, in the foreseeable future, baseline and threshold data (see 6.3) will be generated for soil organism communities (ideally, not only covering invertebrates, but selected groups of microorganisms as well) all over Europe.

6.2.2 Experiences from applying soil biological parameters in soil monitoring

Bispo et al. (2009) successfully tested the three ENVASSO indicators in France, Ireland, Portugal, and Hungary (namely: diversity of earthworms, diversity of collembolans, and soil microbial respiration). In order to assess diversity and abundance of the 3 indicators at European scale, reference values or baselines are needed. They refer to results for the Netherlands (Rutgers et al. 2009) where ranges for selected land use and soil type categories were developed. In the Netherlands, 12 biological indicators are measured at 300 locations in a six-year cycle (Rutgers et al., 2009). Biological parameters included abundance of earthworms, nematodes, micro-arthropods and enchytraeids, bacterial biomass and DNA diversity; most parameters performed a clear pattern across gradients of land-use intensity and soils. Table 6-2 presents results from the French Bioindicator programme.

Table 6-2: Monitoring of biological groups In the French soil monitoring network

Parameters/Indicators	Indicator value	Sites	Source
Abundances of earthworms, Nematodes, Acari and the bacterial community, microbial biomass and earthworm species richness	main land use (grass-land, cropland, forest)	109 sites	Cluzeau et al. (2012)
Macro-invertebrate abundance, Collembola abundance and richness, nematode richness	agricultural practice		
Biological soil quality index based on soil macro-invertebrate community pattern	agricultural practice	22 sites	Ruiz et al. (2011)
Earth worm community and species (abundance. biomass, functional structure, and ecological trait)	main land use. level of contamination	13 sites	Pérès et al. (2011)

Krüger et al. (2017) have measured six biological indicators at 60 sites in two different landscape units in Wallonia (Belgium): Respiration potential (incubation), microbial biomass (fumigation extraction), carbon and nitrogen (dry combustion), net nitrogen mineralization (laboratory incubation), metabolic potential of soil bacteria (physiological profiling), and earthworm abundance (extraction). They demonstrate that all tested indicators discriminate main land use types and enable a fast assessment of biological soil quality at the regional scale. The higher the small-scale spatial variability of the site (larger in grassland, smaller in cropland), the higher the variability of the indicator values.

According to Römbke et al. (2016), one or several soil biological parameters are measured in monitoring programmes of 15 European countries.

As a new data source, the current LUCAS Soil survey 2018includes the additional analyses of parameters related to soil biodiversity: DNA metabarcoding of bacteria, archaea, fungi as well as other eukaryotes (e.g. invertebrates). The final aim is the characterization of soil organism communities.

6.2.3 Data bases in support of baselines

Rutgers et al. (2016) collected and harmonized existing earthworm community data from several European countries and combined these measured occurrences of the earthworm taxa with environmental and climatic variables. They could thus predict earthworm abundance and produce a biodiversity map of earthworms. The resulting maps of earthworm abundance and number of taxa could serve as a reference layer for monitoring.

Van den Hoogen et al. (2020) compiled a global nematode database. Soil nematodes are a good indicator because they play a central role in regulating carbon and nutrient dynamics, and control soil microorganism populations. Tundra, boreal and temperate forests have the highest abundances (> 2000 nematodes /100 gr dry soil).

A new archive about the distribution and ecology of soil animals (earthworms, small earthworms, nematodes, springtails, mites, centipedes, millipedes, and woodlice) is Edaphobase[45] (a project under the German contribution to the Global Biodiversity Information Facility, GBIF-D) (Burkhardt et al. 2014). Up to now Edaphobase contains more than 500.000 observations, about 300.000 sites, and 140.000 taxa (Römbke et al. 2012). Currently, this approach is going to be modified in order to collect the available information on soil biodiversity in an extended version by connecting Edaphobase with the respective databases of other (mainly European) countries. This work is done in a project entitled EUdaphobase; results will be available in about four years.

The LUCAS Soil survey, coordinated by the European Commission's Joint Research Centre, offers an open-access database including soil physico-chemical properties collected every three years, as of 2009, in over 20,000 locations across Europe. As of 2018, a soil biodiversity component was included into the survey scheme. Soil biodiversity data (i.e. DNA) from 1,000 points were generated.

6.2.4 Concepts for proxy indicators for spatial mapping and combined approaches

Several approaches have been developed to map and assess soil biodiversity in Europe.

Aksoy et al. (2017) assessed and mapped the overall potential for soil biodiversity throughout Europe using proxy indicators about the expected effect of soil biota at good condition (pH, soil texture, soil organic matter, potential evapotranspiration, average temperature, soil biomass productivity, land use). Such an indirect approach seems feasible for macrofaunal groups such as earthworms, which are known for its correlation between ecological niche and environmental parameters; thus, their geographical distribution can potentially be predicted from environmental data (Rutgers et al. 2016). Aksoy et al. (2017) provided a first overview of the potential diversity of soil animals and organisms in relation to the existing diversity of soils and its properties (see also 6.3.2).

Rutgers et al. (2018) selected 37 soil, environmental and management attributes in order to quantify the function of soil biodiversity; they distinguished 4 categories: i) soil nutrients; ii) soil biology; iii) soil

45 Edaphobase, a project under the German contribution to the Global Biodiversity Information Facility, GBIF-D; http://www.portal.edaphobase.org

structure; and iv) soil hydrology. These 37 attributes were used in a decision model to derive a qualitative assessment of the soil biodiversity function of soils. Given the wide number of required attributes, data availability certainly constitutes a limitation of the application of the model. Attributes are assessed in qualitative terms (high, medium, low categories), making the data reliability less critical on the one hand, while on the other hand, expert knowledge for setting thresholds is needed.

Creamer et al. (2019) proposed and tested a monitoring scheme for five soil functions, including habitat for biodiversity which uses soil attributes to calculate the functional capacity of soils. For soil biodiversity, the following 14 attributes are analysed: soil texture, bulk density, groundwater table depth, pH, C:N ratio, N:P ratio, soil organic matter (SOM), organic C content, earthworm and nematode abundance and richness as well as bacterial and fungal biomass. These attributes were measured from soil samples of different sites in Europe and combined with site, management and environmental attributes to quantify the functional capacity of the soils evidencing the difficulty, but still feasibility, of monitoring soil biodiversity across Europe. However, there is still a lack of standardized functional methods (including biodiversity) in order to monitor and to quantify ecosystem functions and services (Rutgers et al. 2012).

Based on experiences made in Germany and focusing on species diversity, Toschki et al. (2020) proposed different invertebrate groups (i.e. enchytraeidae, collembola, chilopoda, diplopda and oribatida) for the characterization of three main land use types: forests, grasslands and cropland sites. Only enchytraeidae were useful in all of them, but surely for the biological characterization of soils more than one group is necessary. This is due to the fact that for natural reasons not all groups do occur at all sites in similar and/or sufficient diversity (e.g. soft-bodied organisms such as earthworms or enchytraeids do not thrive well in permanently dry soils).

Table 6-3: Indicators proposed for soil biodiversity monitoring

Indicator	Creamer et al. (2019)	Huber et al. (2008)	Breure (2004)
Diversity of earthworms	●	●	●
Diversity of collembola		●	
Microbial biomass	●	●	●
Diversity of nematodes	●		●
Soil texture	●		
Bulk density	●		
Groundwater table depth	●		
pH	●		
C:N ratio	●		
N:P ratio	●		
Soil organic matter	●		
Organic C content	●		

Most of the methods used to determine the indicators have been standardized by the International Standardization Organisation (ISO) (Römbke et al. 2018). In order to assess the ecological condition of a given site, the so-called **reference approach** is recommended: a reference data base is developed which contains the respective assemblages of species (earthworms, collembola, and enchytraeids) and functional groups (nematodes, fungi, bacteria), by land use, soil properties and other environmental factors (e.g. climate). The ecological condition of a given site can then be determined based on comparison to such reference assemblages. Details for such an approach are presented by Römbke et al. (2012), and Toschki et al. (2020), and are mainly based on a review of large research projects within the last 20 years. The approach is operational, with two conditions: (a) the exact site-specific parameters for such a reference data base need to be agreed upon, (b) existing national monitoring pilot studies (see above) need to be extended, and established in all countries, so that as many as possible and representative soil conditions and their respective organism communities are covered in that database. Edaphobase and LUCAS Soil could serve as reference data bases.

6.2.5 Additional aspects to consider during soil biodiversity monitoring

Besides any standard documentation of the site of soil sampling, the following additional information is recommended to collect:

- Fabric of organic horizons (i.e. peat, forest floor): nature and arrangement of humus constituents (structure, consistence, character) (see Green et al. 1993)
- Type of litter: plant species of origin, plant part (woody, leaf/needle, root), decompositional status
- Fungal mycelia and faunal droppings: distribution and abundance
- Roots: abundance and size
- Presence/abundance of common soil fauna (in particular: earthworms, separating the functional groups endogeics (i.e. dwellers in the mineral layer), epigeics (dwellers in the litter layer) and anecics (vertical burrowers)
- Horizon boundaries (shape and width)

Depending on measurement intensity, Bispo et al. (2009) suggest different monitoring levels: with species counts for earthworms and Collembola at a fairly high density of plots (Level I)and functional diversity and DNA analysis at fewer plots (Level II), and the measurement of parameters about complex biological functions at Level III.

6.3 Baseline and threshold values

6.3.1 Definitions

It has been demonstrated above that a baseline and threshold values are needed to monitor soil biological diversity. It has been demonstrated above that such values have hardly been achieved yet. Huber et al. (2008) suggests that both, baseline and thresholds, need to be stratified by soil type and land use; Rutgers et al. (2018) introduce additional stratifies by climatic zone and management practice. Within the ENVASSO project (Huber et al. 2008), a common approach to the derivation of baseline and thresholds is proposed.

- **Baseline values**. Many scientists from different disciplines have tried to define the highest score for biodiversity corresponding with the pristine or natural state or state of reference. These reference values have entered the policy process as “ecological status” (Water Framework Directive) or “conservation status” (Habitat Directive). This concept has been tested for the region of Flanders, using vascular plant species as the most suitable indicator (Schneiders et al., 2012).

  Huber et al. (2008) suggest the calculation of **reference scenarios** as a baseline, consisting of <u>minimum, maximum and mean values for each indicator,</u> by land use, soil type, climatic condition/biogeographical region. Cluzeau et al. (2012) applied this approach to France, determining baseline values for different biological groups (i.e., soil microbial biomass, nematodes, earthworms, soil macro-invertebrates) under different land uses (e.g. cropland and grassland). They highlight that soil fauna and microbial biomass can be used as bioindicators.

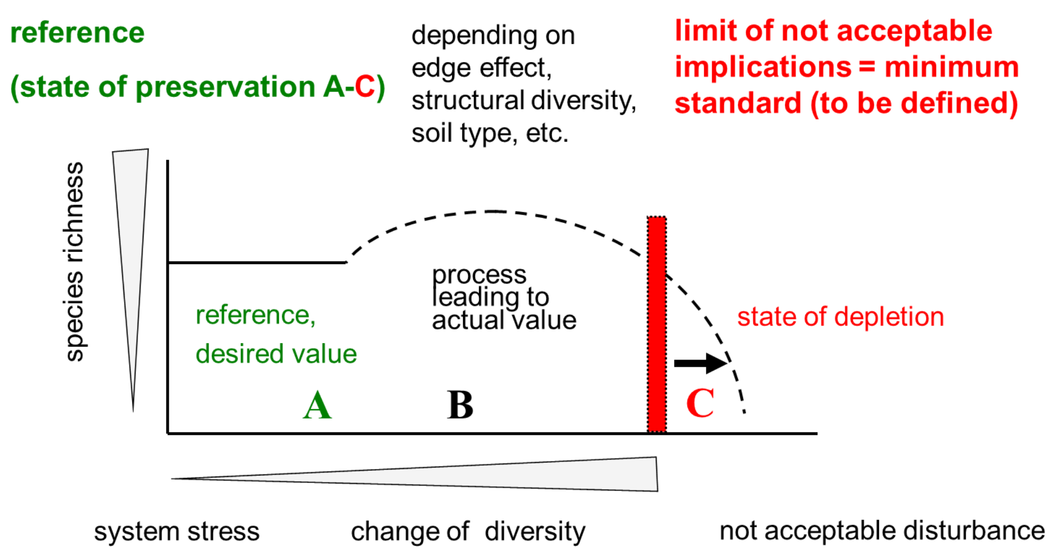
- **Threshold values**. Thresholds are defined by Schneiders et al. (2012), as '**safe minimum standard of conservation**': exceedance of this threshold implies irreversible change of ecosystem condition and may impose unacceptable social or economic costs.

  According to Huber et al. (2008), the simplest threshold would be nil: no organisms belonging to the target group are found at specific sites (in some cases this may be naturally the case, depending on the soil characteristics or on the season, e.g., earthworms in very acidic soils or in the topsoil during the summer months).

A more elaborated approach would aim to define a threshold as an unacceptable deviation from the baseline value or from the first measurement. In the latter case, natural variations must be taken into account.
The same approach is suggested by Breure et al. (2005): initial information on status and trend from monitoring can be combined with ecological know-how and serve as baseline/standard value for each indicator. The definition of unacceptable (and natural) variations could be made based on variations monitored under regional, national, and international monitoring networks. These datasets should be collated according to soil type, land use and climate.

In **Fehler! Verweisquelle konnte nicht gefunden werden.** the theoretical approach is briefly summarized (source: Römbke et al. 2012). It indicates that the reference condition could be determined as species richness, but other biological endpoints or indices are possible. From a biological point of view, the limit of unacceptable implications is not easy to determine, especially when looking not at one species but whole communities.

Figure 6-1: Baseline and threshold approach for soil biodiversity monitoring



Source: Roemke et al. (2012, 2016)

Fehler! Verweisquelle konnte nicht gefunden werden. indicates the general relationship between species richness and the change of the diversity of soil organism groups at different levels of stress, ranging from an unaffected (reference) site to a site which has been severely impacted by (often anthropogenic) stress. The red column indicates the point of unacceptable depletion (Römbke et al., 2012).

The concept of **reference condition** has been successfully tested in Germany. There, ranges of, e.g., species numbers for a given site or soil type, were compiled (example: earthworms). The quality of the approach improves with increasing number of well-documented observations (at best, at international level). As soon as an impact to soil biodiversity is observed (in this example: earthworm as flagship species), more detailed investigations are necessary, such as repeated samplings in different seasons.

6.3.2 Operating ranges for key soil organisms

A first approach of mapping the abundance and diversity of an important soil invertebrate group has been published by Rutgers et al. (2016) for earthworms. The approach has been now also developed at global level by Philipps et al. (2019). It was found that earthworm diversity does not necessarily follow above ground vegetation pattern (composition, density, vigour), but rather regional pattern, such as latitude in the case of Europe (Rutgers et al. 2016). Furthermore, maps have been developed in the last years for microbiological endpoints; see for example Griffiths et al. (2016). There, bacterial community structure could be largely explained through the pH value. However, no critical range of pH values for bacterial diversity has been provided.

According to Aksoy et al. (2017), the potential of soil biodiversity can be mapped indirectly via proxies. This is based on defined levels of soil parameters unsuitable for earthworms, mesofauna or microorganisms. Thresholds for temperature, texture, electronical conductivity, pH and LU change, were used in order to spatially delineate "risk" areas for earthworms but this approach could be extended to collembolans and probably all soil faunal species. This seems that this approach is suitable for the mapping of soil biodiversity hotspots. However, considering the large ranges and a large amount of species presence and abundance which cannot be explained by the scoring, a more differentiated approach is needed. Possibly, by overlaying the earthworm abundance maps by Rutgers et al. (2016) with environmental spatial covariates, the Table 6-4 could be refined for earthworm diversity, thus provide a hypothetical expectation value for earthworm diversity and abundance.

Table 6-4: Thresholds of environmental variables which might have strong effect on soil biodiversity

Variable	Classes of parameters as the basis for the scoring of soil biodiversity potential			
pH	<4	4–5.2	5.2–8.2	>8.2
Soil textural class	coarse	medium	medium-fine	fine
Organic matter (%)	<1	1–2	2–4	>4
potential evapotranspiration (mm)	< -500	-500 - 500	>500	
Annual average temperature (°)	<5	5–20	>20	
Soil biomass productivity	Poor	average	good	
Land Use/Land cover	Artificial	Arable	Permanent crops	Others

Source: Aksoy et al., (2017)

A similar approach has also been suggested by Römbke et al. (2016) and Hallin et al. (2016). However, they propose Operating Ranges (OR) for specific soil animals and microorganisms, defined as a range for a specific parameter (e.g., temperature, pH), which is tolerated by that species. Romeu et al. (2016) provide an example of establishing these ranges across Europe under different land uses and in different biogeographical regions. The operating ranges for key soil organisms (species, groups) or functions could be adopted to evaluate the environmental performance of farms and the environmental efficiency of different agronomic practices and different environmental gradients. The cited studies have shown evidence that diversity and composition of faunal and microbial communities affect ecosystem functioning under fluctuating conditions.

The living conditions for ectomycorrhizal fungi in European forests were studied by Van der Linde et al (2018). Because ectomycorrhizal fungi are strongly determined by the soil environment, thresholds could be determined for key environmental variables, correlated with the abundance of the studied fungi. The authors investigated 1,406 operational taxonomic units (OTUs), from a total of 25,196 samples. Table 6-5 presents the results, which may serve as a baseline to assess future change and resilience of forest fungi.

Table 6-5: Environmental thresholds for operational taxonomic units (OTUs) of ectomycorrhizal fungi

Variable	Decreasing OTUs	Increasing OTUs
Nitrogen throughfall deposition	5.8 kg N $ha^{-1}yr^{-1}$	15.5 kg N $ha^{-1}yr^{-1}$
Forest floor pH	3.8	
Mean annual air temperature	7.4°C	9.1°C
Potassium throughfall deposition	6.9 kg K $ha^{-1}yr^{-1}$	21.7 kg K $ha^{-1}yr^{-1}$
Foliar N:P	10.2	13.3

Source: Van der Linde et al., 2018

7 Soil erosion

In this chapter the assessment of soil erosion by water is discussed. It specifically addresses rill- and interill erosion and ephemeral gullying, which can be observed through large scale soil monitoring. In some parts of Europe, particularly the Mediterranean basin, permanent gullying and badlands are also important degradation forms. Soil erosion refers to the loss of fertile topsoil material after erosive rain event on sensitive soils, largely in the absence of sufficient vegetation cover. About 13% of arable soils in the European Union are affected by medium to high soil erosion rates. The main indicator for soil erosion is the rate of topsoil mass loss, which is usually expressed in Mg ha^{-1} y^{-1}. While several generalized thresholds for unacceptable erosion levels were developed, the concept of soil loss tolerance and the implementation of a tiered monitoring is recommended here.

Soil erosion is the detachment, transport and sedimentation of soil particles by water or wind; it has negative impacts on all societal needs, soil functions and soil-related ecosystem services. Soil erosion is itself an important driver for other soil threats especially the increase of flood risk and the decrease of biodiversity and soil organic matter (Stolte, 2016), playing also an important but underestimated role in soil organic carbon cycling (Chappell et al., 2016). While climate, and in particular heavy rainfall, is the trigger for soil erosion by water, in modern times agriculture, overgrazing, mining, and infrastructures are the drivers for severe soil erosion (Stolte et al. 2016) due to the non-sustainable soil management. Soil erosion is not a problem if the rates are within the geological soil erosion rates; climate change (increasing weather extremes) and unsustainable land management induce the acceleration of the soil erosion rates. Onsite effects of erosion often cause – among others – yield losses, up to 4% per 10 cm soil loss (Bakker et al., 2004). Large erosion events are often accompanied by substantial offsite damages (river and dam sedimentation, including offsite pollution (Boardman 2006).

Table 7-1: Relationship between soil erosion and key societal needs and soil functions

Soil erosion		
Societal need	**Soil service**	**impact**
Biomass	Wood & fibre production	-
	Growth of crops	-
Water	Filtering of contaminants	-
	Water storage	-
Climate	Carbon storage	-
Biodiversity	Habitat for plants. insects. microbes. funghi	-
Infrastructure	Platform for infrastructure	indifferent
	Storage of relocated material or artefacts (excavated geological material, sediments, cables and pipelines, archaeological material)	indifferent

7.1 Erosion processes and challenges for soil monitoring

7.1.1 Types of soil erosion

Huber et al. (2008) distinguishes various forms of erosion and processes leading to it:

- Water erosion: interrill (sheetwash), rill, ephemeral gully and piping as well as ephemeral gullies (subsurface) erosion resulting from surface runoff of excess rain water or subsurface flow.
- Gully erosion
- Wind erosion: strong air movements displacing loose soil particles

- Anthropogenic (Technic) erosion: i.e. tillage erosion (on-site soil loss after tillage of sloping land), harvesting erosion (off-site losses of soil adhering to the crop during harvest, mainly of root and tuber crops), erosion caused by trampling of livestock (on-site soil loss in combination with overgrazing and removal or reduction of vegetation cover, and especially on steep slopes, with subsequent soil displacement)

Monitoring these different forms of erosion is challenging, because they operate at different spatial and temportal scales (Stroosnijder 2005). In addition, several processes can also occur in parallel, or trigger each other, such as trampling and water erosion, water and tillage erosion, or water erosion and harvesting erosion. Boardman and Poesen (2006) summarize the relative importance of the main soil erosion processes for Europe (typical erosion rates for Belgium are presented in Table 7-2). The most extensive form of erosion in Europe is **water erosion**, in particular **rill erosion**. For soil loss due to crop harvest (sugar beet and potato), see Panagos et al. (2020). Poesen et al. (2018) also list other forms of erosion which deserve attention: subsurface erosion resulting in piping and tunnelling, land levelling, soil quarrying and trench digging. Some of these forms are related to the urban sprawl, and are often combined with soil relocation for construction projects. Soil erosion also occurs along the embankments of road and railway, as well as touristic, infrastructure (Seutloali and Beckedahl, 2015; Salesa and Cerdà, 2020).

Table 7-2: Mean annual rates of soil loss from different erosion processes for cropland in central Belgium

Process	Soil loss (t/ha/yr)	Fraction of total soil loss (%)
Water erosion		
Sheet and rill erosion	6.9	26.5
Ephemeral gully erosion	5.4	20.8
Tillage erosion	8.7	33.5
Soil loss by crop harvesting	5.0	19.2
Total	26.0	100.0

Source: Poesen et al. (2018)

The different erosion types vary considerably between locations. According to Boardman and Poesen (2006), gully erosion in the Mediterranean region can account for 10–80% of the total erosion on cultivated and grazed land, whereas rill erosion can clearly dominate in temperate areas where erosive rainfall events are less frequent and strong; this has consequences for the development of an effective and reliable monitoring system. However, erosive rainfalls are believed to increase with climate change in all parts of Europe (see also Burt et al., 2015, for a case study in England). The type of soil erosion which prevails at specific climatic conditions, management type (e.g., root crops), site or region, how erosion can be best monitored.

7.1.2 Soil erosion and ecosystem services

The role of soils and ecosystem services has been introduced in chapter 1.2. With regard to soil erosion, specific ecosystem services affected include crop growth, water filtration and water flow regulation and fresh water provision. The control of erosion rates is a regulating ecosystem service by itself, representing the reduction of soil loss by ecosystems, more particularly of the vegetation covering the ground, compared to bare soil (Guerra et al. 2014). Impacts of soil erosion on soil-related ecosystem services can be monitored or modelled, and then serves as proxy to estimate the negative effects of soil erosion, and to define thresholds on tolerable soil loss rates.

Soil erosion causes the thinning of top soil, with subsequent decreases in SOM and waterholding capacity and subsequent negative long-term effects on various biological and chemical soil processes. Both parameters are key for the before-mentioned soil-related ecosystem services. This is reflected also in the study of erosion effects on ecosystem services by Steinhoff-Knopp et al. (2020). There, subindicators for the soil-related ecosystem services were quantified (crop provision, water filtration, water flow regulation, fresh water provisioning), using pedotransfer functions based on local soil properties, management and climate (Mueller and Waldeck, 2011; Mueller et al., 2013). Therefore, the approach is site-specific and takes into account the spatial variability of covariates which determine the degree of soil erosion. With the help of an evaluation matrix, the authors could quantify the potential supply of soil-related ecosystem services of degraded soils (Table 7-3) and evaluate the impact of soil erosion on soil-related ecosystem services.

Within this concept, the reduction of ecosystem service supply after erosion can be quantified based on the changes of soil properties caused by the loss of fertile topsoil from erosion. If a minimum good status of potential ecosystem service supply is set as a target, site-specific **limits for tolerable erosion rates** can be derived (see chapter 7.3). It has to be emphasised, that the underlaying matrix of subindicators requires validation for soil and climatic conditions outside the case study in Northern Germany. The approach can be easily extended to other parts of Europe where spatial estimates of erosion rates are available.

Table 7-3: Indicators for soil-related ecosystem services in Northern Germany as affected by soil erosion

Ecosystem service	**Indicator**	**Specifi-cation**	**Status ecosystem service supply**					
			0 no	**1** very low	**2** low	**3** medium	**4** high	**5** very high
Crop provision	potential arable yield	Potential yield winter barley [t/ha]	0	≤ 2500	2500 - 2875	2875 - 3250	3250 - 3625	≥ 3625
Water filtration	Nitrate leaching vulnerability	Water exchange rate [%/a]	0	≥ 250	150 - 250	100 - 150	70 - 100	< 70
Water flow regulation	Water storage capacity	potential storable water [mm]	0	< 50	50 - 90	90 - 140	140 - 200	≥ 200
Fresh water provision	Percolation rate	Percolated water [mm/a]	0	< 200	200 to < 250	250 to < 300	300 to < 350	≥ 350

Source: modified from Steinhoff-Knopp et al. (2020)

With these subindicators spatially quantified, the effect of erosion on ecosystem services can be derived after GIS overlay with spatial erosion surveys. Erosion rates can be assigned to each class of ecosystem services supply, thus be used as thresholds.

7.1.3 On-site and off-site effects of soil erosion

The main **on-site effect** is the reduction of soil quality, induced by the loss of fertile top soil (see 7.1.2). However, these on-site effects are typically accompanied by off-site effects: the eroded material is transported by wind and water to adjacent and remote locations, along spatial gradients such as slopes and water runoff channels, sedimenting in catchments and river deltas, dams and other water harvesting installations, harbours, and – as muddy floods – damaging buildings, adjacent properties, and other infrastructure (e.g., Verstaeten et al., 2006). Eroded soil is also known to clog up drainage systems, which could result in overflow and washout, and subsequently its failure (WHO 1991). It can even contribute to the pollution of public water supply. If coupled with poor drainage maintenance, the lifetime of pavements can be affected (e.g., deformations, frost heave). Road and sedimentation clearance operations are needed, causing significant cost. A recent regional scale modelling study on lateral C fluxes (Nadeu et al., 2015) indicates that while lateral export of C from cropland through erosion may have approximately the same magnitude as additional C sequestration in C depleted eroded soils.

7.1.4 Status of soil erosion by water

Soil erosion is among the eight soil threats listed within the Soil Thematic Strategy of the European Commission (EC, 2006); it is one of most widespread forms of soil degradation especially in agricultural areas. Hot spots include the south-eastern and eastern European as well as the Mediterranean region (Kirkby et al. 2004, based on predictions using the PESERA model). In the latter, this is because of the high rain erosivity during winter months (when coupled with lack of proper soil cover by crops).

Several recent publications summarize the state of erosion for the agricultural area for the whole of the EU (Veerman et al.,-2020; EEA, 2019). These reports essentially go back to the works of Panagos et al. (2015 and 2020). There, soil loss has been estimated using the empirical prediction model RUSLE (Revised Universal Soil Loss Equation). According to Panagos et al. (2015), the mean soil loss rate in the European Union's water erosion-prone lands (agricultural, forests and semi-natural areas) was found to be 2.46 t/ha/yr, resulting in a total soil loss of 970 Mt annually. Roughly 25% of the EU land are shows erosion rates > 2 t ha/yr, while ca. 6% of agricultural area shows severe erosion (11 t ha/yr) (Panagos et al. 2020).

Several authors have summarized erosion rates from upscaled local observations or plot measurements (e.g., Gobin et al. 2004). These results show that it is inaccurate to operate with average values rather than ranges of values, provided that few critical erosive events increase the medians for otherwise insignificant erosion levels. Nevertheless, Cerdan et al. (2010) conclude an average erosion rate for arable land across Europe to be 3.6 t/ha/year[46], with a peak of 17.4 t/ha/year for vineyards. Assuming an average soil bulk density of 1.5 g/cm^3, this translates respectively to a loss of 0.2 mm and 1.2 mm of topsoil loss per hectare per year. Considering the typically slow rates of natural soil development from weathering and soil formation processes, estimated to be 0.05–0.5mm year (Wakatsuki and Rasyidin, 1992, cited from Gobin et al. 2004), any soil loss of more than 1 t/ha year can be considered as irreversible. Cerdan et al. (2010) estimate that 70% of the total erosion occurs in 15% of the area; thus, the erosion rate strongly varies across Europe and is likely to occur in hotspots at much higher rates than the average.

Darmendrail et al. (2004) present the results for 14 surveyed localities in the UK, totalling 4.8 Mio ha, and an average annual erosion rate of 0.86 t/ha. For comparison, the modelled rates of erosion in the UK, based on Panagos et al. (2015) amounts to 2.38 t/ha/yr across all land uses, and 1.04 t/ha/yr for arable land (cited from Evans and Boardman 2016; higher rates from modelling did not spatially correspond to the field measurements). For seven investigation areas in Northern Germany, an average 0.85 t/ha/yr was found from field observations (Steinhoff-Knopp and Burkhard 2018). A review for Spain, Solé-Benet (2006) cite 36 studies with a huge variability of erosion rates, indicating the need to harmonize and agree upon methods to measure and model erosion rates.

In silty soils in some Swiss regions, average mass losses of 20 t/event were observed, under extreme weather conditions up to 95 t. Clayey soils reach average rates of 0.3 t, in exceptional cases of up to 10 t (BAFU 2017). On grassland of the same case study, the average loss rate was 1.8 t/ha/yr, up to 30 t on local hot spots, whereas only 0.3 to 1 t/ha/yr would be tolerable given the typically slow soil formation rate (BAFU 2017). Prasuhn (2020), based on long-term monitoring of 203 fields in Switzerland, has reported mean soil loss rates of 0.74 t/ha/year (monitoring from 1997 to 2007) and 0.20 t/ha/year (2007 to 2017). He attributes this reduction to changes in soil tillage practice and erosion control. Considering

46 The estimate is based on field measurements (see Level III, Table 7-4) from 81 experimental sites in 19 countries

the variability of triggering weather events, accurate erosion monitoring of larger areas is difficult: in a watershed in central Switzerland, between 1998 and 2007, 50 % of all eroded material was related to 6 events (BAFU 2017). For English lowlands, Evans and Boardman (2016) report that only ca. 5 % of the observed area suffers from any level of erosion per year – depending on the risk category (the value is higher for more erosive soils), and very few events dominate the long-term averages.

7.2 Indicator specifications

7.2.1 Indicator definition

This indicator aims to delimit and quantify the extent of land which is suffering from soil loss due water erosion at such a level that the proper soil functioning, and the supply of soil-related ecosystem services is impaired. The current knowledge about indicators is summarized so that a harmonized approach for Europe leading to a tiered monitoring can be developed.

The main indicator for soil erosion is the **rate of topsoil mass loss**, which is usually expressed in t/ha/yr. In the absence of data on the actual soil erosion rate, various **proxy or impact (sub-)indicators** are often used to estimate information on the **severity of erosion**, and/or to estimate a **potential soil erosion rate**. These may include for example, dimensions of erosion features, increased turbidity in runoff, amount of deposition exposure of sub-soil, or other indirect parameters such as changed soil depth, reduced organic matter content, exposure of plant roots, and changes in soil texture. Also, reduced rainwater infiltration and changed water holding capacity, as they are affected by compaction, are conditioning the triggering conditions for erosion, and are thus also important to monitor. Besides, the set of (R)USLE factors can also be considered as sub-indicators and mapped separately to indicate potential erosion effects.

Table 7-4: USLE factors as subindicators

(R)USLE factor		Description and Critical values
R	Rainfall and runoff erosivity	Effects of rain drops on soil particle detachment and rate of run-off per rain event; erosive thresholds see Todisco et al. (2019): > 12.7 mm/event
K	Soil erodibility	Inherent property of soil particles to be detached and transported; depends on textural class, SOC content, stability of macroaggregates Soils below a critical shear stress < 0.4 Pa (Gibbs 1962) are more stable (threshold for the motion of soil particles)
L/S	Slope length and slope steepness	Decreased run-off with increasing slope length and decreasing steepness 5° as critical slope gradient (Zhang et al. 2015)
C	Soil cover (vegetation, residues)	
P	Management practices to prevent erosion	reduced/no tillage, contour farming, terracing and strip cropping, green manure, fallow with seeding

An overview of soil-related indicators derived under different initiatives to support agri-environmental policies at the European and global level is given in Panagos et al. (2020). At the European level, CAP addresses soil erosion by means of **two sub-indicators** related to the state of water erosion:

- estimated rate of (potential) soil loss by water erosion;
- estimated agricultural area affected by a certain rate of soil erosion by water.

According to OECD (2008), soil erosion is considered as moderate-severe when the rate is above 11 t/ha/year. For Europe- specifically – the level of severe erosion is lower. The Resource Efficiency

Scoreboard (Roadmap to a Resource Efficient Europe, EC 2011) looks at soil erosion by water by means of an estimate of the **area affected by severe erosion rate** (see also OECD 2013)[47], an indicator which is also part of the EU SDG (Eurostat, 2020). Soil erosion is also included as one of the 28 Agri-Environmental Indicator (AEI) and it is expressed as **erosion rate at different administrative levels** since it is intended to monitor the integration of environmental concerns into the CAP at EU, national and regional levels. It is focused on agricultural areas and natural grassland and distinguishes between moderate (5-10 t/ha/y) and severe (>10 t/ha/y) erosion.

Three major approaches can be applied in order to estimate the topsoil mass loss,

1. predictions using models (chapter 7.2.2),
2. direct measurements in the field (chapter 7.2.3) and
3. measurements on run-off plots (chapter 7.2.4).

While prediction models are capable to produce large-scale maps on national and European level (e.g., Panagos et al. 2015), direct measurements are limited to plots, fields and smaller investigation areas. All three approaches can be combined in a tiered monitoring to produce a sound, reliable database on the current status of soil erosion in the EU based on ground-truth and modelling (chapter 7.2.5).

7.2.2 Model-based predictions to identify target area for monitoring

Soil erosion models predict soil mass loss by simulating the effect of triggering events under specific soil and vegetation conditions. Types of models differ according to spatial scale, process, duration, hydrological processes, and model output. The empirically derived Universal Soil Loss Equation (USLE – later "revised" RUSLE), realized with GIS, is a wide-spread model to estimate long term average annual soil erosion at the field or other aggregated scale (for details see Panagos et al. 2017). It is based on the linear relationship between soil loss and several controlling factors, and thus includes rainfall erosivity, soil erodibility, slope, crop cover and management practice. Process-based models simulate and quantify erosion processes based on triggering (erosive) precipitation events; well-known process-based simulation models are MADALUS, EUROSEM and SWAT. The most recent erosion estimate for Europe is RUSLE-based (Panagos et al. 2017).

The (R)USLE-based models require intensive calibration (weighting of factors) to local conditions (Evans and Boardman 2016). It is commonly observed that model-based potential erosion rates tend to overestimate the real soil loss. Also the spatial accuracy of model-based predictions are questioned. It is assumed that the role of rainfall and slope steepness for the severity of erosion (at least in Europe) is overestimated. This is acknowledged by Panagos et al. (2015), who believe that with improving spatial accuracy of land use data, soil properties, and climatic data, model-based erosion estimates improve. RUSLE typically provides very long-term averages. It seems weak when modelling the erosivity of intense precipitation events (Avwunudiogba and Hudson 2014); a phenomenon quite known for Mediterranean areas, as documented for Spain (Benet 2006). However, it needs to be considered that observed erosion sometimes shows low correlation for some of the factors (e.g., erosive rain pattern and observe events); this limits its predictive power. Soil erosion processes are highly variable, and it is difficult to observe and to predict the actual erosion pattern.

Keizer et al. (2016) observed that the results of various erosion risk models applied at the European-scale differ considerably. This is mostly related to the quality of the input data and the spatial resolution of the results. Recently, the use of model-based predictions has been criticized, among others, because the averaging of model input data over large regions does not properly address the interactions between erosivity and land management, in addition the effects of crop dynamics (rotation systems) and seasonality

47 OECD's agri-environmental indicators contain "% of agricultural land having moderate to severe water erosion risk" (OECD 2013)

are not addressed (Fiener and Auerswald 2016). Also for high mountain ranges, USLE-based modelling is critical.

Measurement data at a certain resolution would be needed in order to improve the performance of the models. All in all, Steinhoff-Knopp and Burkhard (2018) conclude that the actual soil loss from long-term monitoring data can be considered more reliable than model-based predictions. This is why soil erosion monitoring should contain real observations, and for larger areas such as country-scale, model-based predictions and field observations could complement each other.

The role of erosion modelling in a tiered monitoring lies in the harmonised identification of areas with increased erosion risk, which could then be targeted with intensive field observations.

7.2.3 Direct, field-based quantification of soil erosion by water at the catchment to landscape scale

According to Boardman and Evans (2020), typically, the following characteristics shall be typically monitored during field-based programmes: presence of rills, ephemeral gullies, fans, wash, crusting, standing water, crop type, crop cover and irrigation. In addition, runoff leaving the field and entering other fields, roads, tracks, ditches or watercourses needs to be noted.

Herweg (1996), and later, Ledermann et al. (2010), provide an overview of more detailed methods to assess soil erosion in the field. There, the dimensions of **rills and gullies** are measured; for example, those which are deeper than 2 cm can be measured with a tape, and the volume of eroded soil is calculated (channel cross section: mean depth, mean width, and total length). The volumes of the individual erosion features would then be added to obtain the total soil loss. It needs to be noted that **interill erosion** is difficult to estimate with that approach (significant measurement error during field work can be expected).

While such direct measurements are difficult to upscale, measurements are often combined with modelling; they are needed to develop, calibrate and validate predictions from modelling (Stroosnijder 2005, Fischer et al. 2017). Long-term monitoring programmes based on visual and volumetric measurements of water erosion have been described by Evans et al. (2016), Prasuhn (2011, 2020) as well as Steinhoff-Knopp and Burkhard (2018).

7.2.4 Runoff plots

Plot measurements seem to be suitable to study rill and interrill erosion (Cerdan et al., 2010). They are often installed in order to compare different land uses and agricultural practices. Data from run-off plots are difficult to extrapolate to larger regions or countries because over-estimation is often observed (Boardman, 1998). However, such data are needed to calibrate erosion models where they are used. According to Boardman and Evans (2019), many models need to be better calibrated against real-world erosion monitoring data. This is especially important in data-poor areas and for predictions (e.g., policy outlook). For monitoring at national level, the authors recommend volumetric erosion measurements at locations where erosion has been observed (see 7.2.3). Plot selection can follow small-scale studies which recorded repeated erosion over a period of time (Boardman and Evans 2020). A data bases of runoff plots for Europe has been compiled by Maetens et al. (2012).

7.2.5 Tiered monitoring

A tiered monitoring of soil erosion in Europe addresses the different climatic, pedogenetic and agricultural conditions and focusses on regional relevant erosion processes. This section outlines a corresponding monitoring approach and suggests long-term observation approaches considering scale, and – subsequently - measurement intensity. The temporal dimension of any erosion monitoring programme must be long term because of its typical discontinuous character.

- Tier I:
  - Modelling soil erosion at national and European level
  - Monitoring rainfall events
  - Monitoring crop type and crop cover
- Tier II:
  - Erosion damage mapping in selected areas
- Tier III:
  - Plot measurements

In Table 7-5, different measurement intensities are suggested. Monitoring at catchment scale or for whole slopes require large installations, which - for now – are outside of the scope for national or EU-level monitoring. However, such long-term observation plots would probably be best suited to observe extremely episodic events, such as those causing gully erosion or landslides. Evans and Boardman (2016) describe field methods for assessing soil loss by water erosion combined with visual interpretations of aerial and terrestrial photos, and statistical upscaling to quantify soil loss rates at pan-European level.

The quantification (extent, frequency and severity) of water erosion (interrill, rill, sheet[48] and gully erosion), mass movements (including landslides) and wind erosion, through monitoring, is difficult. Usually, long-term monitoring of runoff plots is used to directly measure soil loss by inter-rill and rill erosion (Maetens et al. 2012). Very few studies have monitored gully erosion or piping erosion, and this could be achieved by making detailed observations in a selection of representative catchments throughout various European regions. However, there is bias for erosion rates determined at plot level, exceeding averages at landscape level by 2-10 times (Evans 1995).

Because of the large variability of soil properties across large landscapes, land use (e.g. tillage system) and climate, a large number of plots is required to build a Tier I and II erosion monitoring. A representative network of monitoring plots is needed. Boardman (2006) suggests to focus on monitoring the effects of moderate to strong erosion events. Considering that ca. 70% of the erosion occurs over only 15% of the total area of Europe (Cerdan et al. 2010), hot spots play an important role when stratifying systematic inventories across large landscapes and regions. Until now, a (trans-)national programme of soil erosion measurement using a standardized procedure is missing. At any rate, monitoring as well as erosion control measures must be precautionary especially in sensitive areas, for example, in the Mediterranean, where extreme events are more frequent.

An additional challenge to representativity is the dominance of different types of erosion in different regions in Europe (see Van Camp et al. 2004, Table 2.3. Types of erosion: occurrence at national level). An analysis and mapping of soil problem areas (hot spots) in Europe was conducted by EEA (2000), where broad zones were identified for which the erosion processes are similar (hot spots map for water and wind erosion).

Provided the characteristics, strengths and weaknesses of field-based erosion monitoring and modelling, and the different measurement intensities in large-scale plot systems versus erosion plots, it seems plausible that any monitoring system may consider an approach where hot spots are identified (weather, soil and management conditions), e.g., via modelling, which is then followed by field-based observations.

48 Surface water run-off is not canalized; water running off uniformly over a surface along slopes

Table 7-5: Design of large-scale soil erosion monitoring

Compartment	Measurement and estimation parameters		
	Level I	**Level II**	**Level III –erosion (run-off) plots(1)**
Direct (measured) and indirect (visual estimation) monitoring of soil erosion	Macromorphological features incl. erosion damages (e.g. pedestals, rills, litter movement, flow patterns, deposition, wind-sourced blowouts, and gully channels)(2) Plot sizes (Stroosnijder 2005): 3–25 m (width) x 10–25 m (length), > 3 replicated plots		Overland flow and sediment measured (collection tanks, and cumulative mechanical stage height counters; splash cups; bottles for creep in wind erosion); rainfall simulators Plot size: 0.001–0.1 ha(3)
Vegetation	Percentage of total plant cover	Percentage area, and time (season, length) of uncovered soil Vegetation height (as a proxy for rooting intensity) (lichens, mosses, herbaceous and shrub canopy, litter cover and bare soil)	
Site	Soil surface: curvature, slope gradient, exposition		
Soil	Estimated soil properties (augers) (grain size distribution, bulk density and porosity, soil resistance, soil organic matter, stoniness, soil moisture)	Measured soil properties (topsoil)	Measured soil properties (topsoil and subsoil)
		Soil profile description/soil type	
Weather	Rainfall and temperature, from the nearest weather station; Radar-meteorological data (e.g. RADOLAN, see also Auerswald et al. 2019 and others)		Continuous monitoring: rainfall, temperature, soil moisture Erosive rainfall events (date, duration, mm/m^2); temporal resolution: at least 5 minutes
Seasonality of damage mapping	End of the dry and wet seasons	After every erosive rainfall, snow melt and at the begin of the vegetation period	
Modelling	Upscaling, validation, calibration, uncertainties, representativity gaps, geospatial erosion statistics: Universal Soil Loss Equation (USLE), Revised Universal Soil Loss Equation (RUSLE), Water Erosion Prediction Project (WEPP)		

Note: (1) in areas with potential erosion problems (potential susceptibility for soil loss from water erosion); plots are useful to study differences in erosion among fields and agricultural practices , especially rill and interrill erosion (Cerdan et al. 2010) and to calibrate models; they are less suitable to derive representative erosion rates over landscapes (Boardman and Evans 2019).
(2) see Jackson et al. (1985), field guide for soil erosion see Mosiman and Sanders (2004), based on keys developed by Herweg (1996), work is usually supported by aerial photo interpretation.
Timing of field work: mapping of visible erosion features ((amount, width, depth, length) either after specified size of precipitation event (e.g. 10 mm/h), and/or once erosion features are visible. This means that Level II may need to be spatially flexible; however, also there, some statistical representativity for upscaling needs to be considered.
(3) Measurement of sediment flow at catchment outlet or hillslopes are not part of considerations here for large-scale (national and EU-wide) erosion monitoring

LUCAS Soil is an example showcasing options for Level I monitoring. During the 2018 survey, a visual assessment of erosion features has been introduced. This includes morphological features such as type of erosion (i.e. sheet, rill, gully, mass movement, re-deposition and wind erosion), number of rills or gullies, as well as distance to the plot center.

A good demonstration of erosion monitoring in the field has been presented by for a pilot area in Switzerland (Ledermann et al. 2010). The authors found that erosion damage mapping is well suited to assess rill erosion.

While soil erosivity can be monitored and updated based on changes in soil texture, bulk density and SOC, the dynamic aspect of the management factor remains a challenge. Vegetation cover could be derived from remotely sensed vegetation indices (vegetation density). Airborne geophysical methods like gamma-ray spectrometry and satellite imagery can be used to estimate morphological soil erosion features, erosion areas and sediment accumulation sites, but also soil exposure (Xu et al., 2019). Examples and more details about the use of satellite imagery for monitoring erosion can be found in King et al. (2005), Sepuru (2018), and Lukyanchuk et al. (2020).

7.3 Critical limits

Since it is almost impossible to stop soil erosion completely, the concept of **soil loss tolerance** is used. Soil loss tolerance can be seen as the maximum acceptable rate of soil loss, which is theoretically equal to the natural rate of soil formation (Morgan, 1986). If soil loss remains below this threshold, soil management can be considered as sustainable with regard to erosion. Natural rates of soil formation are variable and hard to measure. Morgan (1986) has proposed a commonly used mean soil loss tolerance of 11 t/ha for deeply developed soil. More recent knowledge has been developed which clearly concludes that this value cannot be considered a proper sustainability limit, at least for Europe (e.g. Steinhoff-Knopp et al. 2020). In Switzerland, regional tolerance threshold values for average soil loss on arable land have been developed (Schweizer Bundesrat, 1998, cited from Ledermann et al. 2008). For shallow soils < 70 cm depth, the maximum tolerable soil loss is 2 t/ha/yr; for deeply developed soils, the threshold is 4 t/ha/yr.

The relationship between soil formation and soil loss tolerance has recently been reviewed by FAO (2019). The study cites Montgomery (2007) who suggests a standard soil lowering of 0.08 mm with a soil loss of 1 t/ha/yr based on an average bulk density of 1.2 g/cm^3. Given the variability of soil structure (texture and packing density) as well as soil organic matter and other important soil properties, the observed soil lowering greatly varies. Based on the same author, the soil naturally develops at an average rate of 0.173 mm yr-1 (2.2 t ha-1 yr-1). Verheijen et al. (2009) has taken European data on soil formation to calculate a tolerable soil loss for Europe of between 0.3 to 1.4 t/ha/yr (soil lowering at 0.02 to 0.11 mm/yr), which reflects the best estimate for mean soil formation rates in Europe. Unfortunately, the authors do not provide guidance to determine the exact site-adapted limit value.

However, Morgan (1986) also recognises several problems with the concept of soil loss tolerance, and he recommends using the **rate of natural erosion** instead. This rate, according to him, would be of the order of 1-2 t/ha per year, while for Europe, it has been estimated to be between 0.3 and 1.4 t/ha per year (Verheijen et al., 2009). The latter range represents limits to maintain the biomass production function. Jones et al (2012) have considered soil loss of more than 1 t/ha/yr to be irreversible on human timescales, but also stresses that the concept of tolerable erosion rates requires further definition: acceptable rates might actually be variable across Europe, e.g. 1 t/ha/yr in one region, but perhaps 2-3 t/ha/yr in other regions. Renard et al. (1997) report that in the USA, tolerable soil loss is considered to be about 2-10 t/ha/yr, depending on soil type. Grimm et al. (2002) estimate a rate between 5 and 20 t/ha per year to cause serious impacts both on-site and off-site (downstream) while bigger soil losses (such as the ones caused by individual storms - 20-40 t/ha per year- or by extreme rainfall events – over 100 t/ha per year) can have catastrophic consequences both on-site and off-site.

As also highlighted by Stolte et al. (2016), the establishment of potential thresholds for tolerable soil loss is still very controversial. There is a noticeable variability in terms of critical values but also lack of clarity on the definition of tolerable and critical soil loss. Further research should be encouraged in this sense in order to build common and more solid basis for both aspects. Given the variability of soil type and climatic features in Europe, ideally, threshold values of erosion rates should be defined for different soil characteristics, land uses and climate zones. Furthermore, the availability of monitored data on erosion rates and the establishment of a comprehensive monitoring network would be ideal for the application of thresholds and development of the indicator.

The Swiss Ordinance on Impacts on the Soil (VBBo 2000) provides orientation values, beyond which site investigations occur and measures are to be taken: for soil solum depth < 70cm, acceptable soil loss is 2 t; at a depth >70 cm, the rate is 4 t/ha/yr (cited from BAFU 2017).

Finally, it must be noted that almost all soil loss estimates focus on soil loss by sheet and rill erosion, either measured on runoff plots or predicted with the RUSLE[49] or a RUSLE-type erosion model. Note that so far soil losses by ephemeral and permanent gully erosion, tillage erosion, harvesting erosion, piping erosion and erosion due to land levelling are not accounted for at the European scale (Poesen, 2018).

49 Revised Universal Soil Loss Equation (RUSLE)

8 Soil compaction

Soil compaction harms the physical structure of soils and thus affects important ecological and economic soil functions, by reducing pore volume and pore continuity as well as particle surface accessibility. As a consequence of compaction, hydraulic conductivity and infiltration are reduced and water logging often occurs, while rooting is hampered and the soil biological habitat is damaged. Operations at critical soil moisture levels, as well as the use of increasingly heavy machinery, cause compaction in the sub-soil. This is particularly critical, since at these depths, compaction is cumulative, persistent and, most likely, irreversible. Soil compaction negatively affects soil biochemical processes including nutrient turnover, greenhouse gas production, and plant health. Eventually, food and fibre production and groundwater recharge are concerned. Compaction is also known to trigger soil erosion.

Table 8-1: Relationship of Soil Compaction to key societal needs and soil functions

Soil compaction		
Societal need	**Soil service**	**impact**
Biomass	Wood & fibre production	-
	Growth and quality of crops	-
Water	Filtering and buffering of contaminants, incl. supply of drinking water	-
	Water storage and availability, groundwater recharge, surface runoff and interflow	-
Air	Composition and exchange of soil gas with the atmosphere	-
Climate	Carbon storage and turn over, Avoidance of climate relevant gas releases (e.g. N_2O, CH_4)	-
Biodiversity	Habitat for plants, insects, microbes, fungi	-
Cultural heritage	Documentation of historical human culture and land management	-
Infrastructure	Platform for infrastructure	indifferent
	Storage of geological material	indifferent

Soil compaction is primarily related to physical soil degradation, but the interactions with chemical and biological properties and functions are evident. Soil compaction occurs primarily if the internal soil strength (the so-called actual precompression stress) is exceeded by additional stress, for example, through heavy machinery, and trafficking at high moisture content. This exceedance results in plastic soil deformation which negatively affects the soil functions and the provision of ecosystem services. Precompression stress indicates the site-specific natural condition to carry and restore external mechanical force; it represents the condition at which soils are resilient and can be sustainably managed. Monitoring focusses on specific soil physical (functional) parameters which describe the mechanical behaviour of the soil.

8.1 Role and assessment of soil compaction

8.1.1 Background and status

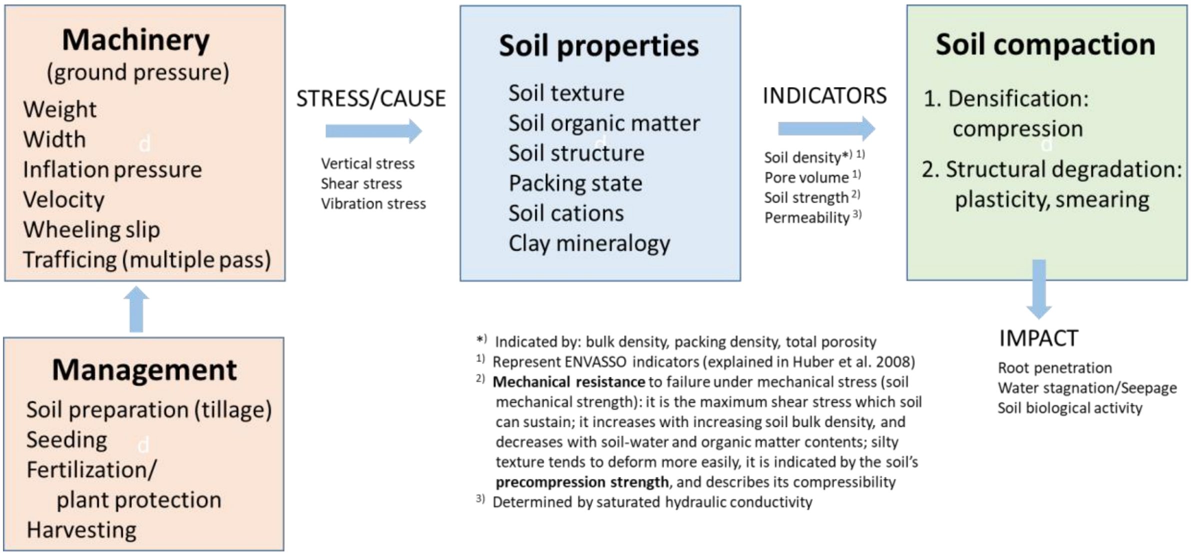
Soil compaction, in particular of the subsoil, is primarily induced by heavy machinery, often paralleled by increasing field size. Seasonal time constraints (independent of the soil moisture level), but also the operational conditions and limited knowledge of service providers (machine parks) appear to be additional pressures. The specific damage on soil is conditioned by the machine weight, contact area, number of passages and area coverage, but also shearing and soil smearing from wheeling slip (Keller et al. 2019, Horn and Peth 2011, Horn 2021). It is mainly the high wheel and axle loads of transportation vehicles and harvesters which result at given contact area to mechanical stresses, which exceed the resisting forces within soil, and which then create irreversible soil deformation and permanent compaction – especially in the subsoil, and when soils are wet and weak, and trafficking efficiency is low (Duttmann et al. 2014). Repeated trafficking generally results in cumulative soil compaction effects down to deeper soil depth and induces subsoil deformation of the pores and their functions if the given soil strength is exceeded by the applied stresses.

Based on the European data base of soil properties -SPADE8- (Koue et al., 2008), Schjønning et al. (2016) estimated that 23% of the total agricultural area of Europe has a critically high level of compaction. Other estimates suggest that between 32% and 36% of European subsoils are highly susceptible to compaction (Jones et al., 2012). Mordhorst et al. (2020) quantified the compaction status of 342 soil profiles, including both natural and potentially anthropogenic compaction. A harmful subsoil compaction between 20-40% was determined for (stagnic) Luvisols and Stagnosols, and an anthropogenic proportion of subsoil compaction with smallest values for air capacity and saturated hydraulic conductivity was found for at least 6%–10% of the area. Van den Akker et al. (2013) calculated that about 43% of the subsoils in the Netherlands are over compacted, while for the agricultural area in Central Switzerland, Widmer (2013) estimates that ca. one-third of area may have critically high soil densities.

Compaction increases the penetration resistance of soil, while root growth and biological activity, like frequency of earthworms, nematodes, or collemboles, are reduced (Gregory et al., 2007; Beylich et al., 2010; Schrader, 1999). The extent to which these changes occur depends on the stress intensity and duration as well as kind of stress applied (static or wheeling induced shear effects). Also, physicochemical processes are affected, such as redox potential, and related effects for the pH value in soil. Consequently, microbial composition can change from oxic to facultative anoxic to anoxic microbiological communities. Stress-induced formation of a platy structure favours horizontal fluxes in slopy area, so that water erosion and stronger and higher floods can occur (Horn et al. 2019; see also review by Alaoui et al., 2018 and Van der Ploeg et al., 1999, 2002).

Trafficking and its effects on compaction receive more and more attention in the development of solutions against compaction. Augustin et al. (2020) concluded from long-term observations, that between 82% (winter wheat) and 100% (sugar beet) of the total infield area is trafficked during a single season. Of that more than 15 % is repeatedly affected. The highest trafficking density is known for sugar beet or corn; there, harvest involves more frequent trafficking with high ground pressures, at a time during late autumn when a higher water content prevails, also at lower depths. During the last 40 years, increases in stress affected soil depth and decreases of the rootability were observed (Keller et al. 2019). The impact of these stresses depends on the soil internal strength (defined as precompression stress), for which site-specific thresholds can be defined so that further deformation at higher stress can be indicated and avoided (see also Figure 8-1).

Figure 8-1: Soil compaction processes, parameters and indicators



Without doubt, soil compaction is the most serious problem, and monitoring of soil compaction based on the indicators and impact effects is urgently needed in order to maintain the sustainable soil properties and their resilience for future generations and to avoid enhanced climate change or soil erosion and surface water pollution effects (Jones et al 2003, Batey 2009, Rogger et al. 2018, Horn 2021)

8.1.2 Soil compaction processes

Observation of soil compaction and the definition and selection of the proper indicator(s) require knowledge of the pressures on soil and its properties, and the spatio-temporal response processes in the soil (Table 8-2).

Table 8-2: Possible indicators for soil (sub) functions directly affected by soil compaction

Soil environment		Properties and indicators
Air regime	Air storage	Air capacity Bulk density
	Air flow	Air permeability O_2-diffusion Pore continuity
Water regime	Water storage	Available water capacity Bulk density
	Water seepage	Hydraulic conductivity (saturated/unsaturated) Pore continuity Flux directions: isotropy/anisotropy
Thermal regime	Heat storage Heat flux	Heat capacity and conductivity, Thermal diffusivity, Water content Pore continuity

Soil environment		Properties and indicators
Biological regime	Microbial composition	Number of species
	Abundance of functional species groups	Oxic/anoxic species and distribution
Physical soil regime: soil strength	Deformation status	Bulk density Proctor density[50] Average mean diameter of aggregates
	Stress-strain[51]	Precompression stress Crushing strength Shear strength Stress propagation Ratio of precompression stress and actually applied stress Changes in air, water, thermal flow processes and biological regimes
Soil production function	Rootability Nutrient availability	Root length - and root surface density Penetration resistance

Sources Amongst others: Horn and Fleige (2003,2009), Jones et al. (2003), Lebert et al. (2007), Lebert (2010), Schjonning et al. (2016), Keller et al. (2019), Horn (2021)

The strength of the soil, hence its capacity to resist to stress which can cause compaction, is greatly influenced by external as well as internal properties and functions – it is obvious that in spring after the snow melting or when soils are wet, soils are physically weak and more susceptible to deformation; during summer, soils dry out and hydraulic stress increases the rigidity of the pore system due to additional structure formation and strengthening.

Depending on the parent material and soil structure formation during its natural development (pedogenesis), soil has a natural range of **rigidity (= strength)**. Compaction occurs when the applied stress overcomes this strength - the **mechanical rigidity limits of the internal soil strength are exceeded**: the soil “fails”. This **internal strength** can be derived from so-called stress-strain curves[52], which define also the **precompression stress** for soils under consideration (Horn and Fleige 2009). This soil-specific relationship between stress and strain characterizes the level of natural compression stress, prior to the current compressed state (due to former stress applications). Only if this soil strength (defined by its precompression stress) is exceeded by actual stresses applied, soil functions deteriorate upon unsustainable soil management. The **precompression stress of a soil (horizon) therefore defines a degradation threshold** because it quantifies the rigidity limits for physical, compaction-related soil functions. The precompression stress as a threshold is consequently the basis for determining or adjusting soil management systems. **Exceedance of the soil’s precompression stress** (i.e., the actual soil strength) can be documented not only by a volume loss (soil subsidence) or increase in bulk density (densification), but more importantly, by changes of sub-indicators (presented here in the overall compaction assessment) which are more directly related to physical soil functions. Such sub-indicators are:

50 Proctor density defines the maximum bulk density at the optimal water content of the soil sample due to a given dynamic energy applied with a Proctor hammer.

51 Stress strain processes define the effects of stress application on the increase in soil particles per soil volume, the changes of pores (diameter and amount) and coinciding air, water, thermal, biological regimes and plant growth.

52 Stiffness of a soil, characterized by a soil-specific relationship between stress (force per unit area) and strain (change in size or shape)

- decreasing with compaction
  - air permeability
  - gas diffusion
  - (saturated) hydraulic conductivity
- increasing with compaction
  - heat flux at a given matrix potential[53]

They depend on natural soil processes including texture, structure, organic carbon, chemical properties and can either be already very low (as a natural site property) or they exceed acceptable values due to soil deformation. Physico-chemical parameters, like redox potential, biological composition, and microbial abundance are also altered (Horn 2021). Consequently, the link between physicochemical and biological processes are indicated through different gas emissions from the soil (e.g., increased N_2O emission). Beyond the threshold value (the soil strength) are all changes irreversible and soil amelioration necessitates decades to improve soil functions, what is documented by long lasting changes of the corresponding values.

Change of soil strength often occurs when soils are moist or wet and/ or under mechanical stress when the natural aggregate strength is exceeded, or, when the soil strength (or: rigidity of the pore and soil structure system) is low compared to the applied stress (Horn et al. 2014). The extent to which these changes occur depends on the stress intensity and duration as well as kind of stress applied (static = vertical loading or wheeling-induced shear and strain effects[54]). Among other effects, plastic deformation and consecutive stress release moreover induce the formation of a platy soil structure which then results in prevailing horizontal water fluxes (Horn et al. 2019). Such decline in soil structure impacts water erosion and flooding risk, especially in areas with prevailing fine textured soils with a typically low infiltration capacity (Alaoui et al. 2018).

8.1.3 Subsoil compaction

When the internal soil strength is exceeded during wheeling, animal trampling, or continuous loading, soil deformation occurs down to depths until an equilibration between external stress and internal strength is reached. Thus, both topsoil and subsoil are affected. The **subsoil** in agriculture is defined as soil below the tillage depth (usually around 20-35 cm). However, while compaction in the topsoil can be mitigated through effective management (like ploughing or chiselling) or through natural processes (like soil biota activity, swelling and shrinkage or temperature changes: freeze, thaw), the damage to the subsoil is particularly relevant since, at these depths, compaction is cumulative and persistent over decades or maybe even centuries (Wiechmann 1995, Keller et al. 2019). Subsoil compaction is hence the main responsible factor for soil degradation, having a persistent impact on other soil threats and functions, too.

8.2 Indicator specifications

8.2.1 Physical soil functional parameters and indicators

Indicators on compaction for soil monitoring were amongst others also suggested by Huber et al. (2008). In the absence of data on actual soil compaction, Huber et al. (2008) have suggested to spatially predict

53 Soil matric potential (SMP) indicates the soil water which is held by the soil matrix (soil particles and pore space), and which is the more negative the finer the pore diameter. It also defines the plant available water range as well as the air capacity or the field capacity.

54 Static (= vertical) loading results in a three-dimensional soil displacement with a preferential dominance in the vertical direction, while wheeling additionally induces three-dimensional displacement: a forced vertical and a more prominent lateral as well as tangential particle movement due to sliding. The latter causes the blockage of pores apart from the reduction in pore diameter or even a complete closure.

the **vulnerability of soils for compaction** by (a) the actual water saturation or its binding forces within the pores (defined as matric potential), (b) the initial drainage condition and (c) the bulk density. However, such estimates provide only very rough information about where soils are over-compacted (van den Akker et al. 2013). Therefore, here, indicators are suggested which are sufficiently sensitive to document and to quantify the intensity of soil compaction and the consecutive effects on soil functions. The application of thresholds available to at least some of these indicators offers orientation to avoid further soil deformation, and to select soil restauration approaches. Indicators are presented in two sets, according to their precision, explanatory power, and easiness of obtaining the input data.

- **Indicator subset I,** indicators can be easily measured or mostly available in regular soil monitoring
  - dry bulk density
  - air-filled pore volume
  - soil texture
  - visual features of compaction

Bulk density

The bulk density D_b defines a mass of dry soil material per volume. The values depend on texture, aggregation, organic carbon content as well as in situ water drainage and anthropogenic, geogenic or pedogenic processes. D_b is a parameter with high spatial and temporal variability. The compaction-sensitive parameter While bulk density (ρ_{Bf}) is compaction-sensitive, it is nevertheless considered a rather unspecific parameter, because it describes only volume changes but do not quantify the potentially negative impacts on pore functions. Thus, there is no direct link to soil strength or compaction. Measurement of D_b can be furthermore misleading because sampling in dry, strongly rooted and stony soils is difficult. Irrespective of these limitations, it is often used to estimate soil compaction. Packing density (PD) is sometimes used instead of D_b it is derived as a function of bulk density and clay content in order to indirectly evaluate the aggregate formation. However, this value has no easily comparable dimension.

Pore volume

The pore volume is directly related to the bulk density given the values for the specific density of the mineral soil components have been previously determined or estimated, depending on the parent material (texture of the weathering product), clay mineralogy, and soil organic carbon content.

Air capacity

Air capacity: the air-filled pore volume *AC (%)* is a measure of the degree of densification which has a strong relationship with aeration and functioning of the root zone. It is most often determined as the difference between water content at saturation (= total pore volume) and the volumetric water content at -6kPa; if other desiccation intensities (e.g. -5kPa) are used, it needs to be documented. Air capacity depends naturally on texture, soil aggregation (structure), and soil organic carbon content, and is further modified by anthropogenic, geogenic and biogenic processes. The air capacity can be monitored by (a) comparison of a current measurement to an initial measurement (as a reference value), (b) comparison to an undisturbed site-specific value, or (c) applying a threshold to be expected at a specific soil (Wösten et al. 1990).

Visual soil evaluation

Penetration resistance (penetrometer): With this approach, the rootability of a soil is described. However, there is no clear dependency between penetration resistance, other visual monitoring, and soil functions: a well-structured soil can have a high penetration resistance although the rootability may be still very good. Nevertheless, penetration resistance can provide a rough estimation of soil compaction effects. For example, it is lower for conservation agriculture, especially under zero tillage, compared to conventional management; soils become better rootable and macroscopically well-aerated, while they are at the same time mechanically very strong. It is important to consider that penetration resistance is best determined at “field-capacity”.

Spade diagnosis (VESS: Visual Evaluation of Soil Structure): VESS is a method to detect changes in **packing density**[55]; the method is described by Diez and Weichelt (1997, in German) and Ball et al. (2017). The soil structure is classified in 5 classes from 1 (= very loose structure) to 5 (= very dense structure).

- **Indicator subset II,** indicators with well-defined physical units like hPa, kPa or MPa, and a strong dependency on the actual water saturation, soil structure, pedo- and anthropogenic processes. These indicators can be linked to the actual and dynamic changes of gas, water and heat fluxes in soils as they are sensitive to document the consequences of soil compaction and soil degradation on physical, chemical and biological functions
  - precompression stress (kPa)
  - contact area pressure (kPa)
  - soil rigidity (-)
  - Shear strength (kPa) (stiffness)
  - Hydraulic conductivity (K) (cm/d) and air permeability (Kl)

Precompression stress

The precompression stress (= **internal soil strength**) is a sensitive and scale-spanning parameter that defines the **rigidity** of soil. It indicates the current state of compaction, as a result of all previous physical, chemical or biological compressive and stabilizing processes as well as natural decompression (loosening such as bioturbation). It is derived from **stress strain curves** as transition from the recompression to the virgin compression range and depends on the soil’s matric potential, as well as former pedo- and anthropogenic processes. The higher the soil strength, the lower the likelihood for additional mechanical stress, and long-term degradation of soil structure (Horn et al. 1989, van den Akker et al. 1998), Trautner et al. (2003), Horn and Fleige (2009), Keller et al. 2019). The values of the precompression stress and the stress-dependent changes of these properties and functions are under laboratory conditions often quantified when the soil is most sensitive (usually in early spring at matric potential values of pF 1.8 =-60 hPa matric potential), or when drying due to evapotranspiration reduces the soil water content (like pF 2.5 or -300 hPa matric potential). The precompression stress i.e. the strength defines the threshold as scale dependent value for single soil horizons to bulk soils, soil distributions within given geological origin up to country or continent scale or e.g. at given land use managements. The PTF´s to quantify the precompression stress are described, amongst others, in Horn and Fleige (2009) and Simota et al. (2005).

Contact area pressure

The contact area pressure defines the stress transmitted into the soil as a function of the load applied (e.g. of the machines, animals etc) and the corresponding contact area of the tires, hooves etc. At a given contact area pressure is the stress transmission into the soil the deeper the greater the contact area (Horn 2015).

Soil rigidity ratio

The ratio between the actual precompression stress (= internal soil strength) and the actual soil stress applied by machines, animals or permanent loads i.e. the contact area pressure, defines the soil sensitivity for changes in the physical, chemical and biological functions. Values > 1.2 define rigid soil structure conditions with no compaction processes, while values < 0.8 define structure as irreversibly deformable. Values in between classify soil properties and functions as very susceptible for further soil deformation. Thus, in order to properly interpret the soil-related indicators (subsets 1-2), the applied external stresses

55 Packing density (a dimensionless value) is defined as the sum of the bulk density and a percentage of clay in order to indirectly include the aggregate formation effects.

by machines need to be monitored and set in relation to the internal soil parameters initially and the changes due to the applied stress. Combining soil strength and management-dependent pressure as an indicator allows to define sustainability or resilience limits like those in Table 8-4 (see Horn et al., 2005, Horn and Fleige 2011).

Shear strength

Shear strength or the stiffness of soil determines the binding forces between particles (texture) or soil aggregates to withstand the rearrangement (= strain[56]) due to smearing (also defined as slip) The pore functions within the soil will be affected due to such particle arrangement.

Hydraulic conductivity (K) and air permeability (K_l)

The saturated or unsaturated hydraulic conductivity as well as the air permeability are sensitive indicators **and** represent the functional quality of soil structure and pore continuity, depending on the matric potential. Both air permeability and hydraulic conductivity can be used to determine trafficability. The saturated hydraulic conductivity (K_s) primarily depends on all saturated macro pores while the unsaturated hydraulic conductivity as well as the air permeability also quantify the fluxes within the various pore diameters. The number of blocked pores which cannot contribute anymore to mass exchanges, can be adducted to also document the slip as well as smearing effects apart from densified aggregates. Data for local sites, for small scale evaluation up to the EU scale can be derived from existing databases like national soil mapping instructions (Ad hoc AG Boden 2005; Simota et al., 2005; Wosten et al., 1990).

8.2.2 Suggestions to include compaction indicators in a Tiered monitoring

Depending on the different sampling and analytical requirements of the indicators mentioned above, different intensity levels for monitoring (Tiers) are recommended.

Table 8-3: Design of large-scale soil compaction monitoring

Compartment	Measurement and estimation parameters		
	Level I	**Level II**	**Level III – Wheeling plots and unloaded reference plots**
Location of sampling	At the field hot spot with visible marks of compaction:		Representative sub-plots throughout a given field surrounding the plot center
	e.g., reduced vegetation cover or growth, puddles	Proportion of affected area around the plot center	
Direct and indirect monitoring of soil compaction	Morphological features (water logging, (platy) soil structure, rooting)		
	Precompression stress (estimated) (1) soil rigidity ratio 2)		Samples are measured at defined matric potential Contact area pressure of the machines and the actual contact area are determined
	Penetration resistance (3) (**estimated** with Pedo-transfer functions PTF)		Measurements of depth dependent PR at given matric potential
Basic soil physical parameters	Saturated hydraulic conductivity, air capacity, plant available water capacity (**estimated** with PTF, soil data	All basic soil physical parameters for PTF are **measured**.	Tensiometer, sensors, actual soil sampling at defined depths Stress dependent changes of the parameters are measured under in field and under lab conditions

56 strain is a measure of deformation representing the displacement between particles at a given stress applied. It is defined e.g. as height change, void ratio

	sets, LUKAS Soil, Wosten, SIDASS)		
	Bulk density (estimated or measured)	Bulk density (measured)	
Basic soil chemical parameters	soil texture/coarse fragments/$CaCO_3$ (estimated – soil auger)	soil texture/coarse fragments/ CaCO3 (measured – soil profiles)	
	soil organic matter (measured)		
Biological parameters	rooting	parameters must be measured	
	Biological activity (bioturbation)		
Depth	Soil surface, upper boundary of lower soil horizons (or simply topsoil and subsoil)	Refined depth classes/by genetic horizon	depths of 40–45 cm and 60–100 cm
Repetitions	4-8 samples per depth		10 - 20 samples per parameter and depth
Operations	Field traffic: percentage of the wheeled area, number of wheel-to-wheel passages		Weight, air pressure, wheel type, axle and tire widths of every vehicle, contact area
Seasonality of monitoring	Spring sampling (soil at field capacity)		Sampling at requested times throughout the year

Note: (1) Precompression stress derived from Pedo-transfer function (PTF) for a given texture and aggregation, acc. to Horn and Fleige (2003): requires pore size distribution, hydraulic conductivity, and soil chemical soil properties, in areas where this approach is not calibrated, horizon specific stress strain measurements of undisturbed soil samples at a given matric potential are needed, and confined shear tests are needed to determine the shear strength of a given structured soil
(2) ratio [precompression stress / actual stress imposed by field traffic] (see also Duttmann et al. 2014)
(3) establish reference sites from undisturbed, uncultivated sites

At Level I, easily and commonly determined soil parameters are used to define the soil compaction probability, while the application of more detailed measurement data appears at a higher Tier (Level II, likely to occur on less plots compared to Level I). At Level III, the most definite estimate of compacted area can be generated, based on more precise measurement techniques and very detailed soil physical analyses. Table 8-4 therefore provides an overview of the different levels to soil compaction monitoring. More detailed descriptions of key indicators are already given in Section 8.2.

8.3 Critical limits

The issue of soil degradation due to compaction and deformation needs to be addressed from two sides:

- by evaluating the soil's state by means of stability or rigidity (precompression stress), as well as physical parameters related to soil functions (hydraulic conductivity, air permeability and air capacity), and
- by determining the ratio of incoming stresses and soil strength, and its effect on physical, chemical and biological properties for the definition of soil degradation (Riggert et al. 2019).

In order to achieve both objectives, the following indicators (s. also Table 8-4) are suggested:

- precompression stress,
- ratio of precompression to actual stress applied,
- air capacity, and
- saturated hydraulic conductivity.

Table 8-4 summarizes both above-mentioned indicators sets. While the first set is based on easily measured or mostly available soil data, the second set refers to well-defined physical units and are closely related to the actual water saturation, soil structure, as well as pedo- and anthropogenic processes. The second set is therefore better suitable to quantify and to also document stress-induced changes in soil functions, such as water, gas and heat fluxes as well as biodiversity effects and physico chemical processes like redox potential changes.

Table 8-4: Soil physical indicators for detecting harmful compaction in the subsoil

Indicator	Explanation and thresholds	Soil sensitivity
Indicator set I		
Bulk density	bulk density values between<1.2 and 1.6 g/cm³ define very loose to normal soil conditions with no or only minor root penetration problems while values > 1.6 - >1.9 g/cm³.represent dense to very impermeable soil conditions.	Soils originating from clay to silt and sand; higher values are due to geological prestressing or anthropogenic impacts.
Air capacity: air-filled pore volume AC pore volume = total volume of pores per bulk soil	A low air capacity impairs root growth and a reduction of oxygen pressure in soil air and an increased formation of climate change relevant gases. Below 5% AC at soil matric potential of -6kPa are aeration or gas diffusion mostly insufficient The pore volume is the greater the finer the particles, the more aggregated and the higher the organic carbon content. (Values below 35% are in generally defined as critical)	Soils originating from clay, loam, silt and sandy loam and sandy loess
Visual soil evaluations	Aggregate type and estimated BD Root growth/ Penetrometer Spade diagnosis The visual assessment of the soil as lose or dense based on aggregate size and strength, pore size and continuity, root density and distribution	Additional assessment for all soils
Indicator set II		
Precompression stress (=internal soil strength)	At low precompression stress (= low internal soil strength e.g. because of weak aggregation or wet soil conditions; very low < 30kPa, low 30-60kPa,) are soils very sensitive to further deformation and decline of physical, biological and physico-chemical functions. Medium: 60-90kPa,or high 90-120kPa allow a more sustainable soil management, if the applied stress is below these values at the depth under consideration.	All soils, but especially loamy, silty and clayey soils
Ratio of precompression stress and actual stress applied	Values > 1.2 define rigid soil structure conditions with no risk to compaction processes, while values < 0.8 define structure as irreversibly deformable. Values in between classify soil properties and functions as very susceptible for further soil deformation.	All soils, but especially loamy, silty and clayey soils, at high water content and weak aggregation
Shear strength	Shear forces due to wheeling result in smearing: the shear strength is smaller for less aggregated soils and decreases with increasing water content. Shearing is more pronounced at higher slip, especially when soils are moist.	All soils, but especially loamy, silty and clayey soils, at high water content and weak aggregation
Saturated /unsaturated hydraulic conductivity	Low conductivity is typical for stagnic soil conditions: delayed percolation impacts soil aeration and groundwater accumulation as well as enhanced surface runoff (critical values are defined below 10cm/d)	

Air permeability and oxygen diffusion	Low air fluxes coincide with retarded gas exchange and the formation of anoxic conditions through CH_4 or N_2O formation	All soils, but especially loamy, silty and clayey soils, at high water content, and weak aggregation due to tillage or soil management
	Critical values for air permeability <12* 10^{-4} cm/s	
	diffusion coefficient (D_s) < 1.5 10^{-8} m^2 s^{-1} (Bakker et al. 1987)	

Source: Lebert et al. (2007), Huber et al. (2008), supplemented with additional information based on the review in this report

8.4 Tools to monitor soil compaction

The following monitoring methods enable the evaluation of soil compaction, its intensity and distribution in space and depth. Besides a pragmatic tool is presented using indicators which can be determined or obtained from national and internationally available data bases, the actual soil functional behaviour to mechanical stresses can be best assessed using in situ and lab measurements or derived pedotransfer functions, which are available for state, national or EU regions like Ad hoc AG Boden (2005), Wösten et al., (1999), Horn et al., (2005), LUKAS Soil data base). They can be used as input parameters for process-based models, which include more detailed mechanical properties.

8.4.1 Soil compaction models

The prediction of soil compaction can be approached by several models, which use soil parameters and indicators to directly describe the mechanical soil properties and related soil processes and functioning. Table 8-5 provides an overview of the most common models.

The FEM coupled process model requires well defined soil mechanical data like bulk modulus, shear modulus or shear strength, which need to be derived site specific either from very sophisticated triaxial tests or derived from stress strain and shear strain curves. It predicts stress distribution as a function of soil strength as well as the soil deformation and changes in pore continuity due to stress propagation.
The following models are all restricted to predict soil stresses under wheel loads including the 3D stress propagation but the consequences of the applied stresses on soil functions are not in the foci.

Table 8-5: Models to predict subsoil compaction

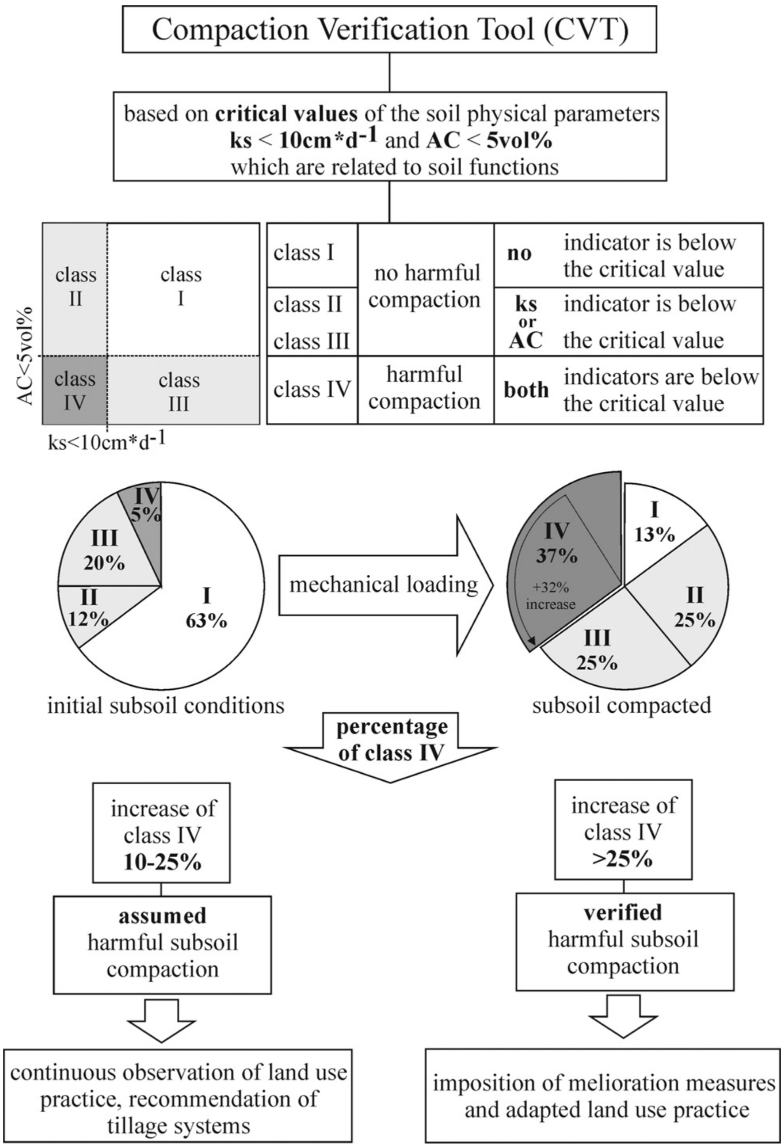
Model	Content	Source
FEM Coupled Process model	Modelling of stress distribution based on mechanical properties and options to link the stresses with physical soil indicators	Richards et al. (1997), Gräsle (1999), Richards and Peth (2006)
SOCOMO	Stress calculation and comparison with internal soil strength	Van der Akker (2004)
Soil flex	Analytical model to predict stress propagation in soils.	Keller et al. (2007)
Terranimo	Open source manual for site specific data analysis for given soil properties and mechanical impacts as a tool for practitioners	Stettler et al. (2014)

8.4.2 Compaction Verification Tool (CVT)

In order to evaluate the actual soil stability and the risk of stress-induced soil degradation, Zink et al. (2011) have developed the Compaction Verification Tool (CVT), which include stress dependent changes in soil functions described as indicators in Figure 8-2. It s based on measurements or estimates of saturated hydraulic conductivity K_s and air capacity (at -60 hPa) as a function of actual stress applied within the virgin compression stress range (see also Table 8-3 and Table 8-4). Suggestions to quantify these sub-indicators are described in Horn and Fleige (2009) for texture classes from sand to clay and based on a large variety of soil profile data.

The proposed minimum values of **class I** (air capacity AC > 5 vol %, saturated hydraulic conductivity K_s > 10 cm/d) represent soils which still function properly; it assumes that the rigidity limits (precompression stress) are not exceeded and/or the texture, organic carbon etc. guarantee these values. The values in **class II** (air capacity >5% and saturated hydraulic conductivity (K_s) < 10cm/d as well as **class III** (AC < 5%, K_s > 10cm/d) define the **"precaution value" (PV)** (no harmful compaction yet), while the values AC< 5% and K_s < 10cm/d in **class IV** are associated with yield decline due to lack of aeration, prevention of gas exchange, and/or stagnant water problems in the soil and, thus, correspond to **"action values" (AV)** (= harmful subsoil compaction).

Figure 8-2: Diagnosis of soil compaction based on threshold exceedance



Source: Zink et al. 2011

In order to promote sustainable soil management practices both in agriculture or forestry, in particular to protect soils from degradation through the actual tillage systems, tree harvesting, and machinery impacts at given water content, the CVT (as '*good*', i.e. class I, or '*acceptable*', like class II and III) can be combined with a traffic light system (Riggert et al. 2019), and can thus be connected with model approaches presented above (e.g., Terranimo) with respect to the applied stresses in relation to soil strength.

It is most likely, that the monitoring is applicable to all scales, and the necessary data for the air capacity as well as the saturated hydraulic conductivity are mostly available for representative soil profiles (level III) or can be derived from existing data bases (e.g. Woesten et al, 1999, or national soil mapping datasets) while the corresponding precompression stress data as threshold values can be derived from PTF´s (Horn and Fleige 2009) or detailed in situ measurements (Level III). The quantification of stress implications on the 2 soil indicators (air capacity and hydraulic conductivity) beyond the precompression stress results from PTF´s (Horn and Fleige 2009) or site-specific measurements in combination with wheeling experiments (Level III).

While CVT allows to map harmful subsoil compaction, the anthropogenic subsoil compaction still needs to be separated from the natural compaction as a result of geogenic and/or pedogenic processes ("initial subsoil condition"; see Figure 8-3). Especially fluvic and stagnic soils tend to have a high degree of natural compaction (46%–65% of the mapped fields in a German case study area), compared to < 13 % for Podsols and Arenosols) (Mordhorst et al., 2020). Anthropogenic compaction is found if the selected compaction-sensitive parameters air capacity (AC) and saturated hydraulic conductivity (Ks) are larger in the subsoil compared to the topsoil. The successful application of this threshold thus requires a horizon-specific analysis (topsoil/subsoil), calibrated with the knowledge from a large regional pedological data base (Level I and/or Level II as presented in Table 8-3).

9 Soil sealing

Soil sealing corresponds to an irreversible loss of soil and its biological functions and loss of biodiversity. Since the turn of the century annual soil loss in Europe ranges between to 300 to 500 km². This chapter presents available indicators and discusses the implications of baseline and threshold definitions for soil sealing. In contrast to all other soil quality indicators presented here in this report baselines and thresholds for soil sealing are not soil science based but policy based.

Soil sealing fulfils the societal need for infrastructure but has negative impacts on all other societal needs and soil functions.

Table 9-1: Relationship of Soil sealing to key societal needs and soil functions

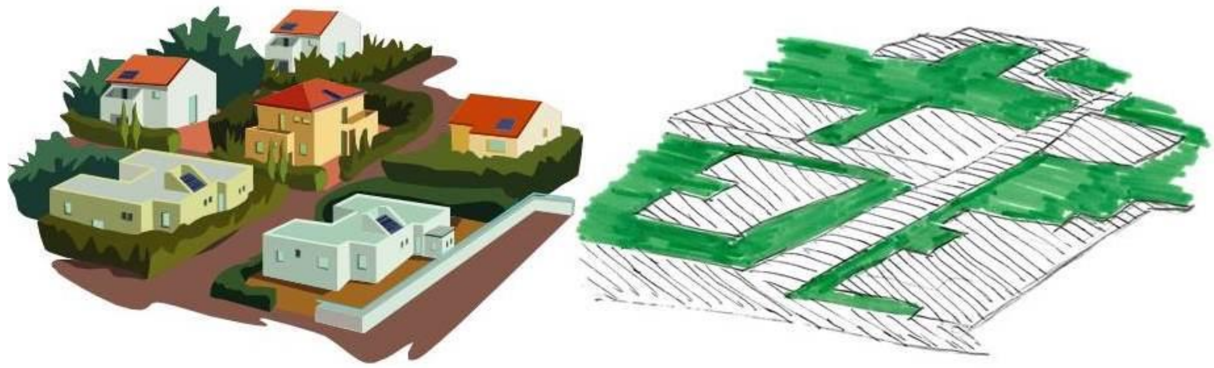
Soil sealing		
Societal need	**Soil service**	**impact**
Biomass	Wood & fibre production	-
	Growth of crops	-
Water	Filtering of contaminants	-
	Water storage	-
Climate	Carbon storage	-
Biodiversity	Habitat for plants. insects. microbes. funghi	-
Infrastructure	Platform for infrastructure	+
	Storage of geological material	+

9.1 Rationale “Soil sealing”

Soil sealing can be defined as the destruction or covering of soils by buildings, constructions and layers of completely or partly impermeable artificial material (asphalt. concrete. etc.). It is the most intense form of soil degradation and is essentially an irreversible process (Prokop et. al. 2011).

Soil sealing accompanies land take; the latter is commonly used to specify urbanisation, expressed as the increase in artificial surface. Land take is usually realised at the expense of cropland or grass land, and in some cases also forest land. The relationship between land take and soil sealing is illustrated in Figure 9-1. Areas subject to land take are not entirely sealed. Sealing rates are usually low in peri-urban areas with on average 10 % and very high in core cities with on average 36 % (Naumann et al 2018).

Figure 9-1: The relationship between land take (left) and soil sealing (right. hatched surfaces)



Urbanization affects soils in different ways: soils can be fully or partially removed, and substituted by construction material, waste, or mixed material (artefacts. debris); it can be sealed (asphalt) or covered with more or less penetrable surfaces. While the urbanization rate (total land take) may be high. the amount of soils sealed, or removed might be decreasing. For this reason it is important to monitor soil sealing with as much spatial accuracy as possible.

9.2 Indicator specification

Soil sealing is usually calculated as percentage (sealed area per total area) or as sealed area per capita for a given region or country. The EEA indicator "Imperviousness in Europe" has been widely used as soil sealing index (impervious soil coverage). Current methods to measure soil sealing are:

9.2.1 Satellite methods

The most common method to measure soil sealing is based on different reflection behaviour of sealed and unsealed surfaces. This so called NDVI method (Normalized Difference Vegetation Index) quantifies vegetation by measuring the difference between near-infrared (which vegetation strongly reflects) and red light (which vegetation absorbs). The NDVI method is used to measure vegetation, drought, but also sealed surfaces.

Healthy vegetation (chlorophyll) reflects more near-infrared (NIR) and green light compared to other wavelengths, but it absorbs more red and blue light (thus. vegetation appears green). Satellite sensors like Landsat and Sentinel-2 both have the necessary data.

Calculations of NDVI for a given pixel always result in a number that ranges from minus one (-1) to plus one (+1); however, no green leaves would provide a value close to zero; zero means no vegetation, approaching +1 (0.8 - 0.9); it indicates the highest possible density of green leaves.

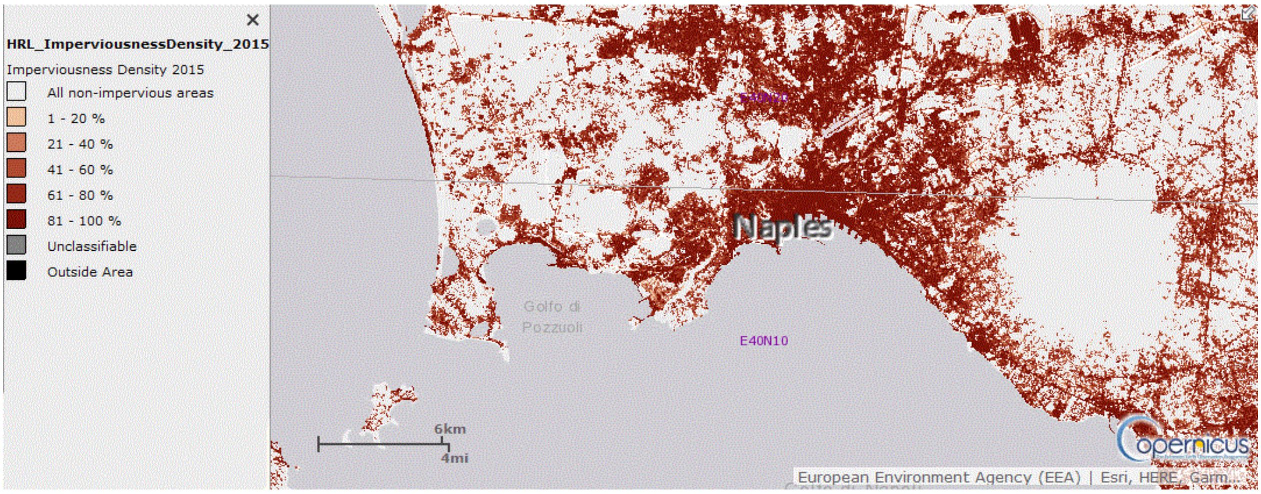
At European level data, readily evaluated data on soil sealing are available through the Copernicus Land Monitoring Service. The indicator "Degree of imperviousness" (or: Imperviousness in Europe) describes the area sealed as defined above. It contains the counts of pixels of impermeable soil cover (thus soil sealing). and are mapped as the degree of imperviousness (0-100%). Imperviousness change layers were produced as a difference between the corresponding reference dates and are presented as degree of imperviousness change (-100% --+100%). Date are available:

- On a three year basis since 2006; namely 2006. 2009. 2012. 2015. and 2018 (in preparation).
- With a resolution of 20 m x 20 m. and from 2018 on with a higher resolution 10 m x 10 m.

- Change layers are available for the periods 2006-2009, 2009-2012, 2012-2015, and 2006-2012. They are however based on a coarser resolution. The imperviousness change value is based on imperviousness layers with a resolution of 100 m x 100 m

Based on the above mentioned data sets from the Copernicus Land Monitoring Service regular European assessments are published by the European Environment Agency under the title "Imperviousness and imperviousness change in Europe" (EEA 2020). Data are available in interactive format as maps and tables for the above mentioned reference years in absolute values or as changes for defined time periods [57].

Figure 9-2: Example from the interactive data platform showing accounts of land surface sealing status and change in Europe (EEA39 and EU28) for every 3 years between 2006 and 2015.



Source: EEA indicator "Imperviousness in Europe"

9.2.2 Computation based on land use data from cadastres or aerial pictures

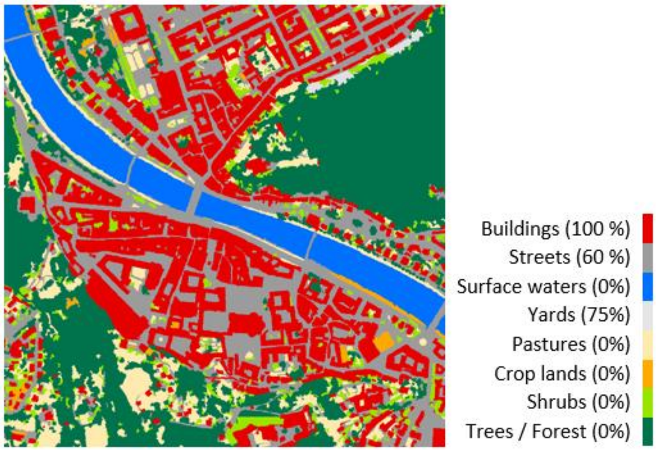
A simpler method to measure soil sealing is to use land use data from cadastres or aerial pictures. and to apply standard sealing indices for specified land use classes. Standard indices are derived from multiple sampling and calculating average values. This method can be easily used for measuring soil sealing in smaller regions or for specific projects but can also be used to perform random tests to validate satellite data for soil sealing.

Table 9-2 shows an example for this method. On the left hand side standard sealing indices for specified land use classes are indicated. On the right side there is a map with the same land use classes visualised as polygons with colour codes. The overall sealing rate can be calculated by summarizing the sealing rate of each polygon.

57 European Environment Agency, interactive data viewer: "Imperviousness in Europe" https://www.eea.europa.eu/data-and-maps/dashboards/imperviousness-in-europe

Table 9-2: Example for computing soil sealing based on land use categories from the cadastre.

Land Use Category	Sealing rate [%]
Buildings	100%
Yards next to buildings	75%
Gardens	0%
Streets	60%
Parking Areas	80%
Rail tracks	50%
Commercial areas	60%
Quarries and waste sites	10%
Recreational areas	20%
Grave yards	35%



Sealing rate = Σ (area buildings) + Σ (area yards) x 0.75 + Σ (area streets) x 0.6 ……./ total area.

Source: Monitoring of soil sealing in Austria

9.2.3 Comparison of national and European monitoring of soil sealing

While land take is regularly monitored at national level for all EU countries, soil sealing is determined by only very few countries with surveys other than Copernicus. Table 9-1 indicates available national soil sealing data for the year 2015. National data refer generally to higher sealing rates, which leads to the conclusion that EEA-Copernicus data do not capture smaller structures and therefore underestimate soil sealing at large. Table 9-3 shows three examples based on aerial pictures and indicates which structures were not captured by the EEA-Copernicus layer.

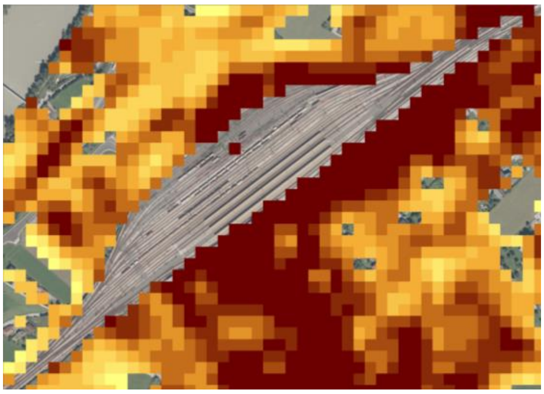
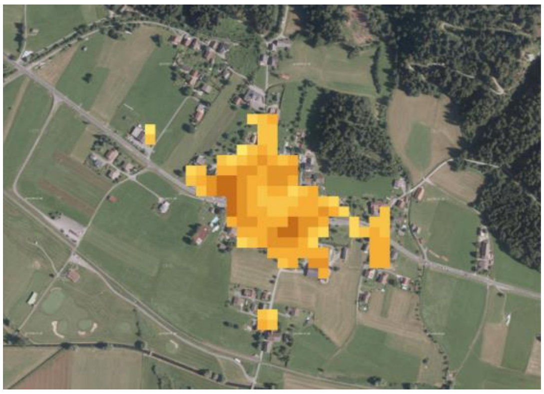
Table 9-3: Available national soil sealing data compared to Copernicus-EEA data (both 2015)

		SOIL SEALING IN 2015			
Country	**country size**	**national method**		**EEA / COPERNICUS**	
Belgium - Flanders	13 625 km²	1 935 km²	14.2 %	12 12 km²	8.9 %
Austria	83 882 km²	2 298 km²	2.7 %	1 475 km²	1.8 %
Luxembourg	2 593 km²	(1)176 km²	(1)6.8 %	49 km²	1.9 %

Note: (1) refers to year 2018 as there is no data for 2015

Figure 9-3 shows a quality check regarding the 2015 high resolution layer for soil sealing. It is obvious that smaller structures, like disperse single family houses and smaller roads are not captured by this data set. However, it can be expected that the higher resolution of the new data set (from 2018 on) will overcome this deficiency.

Figure 9-3: Documentation errors of EEA-Copernicus high resolution layer "Degree of Imperviousness 2015"

Aerial picture	EEA-Copernicus data set 2015 (yellow-red)	Comment
		Railway tracks associated to other impervious surfaces are not mapped
		Scattered houses around a village are not or only randomly selected
		Typical landscape with scattered small settlements and farm houses, only a small part of farm-houses mapped

Note: The squares on the right hand side indicate the sealing density, ranging from pale yellow (1 – 20%) to dark red (>80% sealing)

Source: National HRL verification report

9.3 Baselines and target values

According to the current state of knowledge. Baselines and thresholds for soil sealing are not soil science based but policy based. They refer to defined geographical regions and a target year. Also it can be observed that soil sealing is usually implicitly included in targets to reduce land take.

The baseline is usually a reference year, and the target value refers to a target year and a defined soil sealing or land take rate for a defined region or country. The rate for soil sealing or land take is usually expressed in “hectares per day” as annual average. To give a few examples see Table 9-4:

Table 9-4: Currently applied targets and baselines for soil sealing/land take in selected European countries.

Target	Indicator	Source
Achieve no net land take by 2050	km^2 land take per 3 year period	European Union. Road Map for a Resource Efficient Europe[58]
To decrease land take gradually: 2016: daily land take 6ha per day (baseline) 2025 interim target 3ha per day 2040 final target 0ha per day / “land take neutral”	average annual land take measured in hectares per day	Flanders. Strategic Vision Spatial Policy Plan of Flanders[59]
To reduce annual land take to a rate of 2.5 hectare per day by 2030 and to compensate unavoidable soil sealing.	average annual land take measured in km^2 per year	Austrian Government Programme 2020-2024[60].
To reduce land take for settlements and traffic routes to less than 30 ha/d by 2030 (at present: about 60 ha/d).	average annual land take measured in hectares per day	German Sustainability Strategy 2016[61]
To reduce land consumption from 1.3 ha/day (average 2000 – 2006) to 1 ha by 2020, and 0 ha by 2050.	average annual land take measured in hectares per day	Luxembourg[62]
To halve land take at the expense of agricultural land until 2020 and reduce urban sprawl	average annual land take measured in thousand hectares per year in metropolitan areas	France[63]

According to the current state of knowledge thresholds for soil sealing, for instance, for a defined land use pattern (core-city, peri-urban area, rural area), have neither been defined nor implemented. In practice soil sealing is monitored through land take, corresponding indicators are indicated in the second column of Table 9-4.

58 p15, milestone 4.6; COM(2011) 571, Roadmap to a Resource Efficient Europe

59 p36, Strategic Vision of the Spatial Policy Plan of Flanders, available at: https://www.vlaanderen.be/publicaties/beleidsplan-ruimte-vlaanderen-strategische-visie-geillustreerde-versie [last accessed on 17 August 2020]

60 p 104, Austrian Government Programme 2020 – 2024, available at: https://www.bundeskanzleramt.gv.at/bundeskanzleramt/die-bundesregierung/regierungsdokumente.html [last accessed on 17 August 2020]

61 German Sustainability Strategy 2016, available at https://sustainabledevelopment-deutschland.github.io/en/11-1-a/ [last accessed on 17 August 2020]

62 p35, Un Luxembourg durable pour une meilleure qualité de vie (2010), available at: https://environnement.public.lu/dam-assets/documents/developpement-durable/Un-Luxembourg-plus-durable-pour-une-meilleure-qualite-de-vie-2010.pdf [last accessed on 17 August 2020]

63 The law of agricultural and fishery modernization, available at: https://artificialisation.biodiversitetousvivants.fr/ [last accessed on 17 August 2020]

10 Multi-domain indicator system for soil observation

This report summarizes the knowledge about key indicators in the context of recent soil-related policies and reporting needs. While the current national and EU-wide monitoring instruments are capable in providing some of the needed soil parameters in a representative manner, for example, about soil carbon, some others are not systematically covered (compaction, erosion). This leaves great knowledge gaps and blank spaces about the state of the environment, while interpretations often depend on highly uncertain predictions. The discussions about data needs under the various Green Deal environmental policies and the 8th Environmental Action Plan emphasize once more the crucial role that soils play in controlling the fate of substances released to air, land and water; soils must be properly recognised as a mediator, bioreactor and buffer for many pressures impacting human health and ecosystem functioning. The required information comes only from soil monitoring, measuring accurately the inputs and outputs as well as the biological, chemical and physical transformation and transport processes. This data evolves into policy-relevant information through indicators, once they become coupled with critical limits about the potential and expected harm to the living environment.

10.1 Indicators and soil monitoring

In the run up and follow up to the EU's Soil Thematic Strategy, existing soil monitoring systems were reviewed, and the challenges towards a common European monitoring system compiled (Van Camp et al. 2004; Huber et al. 2008). Van Camp et al. (2004) emphasized the need to develop a common baseline, to decide on a minimum parameter set, quality control, reporting and EU coordination. The suggested measuring parameters followed a tiered approach, covering all soil threats (see Table 10-1).

Van Camp et al. (2004) distinguish between **basic and specific soil parameters**: basic parameters are prerequisites for the soil typological classification (mainly morphological and physical soil parameters), whereas specific soil parameters address specific threats, hot spots, functions (obligatory and facultative parameters). Three sampling intensities are distinguished:

- **Level I**: sites whether all general parameters are measured
- **Level II**: investigations and monitoring of specific parameters and soil threats, e.g. erosion mechanisms or biodiversity, likely linked with research
- **Level III**: related to very specific problems, e.g. radio-nuclides, military sites, decontamination of specific industrial residues, 'hot-spots' of anthropogenic or natural processes

Table 10-1: Parameters for soil monitoring at different measurement levels SOC, erosion, contamination (for other soil threats, salinization, floods, landslides, soil sealing.

Monitoring levels / *Soil threat*	Level I	Level II	Level III	Indicators (Huber et al. 2008)
Soil organic matter and biodiversity	- Total organic carbon - Total (organic) nitrogen - C:N ratio - Bulk density	- SOM compartments and pools - Bioavailability of nutrients and pollutants - GHG emissions - Exogenous organic matter input - Carbon hot spot monitoring: SOC-rich soils	- Microflora; microbial biomass - Biological functions (e.g. respiration, N and C mineralization) - Soil biodiversity (molecular signature	- Topsoil organic carbon content (measured) - Soil organic carbon stocks (measured) - - Earthworm diversity and biomass - Collembola diversity

			- Diversity: Nematodes, Earthworms)	- Soil microbial respiration
Soil erosion	- Modelling - (using data on land cover/land use, geomorphological data, existing national soil data, rainfall data	- **Mandatory physical** parameters: all basic parameters - **Optional physical** parameters: - Unsaturated hydraulic conductivity (laboratory) - Hydraulic conductivity (field) - Penetrometric resistance - Aggregate stability - Soil-water content, volumetric - Soil-water tension - **Mandatory chemical** parameter: SOC	- Monitoring (measurements) of soil erosion - at the plot scale - at the catchment scale - Mapping visible soil erosion features - Continuous measurement of sediment loads at the outlet of small catchments - Measurement of sediment deposition in ponds, lakes or reservoirs	- Estimated soil loss by rill, inter-rill, and sheet erosion
Soil contamination	- Total element concentrations (aqua regia extractable fraction of heavy metals) - Natural background (at least at a subset of sampling points) - Organic compounds, such as persistent organic pollutants (POPs)	- Progress in contaminated site management (per site class)	- very specific contamination problems, e.g. radio-nuclides, military contamination, major chemical facilities	- Heavy metal contents in soils - Critical load exceedance by S and N - Progress in management of contaminated sites

Source: Van Camp et al. 2004

Huber et al. (2008) then presented an overview of 290 indicators, condensed to 60 selected priority indicators for all soil threats as identified by the Soil Thematic Strategy. 27 of these indicators were tested against existing soil monitoring, with 20 being qualified for entering the envisioned European monitoring system (Table 10-1). Correspondingly, performance criteria were provided by Arrouays et al. (2008), including minimum detectable change, background values and indicator thresholds, where available.

More recent evaluations of existing monitoring systems were conducted by Stolte et al. (2016), Van Leeuwen et al. (2017) and Creamer et al. 2019). The research work of the latter two (Landmark project) focused on soil functions in agricultural soils across Europe in support of the LUCAS Soil survey. A set of soil parameters were identified for the application of pedo-transfer rules, by soil function. “Indicator” in the Landmark project was used to indicate a change in the status of soil functions from one sampling period to another. Main methodical focus was then the determination of optimal sampling densities to improve the representativeness of LUCAS.

The concept of sampling levels (Tiers) in soil monitoring was also discussed by the EIONET Task Force Soil Monitoring, summarized here as follows:

- **Level I** could correspond to the Europe-wide sampling networks, such as LUCAS Soil and ICP Forests Level I. Representativeness gaps could be filled by densifying the existing sampling grids. Sampling is limited to the **topsoil** and includes **basic parameters** but also key nutrients and metals; sampling density would be representative for all land use-soil (1:1Mio) combinations across Europe, based on international standards (CEN/ISO). Some countries could improve their national morning with LUCAS Soil thus benefit from this Level I, others could at least improve the representativeness of their existing system provided comparable sampling and analysis protocols (or conversion factors).

- **Level II** could then correspond to national monitoring networks, where there is high sampling density thus improved representativeness (e.g. **hot spots** such as organic soils), **lower depth**, and **more parameters** (including organic contaminants, emerging contaminants, soil biological parameters). Ideally, Level II offers a representative subset of points sampled and analyzed similarly to Level I, based on agreed European protocols for sampling and analysis. Experiences which a combined scheme for sampling and analysis (EU-standards/national standards) were collected during the repetition of the ICP Forests Level I survey (Biosoil project under the Forest Focus Regulation (EC No 2152/2003).
- In reality, the representativeness of LUCAS Soil may be continuously improved, while national experts and extension are needed to cover hotspots, deeper soil layers and other integrated aspects (e.g., crop quality, groundwater reproduction, vegetation composition, soil fauna).

The monitoring levels suggested above do not fully correspond to the ICP Forests Level I and II; while ICP Forests Level I would resemble the above-mentioned approach oriented towards LUCAS Soil, Level II for forest soils involves intensive monitoring of forest sites/forest stands with representative local sampling regimes (ecosystem monitoring) which are able to develop site-related element balances.

Some soil threats such as compaction and erosion, involve **modelling**, as well as other monitoring techniques such as **remote sensing**, but also more intensive sampling schemes to calibrate and validate the models while considering current land use and climate (for erosion, see Table 7-5, for compaction see Table 8-3). Also, **soil biodiversity monitoring and organic pollutant monitoring** have higher requirements to sampling, transport, storage and analysis – difficult to apply at high sampling densities such as with Level I (reference towards monitoring levels for soil biodiversity, see section 6.2). Thus, different intensity levels of monitoring need to be possible – designed in a way that data integration like a **"nested" system** approach becomes possible. In this report, the monitoring parameters for monitoring Tiers are elaborated for the soil threats compaction and erosion.

Once the necessary indicators as well as the corresponding data needs are agreed and defined (this report offers proposals), an integrated monitoring can be designed for the EU and its Member States (and neighbours), considering the conditions of LUCAS Soil combined with each national monitoring system. Methodical guidelines are needed for sampling and analysis, as well as for the necessary data integration steps (to be decided: at indicator or parameter level, or both). While there will be a mosaic of solutions, the resulting EU-wide data base for indicators must be homogenous enough so that the national and EU-wide results supplement each other, are quality assured, comparable, and uncertainties determined.

10.2 Overview of indicators and thresholds

Table 10-2 provides an overview of the findings of this report. All selected indicators respond to the policy needs as described in this report (see also Table 1-6). Healthy soils are understood to have full capacity of their functions and do not exceed the recommended thresholds. This approach is essential in order to assess soil health in the context of policy challenges and societal needs:

- soil health as an element of the good conservation status of ecosystems,
- degraded soils as priority areas for soil restauration, and
- restauration and mitigation action through sustainable land management. In addition, the reporting under various policies (e.g. NEC, SDG) can be greatly improved.

Some of the thresholds found here, can be applied across larger gradients, countries, and soil types (compaction, erosion, nutrients, acidification). Others such as for optimal SOC and contamination have limited regional validity, due to the diversity of policy schemes, monitoring methods, and site/land-use and climatic conditions. In those cases, due to the lack of harmonization across Europe, current continent-wide assessments are hardly comparable and need harmonization (e.g. the national SOC maps generated under the Global Soil Partnership). Provided the enhanced policy incentives under the Green Deal, more engagement in international cooperation and harmonization, as well as intensified monitoring activities at member state level, are needed and can be expected.

Table 10-2: Overview of soil threats and indicators investigated in this report

Soil threat	Indicator	Thresholds	Comment
Soil organic carbon			
Cropland	Deceedance of optimal SOC	Sand: 1,5 (1,0-2,0) [% SOC] Silt: 1,9 (1,4-2,4) Loam and clay: 1,6 (1,0-2,8)	Values for extreme summer-dry areas can be lower (< -100 climate water balance) Values for optimal fertilizer management (Wessolek et al. 2008) Proxy: sequestration potential
Nutrients			
Agriculture	Exceedance of critical levels of mineral nitrogen (agricultural land)	NH_3 in air: 1 – 3 [mg NH_3 m^{-3}] NO_3 in ground water: 50 [mg NO_3 l^{-1}] N in surface water: 1.0 to 2.5 [mg N l^{-1}]	Mineral N: sum of available NH_4 and NO_3
Forest	N limitation based on exceedance of C/N ratio	C/N 20-25 leakage from forests: 1 [mg N l^{-1}]	in the organic layer
Agriculture	Deceedance of optimal phosphorus	P concentration 25-35 (optimal P fertility class)	Extractable P concentration < optimum (value range refers to Mehlich 3-ICP; also available: P-Bray P1 and Olsen P)
Forest land	P limitation based on exceedance of N/P ratio	N/P ratio > 18 (coniferous forests) N/P ratio > 25 (deciduous forests)	in the organic layer
Acidification			
Agriculture	Critical pH levels	pH < 4.5 - 4.7	
Forest land	Critical inorganic Al levels	base cation/aluminium ratio = 1 (0.5-2.0)	Bc: Ca+Mg+K
Soil pollution			
Cropland	Exceedance of screening values for critical risk from heavy metal pollution	Cd, Cu, Pb and Zn by country [mg/kg] *(Arsenic could be added; others?)*	Country-specific values vary broadly and are not necessarily comparable Stratification by land use and soil texture
Soil erosion			
Agriculture	Actual rate of soil loss by water erosion	2 [t ha^{-1} yr^{-1}] (soil loss tolerance)	Threshold for shallow soils < 70 cm: 2 t/ha/yr (Switzerland) Soil formation rate: 0.3 to 1.4 t/ha/yr (Verheijen et al. 2009) All erosion types
Soil biodiversity			
	Loss of soil biodiversity (subindicators)	to be developed: (a) safe minimum standard of conser-vation (b) Operating Ranges (OR) for specific soil animals and microorganisms	requires sub-indicators by species (functional) group
Soil compaction			
	Harmful subsoil compaction (subindicators)	Priority (sub) indicators: Saturated hydraulic conductivity (Ks) < 10 [cm/d] Air capacity (AC) < 5 [%]	Exceedance of "action values" (Zink et al. 2011) Secondary subindicators with available thresholds: bulk density, internal soil strength, air permeability and oxygen diffusion
Soil sealing			
	Sealed area per total area	National targets to achieve No Net Land Take	

10.3 Recommendations for soil monitoring and policy application of soil-related indicators

Recommendation 1: ***C*ontinue and intensify national and EU-wide soil surveys for improved reporting of soil indicators, while establishing a coordination and harmonization mechanism**

Soil monitoring here is targeted to provide data about soil properties in a representative spatial-temporal approach, allowing the quantification and observation of pressure, status, and impact indicators. Any soil monitoring system must be sufficiently robust and pragmatic as to derive the necessary indicators. Because the different soil threats (as well as the underlying soil functions) require different sampling approaches, not all indicators can be served by one identical sampling design, rather the sampling requirements for a core set of parameters needs to be agreed upon (see Craemer et al. 2019). What may look like intensity levels of some national and European monitoring systems (in particular: forests), may appear more challenging now with LUCAS Soil and its growing sampling densities, while various national systems stopped operating. It seems that a European system integrates much deeper now with LUCAS Soil, than it is with the forests Level I and II concept. However, it can be seen for compaction and erosion, that not all parameters can be sampled in high densities of sampling points.
Minimum conditions for a European soil monitoring system:

- **Indicators** must be clearly defined and comparable between any Europe-wide system (e.g. LUCAS Soil) and those at national level. As discussed, any Level I may be a mix of LUCAS Soil and national sampling depending on the national circumstances. Many countries have their own monitoring systems, however, the return intervals, the design, the measuring and updating intensities differ to various degrees (an exception is forest soil monitoring, because there, countries largely follow European protocols for Level I and II; however, most countries have stopped their monitoring of forest soils due to lack of funding – maybe with options evolving by new protection strategies such as those for forests and soils 2021). **Definitions** of indicators, and how they are determined (sampling, analysis, evaluation method) **must be identical** between LUCAS Soil and at least a subsample of national soil monitoring! This requires improved coordination.

- National and EU-wide **data exchange** about soil indicators need to be comparable; there may be less focus on the exchange of measured soil properties than on readily evaluated indicators; however, any soil monitoring shall be well described for allowing post-survey harmonization processes.

- Soil monitoring provides the data to generate and report spatially-explicit **policy-relevant indicators** (an exchange of original measurement data may not needed, although such data are extremely important for improved soil research, for developing harmonization procedures, and for improving Europe-wide harmonized indicators). The resolutions of the corresponding indicator maps depend on the national and European plot densities of the respective surveys, and eventually this contributes to the overall error when data layers from different countries are spatially combined (as done for example in the Global Soil Information System – GLOSIS, which is fed by national data layers, or through INSPIRE).

- **Expert networks**: the community of soil monitoring experts in Europe has a long tradition of continued cooperation and information exchange in different networks (EIONET, EJP Soil project, European Soil Partnership, ICP Forests Soil Expert Panel, and in the future, the EUSO Stakeholder Forum). Development and agreement of technical specifications must be coordinated and jointly developed.

Recommendation 2: Improve indicator fitness for policy purposes through research and investments in soil monitoring

- The research challenge: Knowledge about the impact of soil degradation on drinking water quality, human health, food quality, soil biodiversity and ecosystem health, is still limited. Existing experiences in the risk assessment of local pollution must be expanded to risk assessment related to diffuse pollution and all other soil threats. The improvement of thresholds, the underlying transfer models towards protection endpoints, must be largely improved. Research as currently – for the first time ever – conducted in a public-public partnership (EJP Soil) must be further expanded so that research data infrastructures and indicator assessment can be improved.

- Financial instruments: reliable and spatially accurate soil indicators are needed for many policy processes (zero pollution, chemicals, circular economy, urban development, climate resilience, ecosystem health, and biodiversity, water, and food security). Soil indicators are essential in all mentioned policy schemes. Agreement between EC and member states is needed to implement the new Soil Strategy with an additional policy tool (for example, comparable to the former Forest Focus Regulation) towards a joint funding scheme for soil monitoring, e.g., per member state decision or other regulatory framework on monitoring. Such a framework would enable member states to safeguard and to build upon their historically built monitoring infrastructure.

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