Soil monitoring in Europe — Indicators and thresholds for soil health assessments

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In 2021/2022, this report was also reviewed by soil experts from the following institutions and networks:

• Eionet Thematic Group on Soil

• Common Forum on Contaminated Land

• European Commission Directorate-General for Environment and Joint Research Centre;

• European research projects (EJP Soils, HoliSoil, WORLDSOILS) and the German BonaRes research programme.

About this report

A tremendous effort has been already invested in soil monitoring in Europe, at country and at EU levels. However, there is no comprehensive and updated body of knowledge for identifying healthy soils and those that are degraded and require protection. For the last few years, the European Topic Centre on Urban Land and Soil Systems (ETC/ULS), followed by the ETC Data Integration and Digitalisation (ETC DI) in 2022, have been tasked with collecting research results on soil indicators in relation to soil functions and soil threats and their mapping and assessment. This report synthesises that knowledge with the objective of identifying criteria for healthy soils across Europe; it may serve to trigger policy and management responses to prevent further degradation of soils. For example, soils with naturally low productivity can be sustainably managed and thus healthy, but it is necessary to detect those soils whose ecosystem services (e.g. productivity) are reduced because of, for example, unsustainable management practices.

The European topic centres are part of the Environmental Information and Observation Network (Eionet): each centre consists of a consortium of experts developing data and information products, which support various activity streams of the EEA. A second core element of Eionet is the thematic networks of national experts, such as the Thematic Group Soil ([1]). The European topic centres' progress and tasks are regularly discussed, reviewed and further supported by the Eionet expert networks. While this report is deeply anchored in this cooperation, scientists across the continent were also consulted to review, supplement and quality assure the content of this report.

This report primarily refers to indicators for soil threats ([2]). These are the result of soil properties being altered under pressure, a form of soil degradation imposed by unsustainable soil management and, increasingly, disturbances triggered or enhanced by climate change. Soil threats can be of predominantly physical, or of chemical and biological nature, and are based on either simple measured or estimated soil parameters or complex ones, as a result of a combination of different parameters and sub-indicators. All soil threats can be linked to one or several soil functions and ecosystems services, and this relationship is considered when various thresholds are presented and recommended. A threshold is always addressing a specific unwanted loss of or deterioration in a soil function.

This report does not set out to design a European soil monitoring system, although it provides much information which can inform such a system, including the implications for soil monitoring and measurement needs. Instead, it intends to create a firm knowledge base on how simple soil indicators can be evaluated, applying thresholds that relate the current condition (health) of soils to the functions to be expected or strived for. The report also provides a way of approaching the remaining questions (representativity of thresholds, effects of land use, etc.) through further research, assessments and policy decisions.

The report focuses on eight soil threats and 12 soil quality indicators, which were selected (see Table 10.1), in view of their appropriateness to assess soil degradation (unhealthy soils) related to various important soil functions or ecosystem services. These are described and discussed in Chapters 2 to 9. In most cases, the indicators selected are well established, data availability at the European level is at least acceptable and they are appropriate to describe the key soil degradation types and the impairment of key soil services. Several indicators, for example soil organic carbon, have multiple functions and are used to assess several forms of soil degradation related to different soil services.

([1]) At the end of 2021, the ETC/ULS ceased to exist, and in 2022 the ETC/DI (ETC on Digital Information) began operations. At the same time, Eionet has been reorganised, and the National Reference Centres Soil (NRC Soil) have now become the Thematic Group Soil.

([2]) Not all soil threats are covered in this report, e.g. wind erosion and salinisation; as a separate report on the mapping of indictors and thresholds is currently being prepared, some indicators can be updated and expanded.

Executive summary

Soil is a finite, non-renewable resource because its regeneration takes longer than a human lifetime. Soil is a fundamental part of Europe's natural capital, and it contributes to basic human needs by supporting, among other things, 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. The key trends are:

- urban sprawl and low land recycling rates, which contribute to the continued loss of soil from sealing and replacement (e.g. by construction) and to pollution from traffic and industrialisation;
- the intensification of agriculture resulting in increasing use of fertilisers and plant protection products and of heavy machinery;
- climate change, which causes weather extremes such as drought, heavy rain, landslides and wildfires.

However, land management also influences soil quality positively. 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 prevented from further degradation, such as erosion and compaction.

Resilient, healthy soils are important to help reduce the ecological and economic impacts of unsustainable, intensive land use and weather extremes induced by climate change.

Healthy soils are an integral element of the European Green Deal and are addressed under the environmental measures of the common agricultural policy. Other policies, such as the Waste Framework Directive and the Industrial Emissions Directive, tackle emissions to soil from landfills and industrial processes. To support protection targets related to soil, its condition and functioning must be assessed using proper indicator sets and thresholds, which can demonstrate to practitioners and policymakers the success of management practices.

The development of adequate and broadly applicable indicators and thresholds is challenged by the great diversity of Europe's soils, biota and climate, as well as the varying political, economic and social conditions that lead to different priorities for settings targets and indicators among countries. There are 23 main soil types (JRC, 2008), four prevailing macroclimatic zones ([3]) and eight recognised soil threats (EC, 2006a), which all together form a complex matrix of different vegetation growth conditions across Europe. Currently, our knowledge of indicators and monitoring is profound; however, the definition and classification of indicators is still diverse, as are the sampling, measuring and evaluation systems.

This report describes the rationale for a series of common and broadly accepted soil health indicators to support policy. The focus is on soil threats, and indicators were selected in view of their appropriateness for assessing the condition of soil, its degradation, its resilience and its valuable services. For each indicator, a rationale is provided for using thresholds as critical limits to indicate that soil is in good condition, i.e. healthy soil, in respect of of specific soil functions and local conditions.

([3]) According to the Köppen–Geiger classification.

1 Soil functions and soil health: objectives, terminology and concepts

This chapter:

- **provides definitions for different types of soil degradation and their indicators;**
- **describes the general criteria for a risk-based methodology for assessing soil health to support the development of soil protection policies and measures, using functional thresholds;**
- **summarises the need for soil indicators in current EU policies, strategies and initiatives for soil protection.**

Soil:

- is the top terrestrial layer of the Earth;
- is composed of a mixture of mineral and organic compounds, water, air and living organisms;
- is one of the most complex biomaterials on earth (Young and Crawford, 2004);
- provides multiple functions that support the delivery of ecosystem services, including the life support function;
- varies naturally in both space and time over a range of scales.

In order to manage soil sustainably, and protect it where necessary, knowledge is needed about the state of soil and how it develops under current and future management and climatic conditions. Healthy soils deliver ecosystem services to the best of their capacity. Unquestionable evidence shows that land cultivation and urbanisation have altered many soil properties, causing reduced soil functioning (JRC, 2012; EC, 2020a). Such soils are then degraded, which consequently harms ecosystems and their life support functions ([4]). In the case of soil sealing, soil functions are largely and irreversibly lost.

Indicators are expected to guide land users about which soils are degraded and why, so that specific conservation or restorative action can be triggered. This requires information about the potential of soil in relation to its properties and the pressures on it, such as a specific types of land use and/or climate change. As will be seen, soil quality describes the inherent capacity of soils to deliver a certain degree of functions and services. Unhealthy soils are deprived of certain parts of this capacity. However, given the considerable variability of soils, quantifying adequate reference levels for soil function indicators in relation to land use and climate is very difficult unless large monitoring databases and land use data become available.

What seems possible, and very useful for soil protection policies, is the identification of critical limits, which inform us of any potential risk to degraded soils and to ecosystems, water and human health. For example, such limits could inform us about the production capacity of soils (e.g. depending on the site-specific clay content) or the amount of organic matter needed to ensure their stability and ability to store sufficient water. Levels of degradation can be approximated using critical limits as trigger points for soil protection measures.

([4]) Land degradation is considered desertification when it occurs in dry lands; desertification includes all forms and levels of land degradation (IPCC, 2019). Loss of soil productivity and an increase in aridity are typical indications of desertification.

1.1 Definitions

1.1.1 Soil health

The concept of soil health is defined in a variety of ways in the literature.

Doran et al. (2002) define soil health as synonymous with soil quality, namely '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'. The definition emphasises the multifunctionality of soil as well as its contribution to ecosystem services ('soil-based ecosystem services').

Bünemann et al. (2018) updated the definition of soil quality and soil health 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'. They distinguish two closely linked dimensions of soil quality:

1. soil capability, i.e., the intrinsic capacity of a soil to contribute to ecosystem services, based on 'inherent' (rather static and less sensitive) attributes of soils versus manageable (dynamic) attributes, according to Schwilch et al. (2016);

2. the capacity of soils to function sustainably (focusing on land use and its impact), including productivity and the soil's contribution to environmental quality, including plant, animal and human health.

Hein et al. (2016) also refer to 'capability' as the ecosystem's potential to sustainably generate a particular service needed under its current condition and type of use. There is broad agreement that well-functioning (healthy) soils support (and provide) ecosystem services.

A differentiation between soil health and soil quality was supported by Bonfante et al. (2020) and Vogel et al (2020). They define soil health as the actual capacity of a soil to perform its core functions and to provide ecosystem services, and soil quality as the inherent capacity which provides the basic frame within which a soil evolves. Vogel et al. (2020) also define intrinsic 'soil potential' as the maximum functionality a soil can offer based on its inherent properties. This entails an optimum state (condition) of soil based on sustainable management (best use of its properties). This is the basis for a third dimension of soil quality:

3. the dynamic element of soil quality, which characterises the actual state of soils and is based on sensitive (responsive), manageable soil attributes.

Several soil threats, such as subsoil compaction, erosion, sealing and salinisation, target both dynamic and intrinsic (static) properties making it even more difficult to reverse degradation.

What stands out from the above definitions is that a holistic concept of 'healthy soils' should encompass both the properties (intrinsic/static and dynamic) of soils and the degree to which soils sustainably perform key functions and ecosystem services (e.g., water retention and filtering, food production). Indicators (based on parameters or attributes) must therefore represent the state and potential of the different soil functions.

Healthy soils perform their functions optimally, conditioned by local conditions under sustainable soil management:

- Current local conditions include the impact of historical land management, and this has affected both dynamic as well as inherent, fairly stable soil properties (e.g. reduced thickness of A horizons from erosion).

- Sustainable soil management ensures that (dynamic) soil properties do not decline further and, where possible, they improve and restore soil quality.

'Healthy soil' is thus a concept and a target for stakeholders (politicians, practitioners, consumers), which indicates the level of soil quality desired to achieve in view of soil functions and ecosystem services. The focus is on both dynamic and responsive properties, as much as the soil's intrinsic capacity and thus its natural boundaries. Therefore, 'healthy' defines what the 'quality' of a specific soil needs to look like in order to maximise all locally possible and necessary functions and ecosystem services.

Healthy soils are also the foundation of a recent policy target proposed by the Mission Board for Soil Health and Food (EC, 2020a), in which soil health describes the 'continued capacity of soils to support ecosystem services'. And, furthermore, 'healthy soils provide ecological functions for all forms of life, in line with the Sustainable Development Goals and the Green Deal'.

1.1.2 Extended terminology

Soil threats

Counteracting soil threats represents the main structural element of soil protection according to the 2006 EU soil thematic strategy. It has also been adopted in the Status of the world's soil resources report (FAO and ITPS, 2015). Soil threats are processes that damage soil and its functional properties. This damage then reduces the soil's capacity to provide ecosystem services. Spatial data on soil threats indicate focal areas for sensitive management and soil restoration (Huber et al., 2008). Soil threats are thus characterised by a negative trend in one or several soil properties (e.g. soil organic carbon (SOC) loss under cultivation, industrial pollutant inputs, water holding capacity), or are indicated directly by features observed in the field (e.g. erosion, sealing). Single threats typically affect different soil properties (physical, chemical and biological) or induce other threats (e.g. soil erosion is accompanied by SOC losses).

Table 1.1 lists soil threat indictors and how they relate to important soil functions (i.e. soil services in the context of ecosystem services; see also Lehmann et al. (2020)).

Table 1.1 Soil threats and their linkage to soil services and key societal needs

	Societal needs				
	Biomass	**Water**	**Climate**	**Biodiversity**	**Infrastructure**
Soil services	Wood and fibre production	Filtering of contaminants	Carbon stage	Habitat for plants, insects, microbes, fungi	Plataform for infrastructure
	Growth of crops	Water storage			Storage of geological material
Soil threat indicators					
Soil organic carbon	+	+	+	+	indiff. **(i)**
Soil nutrient statuts	+	- **(ii)**	indiff.	+	indiff.
Soil acidification	-	-	indiff. **(iii)**	-	indiff.
Soil pollution	-	-	-	-	indiff. **(iv)**
Soil biodiversity	+	+	+	+	indiff.
Soil erosion	-	-	-	-	indiff.
Soil compaction	-	-	-	-	indiff.
Soil sealing	-	-	-	-	+

Legend

+	Positive impact
-	Negative impact on soil service
indiff.	Neutral or unknown impact

(i) Soil organic carbon/infrastructure: organic soils are unstable as platform for infastructure.

(ii) The filtering capacity of soils prevents of buffers eutrophication and acidification.

(iii) Soil organic /carbon storage: fulvic acid (from acidified forest floors) engances bleaching and nutrient loss, as well as loss of dissolved organic carbon; acidic soils slow down decomposition. From a climate point of view, soil acidification could favour carbon storage, as it leads to a lower biological activity and hence accumulation of dead biomass.

(iv) Land prices are lower if the soil is polluted, as remediation costs are incurred.

Soil degradation

Soil degradation can be defined as a decline in soil quality (Bone et al., 2010), resulting in the reduced functioning of the soil. This includes nutrient limitations and excesses, limited productivity, reduced water conservation and reduced resilience to drought and extreme rainfall. Soils are in good condition when not subject to degrading processes (indicated by soil threats or declining soil function indicators). Minimising 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 and ITPS, 2015). Therefore, soil degradation is 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 level to which soil functions are affected. Soil functions can be assessed by defining a desired status (health) of ecosystem services performed by a given soil. This includes the application of thresholds in view of the protection of so-called endpoints (i.e. food quality, human health, water and air quality, and soil biodiversity), from any harm through soil degradation.

Thresholds

In general, thresholds are perceived as values above or below which a significant shift or rapid negative change takes place (Van Lynden et al., 2004). This can be a single critical value or the critical limits of a range of values (if the variability of soil conditions so requires). In the context of soil protection, the following information is needed from any such thresholds:

- the critical level at which deteriorating or lost soil functions have unwanted effects on ecosystem services;
- the critical level at which a specific preventive or restorative activity is needed.

There are some issues for consideration:

- Some trade-offs between soil functions occur, and any threshold must consider the historical and intended land use. Historical land use and disturbances are likely to have affected intrinsic soil properties (compared with more responsive properties), thus reducing soil quality. This kind of soil degradation is more difficult to reverse (e.g. subsoil compaction, soil sealing).
- The relationship between soil quality and soil health is land use- and site-specific: a soil under semi-natural vegetation can have a lower intrinsic quality than a cultivated soil; on the other hand, both soils can be healthy and fulfil the functions expected as long as the cultivated soil is sustainably managed.
- Determining a positive or negative trend may be sufficient to inform policies; however, a decision is needed about where and under which conditions specific land use changes, protective measures, etc., are needed. Thresholds serve this purpose.
- Any deterioration in soil functions can be considered negative and indicates the impact of unsustainable land management. Thresholds as defined here allow an acceptable range of degradation ('light' degradation until a threshold is reached), as long as ecosystem services are not significantly affected.

Here, we pursue critical limits to:

- identify tipping points between soil degradation as the reduction in or loss of soil functions and ecosystem services;
- identify thresholds to trigger action (prevention, restoration, remediation).

Soil functions

Soil functions describe the soil's capacity to support the ecosystem services essential for human well-being ([5]). Soil processes enable the provision of such services (Schwilch et al., 2016). Bünemann et al. (2018) define soil functions as bundles of soil processes that underpin the delivery of ecosystem services.

The functions listed in Table 1.2 are a selection for the purpose of testing soil functional indicators for soil quality assessments. Other functions, such as the filtering of polluting substances or soil as a source of raw materials (as stated in EC, 2006b) also need consideration when developing a holistic view on soil's functions and its services to ecosystems.

Researchers at the Landmark project ([6]) selected five major soil functions and derived a hierarchy of four levels of soil and non-soil (environment, management) attributes to describe them; they then selected 29-40 attributes for each process. Vogel et al. (2020) used the same functions and suggest a matrix of 18 suitable and observable soil attributes (Table 1.2). They suggest eight physical, chemical, and biological attributes, related to the state of soils and affected by soil management, and 10 intrinsic attributes of the hydrology, site and soil. In the case of complex attributes that are difficult to measure, they suggest using pedotransfer functions to predict them.

([5]) www.fao.org/soils-portal/soil-degradation-restoration/en

([6]) https://landmark2020.eu/project-details

Table 1.2 Important soil functions

Soil function	Description
Production	Capacity to produce biomass
Water storage and quality	Capacity to store precipitation water and filter for soil pollutants
Carbon storage	Capacity to store and stabilise SOC
Nutrient cycling	i. Capacity to provide nutrients from mineral and organic soil resources in available form (nutrient mobilisation capacity) ii. The capacity to store mobile nutrients within the root zone to avoid losses by leaching and gaseous emissions (nutrient buffering capacity)
Habitat for biological activity	Provision of a species (gene) pool that can buffer ecosystem functions against species extinction (assumption: loss of soil function is more likely with low species diversity in each functional group)

Source: Modified from Vogel et al. (2020).

Ecosystem services

Ecosystem services can be summarised as the goods and benefits people and societies receive from ecosystems. Soil as the below-ground 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) consider 29 of 83 ecosystem service classes in the Common International Classification of Ecosystem Services (CICES 5.1; Haines-Young and Potschin, 2018) to be related to soil and 40 classes to be affected by agricultural soil management.

Soil indicators

According to Bünemann et al. (2018), soil quality assessment relies on a set of 'sensitive soil attributes that reflect the capacity of a soil to function'. Vogel et al. (2020) suggest that useful indicators are soil attributes that provide substantial information on soil functions (e.g. water capacity), with 'soil attributes' being measurable soil properties. However, they also state that the knowledge or recommendations necessary to interpret indicators is still scarce, and this is because of the lack of 'clear conceptual or mechanistic relationships between indicators and soil functions'. Compared with a parameter (see Figure 1.1), an indicator is embedded in a well-developed interpretative framework and has meaning beyond the measure it represents.

Doran et al. (2002) provide performance criteria for indicators for soil quality and health; they are capable:

- of defining ecosystem processes;
- of integrating physical, chemical and biological properties;
- of being sensitive to management and climatic variations;
- of being accessible and practicable for agricultural specialists, producers, conservationists and policymakers.

Accessibility requires the availability of analytical methods, and the indicator must be interpretable by the end user (Bloem et al., 2006a). It must be reliable, reproducible and applicable to a range of sites.

Indicators therefore act as a link between soil services, as determined by one or more soil functions on the one hand, and the degree to which a soil can actually perform the functions or deliver services on the other. Soil health in the context of the indicators presented in this report can be interpreted as the resulting state of soil, relative to critical limits not to be exceeded.

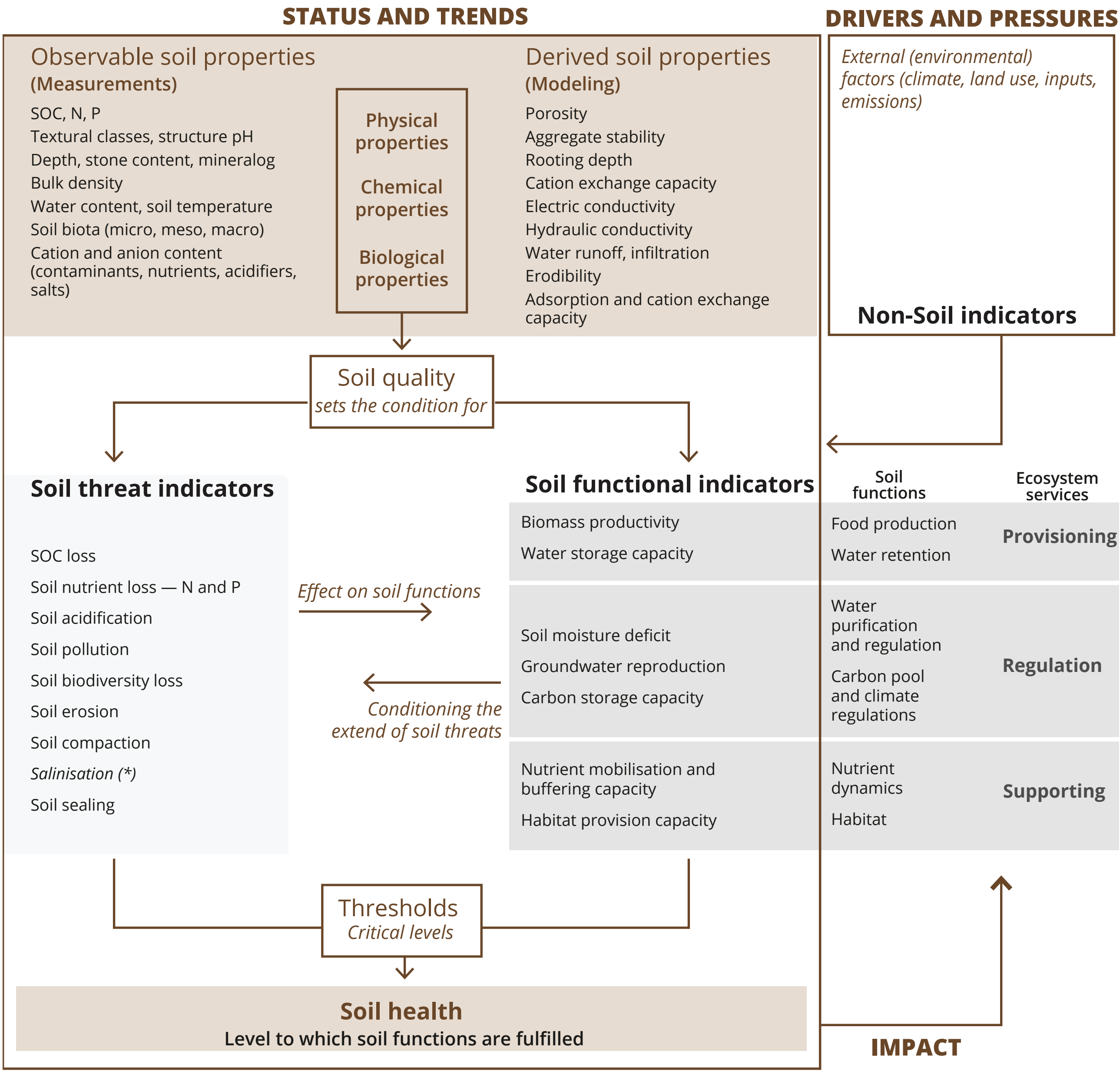
Figure 1.1 presents an overview of soil threat and soil function indicators and how they relate to soil functions and ecosystem services. Soil function indicators have been developed in various soil functional assessment and soil evaluation systems (ad hoc AG Boden, 2007; Siemer et al., 2009; Lehmann and Stahr, 2010; Lehmann et al., 2013), with the objective of identifying soils that need particular focus in land use planning; this can be achieved by categorising soils according to their sensitivity to pressures and their protection level. Greiner et al. (2017) conducted a profound review of soil function indicators, and methods for quantifying the contributions of soils to ecosystem services.

Soil parameters

Van-Camp et al. (2004c) distinguish general and specific soil parameters: basic parameters characterise the soil as required for its typological classification (mainly morphological and physical soil parameters), whereas specific soil parameters address specific threats, hot spots and functions (obligatory and facultative parameters). Vogel et al. (2020) have identified and classified soil attributes (i.e. measured soil parameters): dynamic parameters which are sensitive to management and disturbance, and rather static parameters which characterise intrinsic soil properties that do not depend on management, so their measurement does not need to be repeated frequently. Complex parameters that are difficult to measure are derived from basic parameters using models (e.g., pedo-transfer functions) (see also Figure 1.1).

Previously a distinction between soil threat indicators and soil functional indicators has been proposed. Soil threat indicators merely quantify the magnitude of one or more threats (see Figure 1.1), whereas soil function indicators are connected to one or more specific soil functions and services. Both threat and function indicators are characterised by thresholds, even though the meaning and interpretation of such thresholds can be very different in that they relate to either specific threats (e.g. critical erosion rates) or entire functions (e.g. biomass production). This also suggests that there is an overlap between both, since erosion (as well as pollution, SOC loss and compaction, to mention a few) all affect biomass productivity.

Figure 1.1 Conceptual visualisation of soil threat indicators versus soil function indicators in relation to soil functions or ecosystem services



Note: Salinisation is not covered in this report. N, nitrogen; P, phosphorus.

Source: EEA.

1.2 Risk-based approach to defining thresholds

To be able to relate soil quality to ecosystem services (or soil functions that make up a specific service), it is imperative to be able to connect a specific service to a specific soil quality standard or limit in the relevant protection targets, such as food quality, human health or drinking water quality.

Figure 1.2 presents a conceptual framework for soil degradation assessment. In order to use this framework to assess nutrient losses and contamination and also physical forms of degradation, such as soil erosion and soil compaction, it is important to understand the relationship between soil dynamics (processes that respond to a pressure, indicated by soil properties that can be monitored) and the critical limits for protection or 'endpoints'. This usually requires models that describe the behaviour of a soil under stress and help to define thresholds. The key principle underlying this approach is that critical limits (thresholds) in endpoints (e.g. water quality, human health, ecosystem functioning) are converted into 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 or change in actual land use.

The level of soil degradation can be quantified locally or regionally as the degree to which the current soil condition exceeds the relevant thresholds for specific functions. This approach is also a key element of risk-based land management (Vegter et al., 2003; see also Section 5.1.4), in particular regarding contaminated land; it is not necessarily in line with other definitions of soil degradation, in which any (undesirable) change in soil properties may be 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 desirable in the case of adding nutrients, such as phosphorus (P) in a situation where phosphorus is limited, but it is considered unwanted in phosphorus-saturated soils or when adding toxic pollutants (e.g. cadmium). From a risk point of view, accumulation can be equivalent to degradation if it leads to exceeding critical limits in relevant endpoints.

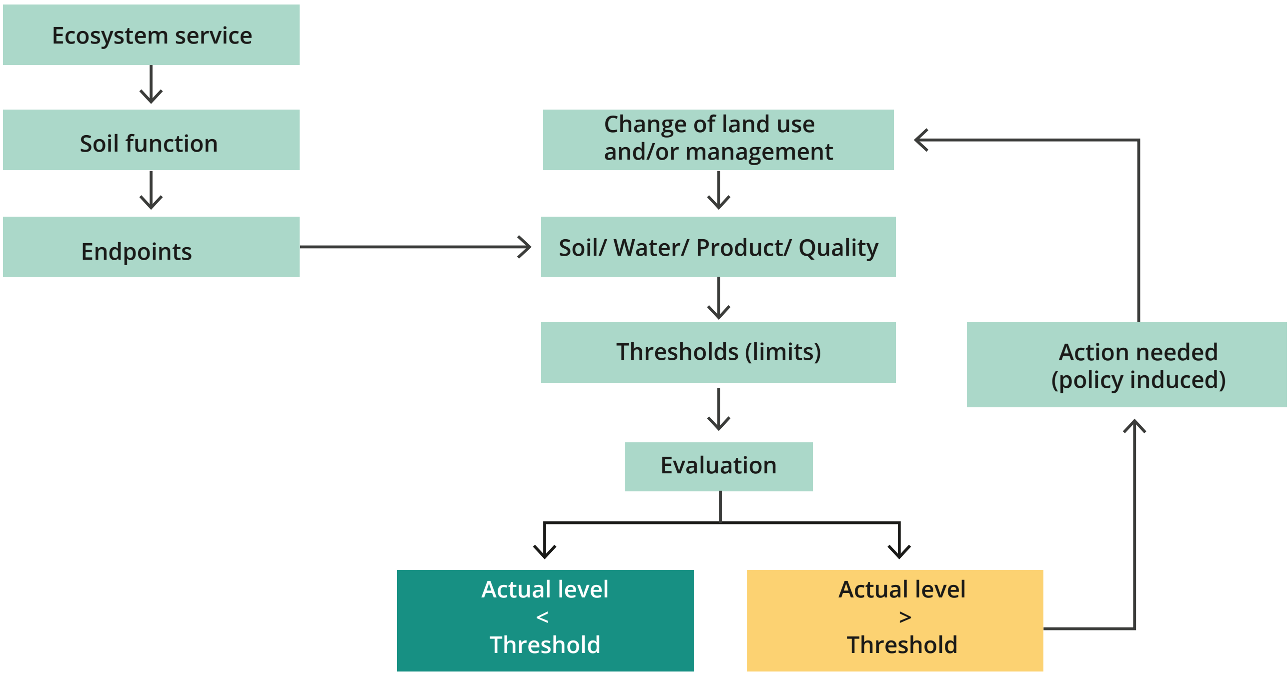
This brings us to the second relevant aspect of risk analysis in accordance with the principle of risk-based land management, which is the dynamic aspect. Current conditions (soil properties or concentrations of unwanted substances) in soil and water can be such that thresholds for relevant endpoints have not yet been exceeded. However, depending on, for example, land use (7) (and changes in inputs and/or atmospheric deposition), conditions can change so that thresholds can be exceeded at a certain point in time (Figure 1.3). Hence, precautionary, prevention and other relevant measures should be taken before the threshold is reached.

At present, many risk assessment models are still being developed, while some are already in use, for example to assess soil condition regarding pollution (e.g. the risk assessment model SansCrit (7) and the Dutch risk assessment toolbox (9); the CLEA model (10) used in the United Kingdom; and the S-Risk model used in Belgium). This means that it is not possible to assess soil degradation overall with one single indicator. At present, assessment of soil degradation according to the current state of research, can be carried out only for specific soil services. For these, it is imperative to consider, in addition to the general soil properties (or indicators) used in these models, specific regional conditions such as climate and crop type (example: regional versions of the S-Risk model).

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 that are valid for all circumstances. The list of parameters needed to develop all indicators and thus to monitor all soil threats can be quite long (see also Section 10.1). 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) to connect the endpoint to the current condition of the system.

(7) Land use is dynamic; it includes inputs to soil and mechanical changes caused by trafficking or different methods of soil preparation (e.g. tillage) and types of sowing systems
(8) SansCrit: https://www.risicotoolboxbodem.nl/sanscrit
(9) www.rivm.nl
(10) CLEA model: https://www.gov.uk/government/publications/contaminated-land-exposure-assessment-clea-tool

Figure 1.2 Conceptual framework for soil degradation assessment



Source: EEA.

1.3 Assessment of soil health

Figure 1.3 presents a generic schema for the assessment of soil health. It is valid for any approach that applies thresholds in relation to soil threat or soil function indicators. Thresholds:

- indicate a critical limit beyond which soil functions are 'significantly' reduced or even lost: degraded/not degraded;
- point out the need for preventive and restorative measures;
- provide a measure to interpret the direction of trend towards soil recovery or degradation;
- indicate a level beyond which potential harm to protection targets (e.g. water quality, biodiversity) can be expected;
- enable adjustments in management practices in response to varying conditions (climate, soil).

Some aspects need to be considered in developing and implementing thresholds:

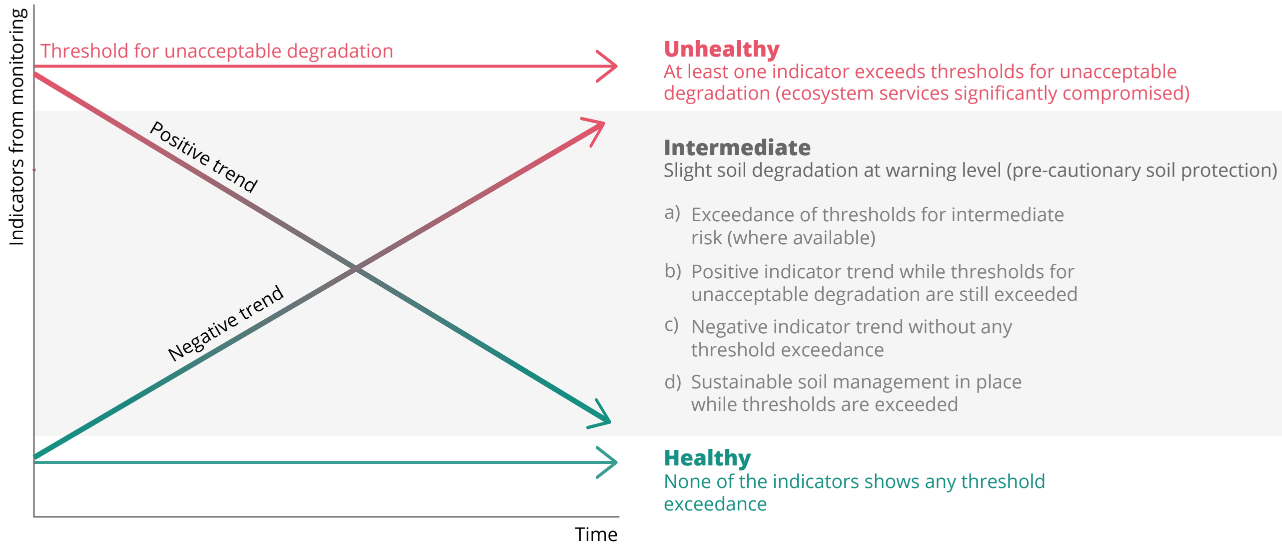
- Thresholds may not be comparable between countries and their risk assessment approaches, despite similar soil and land use conditions — see Chapter 5.
- Thresholds require updating as progress is made in risk assessment and research.
- If thresholds are not available, benchmarks or reference values may provide orientation values to identify degraded, unhealthy soils (see Chapter 2 for an overview of different approaches to defining SOC thresholds).

Soil degradation and the impact on soil functions can be characterised by three possible scenarios. In Figure 1.3, the green line represents the relevant threshold as a basic criterium for healthy soils. 'Healthy' 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 in any relevant time frame considered. Opposite this is the 'unhealthy' scenario in which current and future conditions are such that the system is at risk. This calls for measures either to reduce the impact or to change land use so that less stringent risk limits can be used.

The 'intermediate' scenario combines four elements:

- a positive trend for a soil indicator while the threshold is still exceeded;
- a negative trend while the threshold is not yet exceeded;
- sustainable soil management is implemented; however, a corresponding positive signal in the soil indicators measured cannot yet be detected;
- specific thresholds for intermediate risk are defined (warning levels).

Figure 1.3 Dynamic assessment of soil health



Source: EEA.

- Four main types of soil degradation 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 run-off, wind and water erosion, greater soil temperature fluctuations and an increased propensity for desertification. It also includes soil excavation and soil sealing.

- **Soil chemical degradation** is characterised by changes in soil processes, including nutrient depletion, acidification, salinisation and contamination, which in turn leads to a reduced cation exchange capacity, increased aluminium or manganese toxicities, calcium or magnesium deficiencies, and leaching of nitrate nitrogen or other essential plant nutrients. For nutrients and contaminants, annual inputs, such as those from agricultural management (inputs of N, P, K and also copper, zinc, cadmium and antibiotics via animal manure, sewage sludge, compost, digestate or mineral fertilisers) or from additional sources, including inputs via air or sedimentation, are also considered chemical degradation.

- **Soil biological degradation** refers to reduced soil biological activity, which can be accompanied by a loss of soil biodiversity. This leads to lower levels of mineralisation and respiration and an accumulation of incompletely decomposed dead organic matter (necromass). Nutrient availability is reduced, and organic matter accumulates in forest topsoils. In peat soils, degradation (cause by drainage) leads to SOC losses.

- **Soil ecological degradation:** although a clear characterisation of 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 element cycling and water infiltration and purification, perturbations in 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 1.3, which combines types of soil degradation with soil threats and soil services.

Table 1.3 Soil degradation types, corresponding soil threats and affected soil services

Degradation type	Impact of threats ([a])	Affected soil services ([b])
Soil physical degradation	Subsoil compaction Soil erosion Landslides Sealing	Growth of crops Wood and 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 and 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 metabolism of contaminants Decline in soil organic matter and/or carbon	Habitat for plants, insects, microbes, etc. Water storage Substance filtering Carbon storage

Note: ([a]) The soil threats listed are a combination of those mentioned in the EU soil thematic strategy and the Recare project according to Stolte et al. (2016).
([b]) According to Adhikari and Hartemink (2016).

1.4 Existing indicator systems, including soil quality

1.4.1 Global and European soil indicator systems

Table 1.4 provides an overview of commonly discussed European and global soil indicators. The sorting element for these indicators is the European Commission's soil thematic strategy (EC, 2006b). It should be mentioned that Eurostat, FAOSTAT and the Organisation for Economic Co-operation and Development (OECD) also maintain indicator systems, which contain soil-related indicators as an element of agri-environmental indicator sets. Soil indicators are also included in the EEA's indicator system, which is populated by the members of the European Environmental Information and Observation Network (Eionet), and which is — among other things — used for the EEA's regular reporting on the status and outlook of the European environment. The EEA's system also includes indicators under various pieces of EU legislation, for which the EEA acts as the knowledge centre and data hub (reporting in the context of soil for the Land Use, Land Use Change and Forestry (LULUCF) Regulation and the National Emission Ceilings Directive — see Table 1.3).

Table 1.4 Overview of broadly discussed soil quality indicators

Degradation types/soil threats	Glasod (a)	EU (b)	Envasso indicators, modified (c)	Indicators in Status of the world's soil resources report (d)	SEEA (e) and FDES (f)(g)
Water erosion	X	X	• Soil loss (t/ha) • Observed erosion features (type/amount per unit area)	Soil loss	Area affected by soil erosion
Wind erosion	X				
Overblowing	X		• Deposited soil (t/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
Salinisation	X	X	• Salinity state: total salt content (% EC) • Exchangeable sodium (pH, ESP %)(h)	Spatial distribution of salt-affected soils	Area affected by salinisation
Acidification	X	(X)	• Top soil pH • %Exchangeable acid cations (Mn, Al, Fe)	pH Acid neutralisation capacity	Area affected by acidification
Loss of nutrients	X	(X)	• %Exchangeable basic cations • %Trace elements (including micronutrients)	Soil fertility: %nutrients, pH Nutrient balances: N, P	Nutrient concentrations: N, P, Ca, Mg, K, Zn,
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 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 in desertified soil		Area affected by desertification
Water cycle				Soil moisture	

Note: (a) Glasod: Global Assessment of Human-induced Soil Degradation: 12 types of human-induced soil degradation recognised.
(b) Soil thematic strategy of the European Commission (EC, 2006a).
(c) Envasso: Environmental Assessment of Soil for Monitoring: Volume 1 identifies 290 potential indicators related to 188 key issues for nine soil threats (Huber et al., 2008).
(d) FAO and ITPS (2015).
(e) SEEA: System of Environmental Economic Accounting: internationally agreed standard for producing comparable statistics and accounts (status and changes in stocks of environmental assets); it follows the same accounting structure as the System of National Accounts. The SEEA is a guide to integrating economic, environmental and social data into a single, coherent framework for holistic decision-making.
(f) FDES: Framework for the Development of Environment Statistics, FDES 2013 (United Nations Statistics Division, 2017); it uses SEEA definitions and classifications.
(g) By soil type, nutrient, national, subnational (in the case of pollution: by location, subnational, type of pollutant, source).
(h) ESP: exchangeable sodium percentage

Source: Baritz, R. (EEA) for FAO- SoilSTAT, internal concept note 2019, adapted for this report.

Table 1.5 Proposed MAES soil indicators

<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 vegetated land</th></tr>
<tr><td rowspan="8">Soil pressures</td><td></td><td colspan="6">Intensive management (e.h. tillage)</td></tr>
<tr><td rowspan="2"></td><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">Imperviouness (%)</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="6">Soil state</td><td rowspan="6">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</td><td colspan="2" rowspan="2"></td><td colspan="2" rowspan="3"></td></tr>
<tr><td colspan="2">Soil productivity</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 biodiversity</td><td rowspan="2">Earthworms diversity/ abundance</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</td></tr>
</table>

Source: Adapted from EC et al. (2017).

The initiative 'Mapping and assessment of ecosystems and their services' (MAES) was launched under the Seventh EU Environment Action Programme. A set of soil-related indicators was proposed at a MAES workshop (EC et al., 2017) (Table 1.5).

A subset of these indicators was considered essential for covering the role of soils in 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 (t/ha or kg/m^2)
- soil biodiversity potential.

The 2020 MAES assessment then presents an updated collection of soil indicators from the Commission's Joint Research Centre (Maes et al., 2020):

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

The report presented additional findings from the literature — without specific reference to indicators — on diffuse pollution, salinisation and desertification.

1.4.2 *Land degradation and SDG indicator 15.3.1*

The context-specific nature of land degradation requires a combination of indicators to fully describe the condition of land and soil. Figure 1.4 presents an overview of processes leading to degradation, and how the current sub-indicators of Sustainable Development Goal (SDG) indicator 15.3.1 (land productivity, land cover change, carbon stocks) relate to ecosystem services as affected by land degradation. Countries are encouraged to use additional indicators. Important land degradation processes are in fact the soil threats mentioned above. Indicators in this context represent 'key processes which underpin land-based natural capital' (Orr et al., 2017).

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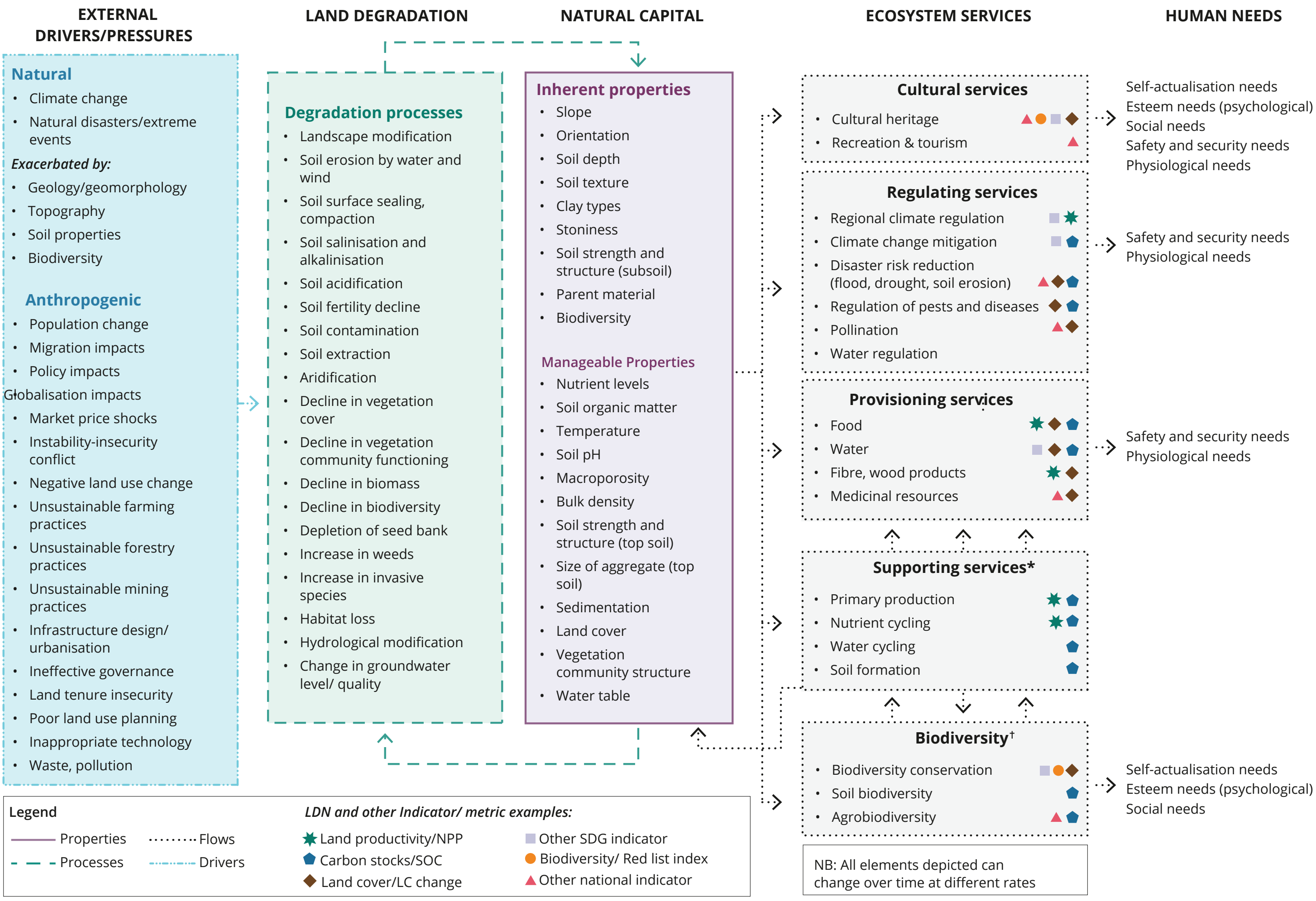
Figure 1.4 Operational definition of land degradation and linkage with sub-indicators



*Services that support all other ecosystem services and also influence natural capital.
† Biodiversity underpins all ecosystem services.

Source: United Nations Statistics Division (2022).

1.5 Soil indicators for EU policy targets

Despite the fact that the proposal for an EU Directive on soil protection was withdrawn in 2014, various aspects of soil protection have been incorporated into sectoral policies or other non-soil-related policies. Most relevant developments regarding the restoration and protection of soil functions are described below in chronological order, and their targets are summarised in Table 1.7.

- The 2000 **Water Framework Directive** (EU, 2000) aims to prevent and reduce pollution of water bodies from agricultural and industrial sources by prescribing specific measures. The directive indirectly regulates diffuse soil contamination because soil pollution is in numerous cases responsible for surface water or groundwater pollution. The directive requires that Member States 'produce River Basin Management Plans' and establish 'programmes of measures'. This includes identifying point sources and diffuse sources of pollution, quantitatively estimating their impact and implementing measures to reduce their impact. The directive is backed up by a clear implementation schedule, including monitoring and evaluation.

- Three non-binding EU policy documents with a clear focus on soil protection were released between 2006 and 2013. Firstly, the **soil thematic strategy** (EC, 2006a), which for the first time identifies and presents the threats to soil in Europe; secondly the **Roadmap to a resource efficient Europe** (EC, 2011), prescribing non-binding targets for land take, soil erosion and local soil contamination; and thirdly the **Seventh Environment Action Programme to 2020** (EU, 2013a), with objectives to reduce land take, manage local soil contamination, prevent soil erosion and increase soil organic matter. All three strategies demand further action: targets, incentives for implementation, and monitoring. While the EU is now working towards its 2030 agenda, the achievements of these earlier strategies are summarised in the EEA's *The European environment — state and outlook 2020* report (EEA, 2019a). With regard to soil, Europe is not on track to meet the above-mentioned targets. Furthermore, some soil threats are not covered by any targets (e.g., compaction), and thus, there is lack of evidence about their status. In the context of the proposal for the **Eighth Environment Action Programme** (EC, 2020b), the European Commission is undertaking a consultation on a monitoring framework with headline indicators (EC, 2021a), among them SOC, and placeholders for healthy soils and soil sealing.

- Regulation EU No 1306/2013 on the **common agricultural policy (CAP)** (EU, 2013b) introduced standards for good agricultural and environmental condition of the land (GAEC), which are linked to agricultural subsidies. Also important are the rural development measures as set out in the regulation. 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 soil surface to reduce erosion by water and wind. Member States set quantitative targets and report progress through annual implementation reports. The implementation of both the GAECs and the rural development measures in support of soil quality has been poor across the EU (EC, 2020a). The 2013 CAP regulation will be replaced by a new regulation for the CAP period 2023-2027. The new CAP lays down a common set of indicators as part of a new performance, monitoring and evaluation framework. The indicators will be monitored through annual performance reports and a biannual review of progress in implementing the CAP strategic plans.

- The 2016 **National Emission Ceilings (NEC) Directive** (EU, 2016) 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 EEA briefings on the NEC Directive reporting status (EEA, 2019b). The five main air pollutants nitrogen oxides, non-methane volatiule organic compounds, sulphur dioxide, ammonia and fine particulate matter, as well as carbon monoxide, are monitored and reported every four years at selected monitoring sites. Since 2018, Member States must implement air pollution control programmes by establishing monitoring sites to assess the impacts of air pollutants on sensitive receiving environments (freshwater, non-forest natural and semi-natural habitats, and forest ecosystem types). The NEC Directive lists the soil indicators for this assessment ([11]).

- With the adoption of the 2018 **LULUCF Regulation** (EU, 2018a, Article 1), greenhouse gas emissions and CO_2 removals from the LULUCF sector 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 LULUCF Regulation sets a binding commitment for each Member State to ensure that accounted emissions from land use are entirely compensated for by an equivalent accounted removal of CO_2 from the atmosphere ('no net debit' rule). Although Member States already partly undertook this commitment individually under the Kyoto Protocol until 2020, the LULUCF 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 SOC stocks, for example restoration of forests and wetlands, and avoid the conversion of grassland to cropland.

- The EU has meanwhile updated the 2030 target to reduce net greenhouse gas emissions to 55% below 1990 levels. This target was set in the **2030 climate target plan** (EC, 2020c) and included in the **European Climate Law** (EC, 2020d), and it is part of the process of achieving a climate-neutral Europe by 2050. This includes recognising the need to enhance the EU's carbon sink through a more ambitious LULUCF Regulation; therefore, the provisions under the European Green Deal also include revising the 2018 LULUCF Regulation (EC, 2021b).

To achieve climate neutrality in 2050, the capacity of land to capture CO_2 will have to increase; this includes soils. Two mechanisms are envisaged:

1. carbon farming (COWI et al., 2021);

2. carbon removal certification mechanism (CRCM) ([12]).

The uptake of carbon removals and increased circularity of carbon is incentivised by the circular economy action plan, while the farm to fork strategy may enable payments to farmers and foresters for the carbon sequestration service they provide. These policy demands will require improvements in SOC stock monitoring with regards to reliability (uncertainties) and spatial and temporal resolution.

([11]) For terrestrial ecosystems an assessment of soil acidity, nutrient loss, nitrogen status and balance, and biodiversity loss is required based on the following indicators: soil acidity (every 10 years); soil nitrate leaching (annual); and carbon-nitrogen ratio (C/N) (every 10 years).

([12]) Study by UBA, Ecologic, Rambøll and Carbon Counts in preparation: https://ec.europa.eu/clima/tenders/2020/305336_de.

Table 1.6 Objectives, targets and recommended indicators of the EU's Mission Board for Soil Health and Food

Objective	Target	Presence of soil pollutants, excess nutrients and salts	Soil organic carbon	Soil structure bulk density & absence of 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 salinisation	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 management 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 24% 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 (EC, 2019) 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, so the importance of soil health is broadly addressed. The following key objectives of the Green Deal include policy developments that are highly relevant for protecting soil in future:

  - Preserving and restoring ecosystems and biodiversity. In May 2020 the European Commission released the new **Biodiversity strategy for 2030** (EC, 2020e). The overall objective is to reverse biodiversity loss in the EU, and to increase resilience to 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 no deteriorating trends in conservation or in the 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 with favourable status fall into that category or 'show a strong positive trend'. The recent EC proposal for a regulation on nature restoration (EC, 2022) calls on Member States to 'achieve an increasing trend at national level' for a set of indicators 'until satisfactory levels' are reached. Indicators describe the condition of ecosystems and include SOC in agroecosystems.

  - In May 2020, the **Farm to fork strategy** (EC, 2020f) was published, which focuses on fair and environmentally friendly food production. The strategy repeats the target for organic farmed land set out in the biodiversity strategy and also 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 their CAP strategic plans.

  - A **Zero pollution ambition for a toxin-free environment** has the overall objective of avoiding 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, and also causes economic losses (e.g. crop yield loss, health-related costs, remediation costs). It includes two actions relevant for soil protection:

    - The **Chemicals strategy for sustainability** (EC, 2020g), 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. A Commission staff working document (EC, 2020h) addresses the hazard from per- and polyfluoroalkyl substance (PFAS) contamination of soils; another staff working document raises the concern about mixtures of chemicals in environmental media (EC, 2020i). Currently, a framework of indicators is being developed to monitor the drivers and impacts of chemical pollution and to measure the effectiveness of legislation on chemicals. It is likely that the chemicals strategy will deliver a list of substances that need to be addressed by soil monitoring.

    - **The Zero pollution action plan for air, soil and water** was published in May 2021 (EC, 2021c). The plan has the ambition to improve Member States' governance framework for preventing pollution. A Commission staff working document outlines a monitoring and outlook framework for the zero pollution ambition (EC, 2021d). It envisages regular reporting on (1) monitoring (relying on indicators on diffuse and local soil pollution) and (2) outlook, including a clean soil outlook.

In addition to the policies listed above, the EEA report *The European environment — state and outlook 2020* (EEA, 2019a) mentions several other policies with indirect effects on soil:

- Nitrates Directive (Directive 91/676/EEC)
- Sustainable Use of Pesticides Directive (Directive 2009/128/EC)
- Sewage Sludge Directive (Directive 86/278/EEC)
- Fertilisers Regulation (Regulation (EU) 2019/1009)
- Mercury Regulation (Regulation (EU) 2017/852)
- Plant Protection Products (Regulation EU 1107/2009).

Other initiatives to be mentioned are:

- The EU **Mission Board for Soil Health and Food** (EC, 2020a) has the ambition of supporting the European Green Deal and in particular the farm to fork strategy. Most noteworthy is the Board's 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 Board sets out eight objectives, which are complemented with quantitative targets. For each quantitative target, indicators for monitoring are specified (see Table 1.6).

- The international **'4 per 1000' initiative** (4 per 1000, 2022) recognises the importance of SOC sequestration in arable soils for climate change mitigation and food security. Initiated by the French Ministry of Agriculture, it was launched during the United Nations Framework Convention on Climate Change (UNFCCC) Conference of the Parties in Paris in 2015 (COP 21). Many European countries are partners of the initiative. The rationale is that an annual growth rate of 4‰ (4 per thousand or 0.4%) in SOC stocks in the top 40cm of all soils over a time frame of 20 years would be equivalent to annual anthropogenic carbon emissions of 8.9Gt, and therefore compensate for the annual increase in atmospheric CO_2 emissions arising from the agricultural sector. While the achievable CO_2 sequestration potential might be an overestimation (e.g. Van Groenigen et al., 2017; De Vries, 2018), the initiative successfully stimulates climate-smart agriculture by focusing on SOC sequestration.

- The **Sustainable Development Goals** (SDGs) were agreed in 2015 as part of the UN 2030 agenda for sustainable development. The EU committed to implementing the SDGs. Based on the Commission communication *Next steps for a sustainable European future* (EC, 2016), in 2017 the European Commission developed a reference indicator framework (EC, 2020j) to monitor the SDGs in the EU and, since then, has reported 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 (i.e. used for more than one goal). The following describes the methodologies of the most important global SDG indicators with regard to soils:

    - **Indicator 2.4.1**, 'Proportion of agricultural area under productive and sustainable agriculture', of SDG 2.4, focusing on sustainable food production systems and resilient agricultural practices. Its scope includes ecosystem maintenance and soils. At global level, it is doubted whether this indicator can be monitored with remote sensing, soil and water sampling. Therefore, the Food and Agricultural Organization of the United Nations (FAO) recommends farm surveys (FAO, 2018). In Europe, farmers may be capable of assessing the environmental impact of their practices. One of the 11 sub-indicators refers to soil health: 'Prevalence of soil degradation'. This sub-indicator follows the guidance set out by FAO and ITPS (2015), and proposes observing 10 soil threats. These are then reduced to four indicators:

        - soil erosion
        - reduction in soil fertility
        - salinisation 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:

        - desirable target: less than 10% of area affected
        - acceptable target: 10-50% of area affected
        - unsustainable target: more than 50% of area 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 worth noting that the 'one out, all out' principle ([13]) is applied, meaning that an area is considered degraded if only one indicator shows a negative trend.

All EU Member States and the European Commission have committed themselves to 'achieve a land degradation-neutral world by 2030'. The indicator on 'land degradation neutrality' includes three sub-indicators, namely land carbon stocks (above and below ground), land productivity and land cover change. Eurostat reports on land degradation in response to SDG target 15.3 using two soil-related indicators, namely 'soil sealing index' and 'estimated soil erosion by water'.

([13]) The 'one out, all out' (1OAO) principle considers changes in the sub-indicators: (1) positive or improving, (2) negative or declining or (3) 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).

Table 1.7 Soil-related policy objectives and targets at EU and global levels (binding and incentive-based non-binding policies and measures)

Policy document	Relevant policy objectives or targets
Water Framework Directive (2000/60/EC)	• Member States to produce river basin management plans, requiring the identification of point sources and their impacts • Member States to establish programmes of measures and implement 'basic' measures, including adapted agricultural production schemes to reduce nitrogen inputs to agricultural soils and as a consequence connected water bodies
Roadmap for a Resource Efficient Europe (EC, 2011)	• Soil erosion is reduced by 2050 • Increase in soil organic matter between 2011 and 2050 • By 2020 remedial work on contaminated sites well under way • Achieve no net land take by 2050
National Emission Ceilings Directive ((EU) 2016/2284) ([a])	• 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 Seventh Environment Action Programme) • To reduce the ecosystem area subject to eutrophication by 35% by 2030, compared with 2005 (Clean air programme for Europe (EC, 2013)) • 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 ecosystems)
LULUCF Regulation ((EU) 2018/841)	• To ensure the contribution of the LULUCF sector to the achievement of the EU's emission reduction target of at least 40% and to the long-term goal of the Paris Agreement in the period 2021-2030 • Member States have binding commitments to compensate CO2 emissions from the land use sector; land management practices that increase soil organic carbon stocks are accountable compensation measures
Regulation (EU) No 1306/2013 on the financing, management and monitoring of the CAP 2021-2027	• CAP post-2020 continues to promote practices beneficial to 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 • GAECs as conditional standards remain valid in the future CAP, including GAECs in support of soil protection and quality: GAEC 6 'Tillage management'; GAEC 7 'No bare soil'; GAEC 8: 'Crop rotation'
Eighth Environment Action Programme 2030 (EC, 2020b)	• Protecting, preserving and restoring biodiversity and enhancing natural capital, notably air, water, soil, and forest, freshwater, wetland and marine ecosystems • Umbrella programme ([b]) 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 (EC, 2020e)	• 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 the 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 (EC, 2020f)	• 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

Table 1.7 Soil-related policy objectives and targets at EU and global levels (binding and incentive-based non-binding policies and measures) (cont.)

Policy document	Relevant policy objectives or targets
Zero pollution action plan for air, soil and water (EC, 2021c)	• A zero pollution ambition for a toxin-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 (1) a monitoring framework on the state of pollution and (2) an outlook report, including a specific assessment of the evolution of human health and environmental impacts
EU Soil Strategy 2030 (EC, 2021e)	• By 2050, all EU soil ecosystems are in healthy condition and are thus more resilient • To combat desertification and to achieve no net land take, to restore degraded ecosystems including soils, • To contribute to (1) land-based climate neutrality by 2035, and (2) reducing the impact of soil pollution (on ecosystems, waters, human health) • To achieve progress in the management of contaminated sites

Notes: ([a]) Ecosystem monitoring under Article 9 and Annex V of the NEC Directive. The extent of the impacts of air pollution on ecosystems 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 Working Group on Effects under the Gothenburg Protocol to the Convention on Long-range Transboundary Air Pollution, including international cooperative programmes on forests, vegetation and integrated monitoring.

([b]) Including (1) 'A clean planet for all' (EC, 2018), followed by the long-term low greenhouse gas emission development strategy (EC, 2020k), (2) the circular economy action plan for a clean and competitive Europe (EC, 2020l), and (3) new strategies under the European Green Deal (biodiversity strategy 2030, farm to fork strategy, zero pollution action plan).

Considering the objectives and targets listed in Table 1.7, it becomes evident that monitoring of soil health indicators and associated evaluation schemes are needed. Assessments of the condition of European soils (JRC, 2012; EEA 2019a, EC 2020a) have been lacking a systematic and complete indicator set, and information about soil functions and its ecosystem services is still limited.

2
Soil organic carbon loss

Sufficient soil organic carbon (SOC) is a key element of healthy soils, affecting the quality of water, air, biodiversity and, ultimately, food and water security. Although SOC content has already been widely used as an indicator of soil health, it is challenging to define thresholds for optimal or critical SOC content below which soil functions are hampered. This is because complex biochemical processes are involved in SOC turnover, including mineralisation and stabilisation. In addition, soil and environmental conditions vary profoundly across Europe. The SOC content reflects an interplay between vegetation, climate and soil; depending on the chemical composition of SOC, its binding with soil minerals and its storage within soil aggregates, SOC content can respond rapidly to climatic changes or changes induced by land management, in particular its labile fraction. Nowadays, a natural equilibrium under undisturbed conditions is rarely seen; however, the thresholds for the lowest SOC contents necessary to ensure soil health are barely understood. This makes it difficult to determine the level at which soils are degraded as a result of loss of SOC content. In this chapter, several approaches to defining such thresholds are summarised. They mostly rely on the relationship between SOC content and crop yield response in agricultural soils but also include the role of SOC in the structural stability of soils.

Conservation of soil organic carbon (SOC) levels, or its increase where degraded, has positive impacts on almost all key societal needs related to soil and almost all soil functions, including achieving climate change mitigation targets and biodiversity conservation (Table 2.1). However, as with other soil threats, trade-offs between functions are possible, even if the underlying quantitative relationships are not always linear, depend on other site factors, and are partly still unresolved (such as that between SOC and productivity, and the role of SOC-rich soils and nutrient losses ([14]). The role of SOC in the context of infrastructure is ambiguous: SOC is generally lost through construction and sealing; however, relictic SOC in partially sealed soils and SOC in soils of green infrastructure fulfil important ecosystem services in urban areas.

Table 2.1 Relationship between SOC and key societal needs and soil functions

Societal need	Soil function	Impact
Biomass	Wood and 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	Indifferent for mineral soils ([a])
	Storage of geological material	Indifferent

Note: ([a]) SOC-rich soils (e.g. drained organic soils in estuaries) are unstable and unsuitable as a platform for construction because of subsidence and natural changes of the soil's drainage status (Trepel, 2015).

([14]) Decomposition of organic nitrogen in SOC-rich soils during phases of reduced nitrogen uptake by plants (e.g. after harvest) can be accompanied by nitrate losses.

2.1 Rationale: role of soil organic carbon in soil productivity and in filtering and storing water, nutrients and pollutants

Soil organic matter (SOM) content is closely related to almost all soil functions: it is a source of energy and carbon for soil organisms and affects the temperature and hydrology of soil; it affects soil aggregation (thus its erodibility), pore volume, the total reactive soil micro-surface, and thus biochemical processes including mineralisation rate and cation exchange, but also nitrogen losses and greenhouse gas emissions. Hence, SOM also affects the storage and release of nutrients and heavy metals, and it contributes to soil acidity (forest floors, Podsols) or its buffering (Blume et al., 2016). With regard to greenhouse gases, soils can, under certain conditions, sequester carbon and thus contribute to climate change mitigation, removing CO_2 from the atmosphere (IPCC, 2019b).

Since there is a close relationship between soil nutrient status and SOC content, it is not surprising that soil productivity can be closely related to SOM levels (Körschens et al., 2005; Feller et al., 2012). For example, Lal et al. (2011) estimate that an increase in SOC content of 1 tonne/ha in the root zone increases annual food production by 24-32 million tonnes of food grains (the authors refer to developing countries). SOM (as much as SOC) is today recognised as critical to preserving food security, and a decline in SOM levels 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 health in response to management impacts under various environmental conditions (Bünemann et al., 2018). While, under stable environmental conditions, the SOC stock develops to achieve a long-term equilibrium of mineralisation and stabilisation, changes in management and natural disturbances affect this equilibrium (Wiesmeier, 2019) and frequently lead to a depletion of the SOC stock, thus causing soil degradation. Likewise, if the SOC content falls below a certain threshold (or critical limit), major soil functions may be impaired to such a degree that a certain type management cannot be maintained.

SOC dynamics is closely related to nitrogen dynamics (Van Groenigen et al., 2015). Assuming that a C/N ratio of 12 is maintained in an agricultural soil, for instance, this means that to sequester 12 tonnes of SOC 1 tonne of nitrogen must be sequestered, or, vice versa, losing SOC inevitably entails a loss of soil nitrogen (Van Groenigen et al., 2017). Nitrogen affects the composition of microbial communities (e.g. proportion of fungi and bacteria), root turnover and the chemical composition of SOM. SOM (research is nowadays focused on SOC) is thus an important indicator to regulate the need for nitrogenous fertilisers. SOM may become a source of nitrogen, particularly during winter in agricultural fields, while the presence of SOM with a higher C/N ratio may also contribute to minimising environmental pollution (Musinguzi et al., 2013) due to microbial immobilisation processes. The C/N ratio of SOM finally affects the need for fertiliser nitrogen (Schjønning et al., 2018) as well as dissolved organic matter formation (Kindler et al., 2011) and nitrous oxide release (Mu et al., 2009). Establishing critical SOM (or SOC) levels or thresholds may help to restore SOC-depleted (and nitrogen-depleted) soils. Also, the relationship between SOC and soil fertility and crop yield (which is predominantly positive) has now become the basis for identifying SOC sequestration potentials (Amelung et al., 2020).

2.2 Indicator specification: 'Loss of SOC below critical levels'

2.2.1 Fractions relevant for characterising SOC dynamics

SOM is the sum of all dead organic components at various stages of decomposition in a soil, which are made from basic elements including carbon, nitrogen, oxygen, hydrogen and an array of attached cations and ions. 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 SOC. Historically, a factor of 1.724 has been used to convert SOC to SOM, 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 on forest floors, because of differences related to the different stages of decomposition and mineralisation. To avoid such uncertainties, we recommend not converting at all but using SOC as measured.

While the loss of total SOC concentration over a monitoring period is often suggested as an indicator, the change in 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 fractions of SOC vary considerably in their physical and chemical properties, resulting in a wide range of turnover (Gobin et al., 2011). Monitoring these different fractions is important, because it helps to understand how carbon dynamics in soil are affected by disturbances and how they can be effectively restored (Lehmann et al., 2008; Poeplau et al., 2018), and thus how to model the dynamics (Herbst et al., 2018; Lavallee et al., 2020).

The existence of humic substances (uncharacterised structural composition, persistent, large-molecular-size constituents) has been questioned (e.g. Schmidt et al., 2011). Rather, humus is perceived as a continuum of progressively decomposing organic compounds (Lehmann and Kleber, 2015): it is not 'humic substances', such as partly decomposed plant compounds, that form the basis for soil's carbon sequestration potential and soil fertility, but necromass (microbial residues and their biomolecular coating of dead fungi and bacteria). The contribution of microbe-derived carbon to SOC could be up to 82% (47-80%) (Liang et al., 2019) (see also Table 2.2). Yet, monitoring microbial necromass, for example on the basis of biomarker analyses (Amelung et al., 2008), is time consuming and not practical for large-scale monitoring. On that basis, Cotrufo et al. (2019) distinguish a mineral-associated organic matter (MAOM) pool, and a particulate organic matter (POM) pool. The latter may even be suitable for spectroscopic assessments (Reeves et al., 2006; Bornemann et al., 2010; Vohland et al., 2011), thus facilitating larger scale monitoring. More recently, a machine-learning model based on Rock-Eval thermal analysis (PartySoc) succeeded in partitioning SOC into its 'centennially stable' and 'centennially active' fractions (Cécillon et al., 2021).

It must also be considered that SOC storage and accumulation requires a specific amount of nitrogen (nitrogen efficiency of carbon sequestration). According to Cotrufo et al. (2019), this amount of nitrogen depends on the share of MAOM and POM, and their respective C/N ratios. POM consists of partly decomposed plant origin (with low nitrogen content), while MAOM is mostly of microbial origin and is chemically bound to minerals and thus physically protected in small aggregates. Cotrufo et al. (2019) hypothesise that any additional carbon storage in soil is realised only through POM accrual. As POM is more susceptible to decomposition and thus loss, assessing these SOC pools helps in determining C sequestration potential and in monitoring SOC in response to management actions (see also Christensen, 1992; Lavallee et al., 2020). However, there is indication that also the MAOM fraction can also be responsive to management (Trigalet et al., 2014).

Moreover, to avoid overestimations of SOC, the amount of inorganic SOC must be determined and removed from any total SOC estimate (in the case of calcareous soils or after liming). In such cases SOC is usually not measured but calculated from total SOC after subtracting the inorganic carbon part. Another critical component is the assessment of rock fragments, which, when present in significant amounts, reduce the available soil volume and thus concentrate SOC in the fine earth for a given SOC stock (Bornemann et al., 2011).

Table 2.2 SOM fractions/pools in soils

Organic matter fraction/pool	Approximate share of total SOM	Characteristics of pool
Microbial biomass C (bacteria, fungi)	1-5%	Labile, active
POM. It can be further subdivided into freshly added plant and animal residues, usually coarse POM with a low degree of decomposition (e.g. >250µm), and partially decomposed residues, frequently finer sized	5-25%	Labile, active
MAOM, largely consisting of microbial necromass and other adsorbed organic soil constituents	50-75%	Stable, activity level depends on degree of organo-mineral complexation
Inert organic matter, black carbon	5-20% or more	Constant, passive

Sources: Skjemstad et al. (2004); Rodionov et al. (2010); Gobin et al. (2011); Cotrufo et al. (2019); Lavallee et al. (2019).

2.2.2 SOC as an indicator

SOC content as an indicator is commonly expressed as concentration or stock (syn. pool size, density), and its quantification refers to a specific depth of soil. In this section, key references have been selected from the vast literature base on SOC measurement and monitoring:

- reference literature on SOC to address policy needs, including greenhouse gas inventories: Bispo et al. (2017), FAO (2019a), IPCC (2019b);
- reference literature on SOC analysis: Nelson and Sommers (1996), standard operating procedures of the Global Soil Laboratory Network (Glosolan) (FAO, 2022);
- reference literature on SOC monitoring: Goidts et al. (2009), Schrumpf et al. (2011), Poeplau et al. (2017), Arrouays et al. (2018);
- reference literature on SOC monitoring from remote sensing: Angelopoulou et al. (2019), Chabrillat et al. (2019),

In the context of assessing 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:

$$\text{Land area}_{\text{SOCdegraded}} = \text{Land area}_{\text{SOC content < threshold}}$$

$SOC_{content}$ is expressed as the concentration of organic carbon in fine soil (fractions <2mm), per mass of soil (expressed as grams C/kg soil, or a percentage), from a sample representing a certain soil layer or soil horizon of a specific depth. Organic soil residues are not included in this estimate when they do not pass through a 2-mm sieve.

While the existing thresholds presented below focus on the SOC content (or SOC concentration), they could be converted to SOC stocks assuming a specific depth and bulk density, although this conversion would be highly error prone. However, in future, the development of thresholds related to SOC stocks may be advisable, since SOC stocks could be more indicative for cropland soils in the case of changes in tillage depth (in particular, using reference and benchmark values from soil monitoring as thresholds; see Table 2.3).

SOC_{stock} represents the pool of organic carbon for a specific layer of soil. The quantification of this pool relies on $SOC_{content}$, bulk density, coarse mineral fragment content and layer thickness, expressed in tonnes C/ha, and calculated as:

$$SOC(stock) = \left(\frac{d * C * (1 - CM) * BD}{100}\right)$$

where SOC is the stock in tonnes C/ha, *d* is the depth (m), *C* is the content of organic carbon (grams C/kg), CM is the fraction of coarse material or rock fragments, by mass, and BD is the bulk density (kg/m^2) (see also Poeplau et al. (2019) about the coarse fraction for calculating SOC stocks).

SOC_{stock} is the reporting unit in greenhouse gas inventories (Goulding et al., 2013). Because bulk density is not always measured, pedotransfer functions are available (e.g. Hollis et al., 2012), although this approach is error prone (Wiesmeier et al., 2012). Farm advisers usually build their recommendations on $SOC_{concentration}$. Regarding the stone content, it is often neglected in agricultural topsoils; however, following recommendations made by the Intergovernmental Panel on Climate Change (IPCC, 2006), it is good practice to quantify carbon stocks in subsoils in order to quantify the amount of carbon vertically redistributed and the part lost (or gained) in the local soil under investigation. For this reason, the weights, or volumes, of fine gravel (inside the sampling cylinders for determining bulk density), coarse gravel and stones need to be estimated, in particular for non-agricultural soils and SOC estimates at greater depths.

When talking about SOC thresholds, we have to distinguish between mineral soils that contain only a few per cent of SOC, but that cover more than 90% of the land surface, and organic soils that are rich in SOC, such as bogs, fens, and folic histosols, which cover only 3% of the land surface but store more than 20% of all SOC on Earth (Yu et al., 2010; Scharlemann et al., 2014; Schimmel and Amelung, 2022). The management of these two categories of soils is also very different. For organic soils, SOC stock estimations are difficult: to monitor carbon sequestration, estimations should include the full depth of the organic layer.

2.3 Critical limits for soil organic carbon

2.3.1 Overview of approaches to determine degradation by SOC

Basing carbon thresholds only on soil fertility hardly accounts for other needs, for example climate change mitigation and other ecosystem services, such as water infiltration and groundwater recharge or biodiversity. In addition, the available research on thresholds frequently excludes non-agricultural land uses. Nevertheless, the effects of soil management and the derivation of SOC thresholds for sustainable (arable) soil management have usually focused on the effect of the decline in SOC levels on crop yield. This focus on yield benefits means that farmers are more likely to accept changes in management practices (e.g. Amelung et al., 2020). Yet, crop yield is also the result of the interaction of many factors, in particular soil fertility, for which SOM and nutrient availability are as important as sufficient water. Hence, correlations between SOC content and yields are not always clear, and the same applies to effects of organic matter additions to yield development (Hijbeek et al., 2017a; Vonk et al., 2020).

Oldfield et al. (2019) explored how SOC relates to crop yield potential in maize and wheat, considering co-varying factors of management, soil texture and climate; SOC was found to have an impact on yield with zero nitrogen inputs. The yields of these two crops are on average higher 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 also been found for cereals, even when fertiliser is applied (Pan et al., 2009). For rice, SOM content is positively correlated with yield, and this explains up to 70% of the yield under fertilization regime (Zhao et al., 2016). While these studies are often local, they nevertheless suggest globally valid thresholds for SOC-yield relationships. Loveland and Webb (2003) conclude that quantitative evidence for single thresholds in relation to crop yields is difficult to find. Rather, any typical SOC content can be determined only if specific soil, management and climatic conditions are considered. Given the diversity of soils and growing factors, a universal value for a critical minimum SOC level may not be appropriate (Goulding et al., 2013). It is possible that relationships between SOC and yield are crop, soil and/or region specific. Nevertheless, and even if the thresholds are also region specific and depend on, for example, soil texture, there are to our knowledge no indications in the literature yet that would support a SOC-induced yield improvement above the 2% SOC level.

The most often mentioned SOC threshold is 2% (Kemper and Koch, 1966; Greenland et al., 1975, both cited from Huber et al., 2008). Below this level, potentially serious degradation of soil can occur. These conclusions are supported by Shi et al. (2020): they defined SOC thresholds for aggregate stability in the Belgian loam belt and found that below 2% SOC soil aggregate stability deteriorates, with a mean weight diameter (MWD) of between 0.4mm and 0.8mm (Le Bissonais, 1996), while below 1.5% SOC soil aggregates were highly unstable with a MWD of less than 0.4mm.

Loveland and Webb (2003) summarised what is known about critical thresholds of SOC for different soil functions, mainly in soils in temperate regions (for tropical soils, see Musinguzi et al., 2013). They concluded that the quantitative evidence for thresholds at that time was still limited and recommended a SOC threshold of 1% as being more appropriate than 2%. Below the 1% SOC 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). Oelofse et al. (2015) also concluded that SOC levels below 1% may be insufficient to sustain yields (based on an evaluation of 869 Danish national field trials). Wessolek et al. (2008) questioned a global 2% threshold for SOC, because it cannot be achieved for soils with naturally low SOC levels — not even with an optimal supply of organic matter (e.g. sandy cropland soils in north-eastern Germany and other marginal soils).

In a review study, Pawar et al. (2017) concluded that existing SOC thresholds are largely located at the levels where soil function indicators perform sufficiently well while optimal yield is achieved. Examples of such soil function indicators are degree of water-stable soil aggregation, soil stability, water-holding capacity, micronutrient availability and cation exchange capacity. When SOC levels in soil are below 1%, soil health may be constrained and potential yields may not be achieved, while a minimum of 2% SOC is necessary to maintain structural soil stability, and, if SOC content is 1.2-1.5%, stability declines rapidly (Kay and Angers, 1999). Lopes et al. (2013) delineated a critical range of SOC for two soil orders: alfisols (corresponding largely to luvisols and lixisols in the World Reference Base for Soil Resources) with 0.5-0.77% SOC; and entisols (leptosols in the World Reference Base) with 1.03-1.16% SOC.

When cross-compliance was introduced to the EU common agricultural policy through Regulation 1782/2003, Member States were asked to define, at national or regional level, minimum requirements for good agricultural and environmental condition (GAEC). In that context, Germany had implemented national SOM thresholds (related to GAEC 2 'Maintenance of organic matter and soil structure', later called GAEC 6 under Council Regulation (EC) No 73/2009). The German implementation sought to conserve the site-typical humus content for sustainable crop growth: SOM must be more than 1% (0.6% SOC) if soil clay content is less than 13%, and SOM must be more than 1.5% (0.9% SOC) if soil clay content is more than 13% (Bundesministerium der Justiz, 2004). This national threshold has been replaced in 2014 because of its arbitrary setting, so that nowadays only the burning of straw is usually prohibited.

Table 2.3 presents an overview of the thresholds for SOC discussed in this report.

Table 2.3 Overview of SOC thresholds

Section of report	Definition	Comments on the practicality of existing thresholds
2.3.2	**Reference values, site-specific,** typical SOC or SOM values under current management (Arshad and Martin, 2002)	Can be derived from existing monitoring systems for different land uses (e.g. as baseline); 50% of the standard deviation of the mean are 'low', indicating deficiency
	Benchmark values	Requires extensive monitoring evaluations and large data sets to sufficiently define site-specific value ranges
	Near-natural forest soils (Arshad and Martin, 2002)	Problematic, because humus dynamics in cropland soils (low C input, high C turnover) are different from forest soils, leading to a lower level C equilibrium at cropland sites
	25th percentile of SOC values for permanent grassland (Sparling et al., 2003)	Pragmatic, but selection of quartile thresholds requires validation
	Modelled SOC steady state (25 years) for grassland (Sparling et al., 2003)	80% of steady state as target SOC value
	12.5th - 87.5th percentiles as upper and lower benchmarks	Example: Drexler et al. (2022)
	Optimal SOC content for soil functioning (based on the role of SOC in soil functional pedotransfer function, combined with data from long-term field experiments) (Wessolek et al., 2008)	Reference values for central European cropland soil and climatic conditions are available Needs to be validated for clay-rich soils and climate regions outside central Europe Values are site specific and ensure sufficient yield while not limiting natural soil functions. If the inert stable SOC fraction is known, Körschens et al. (1998) suggest that optimal SOC is C_{inert} + 0.62%(C_{inert}); a separate pedotransfer function is suggested to identify the inert fraction based on the clay content (Körschens and Schulz (1999); validity range by Körschens (1999): 400-800mm precipitation and 6-10°C average annual temperature). Note that C_{inert} as defined by Körschens et al. (1998), residual C at fallow trials, differs from C_{inert} commonly used in soil modelling (e.g. black carbon; Skjemstad et al., 2004)
2.3.3	Soil **vulnerability** index based on the SOC/clay ratio	Optimum SOC content, defined as 10% of the observed clay content (piloted in Switzerland, England and Wales)
2.3.4	Reciprocal SOC **sequestration** potential	Optimum SOC content for the 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); values are conceptually comparable to optimal values (see Section 2.3.2)
2.3.6	**Farmers' perspective** on deficient SOC	Degraded SOC levels according to farmers' perceptions (values for Europe)

Note: Benchmark sites reflect environmental and management conditions that are representative of a large area (Van Lynden et al., 2004). Each site represents a very specific set of local conditions that 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 inadequate, such that crop yields are affected even at optimal nutrient fertilisation rates. This concept is based on the fact that SOM provides and represents key properties of soils, while depending on and regulating various biologically mediated soil processes and functions.

In a simplified approach to thresholds, Arshad and Martin (2002) suggest deriving site-specific SOM values as references for monitoring and as proxies 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 would theoretically represent the highest SOC stock a given soil can achieve ('reference SOC stocks', according to Batjes (2011)). Yet, taking this approach, nearly all arable soils would be classified as degraded, because SOC is inevitably lost when breaking a native sod. 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. For New Zealand, Sparling et al. (2003) proposed the median SOC for permanent grassland as a target value, and its 25th percentile as a minimum value. This is a pragmatic solution and can be easily determined. The 25th percentile is conservative and seems quite realistic. An example of derive modelled reference SOC stocks is Lugato et al. (2015), who produced a spatially explicit estimation of soil carbon storage potential in European arable soils by 2050, applying different management scenarios to the Century model framework. Site-specific benchmarking is gaining increasing importance for model validation and threshold setting.

Wessolek et al. (2008) reviewed a great variety of soil models, long-term agricultural experiments, and ancillary soil analytical data sets (containing SOC, soil properties, crop type and crop yield, fertiliser application, atmospheric nitrogen deposition, nitrate loss). Pedotransfer functions were analysed for the models, which predict soil functions and potential threats (e.g. soil water storage, cation exchange capacity) and which have SOM as a driver. Despite the long history and vast amount of research invested in SOC dynamics, the derivation of site-specific SOC or SOM content in relation to soil function 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 long-term German field trials, the authors developed a matrix of SOM concentrations, depending on soil texture, climatic water balance and management intensity (type of fertilisation).

Tables 2.4 and 2.5 present matrixes of site- and management-specific reference values as derived by Wessolek et al. (2008). The approach seems promising and could serve as a proxy for minimum SOC values in soils. However, values need to be validated for different European regions and site conditions which are not well represented in Wessolek et al. (2008).

The validity range for the values in Table 2.4 is roughly 400-800mm precipitation and 6-10°C average annual temperature. In order to develop minimum SOC levels, a long-term extensive management regime is assumed (no fertilisation), and the values in Table 2.4 were reduced by 50% of the standard deviation (Table 2.5). Maximum SOC levels were defined to inform the condition under which intensive management can lead to nitrate losses as a trade-off between fertilisation and accumulation of soil organic matter; in particular, this concerns sites with naturally high SOC values once they are managed under an intensive fertilisation regime. While the initial focus of Wessolek et al. (2008) was to define the optimal SOC content for soil functioning, the current approach aims to define site-specific reference values, as suggested by Arshad and Martin (2002).

Table 2.4 Matrix of mean SOC target values (% soil mass) for mineral cropland soils based on extensive national soil data sets

Soil texture class	Management intensity (fertiliser)	Climatic water balance (mm) during summer ([a])		
		Less than -100	-100 to 0	More than 0
Sand	Max. both ([b])	1.01	1.51	2.01
	Organic and mineral	0.95	1.45	1.95
	Organic	0.83	1.33	1.83
	Mineral	0.73	1.23	1.73
	Null ([c])	0.70	1.2	1.7
Silt	Max. both	2.37	1.92	1.44
	Organic and mineral	2.19	1.72	1.24
	Organic	2.07	1.61	1.18
	Mineral	1.89	1.5	1.11
	Null	1.71	1.24	0.77
Loam and clay	Max. both	0.99	1.64	2.8
	Organic and mineral	0.95	1.2	2.67
	Organic	0.91	1.12	2.63
	Mineral	0.87	1.07	2.59
	Null	0.82	1.16	2.46

Notes: ([a]) Negative water balance: potential evapotranspiration more than precipitation during summer. Positive values indicate climate- induced surplus in the water budget from April to September.
([b]) Maximal application of organic and mineral fertiliser.
([c]) Null = no fertiliser applied

Source: Compiled from Wessolek et al. (2008). Data are valid for Germany and neighbouring countries for different soil textures, climatic conditions and fertilisation regimes.

Table 2.5 Matrix of mean SOC minimum and maximum thresholds for cropland soils (% soil mass)

Soil texture class	Climatic water balance (mm) summer					
	Less than -100		-100 to 0		More than 0	
	Min.	Max.	Min.	Max.	Min.	Max.
Sand	0.5	1.23	0.9	1.73	1.2	2.23
Silt	1.5	2.53	1.0	2.07	0.8	1.59
Loam and clay	0.6	1.47	0.9	1.92	1.9	3.23

Source: Compiled from Wessolek et al. (2008).

Minimum values in Table 2.5 present an approximation of a threshold for SOC deficiency. These values still require additional validation outside their representativity range. Moreover, the extent to which specific soil functions are lost once these thresholds are exceeded is not yet fully understood. In dry climatic regions, silty soils have the highest thresholds: Wessolek et al. (2008) explains this based on the high water storage capacity of silt, which allows high yields during the summer period.

Drexler et al. (2022) used the German inventory of agricultural soils first to define typical, site-specific SOC contents (taken from 0-30cm samples) and then to derive benchmark SOC values to guide agricultural management. Typical site-specific SOC values range from 0.89% to 2.3% for light and heavy arable soils, and from 2.2% to 6.0% for grassland soils. The data set was optimised by stratifying it into 33 strata: land use (cropland, permanent grassland and ley-arable rotation), soil texture and mean annual precipitation (see Table 2.6), C/N ratio (C_{org}/N_t ratio <13, 13-15, >15). Lower and upper benchmarks are defined as the 12.5% and 87.5% quartiles for each stratum, which then excludes 25% of sites with extreme SOC values (as non-site-specific). Of particular interest here are values below the lower benchmark which could be due to unsustainable management (the authors also found such sites with low mean annual carbon inputs from organic fertilisers). The effects of soil threats were not investigated, so a clear relationship with soil degradation cannot be determined.

National estimations for site-typical SOC contents exist for many other European countries, allowing national benchmark SOC values to be implemented in the same way as Drexler et al. (2022): for example for the United Kingdom, Verheijen et al. (2005), with 1.1% SOC for drier conditions and low clay content and up to 4% for wetter conditions and high clay content; and for Luxembourg, Chartin et al. (2020). It should also be noted that values change under the influence of groundwater (Wessolek et al., 2008; Drexler et al., 2020). For the Mediterranean region, Grilli et al. (2021) identified maximum and mean values of SOC content for soils at risk of desertification, i.e. 1.5% for cropland.

Such site-specific SOC ranges could help identify SOC-depleted sites that clearly fall below an agreed threshold calculated from representative soil monitoring. Such site-typical SOC ranges are already being used by farmers as reference values (e.g. Wiesmeier et al., 2019). Such values could be modified based on typical SOC levels relative to concentrations under best practice management for comparable site conditions (to establish benchmark values), thus defining and quantifying a desirable target condition (e.g. healthy soils have at least 75% of the benchmark SOC content). However, such benchmark values do not allow direct evaluation of specific soil functions.

Table 2.6 Site-specific lower SOC benchmarks for German agricultural soils across different strata

Texture ([a])	Arable soil				Permanent grassland			
	Low MAP	Lower benchmark	High MAP	Lower benchmark	Low MAP	Lower benchmark	High MAP	Lower benchmark
Light	<700	0.7	>700	0.9	<650	1.4	>650	2.0
Medium	<850	1.0	>850	1.1	<950	2.3	<950	3.0
Medium II	<1,000	1.1	>1,000	1.4	<1,250	2.7	>1,250	3.1
Heavy I	<900	1.2	>900	1.4	<750	3.3	>750	3.7
Heavy II	<900	1.5	>900	2.0	<750	3.7	>750	4.3

Notes: ([a]) Light <12% clay, <50% silt; medium I 12-17% clay, <50% silt; medium II 17-25% clay; heavy I 25-35% clay; heavy II >35% clay. MAP, mean annual precipitation (mm).

Source: Compiled from Drexler et al. (2002).

2.3.3 SOC/clay ratio

As stated above, SOC content is strongly correlated with various other soil properties, in particular, the clay content. With increasing clay content, an increasing amount of SOC is stabilised against decay and thus protected from decomposition. Therefore, the higher the clay content, the higher the threshold for a SOC value that can be sustainably achieved.

In a recent study, Johannes et al. (2017) reviewed and investigated the role of soil structural parameters (aggregate stability, porosity, mechanical properties, penetration resistance) and soil texture and its relationship with soil organic matter. Their study was inspired by the work of Dexter et al. (2008), who studied the relationship between soil texture, in particular the clay content, and SOC content. They proposed that the optimum SOC content was 10% of the clay content, later specified by others as dispersible clay rather than total clay (e.g. Schjønning et al., 2012). 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 2.7). The threshold translates into a vulnerability limit: %SOC = 0.1×%clay. Prout et al. (2020) suggest a vulnerability limit of less than 1/13 as the threshold to indicate degradation because hardly any grassland and woodland sites fall into that category. Nevertheless, it seems that the SOC/clay ratio as an indicator of good soil structure applies to a wide range of soils and land uses (arable land, grassland and woodland), and can be used to monitor and understand the state of soils at larger scales. For England and Wales, Prout et al. (2020) found that 38.2%, 6.6% and 5.6% of arable, grassland and woodland sites, respectively, were degraded as a result of loss of SOC. The data correspond well with preliminary observations for Wallonia indicating degraded land with a SOC/clay ratio of 0.08, transitional land with SOC/clay ratios of between 0.08 and 0.1, and favourable and highly favourable land with SOC/clay ratios above 0.1 and 0.12, respectively (C. B. Chartin, Université Catholique de Louvain, personal communication, August, 2022).

It can be expected that any increase in SOC has a positive effect on the recovery of soils from degradation due to 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 of soils (Fell et al., 2018).

The present few studies indicate that the SOC/clay ratio seems to be a valid threshold for soil structural stability under at least all western and central European conditions, with soils typically dominated by 2:1 layer clay minerals. The ratio will probably have to be translated into a different threshold for the tropics and for volcanic soils with different clay mineralogy.

Table 2.7 SOC/clay ratio as an index of good soil structure

<table>
<tr><th>SOC/clay ratio</th><th>Explanation</th><th>Soil structure ([a])</th><th>Explanation</th></tr>
<tr><td>>0.125 (1/8)</td><td>Field-level optimum for good structural quality ([b])</td><td rowspan="2">>1/10 (VESS <3)</td><td rowspan="2">Acceptable or good soil 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 likely to be unacceptable ([c])</td><td>< 1/10 (VESS >3)</td><td>Degraded soil structure</td></tr>
</table>

Notes: ([a]) VESS, visual evaluation of soil structure ([15]) (Ball et al., 2017; see also Chapter 8); the score ranges from 1 (good structure) to 5 (poor structure).
([b]) SOC enriched relative to the clay content.
([c]) SOC depleted relative to the clay content.

Source: Compiled from Johannes et al. (2017).

([15]) A simple description of the method and a video are available at https://www.sruc.ac.uk/info/120625/visual_evaluation_of_soil_structure/1553/visual_evaluation_of_soil_structure_-_method_description

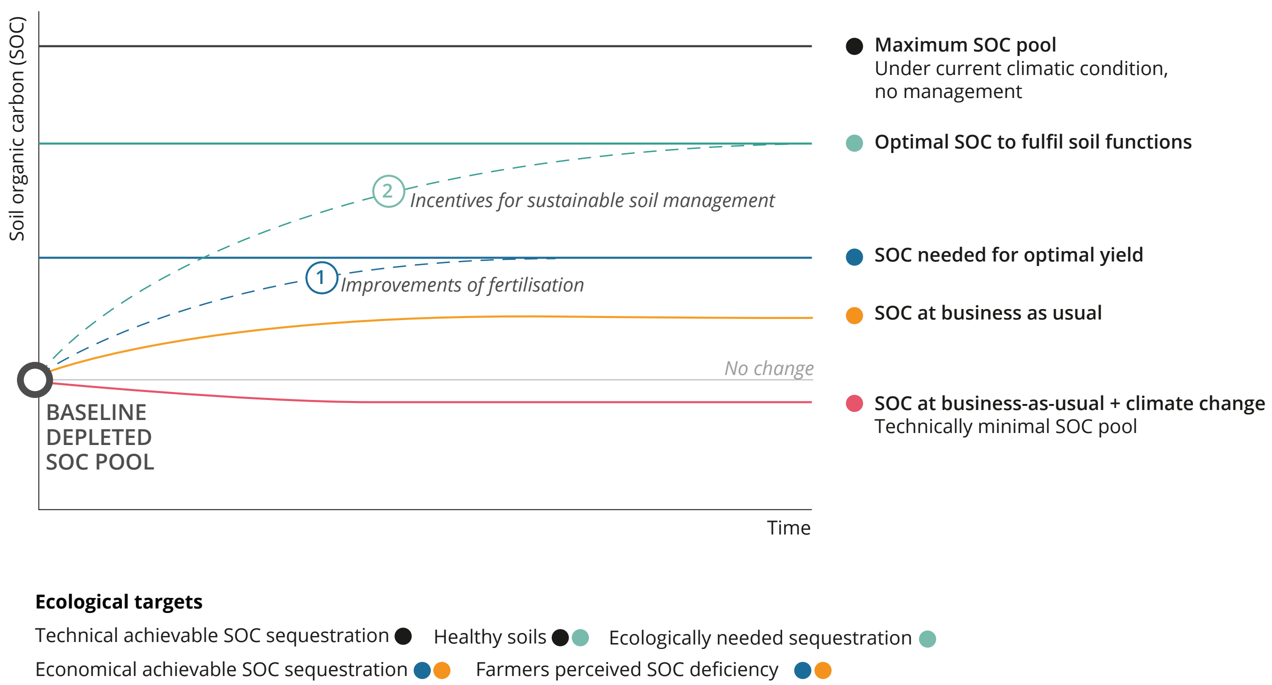
2.3.4 *SOC critical limits as a reciprocal of the SOC sequestration potential*

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). Soils' potential to sequester carbon arises because historical management has depleted the carbon pool of many soils. And even nowadays, certain soils still lose carbon under current management in some areas of Europe, especially cultivated carbon-rich soils (drained organic soils) and soils that have undergone forest and grassland conversions.

Among the many benefits of SOC, Amelung et al. (2020) emphasise the potential contribution of soil to mitigating CO_2 increases in the atmosphere and thus its potential contribution to stabilising the climate. The major potential for carbon sequestration is in cropland soils, especially where large yield differences still exist and/or where large historical SOC losses have occurred (Amelung et al., 2020; Lessmann et al., 2021). A map of yield differences could then indirectly provide reference values for carbon sequestration potential, hence SOC limit values. It would basically correspond to the SOC level achieved with optimal fertiliser management.

Figure 2.1 presents the theoretical concepts underlying SOC thresholds and how they relate to soil degradation and carbon sequestration. In agricultural systems the achievable optimal level of SOC differs for arable cropping systems with annual crops and perennial systems, such as grassland and forest ecosystems. Additionally, some aspects of SOC dynamics under different management regimes and climate change remain uncertain, for example how optimal SOC increases the resilience of soils to climate change and how this could help mitigate 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 focused primarily on the productivity-related function of SOC (yield), which is not sufficient to prepare soils for future arid 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 the conditions under which SOC sequestration is realistically possible (see also Amundson and Biardeau, 2018).

Figure 2.1 Conceptual overview of SOC thresholds and carbon sequestration



Source: EEA.

It becomes clear from Figure 2.1 that some thresholds identified in this report correspond to the SOC sequestration potential of soils. In particular the optimal SOC content, which corresponds to healthy soil, 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 a guide for target values for optimal SOC content, to be derived from modelling or sampling undisturbed locations.

When soils remained undisturbed, i.e. under prolonged native site conditions with climax vegetation, we can assume that the maximum level of SOC is stored. 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, soil degradation from SOC decline becomes apparent through yield gaps or yield decreases relative to a benchmark; however, there are various causes of such yield effects, as the decline in SOC content depends significantly on local conditions. Commonly, especially amongst farmers, the full range of SOC's ecosystem services (including stabilisation, protection from climate change and the improved water dynamics of SOC-enriched soils) is still hardly accounted for, and any SOC loss is rarely noticed (Hijbeek et al., 2017b). This is probably the reason why available soil degradation maps which focus only on the production functions of soils, are not considered reliable (Gibbs and Salmon, 2015; Amelung et al., 2020).

The difference between the actual and the potential SOC stock can be used as an indicator of the potential SOC sink capacity. It is larger than the true SOC sequestration potential, at least in the time scales under consideration (e.g. to be achieved by 2050), because even reconversion of arable land to grassland or forest might fail to sequester SOC to the level of the native ecosystem within a century (Lugato et al., 2015). Hence, the technical achievable SOC sequestration (technical ASC; Figure 2.1) will not necessarily reach the SOC levels of a native, undisturbed ecosystem with climax vegetation unless the time to reach that point were allowed to span millennia. On the other hand, and depending on the SOC management scenario chosen for the modelling of the potential future SOC stock and the level of degradation of the current SOC levels for cropland, the overall SOC sequestration potential is likely to be higher than the 'optimum SOC' for soils (compare with Section 2.3.2).

Current methods to calculate the SOC sequestration potential use models to compare baselines with assumed management scenarios: business-as-usual (BAU) and sustainable soil management methods (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 former SOC loss, the higher the current carbon SOC sequestration potential.

The approach has not been tested or validated for organic, sandy (>90% sand), saline (>4dS/m in the top 30cm), and waterlogged soils.

In mineral soils, the soil matrix is believed to have a finite storage capacity for organic carbon, thus limiting SOC storage in certain pools such as the silt-clay fraction (Hassink, 1997). This is based on the observation that stable SOC compounds are particularly adsorbed onto the reactive mineral surface of the smallest soil particles (see also Kleber et al., 2015). These findings were taken further by Wiesmeier et al. (2014), who reported significant amounts of carbon stored in the >20µm fraction (>20% in the analysed literature, 40-60% in his study). Moreover, a more recent report by Lawrence et al. (2018) stresses that it is the fine fraction carbon that should be used as an indicator of soils' capacity to stabilise carbon. Six et al. (2002) thus proposed a SOC saturation limit that includes the unprotected carbon. Angers et al. (2011) finally defined the SOC saturation deficit as the difference between theoretical carbon saturation (calculated according to Hassink's regression function: C_{sat} = 4.09-+ 0.37(%clay+fine silt), C in g/kg soil) minus the measured content of organic carbon in the clay + fine silt fraction (<20µm) of soil. Stewart et al. (2007) concluded that the greatest efficiency of soil carbon sequestration will occur in soils far from carbon saturation. This applies to nearly all arable soils. However, considering that a large proportion of the additional increase in SOC (e.g. through carbon farming and elevated carbon inputs) might also occur in coarse soil fractions such as POM (Gulde et al., 2008), and considering that this SOC is less stable, any such newly stored SOC may be rapidly lost if the change in farming practice is not 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 study the influence of a large spectrum of SOM contents on yield and C and N dynamics. The authors propose an upper limit for SOM content, above which there is an increased risk of nitrogen and CO_2 loss; lower limits represent the SOM level required 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 2.8). This means that, even with the addition of fertiliser, below 1% SOM, mineralisable nitrogen is so low that potential optimal yields can no longer be achieved. Soils under the influence of groundwater have been excluded from these considerations. For comparability, initial SOM values were converted to SOC using the factor 1.724.

Table 2.8 **Guideline ranges for SOC content of sandy and loamy soils without groundwater influence (% SOC in plough layer) depending on fine silt (<6.3µm) and clay content (Pipette method)**

Clay + fine silt (%)	SOC in sandy soils (%)		SOC in loamy soils (%)	
	Upper value	Lower value	Upper value	Lower value
4	0.9	0.6		
5	0.9	0.6		
6	0.9	0.6		
7	0.9	0.6		
8	0.9	0.6		
9	1.0	0.7		
10	1.0	0.7	1.2	0.8
11	1.0	0.8	1.2	0.8
12	1.1	0.8	1.3	0.8
13	1.1	0.8	1.3	0.9
14	1.2	0.9	1.3	0.9
15	1.2	0.9	1.4	1,0
16	1.2	0.9	1.5	1.0
17	1.3	1.0	1.5	1.0
18	1.3	1.0	1.6	1.1
19	1.3	1.0	1.6	1.2
20	1.4	1.1	1.6	1.2
21	1.5	1.2	1.7	1.2
22	1.5	1.2	1.7	1.3
23	1.5	1.2	1.8	1.3
24	1.6	1.3	1.9	1.4
25	1.6	1.3	1.9	1.5
26			2.0	1.5
27			2.0	1.5
28			2.0	1.6
29			2.1	1.6
30			2.1	1.6
31			2.2	1.7
32			2.3	1.7
33			2.3	1.8
34			2.4	1.9
35			2.4	1.9
36			2.4	1.9
37			2.5	2.0
38			2.6	2.0

Source: Compiled from Körschens et al. (1998).

Table 2.9 Aggregation of SOC thresholds for soil groups

Soil group	Clay/fine silt (%)	Körschens et al. (1998)		BMLFUW (2017) Minimum SOC threshold	SOC/clay ratio		
		Sandy soils	Loamy soils		1/8 (optimum for good structural quality)	1/10 (minimum desired SOC level for farmers)	1/13 (structural soil quality is unacceptable)
Light (<15% clay)	4-7	<0.6	-	1.2	<1.9	<1.5	<1.2
	8-14	0.6-0.9	0.8-0.9				
Medium (15-25% clay)	15-22	0.9-1.2	1.0-1.3	1.5	1.9-3.1	1.5-2.5	1.2-1.9
	23-25	1.3-1.6	1.3-1.5				
Heavy (>25% clay)	25-32	-	1.5-1.7	1.7	>3.1	>2.5	>1.9
	>32	-	1.7-2.0				

The thresholds developed by Körschens et al. (1998) demonstrate how the existing variability in soil properties affects the development of thresholds. As soon as values become simplified or grouped for easier application, ranges of values apply (see Table 2.9). 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 Körschens et al. (1998), could be more easily implemented to provide guidance for optimal fertiliser application on arable land and pastures (Table 2.9).

It is currently difficult to apply the thresholds defined by Körschens et al. (1998) in Europe with the available Europe-wide texture data: the 'fine silt' class (6.3-2μm) cannot be isolated. Besides, no definition is given for 'sandy soils' or 'loamy soils'. Like the results in Tables 2.4 and 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 a guide to evaluating SOC measurements from monitoring.

2.3.6 SOM thresholds from a farmer survey

Farmers' perceptions of the levels of SOM needed to maintain agricultural production levels in a sustainable manner were investigated by Hijbeek et al. (2017b) and Vonk et al. (2020). Besides a literature review, an extensive farm survey was conducted involving 1,452 arable farmers in five European countries (Austria, Belgium, Germany, Italy and Spain). Thresholds were derived based on a subset of 635 farmers who also reported an average farm-level SOM content (farmers reporting peat soils or reporting an average SOM content above 12% were excluded; land use included the percentage of farmland under cereals and/or grass, and the percentage under specialised and/or horticultural crops, and a mixture of these two categories and forage crops). Frequency distributions were stratified by soil texture and macroclimatic region (Hijbeek and Trombetti 2020, for this report). Due to the 'fuzziness' of the responses to the questionnaire (some farmers perceive a given level of SOM as deficient, whereas others perceive it as adequate), and the corresponding statistical weakness, only two thresholds were derived in order to take a conservative approach (see Tables 2.10 and 2.11).

Table 2.10 Definition of SOM thresholds derived from a farmer questionnaire

Threshold 1	Threshold 2
Lower end of SOM value range (10th percentile) at which farmers judge their current SOM content sufficient	Upper end of the range of values (90th percentile) at which farmers perceive a high or very high deficiency of SOM content
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
1.1% of the European cropland area is concerned (in particular northern France)	14.3% of the cropland area (mostly central/eastern Europe and southern Europe)

Sources: Elaborated based on Hijbeek et al. (2017b); SOC data for the area estimates are from De Brogniez et al. (2015).

Table 2.11 SOC thresholds for cropland (% soil mass) derived from a farmer questionnaire

Climate ([a])	Representativity	Texture	Threshold 1	Threshold 2
Atlantic	*n*=358 (Belgium, Germany)	Coarse	1.2	2.0
		Medium	1.0	1.5
		Medium fine	1.6 ([b])	-
Continental	*n*=167 (Austria, Germany)	Coarse	1.0	1.2
		Medium	1.3	1.9 ([b])
		Medium fine	1.2	1.4 ([b])
Mediterranean	*n*=110 (Italy, Spain)	Coarse	-	-
		Medium	0.6	1.2
		Medium fine	0.8	0.8 ([b])

Notes: ([a]) Climate regions according to Metzger et al. (2005); No farm has been included for the Boreal region: for this region, thresholds have been derived from literature (Soinne et al., 2016): there, a threshold of 4% for SOC (8% SOM) can be considered for finer soils (clay > 30%) while, for coarser soils, a 2% threshold for SOC (4% SOM) could be considered based on expert judgement.
([b]) particular high uncertainty due to low number of observations

Source: Compiled from Hijbeek et al. (2017b); SOC values have been recalculated from original SOM values by applying the conversion factor of 1.724.

When applying farmers' perception of SOC depletion as a threshold, very little cropland is identified as 'degraded' in most of the intensively managed agricultural areas in Europe. This low threshold largely considers SOC in relation to yield dynamics, and excludes the role of SOC for other ecosystem services. It is obvious that the initial SOC content of uncultivated soils under permanent vegetation cover was originally much higher. 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. However, this loss of soil function, and the regaining of soil functioning with increasing SOC, is still difficult to quantify (see also Wiesmeier et al., 2019).

2.4 Conclusions for soil organic carbon monitoring

There is a wealth of literature investigating critical limits for SOC content. At the same time, in the context of developing national greenhouse gas inventories, and in the context of many national soil policies and EU-wide initiatives (e.g. LUCAS Soil, see also Orgiazzi et al., 2018), soil monitoring is now capable of producing representative SOC data, although the resolution is frequently still too coarse for site-specific recommendations. Many authors conclude that quantitative evidence for single thresholds needs to consider site-specific conditions (at least soil texture and climate, but also current and historical land use, distance to groundwater, slope, etc.), and so a universal SOC threshold seems meaningless. The critical limits compiled in this report are largely limited to cropland, while some also include grassland. Current values largely focus on central Europe, and knowledge of Mediterranean conditions in particular is limited.

Several minimum SOC levels have been suggested for cropland, frequently as deviations from typical site-specific values and in relation to other soil properties such as clay or clay + fine silt content. When applying these thresholds, SOC degradation is evident: for example, in Germany, 13% of the cropland was found to be below the benchmark SOC content; and in the EU, using the SOC/clay ratio, 37.1% of agricultural land appears to be SOC degraded.

[The results of applying the indicators presented in this report will be presented in a separate report by the European Topic Centre on Data Integration and Digitization (ETC/DI), in 2023); the current spatial estimate of SOC deficiency requires regional validation, in particular for the Mediterranean region.]

This review has also revealed that upper limit values for SOC also need to be considered, because excess SOC content introduces the risk of nitrate loss or greenhouse gas emissions, especially during winter. The definition of optimum SOC stocks is difficult, even if achievable, because optimum SOC content is not only site specific but possibly also different for the various soil functions.

Specific monitoring programmes are needed for organic soils and their associated depth of accrued organic matter. The programmes for SOC monitoring in mineral soils are useful, but they should aim to increase their spatial resolution at the EU level and include not only assessment of bulk density, texture and possibly fine fraction carbon (<20µm, MAOM) but also yield differences and information on past and current land use.

3 Soil nutrient loss: nitrogen and phosphorus

Soil nutrient status affects biomass production in natural soils and crop yields in agricultural soils, although the impact is less in fertilised 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, phosphorus, calcium, magnesium, potassium and sulphur, and micronutrients, i.e. boron, zinc, manganese, iron, copper and molybdenum. In addition, it affects the diversity of soil microorganisms, soil animals and plant species, which can be positive or negative. In general, a high nitrogen status reduces the diversity but the increase in other nutrients may either increase or decrease diversity. A higher nutrient status further increases carbon storage, especially by the enhanced crop residue input. Finally, a higher nutrient status may reduce water quality, especially in the case of a high phosphorus status, which increases the risk of phosphorus run-off to surface water.

The most important soil nutrients are nitrogen (N) and phosphorus (P). Together with soil pH, which is strongly related to the availability of the base cations Ca^{2+}, Mg^{2+} and K^+ and also to the availability of micronutrients (especially iron, zinc and magnesium) and of toxic aluminium, they are the main determinants of soil fertility. These soil parameters, i.e. N, P and pH, with their related impact on other elements, are affected by inputs from fertiliser and manure application in agriculture and by atmospheric deposition — the latter being the main source of inputs in non-agricultural soil.

This section focuses on the importance of N and P monitoring. The impact of soil N and P status and inputs are described, considering biomass production and crop growth, soil and plant biodiversity and water quality (see also Table 3.1). This is followed by an overview of indicators for soil N and P status related to those impacts. Finally, **thresholds** are described, below which the nutrient status should not drop to avoid a decline in crop yield (**target levels**), or above which the nutrient status should not rise to avoid adverse impacts on biodiversity and water quality (**critical levels**). In Chapter 4, the same is done for soil acidity.

Table 3.1 Relationship of soil nutrient status to key societal needs and soil functions

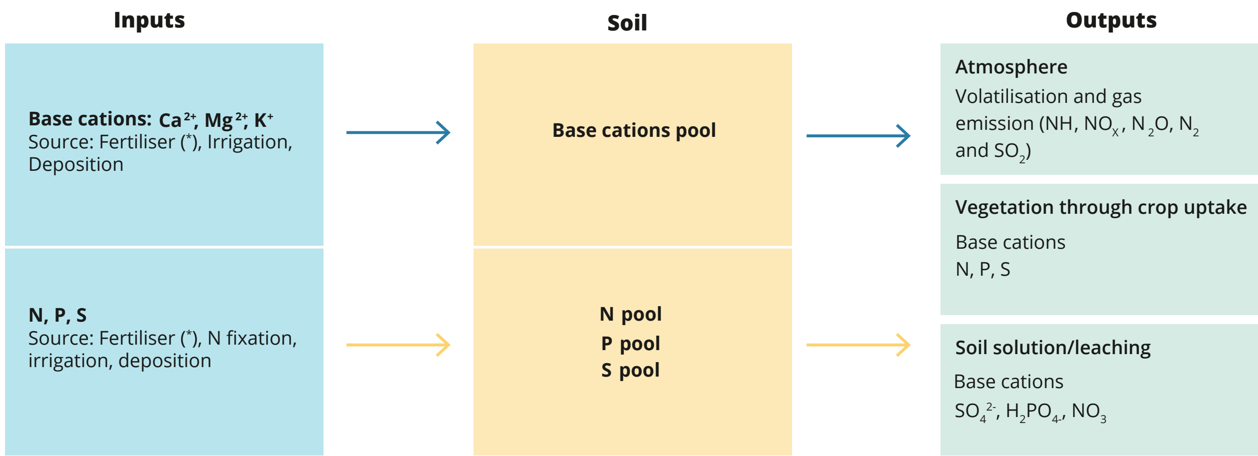
Societal need	Soil service	Impact
Biomass	Wood and fibre production	+
	Growth of crops	+
Water	Filtering of contaminants: water quality	-
	Water storage	Indifferent
Climate	Carbon storage	+
Biodiversity	Habitat for plants, insects, microbes, fungi	+ or -
Infrastructure	Platform for infrastructure	Indifferent
	Storage of relocated material or artefacts (excavated geological material, sediments, cables and pipelines, archaeological material)	Indifferent

3.1 Rationale: impacts of soil nitrogen and phosphorus 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 (contents and pools), which in turn affects the output of nutrients in vegetation (nutrient uptake due to biomass production, such as tree growth and crop growth) and to air and water (Figure 3.1). N and 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 they may cause eutrophication (see Section 3.1.3), and growth may even be reduced at very high levels of inputs (see Section 3.1.2). In agricultural soils, crop growth is strongly stimulated by N and P fertiliser and manure inputs, but the risk of N and P losses to water and the related impacts of eutrophication is high. The losses of N and P to air, and N and P to water in response to their inputs, is highly variable because of the differences in soil properties and corresponding N and P dynamics, as explained below.

Figure 3.1 Link between nutrient inputs and soil nutrient pools and nutrient outputs to air, vegetation and water



Note: (*) Including sewage sludge, waste waters, composts, digestates.

Source: EEA.

Nitrogen

Unlike other nutrients, such as P, calcium, magnesium, potassium and sulphur, N in (agricultural) soils is only organically derived, since there are hardly any or no N-containing minerals in (agricultural) soils. Processes such as N dissolution/weathering, or inversely precipitation, thus do not occur, and there is also very limited sorption of nitrate to either clay or organic matter, although it has been found to occur on positively charged soil particles (Wong et al., 1990, 2009), while ammonium has been found to adsorb in heavy clay soils but is generally transformed into nitrate in agricultural soils. 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 availability of N to crops in agricultural soils. Important soil parameters affecting soil N availability include pH, organic matter and clay content. When N is added to soil, a part is consumed by vegetation, and the N surplus is emitted to air, as ammonia (NH_3), nitric oxide (NO), nitrous oxide (N_2O) or dinitrogen (N_2), or to groundwater and surface water, mainly as nitrate (NO_3^-), or it accumulates in soil (Figure 3.1); however, the long-term change in the stock of organic soil N is very limited. Due to this behaviour, changes in N inputs from fertiliser and manure directly affect crop yields, even in soils with a high soil N status.

Phosphorus

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 groundwater and surface water. Because of this, changes in P inputs from fertiliser and manure have smaller impacts on crop yields in soils with a high P content. A meta-analysis of the yield response to long-term phosphorous fertilisation from 30 field experiments in Germany and Austria showed a strong decline in the magnitude of the yield response with increasing soil P content, with no effect at high soil P status (Buczko et al., 2018). Important soil parameters affecting the availability of soil P include the aluminium and iron oxide content.

3.1.2 Impact of N and P on biomass production and soil carbon sequestration

Agricultural soils and crop growth/soil carbon sequestration

In unfertilised soils, crop growth is determined by the soil N status and specifically the soil P status, as described above. In case of fertilisation, however, the N status of the soil is not very relevant, as N is readily available, and N limitation due to the soil N status can be compensated for by N fertilisation. In agricultural systems, N inputs from fertiliser and manure thus mainly affect crop yield, thereby also increasing the carbon (C) input by crop residues. On the other hand, soil N input may enhance soil C decomposition, and it has been claimed that inorganic N fertiliser leads to a decline in both the organic C (Khan et al., 2007) and organic N (Mulvaney et al., 2009) content based on an analysis of soil C and N data from long-term field experiments. However, both assessments were heavily criticised in the literature (Reid, 2008; Powlson et al., 2010) and there are various data sets indicating that enhanced N inputs lead to an increase in soil C in agricultural soils with N limitation (Amelung et al., 2020).

The P status may highly affect long-term crop yield, and crop yield increases in response to P fertilisation until a threshold is reached, above which crop yield no longer responds to P application (Mallarino and Blackmer, 1992). For example, Jungk et al. (1993) found that 14 years of different P fertiliser application rates varying between zero and 180kg P_2O_5 per year at four equal intervals hardly affected the yield or 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 accumulated from previous soil P applications in excess of plant demand. This critical P level is affected, however, by soil properties such as clay and organic matter content and also by climatic variables, such as temperature (Buczko et al., 2018; Hirte et al., 2021). In summary, N fertilisation mainly affects crop growth, but the impact of soil N status is limited, while soil P status affects crop yield up to a critical level.

Forest soils and forest growth/soil carbon sequestration

In unfertilised soils, such as terrestrial ecosystems, biomass production is generally N and/or P limited. This holds specifically for forest ecosystems, in which tree growth is generally limited by the N availability, which in turn is affected by the soil N status (especially the soil C/N ratio and the P availability). Increased N deposition often thus stimulates forest growth and hence C sequestration (Högberg, 2007; De Vries et al., 2009; Thomas et al., 2010), considering that in forests SOC decomposition is often reduced in response to (high) N deposition (e.g. Janssens et al., 2010).The N-induced increase in growth can be diminished, however, when the accompanying 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, which is also generally low. Newman (1995) reviewed P deposition and weathering in global terrestrial ecosystems and estimated a range of 0.07-1.7kg/ha/year for P deposition and 0.01-1.0kg/ha/year 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 area of ecosystems where P is limited. Using the leaf N/P ratio of 15 dominant tree species as an indicator, the spatial variation in plot-level shifts towards N or P limitation across 163 European forest plots during the period 1995-2017 has been demonstrated. In total, 38% of the plots studied shifted towards P limitation, while only 6% of the plots shifted towards N limitation, as indicated by a significant increase and decrease in leaf N/P ratio, respectively. Forests are thus increasingly suffering from P deficiency (Talkner et al., 2020). Beech forests especially 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.

N is a very important driver whose impacts on C sequestration is still rather controversial. Various studies suggest that leaf litter with high N concentrations (or high N/lignin ratios) decomposes faster than litter with a lower N content, but this difference in decomposition rate reverts during the later stages of the decomposition process. N addition thus seems to accelerate decomposition of low-lignin 'easily degradable' litter but to reduce decomposition rates of high-lignin 'recalcitrant' substrates. Whether the addition of N ultimately causes an increase or decrease in soil C sequestration then depends on the ratio of low-lignin to high-lignin litter.

In natural systems (especially forests), the addition of N retards below-ground C cycling, thus leading to N induced increases in (soil) C sequestration (e.g. Janssens et al., 2010). There are also conditions under which N addition accelerates decomposition and soil respiration, thus decreasing soil C sequestration, such as forests with severe N limitation or, conversely, strong N saturation, but this seems to be a minority of systems. In agricultural systems, however, the impacts of N inputs on soil C sequestration are less clear.

3.1.3 Impact of N and P on biodiversity and water quality

Soil nutrient (N and P) status affects not only crop yields, but N in soil also affects the biodiversity of soil organisms, whereas the soil P content has an impact on surface water quality by affecting P accumulation and losses to surface water. Finally, both N and P affect biodiversity, especially plant species diversity, in non-agricultural ecosystems, as discussed below.

N and soil biodiversity

Ample available N, induced by increased atmospheric N deposition and, more specifically, by N fertilisation in agricultural soils, affects terrestrial ecosystems. These N inputs reduce the abundance, activity and composition of soil fungi, saprotrophic decomposers, mycorrhizal fungi and N fixing bacteria (Streeter, 1988; Johansson et al., 2004; De Vries et al., 2006). Both plant litter and microorganisms are the food source of detritivores in the soil food webs; thus, excess N leads to bottom-up effects on the whole below-ground food web, on plants and eventually also on the above-ground food web (Wardle et al., 2004).

Agricultural systems

The impact of N on soil biodiversity is mostly described in studies in which organic farming systems were compared with conventional, intensive farming systems, but the interpretation is partly hampered by other differences, such as avoiding using pesticides in organic farming systems (see Velthof et al., 2011). An overview of the impact of N on soil biodiversity indicators such as soil microbial biomass, activity, N mineralisation and diversity (genetic diversity, number of genotypes or species of bacteria) is given in Velthof et al. (2011) (see also Chapter 6).

Terrestrial ecosystems

In terrestrial ecosystems, including forests, N deposition increases not only N availability but also soil acidification, thereby changing soil life, including fungi and bacteria, nematodes and springtails. N specifically reduces the occurrence and activity of certain soil fungi. For example, Van Geel et al. (2020) observed a 40% potential loss of mycorrhizal fungal species richness along an N deposition gradient in western Europe in both dry and moist heathlands. Such decreases have implications for nutrient acquisition from recalcitrant organic sources and for pathogen resistance, while plants invest less in root growth and collaborations with microorganisms, as ample N is available. Soil acidification does not so much affect soil fungi but decreases bacterial community richness and diversity, as observed in long-term N and sulphur manipulation experiments (Choma et al., 2020). Overall, nitrogen input and related soil acidification negatively affects soil life.

N (and P) and plant species diversity in non-agricultural ecosystems

Both N and, to a lesser extent P, affect biodiversity, especially plant species diversity, in non-agricultural ecosystems. In this context, forest ecosystems are the largest non-agricultural form of land use. In forests, N is generally a limiting nutrient, and deposition may first increase growth and productivity through enhanced N availability, but at a later stage it may cause eutrophication and acidification, negatively affecting nutrient balances and increasing trees' 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, increased growth is not a benefit but a threat, as it causes a decrease in plant species diversity, which is also a trade-off in forest ecosystems. Atmospheric N deposition thus affects various ecosystem services through its impacts on the fertility (quality) of forest soils and thereby the forests' capacity to provide services such as wood production (provision service), C sequestration (climate regulation service), buffer capacity (water quality regulation service) and pest/disease regulation (Erisman et al., 2014; De Vries et al., 2014b). 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 affects plant species diversity. With respect to N, the concept of a critical load is crucial, namely the load that affects non-agricultural ecosystems, especially through impacts on plant species diversity (e.g. De Vries et al., 2015a; UBA, 2021).

N and P and water quality

Agricultural systems

Elevated N and P concentrations in surface waters, especially rivers, and in coastal and marine waters contribute to the phenomenon of eutrophication, which has impacts on the biocoenosis of freshwater ecosystems and coastal and marine ecosystems. The enrichment of N and P in freshwater is largely due to surface run-off from agricultural soils, while the N and P loss from terrestrial ecosystems generally dilutes the concentrations of N and P in surface waters. Specifically, in marine ecosystems, where N is considered to be the most important element in limiting phytoplankton growth, the effects can be considerable and often negative.

Terrestrial ecosystems

In terrestrial ecosystems, elevated nitrate and sulphate concentrations can affect the acidity of surface waters, especially of soft water lakes, since those systems are not limed (as is the case in agriculture). In acidic soils, N- and sulphur-induced acid deposition can cause aluminium to be released from soils in the watershed, which leaches into lakes and streams. There were well-documented, large-scale effects of acid deposition on aquatic ecosystems in the early 1970s in both Scandinavia and North America, including mortality of adult fish and reproductive failure (e.g. De Vries et al., 2015a). However, there is clear evidence of chemical recovery from surface water acidification, and also biological recovery (e.g. Hesthagen et al. 2011). This aspect overlaps with the topic of soil (and surface water) acidification (see also Chapter 4).

3.2 Indicators of nitrogen and phosphorus status of soils

Indicators for 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 into the soil fertility status and the need for N and P fertilisation to support crop growth.

- For N in agricultural soils, target levels or critical levels are not defined, but the total concentration of mineral N (N_{min}, i.e. the sum of nitrate N and ammonium N) is used for N fertiliser recommendations, as it affects the N mineralisation rate and thereby soil N availability (see Section 3.2.1). There are indicators, however, for forest soils, including the C/N ratio in the organic layer and the dissolved total inorganic N concentration (see also Section 3.2.1). Critical limits for dissolved total inorganic N concentration are used to derive critical N loads by multiplying those values with a water flux, thus deriving a critical N leaching rate, and adding values for the related N uptake, N immobilisation and denitrification (e.g. Posch et al., 2015).

- For P, target and critical levels can be defined, making the following distinction between the two: below the target level, the soil P status should be increased because P is limiting for crop growth; above the critical level**,** there is an enhanced risk of negative effects on water quality because of increased run-off of P to surface waters. Details of indicators are described below.

3.2.1 Indicators of the N status of soils

Mineral N in agricultural soils

In agricultural soils, the total concentration of mineral N (N_{min}), determined by extraction of 1M KCl, is the most relevant indicator of the N status of an agricultural soil in relation to crop yield. It is an indicator of potentially available N, because of its relationship with N mineralisation, which increases the sum of dissolved ammonium N and nitrate N in the soil solution. Only this fraction of N is directly available to plants. In several countries, the concentration of N_{min} is assessed each year in the topsoil (plough layer) by agricultural extension services, since N_{min} is highly variable, depending on soil and crop properties and climate.

Advice about N fertilisation aims to achieve 'balanced N fertilisation'. Based on a target crop yield and the N content in the harvested crop, the required N uptake in plants is derived. Then, the effective N input is calculated to achieve the required uptake, taking into account not only N fertiliser but also other N sources. This includes mineralisation of mineral soil N, as explained above, and external N inputs from manure, crop residues, N fixation and atmospheric N deposition. 'Effective' relates to the fact that the availability of N for plant uptake from other N sources is lower than from N fertilizer, since there are larger unavoidable losses of N to air and water, for example because N that is mineralised or comes in from manure and deposition becomes available partly outside the growing season. The gap between the required N uptake and the effective N input from sources other than fertiliser is then used to calculate a site-specific N fertiliser recommendation.

C/N ratio in the organic layer of non-agricultural (forest) soils

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

- impacts on soil chemical processes such as mineralisation, immobilisation, nitrification, leaching, acidification;
- plant nutrition and forest growth;
- plant species diversity.

Below a specific threshold, terrestrial ecosystems will react to additional N inputs by increasing 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 and 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 a shift towards more nitrophilic species are also observed (e.g. Bobbink and Hettelingh 2011).

One possible indicator for the impact of N eutrophication in forests is the C/N ratio for either the highly humified organic layer (H horizon, for moderate to nutrient-poor forest soils) or the top few centimetres of mineral soils (nutrient-rich forest soils without 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 us to derive a critical C/N ratio in these soils, as discussed below in Section 3.3.

Dissolved total inorganic N concentration in non-agricultural (forest) soils

Dissolved total inorganic N concentration in non-agricultural soils has been used as an indicator for various adverse impacts, including vegetation changes, nutrient imbalances, N leaching, impacts on root growth, and increased sensitivity to frost and diseases. Critical limits for those indicators are given below (Section 3.3.1) and are used to calculate critical N loads on forest ecosystems. They have thus been incorporated in the critical load mapping manual (see table V.5 on critical (acceptable) N concentrations in soil solution for calculating $CL_{nut}(N)$; CLRTAP, 2017).

Output indicator: N concentrations in air and water linked to losses from agricultural soils

At high target crop yields and/or in soils with limited possibilities for denitrification (e.g. well-drained sandy soils), current N inputs may result in an exceedance of the critical limits for NO_3 in groundwater or total N in surface water (see Section 3.3 for critical limits). Similarly, at high manure N inputs, typical of areas with intensive livestock husbandry, NH_3 emissions may be such that they exceed critical levels for NH_3 in air or critical loads of N to ecosystems. It is thus 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 those that result in concentrations of NO_3 in groundwater, total N in surface water or NH_3 in air that are equal to the critical levels of those N compounds (De Vries and Schulte-Uebbing, 2020; De Vries et al., 2021). In Europe, there are many regions where current inputs exceed those critical N inputs; this cannot be monitored by a soil N indicator but it can through air and water quality indicators. Note that there are also emissions of N_2O, but these have not been included as a criterion in the above-mentioned studies, since there is no clear limit for N_2O in the atmosphere. One could use radiative forcing as a criterion, but N_2O is only one of the greenhouse gases that affect radiative forcing (others being CO_2, methane, ozone, etc.), making the assessment of a critical load very difficult. Furthermore, NH_3 emissions from agricultural land also reduce radiative forcing by increasing forest growth and enhancing CO_2 uptake, thereby almost completely counteracting the warming effect of N_2O (De Vries et al., 2011).

3.2.2 Indicators of the P status of soils

Crop yields and available soil P contents in agricultural soils

The indicator used in agricultural soil is the available P concentration. The P concentration in the rooted topsoil is derived from soil P tests (extractants); these indicate the availability of P and are used to make P fertiliser recommendations, based on the relationship between P fertilisation and crop yield (Jordan-Meille et al., 2012). Many extractants are used to assess the available soil P level and all extract a different soil P pool. Examples of available soil P parameters 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), P-CAL (calcium- acetate-lactate extract; Schüller, 1969) 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, 14 extraction methods in Europe were evaluated by Jordan-Meille et al. (2012), and they concluded that the extraction methods tested yielded different amounts of P from different P pools. Of the above-mentioned methods, they found that, in terms of the amount of P extracted, the extraction methods could be ranked in the following order:

P-Olsen < P-ammonium lactate < P-Mehlich 3 < P-Bray II < P-oxalate < P-total

For thresholds, it would be advisable to use a harmonised extraction method, but unfortunately there is currently no agreement of such a method, since many countries have related the P concentration obtained from a given soil extraction method to crop yield and do prefer to keep that approach. It would be even better if a harmonised approach was taken to determining both the reactive (long-term availability) soil P pool (e.g. ammonium lactate) and the dissolved P concentration (such as P in water or 0.01M $CaCl_2$), as this gives information on the soil P buffering capacity, i.e. the speed with which P in solution is replenished from the available pool after P uptake or 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 (P_w). 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 P saturation index, which is defined as the ratio $P_{ox}/(Fe+Al)_{ox}$, where P_{ox} and $(Fe+Al)_{ox}$ stand for the P, aluminium and iron 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), related to specific soil functions, are difficult to define. Total N and available N do affect crop growth in unfertilised soils by affecting N mineralisation (see above), but there is no critical limit for it, because N fertilisation ensures a high N supply. Furthermore, excess N does not limit crop growth when the related soil acidification is properly counteracted by liming (see Chapter 4). A high N_{min} content may negatively affect soil biodiversity, but limit values cannot be defined, because impacts are not related to differences in soil N status but to the effects of adding N to the soil (via fertiliser), 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 groundwater level) and N emissions, rather than the actual soil N status. Minimising the N surplus 'at the farm gate', i.e. the N input from feed and fertiliser minus the output from plant and/or animal products, is one way of minimising N losses to water and air.

Critical limits for the C/N ratio in the organic layer of forest soils (H horizon) ([16])

The N retention capacity of forest soils is strongly affected by N transformation (mineralisation and immobilisation) processes in the organic layer (litter, fermented Of and humic Oh horizon) and to a lesser extent in the mineral topsoil. At high soil C/N ratios, most incoming N is retained by microbial immobilisation and limited N is available to plants. When more N is stored, the C/N ratio declines, and more N becomes available by mineralisation for plant uptake and leaching. Based on the relationship between N leaching and the C/N ratio in the organic layer of forests, C/N ratios of around 25 (between 20 and 30) are considered critical, with a very high N retention fraction and thus limited leaching risk at a C/N ratio above 30, while the N retention fraction is low and the leaching risk is high at a C/N ratio below 20; in between, there is strong variation as shown in Table 3.2.

Table 3.2 illustrates that there is very limited leaching risk at a C/N ratio of above 30, while it is high at ratios below 20 and in between there is strong variation. In more detail, De Vries et al. (2006) derived an N retention fraction based on the ammonium 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 increased leaching. For example, Gundersen et al. (1998) presented a very limited C/N range in the organic layers (30-25) to distinguish sites with high N retention and thus low leaching potential (C/N ratio >30) from those with low N retention and thus high leaching potential (C/N ratio <25). Using a data set of published N budgets and C/N ratios in the organic layer, MacDonald et al. (2002) found the strongest relationships between N output and N input when the data were divided into 'N-rich' sites (C/N ratio ≤25) and 'C-rich' sites (C/N ratio >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 ([17]) level II database, Van der Salm et al. (2007) found that N leaching was best explained if the C/N ratio was further refined based on annual average temperature and N throughfall.

([16]) H horizon: part of the forest floor, commonly understood to be dominated by humified organic matter (> 60%).
([17]) International Co-operative Programme of Assessment and Monitoring of Air Pollution Effects on Forests

Table 3.2 N retention and related N leaching risk versus C/N ratio in the organic layer of forest soils

Indication	C/N ratio 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

Critical limits for N concentrations in air and water

As stated above, N inputs in excess of N uptake, called the N surplus, cause emissions of NH_3 to air, leaching of NO_3 to groundwater, and run-off of total N to surface water, and there are critical limits for these concentrations in air and water in view of their impacts on ecosystems and health.

- NH_3 in air: 1-3mg NH_3/m^3
  Evidence shows that NH_3 in air can have significant toxic impacts on plants as a result of direct uptake through the foliage above a threshold level (for an overview of direct effects of atmospheric NH_3 on terrestrial vegetation, see Krupa (2003) and Cape et al. (2009)). The sensitivity of (plant) species to NH_3 increases from lichens to higher plants: lichens > native vegetation > forests > agricultural crops. Cape et al. (2009) reviewed methods to set a critical level for NH_3 and collated the evidence available 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 1mg NH_3/m^3 is now proposed for lichens and bryophytes and of 3mg NH_3/m^3 for higher plants, including forests.

- N in soil solution: leakage from forests: 1mg N/l
  De Vries et al. (2007) suggest an upper limit of 1mg N/l to differentiate between undisturbed and 'leaky' N-saturated forest sites, based on the findings of Gundersen et al. (2006), who gave an overview of current water quality in forests by compiling a list of studies from the 1990s on nitrate concentrations in seepage water from temperate forests, including >500 sites in Europe. From the survey data, it is difficult to conclude at exactly what level a forest ecosystem can be considered 'leaky', but the authors suggest an annual average N concentration of 1mg N/l for seepage water and 0.5mg N/l for streams and catchments. Stoddard (1994) characterised four progressive stages of N saturation based on changes in seasonality and levels of NO_3 leaching in streams, and a value of 1mg N/l coincides with the limit for the near final stage.

- N in soil solution: impacts on forests: 1-5mg N/l
  Empirical data suggest that critical dissolved N concentrations for adverse impacts on fine root biomass/ root length and increased sensitivity to frost and fungal diseases vary between 1-3mg N/l and 3-5mg N/l, respectively (De Vries et al., 2007)). The critical values for impacts on fine root biomass and root length are based on Matzner and Murach (1995), who found that the total fine root biomass of Norway spruce saplings decreased significantly when the dissolved N (NO_3+NH_4) concentration was >2mg N/l. Critical dissolved N concentrations for increased sensitivity to frost and fungal diseases have been derived from a critical N concentration in the needles of 18g/kg, above which sensitivity increases. De Vries et al. (2007) derived a relationship between foliar N content and dissolved annual average N concentration, based on the results from 120 intensive monitoring plots in Europe. Below 3mg N/l, the N content in foliage was always below 18g/kg, while above 5mg N/l, values were nearly always above this 18g/kg. In this range, adverse vegetation changes are also found (De Vries et al., 2007). In this context, the potentially leachable N can also be used by extraction with 0.1M KCl or 0.005M $CaCl_2$, which correlates well with dissolved N.

- NO_3 in groundwater: 50mg NO_3/l^-
  The critical NO_3 concentration in groundwater is generally set to the World Health Organization (WHO) drinking water limit of 50mg NO_3/l or 11.3mg NO_3-N/l. This limit is based on epidemiological evidence for methaemoglobinemia in infants (WHO, 2011).

- N in surface water: 1.0-2.5mg N/l
  Critical limits for dissolved total N in surface water, as an indicator for eutrophication of aquatic ecosystems, vary in the range 1.0-2.5mg N/l. This is based on (1) an extensive study on the ecological and toxicological effects of inorganic N pollution (Camargo and Alonso, 2006) and (2) an overview of maximum allowable N concentrations in surface waters in national surface water quality standards (Liu et al., 2011).

For comparison: critical loads refer to the critical level of deposition from the atmosphere, and can be calculated with models that make use of critical limits of N in soil solution (see De Vries et al., 2015a). In addition, critical limits for N in air, groundwater and surface water are used to assess critical N inputs in agricultural soils (e.g. De Vries and Schulte-Uebbing, 2020).

3.3.2 *Target levels and critical limits for P status indicators*

Relationships between target levels and critical levels for available P for crop yield and water quality in agricultural soils

To avoid losses in crop production and negative environmental impacts, available soil P levels should ideally stay:

- above a target level below which crop yield is limited (Mallarino and Blackmer 1992);
- below a critical level above which P leaching and run-off is significantly enhanced (e.g. Li et al., 2011).

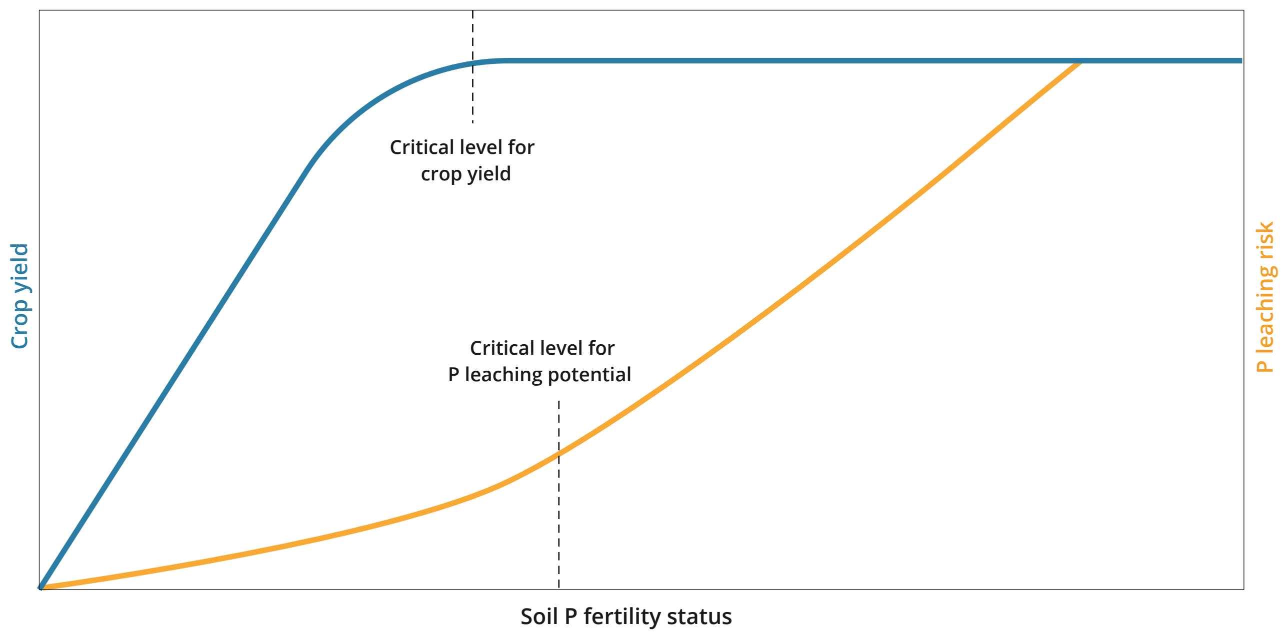
This principle is illustrated in Figure 3.2. Both crop yield, with related plant P uptake, and P leaching are affected by the soil P fertility status. The soil fertility status is approximated by an extractant that assess an available soil P level, such as P-Bray, P-Olsen, P-oxalate, P-ammonium lactate and P-Mehlich. The figure shows a critical available P level above which crop yield no longer responds (target level for crop yield in Figure 3.2), and a critical available P level above which the risk of P leaching increases (critical level for P leaching potential in Figure 3.2). The latter level is defined as 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, mostly approximated by P-CaCl2 as an indicator of risk (see Heckrath et al., 1995; Hesketh and Brookes, 2000). In the figure, the critical P level (or target level) for crop yield is lower that the critical P level for leaching, but the reverse may also be true. Critical limits for crop yields and water quality are given below based on this principle.

Target levels for available P for crop yields in agricultural soils

The concept of thresholds for P has been widely applied in fertiliser recommendations: a common practice is the 'build-up and maintenance' approach. The principle of this approach is that P application should:

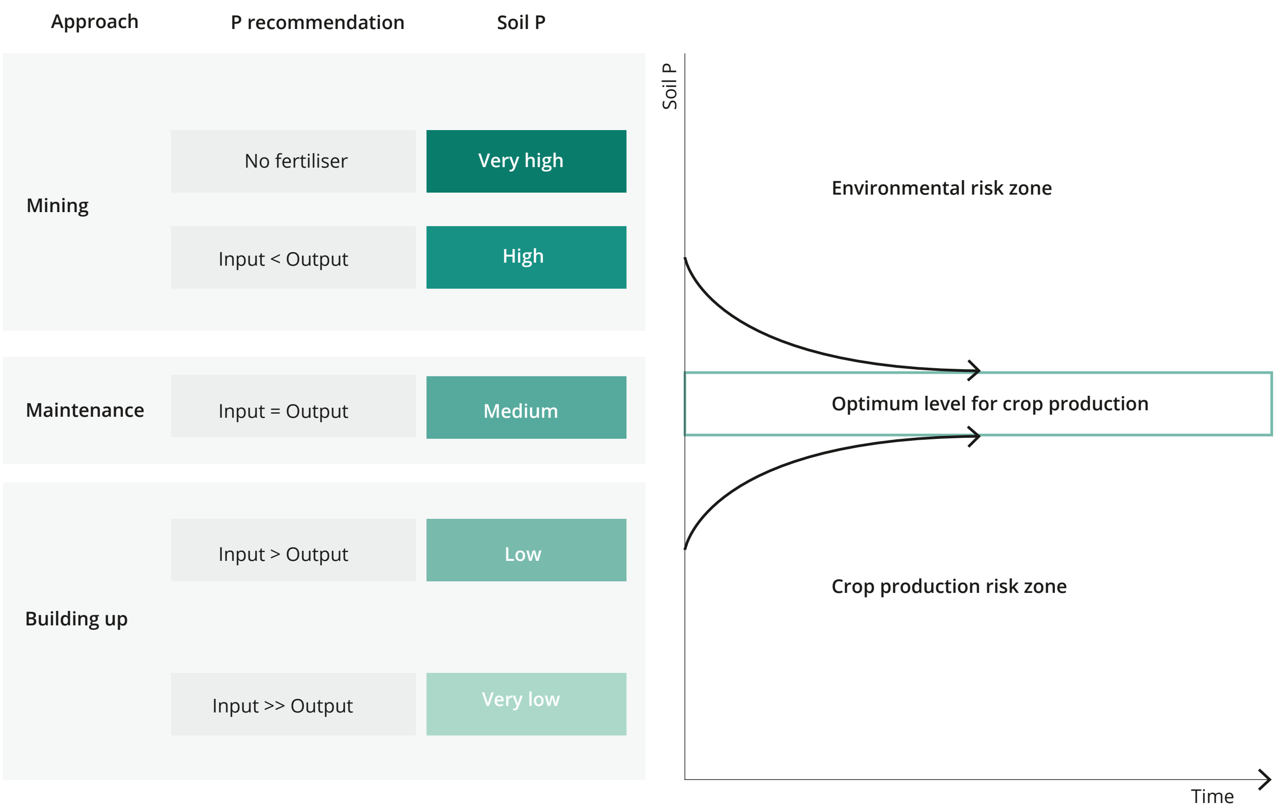
- not be made in soils with available soil P levels above the change point (threshold) for P leaching;
- equal the P withdrawal in harvested crops, if:
  - available soil P > target level for crop yield;
  - available soil P < critical level for P leaching;
- equal the P withdrawal in harvested crop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

Figure 3.2 Relationships between crop yield (left y-axis) and P leaching risk (right y-axis) and soil P fertility status



Source: Bai et al. (2013).

Figure 3.3 **Principle of soil P status and environmental risk**

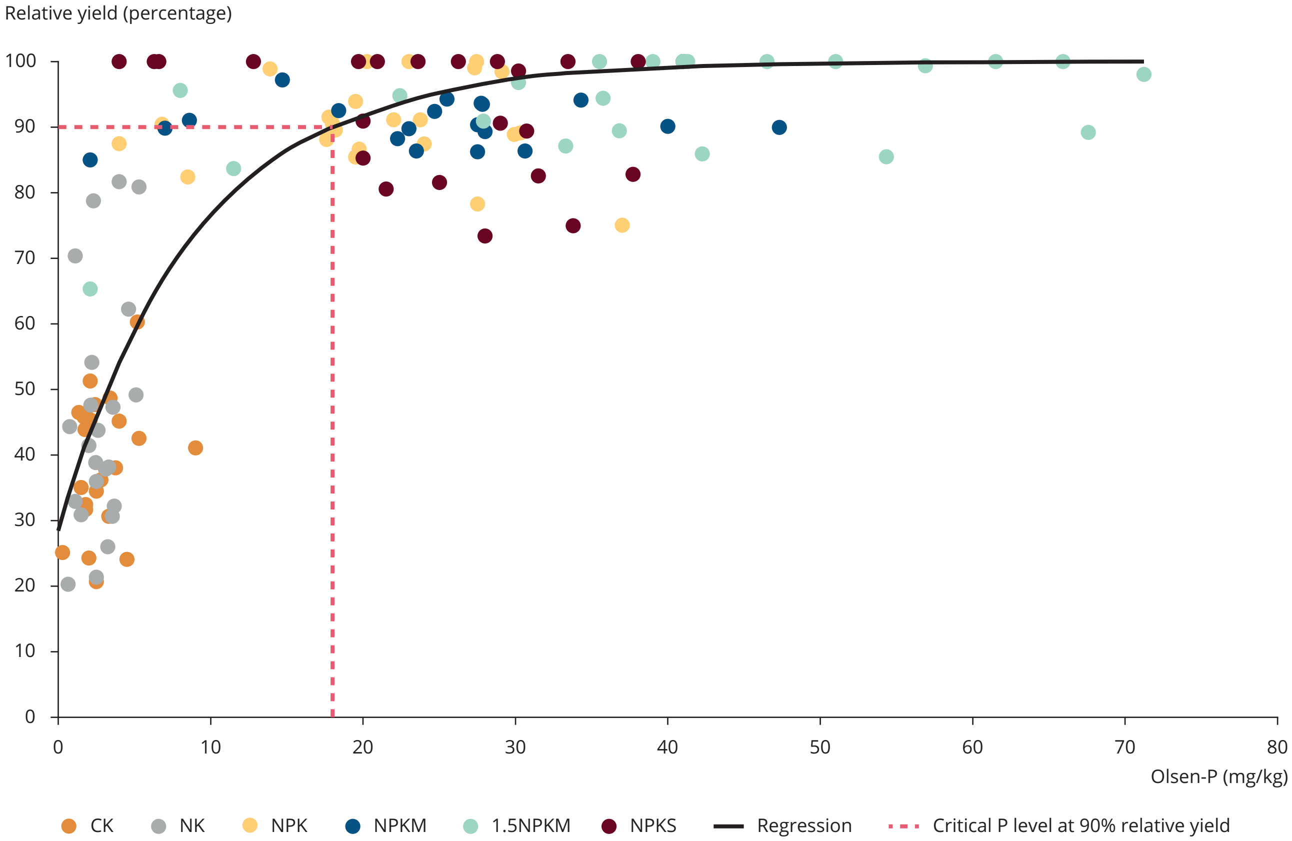


Source: Adapted from Li et al. (2011).

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).The critical (target) P level for crop yield can be derived from short-term (months) pot experiments (in a laboratory greenhouse) and long-term (years) field experiments in which P fertiliser is added to soil and crop yields are recorded while taking account of differences in soil P status. The advantage of short-term (months) pot experiments is that soil P level is the only variable element, while all other circumstances are equal; the disadvantage is the difference in environmental conditions between laboratory and field. Conversely, the advantage of long-term field experiments is that the impacts are measured under field conditions, but the disadvantage is that other factors affecting crop yield, such as climatic variables, may also change over time, which should be accounted for when deriving a critical (target) P level.

A critical level is defined as the level determined by a soil test below which a crop yield response to additional nutrients is expected and above which crop yield does not respond to nutrient application (Voss, 1998). The critical P-Olsen is generally related to an expected yield loss of 5-10%, i.e. a crop yield that equals 90-95% of the maximum yield values (see also Figure 3.4 with results by Chen, X. et al., unpublished data for wheat). For example, Bai et al. (2013) in long-term experiments found critical Olsen-P values for maize, wheat and rice, of 18mg/kg, 14mg/kg and 11mg/kg, respectively, based on relationships between crop yield and soil Olsen-P values (Figure 3.4 left). Overall, critical Olsen-P values ranged from 7mg/kg to 18mg/kg. Critical limits vary significantly, depending on the type of analysis (extraction method), crop type and soil properties, and thus must be experimentally derived.

Figure 3.4 **The relationship between soil Olsen-P contents and relative yield of wheat in long-term experiments in China**



Note: CK: control plot without nutrient additions; N additions of nitrogen, P for phosphorus, K for potassium and S for sulphur; M: addition of animal manure; 1.5 NPKM: 50% higher addition of N, P, K and manure than in the standard treatment.

Source: Chen, X. et al., unpublished data.

Critical limits for dissolved P and soil P saturation index in agricultural soils in relation to water quality

Thresholds for P in the soil are also important to protect surface waters and groundwaters from eutrophication. For that purpose, the indicator 'P saturation index' was introduced above, i.e. the ratio: $P_{ox}/(Fe+Al)_{ox}$.

The critical P saturation index (PSI) is 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 expressed as 25-35% of the P sorption capacity, which in turn is calculated as $0.5\times(Al+Fe)_{ox}$ for sandy soils and non-calcareous clay soils. The critical PSI can be related to a critical value for P in soil water (P_w) according to Chardon (1994):

$$P_w=481\times PSI^{1.433}$$

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

Using a PSI of 0.15 would lead to a critical P_w level of nearly 20mg P/l, which is slightly higher than the agronomic optimum P_w level for crop yield found by Jungk et al. (1993) (nearly 10mg P/l) and lower than the agronomic optimum P_w of nearly 35mg P/l 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 run-off that exceeds 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 thresholds for P fertilisation.

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 on the growth or nutritional quality of forests. Instead, data are given for the P concentration and N/P ratios in foliage (needles and leaves), which indicate P limitation (P concentration) or imbalanced growth (too low N/P ratios). Critical N/P ratios vary strongly between coniferous and deciduous tree species: for conifers an N/P ratio <12 indicates N limitation and a N/P ratio >18 indicates P limitation, while for deciduous trees an N/P ratio <17 indicates N limitation and a N/P ratio >25 indicates P limitation (Mellert and Gottlein, 2012). One could use these values as indictors for the N/P ratio in the organic layer, since that layer reflects the N/P ratio in foliage, so: N/P ratio in organic layer >18 (coniferous forests) and 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 from 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 with deciduous trees (De Vries and Leeters, 2001).

4
Soil acidification

Soil acidification occurs when the acid neutralising capacity of the soil is reduced, which leads to a decrease in pH. Acidification can be caused by acidic precipitation of sulphur dioxide, ammonia and nitric acid, and it has historically affected both forest and agricultural soils. Nowadays, the most significant effect is seen on unlimed agricultural land as a result of the application of ammonium-based fertilisers 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, calcium and magnesium (base cations). At very low pH (<4.5) it increases the availability of elements such as aluminium and manganese, sometimes even to toxic levels. As a consequence, crop yields decline. Soil acidification can be counteracted by liming.

Table 4.1 provides an overview of the impact of soil acidification on soil functions and services. Input of acids (sulphuric and nitric acids, amino acids and nitrate) through atmospheric deposition is closely linked to the loss of base cations (e.g. calcium, magnesium, potassium, sodium) through cation exchange processes in soils (acid neutralization capacity until the soil's buffer capacity is exhausted), and subsequent leaching of mineral nutrients. This process negatively affects water quality, biological activity, and plant growth. The impacts of soil acidification on soil carbon pools is uncertain as it can decrease forest growth and related litterfall and thereby carbon input but simultaneously, it reduces decomposer activity. In the case of acid forest soils, low pH tends to favour the accumulation of organic matter through reduced decomposition.

Table 4.1 Relationship of soil acidification to key societal needs and soil functions

Societal need	Soil service	Impact
Biomass	Wood and 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	Indifferent
	Storage of geological material	Indifferent

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

4.1.1 Impacts of soil acidification on soil fertility and crop growth in agricultural systems

Nitrogen generally has a positive effect on soil fertility and conditions for crop growth, but the overuse of nitrogen fertiliser can also lead to significant acidification of cropland, reflected by a decline in pH (Guo et al., 2010), unless soils are properly managed (e.g. limed). In non-calcareous acidic soils (pH between 4.5 and 7.0), base cation nutrients, i.e. calcium (Ca^{2+}), magnesium (Mg^{2+}) and potassium (K^{+}), adsorbed onto soil organic matter and clay, are crucial in buffering the protons produced by elevated nitrogen inputs (De Vries et al., 2015b). During acidification, these base cations are replaced by protons and subsequently leached from the rooting zone, accompanied by nitrate (De Vries et al., 1989; Lucas et al., 2011), which decreases their availability. This is an adverse effect, since this loss of base cations implies a loss of the acid neutralisation capacity, and it may affect plant growth at low base saturation levels (i.e. the ratio of adsorbed base cations on clay and organic matter to the cation exchange capacity). At very low pH levels, there is not only more limited availability of base cation nutrients, such as calcium, magnesium and potassium, but also elevated concentrations of toxic elements, such as aluminium, manganese and toxic metals, such as cadmium, which can restrict plant and soil biota growth due to nutrient deficiency and metal toxicity (Rengel, 1992; Kochian et al., 2004; Wang et al., 2007). In addition, pH can also affect the availability of zinc with available zinc increasing at low pH (pH < 5.5) and, conversely, a lower availability or even deficit at pH levels higher than pH 7. For phosphorus the impact of pH on availability is more complex and plant availability decreases both at low (pH < 5) and high (pH > 6-7) due to interaction and precipitation with iron and aluminium (at low pH) and calcium at high pH (Barrow, 2017). There are indications that crop growth can be limited in acidic soils (Lucas and Davis, 1961; Baquy et al., 2017). Aluminium 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 forming aluminium-phosphorus precipitates (Hinsinger, 2001).

4.1.2 Impacts of soil acidification on soil fertility, tree vitality and biodiversity in forest ecosystems

In forest soils, the link between nitrogen- and sulphur-induced acid deposition and changes in soil and soil solution chemistry is well documented. In calcareous soils, the input of acidifying compounds (nitrogen and sulphur) 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 (HCO_3^-) and calcium (Ca^{2+}) from calcium carbonate, with HCO_3^- and Ca^{2+} ions leaching from the system, while the pH remains the same. In non-calcareous soils, buffering is taken over by the weathering of silicate minerals and by the cation exchange processes of the soil adsorption complexes. In these soils, protons are exchanged for calcium (Ca^{2+}), magnesium (Mg^{2+}) and potassium (K^{+}), and these cations are leached from the soil together with anions (mostly nitrate or sulphate). Subsequent leaching of Ca^{2+}, Mg^{2+} and K^{+} leads to loss of the soil's base cation buffering capacity and to imbalances in the nutrients needed for plant growth. Because of the restricted capacity of this buffering system, soil pH will decrease. It has been shown that acid deposition in many forested catchments has caused prolonged export of base cations, such as Ca^{2+} and Mg^{2+}, from forest soils, resulting in base cation nutrient depletion (Watmough et al., 2005; Sverdrup et al., 2006; Akselsson et al., 2007). When the soil pH drops below 4.5, the acid input is also buffered by aluminium release, causing aluminium toxicity. Significant correlations between sulphur and nitrogen deposition and enhanced concentrations of Al^{3+} in soil solutions have been demonstrated in acidic forest soils in Europe (De Vries et al., 2003).

Soil acidification is correlated not so much with tree growth but rather with a decline in tree vitality, disturbed tree nutrition, enhanced tree mortality and reduced plant species diversity in the forest undergrowth (De Vries et al., 2014a; Schmitz et al., 2019). This has led to liming campaigns for forest soils, which have — in combination with decreasing acid deposition — significantly increased soil pH. Exceedances of sulphur critical loads have been reduced but are still observed in forests (Forsius et al., 2021).

4.2 Indicators for acidity status of soils

There are various indicators for soil acidification, including pH, base saturation, aluminium concentration and the ratio of aluminium to base cations (De Vries et al., 2015b).

Agricultural soils: pH and base saturation
In agricultural soils, pH and base saturation is the indicator that is used to assess the soil acidity status and the need for liming. Dissolved aluminium concentrations or the ratio of aluminium to base cations are never used as indicators, since aluminium release happens at pH values below 4.5 and a base saturation level of below 25% is considered (far) too low for agricultural soils, since crop yield is clearly affected below such values (see Section 4.3). Overall, pH is the key indicator used in agricultural soils, since it is best related to the availability of nutrients and crop yield.

Forest soils: aluminium concentrations and base cation to aluminium ratios
Ulrich and co-workers (e.g. Ulrich and Matzner, 1983) were among the first to postulate that increased aluminium concentrations, specifically inorganic aluminium, and elevated Al/Ca ratios in soil solution are a major cause of forest dieback, because they damage the root systems of tree species. The effects of high concentrations of aluminium on trees were tested with seedlings, grown in water cultures, in pot trials (greenhouse), mainly in the 1980s (see Rengel (1992) and

Kinraide (2003) for overviews). The hypothesised mechanisms of aluminium toxicity include hampered root growth and inhibition of the uptake of nutrients (Schulze, 1989; Sverdrup et al., 1990, 1992; Sverdrup and Warfvinge, 1993; Warfvinge et al., 1993; Matzner and Murach, 1995). Furthermore, several authors (e.g. Roelofs et al., 1985) showed that release of aluminium by soil acidification and imbalances in the ratio of ammonium to base cations, due to excessive nitrogen inputs and reduced nitrification, may cause nutrient deficiencies, which may be aggravated by a loss of mycorrhiza or plant root damage. This coincided temporally with field observations and foliage analyses in which deficiencies of magnesium and potassium caused yellowing of the needles in Norway spruce (Zöttl and Mies, 1983). In the 1980s, several authors (e.g. Ulrich and Pankrath, 1983; Hutchinson et al., 1986) considered soil acidification, especially the increase in the concentration of Al^{3+} in soil solution, responsible for forest decline, since Al^{3+} is very likely to be toxic to plant roots (Marschner, 1990; Mengel, 1991; Sverdrup and Warfvinge, 1992; Cronan and Grigal 1995). The risk from Al^{3+} to 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 aluminium concentrations and for the aluminium to base cation ratio have been derived, and an overview of these levels is given in De Vries et al (2015b). However, the standard indicator for soil acidity is the pH level, which is the indicator used in this study.

4.3 Critical limits for pH in agricultural soils

4.3.1 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 the critical level below which crop yield is limited, generally a level at which the availability of heavy metals is also limited. As with phosphorus, the critical pH level can be derived by:

- Short-term manipulation experiments in which the soil pH value is manipulated by adding H^+ or OH^- and then growing a crop in it (pot experiments in a laboratory or greenhouse). The advantage is that soil pH is the only variable, while all other factors, such as soil type, temperature, water availability, nutrient availability, are kept equal. The disadvantage is that differences in environmental conditions in the laboratory or greenhouse on one hand and those under field conditions on the other, may affect the response of the experiment. Furthermore, adjusting soil pH by adding acid or alkali can strongly affect the soil microorganism community, as well as nutrient availability and biomass accumulation.

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

An example of the 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 the current yield divided by the maximum crop yield without acidification impacts, in both short-term manipulated experiments and long-term experiments in wheat, maize and rice. The critical pH value (threshold) corresponds to the pH at an expected yield loss of 5%, i.e. a crop yield that equals 95% of the maximum yield. In short-term experiments, critical values ranged between 4.5 and 4.7 for all three cereal crops (see Table 4.3), being close to the pH value of 4.5 at which aluminium release starts to occur. In long-term observations, the critical pH values related to an expected yield loss of 5% are, however, higher (5.0-5.9).

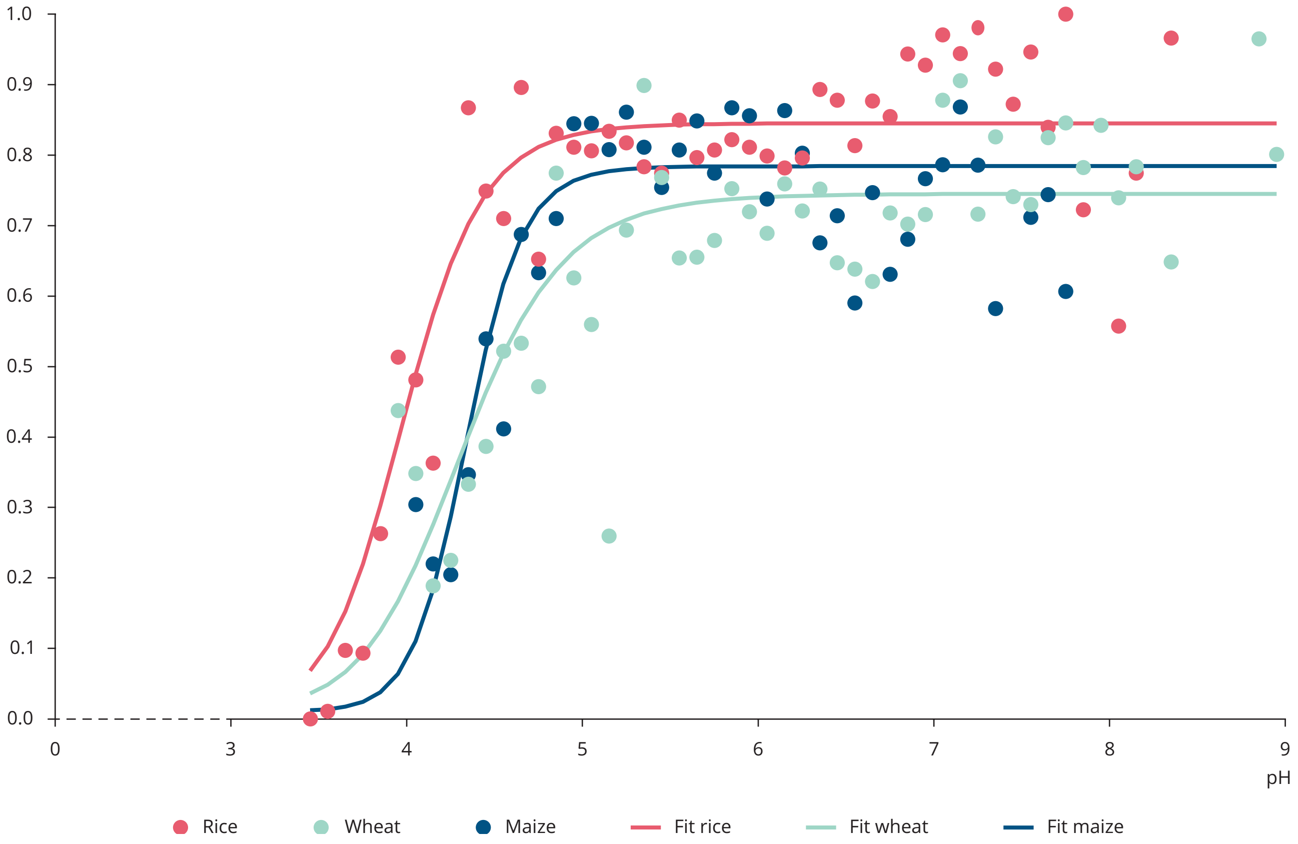
In liming recommendations, the optimal target soil pH for a range of crops is often given with the aim of maintaining soil pH close to a level at which overall nutrient availability for crop uptake is optimal. An example of such target levels is given in Table 4.2.

Table 4.2 Example of optimal pH values for different crops as given in the literature

Crop	Optimum pH
Beet, beans, peas and oilseeds	7.0
Cereals and maize	6.5
Grassland	6.3
Grassland (high molybdenum)	<6.2
Potatoes	6.0

Source: Teagasc (2022).

Figure 4.1 **Impact of soil pH on the relative yields of wheat, maize and rice using combined short-term pH manipulation experiments and long-term observations**



Source: Adapted after Zhu et al. (2020).

Table 4.3 **Summarised critical pH values for wheat, maize and rice derived from short-term manipulation experiments (STE) and long-term observations (LTE)**

Crop	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

Source: Compiled from Zhu et al. (2020).

Various studies indicate that crop production is already constrained at pH values below 5.5-6.0 due to the limited availability of calcium, magnesium, potassium and phosphorus (Walker et al., 2011; Holland et al., 2019). The results at least indicate that a pH value below 5 should certainly be avoided, while 4.5 is really critical in view of aluminium toxicity. Ideally, the pH should stay above 5.5 or even 6, as indicated in the examples in Table 4.3.

4.4 Critical limits for dissolved free aluminium and the molar base cation/aluminium ratio in forest soils

4.4.1 Free aluminium concentration of 2mg/l

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

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

The results from a variety of laboratory experiments described above showed that the Ca/Al ratio was a better indicator of root impacts than inorganic aluminium (Sverdrup et al., 1992; Sverdrup and Warfvinge 1993; Cronan and Grigal 1995). As with aluminium, a wide range of toxicity thresholds for the Al/Ca ratio has been reported. Sverdrup and Warfvinge (1993) carried out a systematic review of the impacts of aluminium on the growth of tree seedlings and plants in laboratory experiments, based on approximately 200 studies. The response in acid soils, expressed by root growth, stem growth or plant growth in experiments, has been determined for various 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 as a function of Al alone or of the Al/Ca ratio. The critical limit was most conveniently expressed as a molar Bc/Al_{tot} ratio, with Al_{tot} being the total (inorganic and organically complexed) aluminium concentration and Bc denoting the base cations Ca^{2}+, Mg^{2+} and K^{+}. Many calculations of critical loads of acid deposition on forest ecosystems use either a general limit value of 1 for the Bc/Al ratio or a tree species specific value, ranging between 0.5 and 2.0.

The relevance of laboratory experiments addressing aluminium toxicity under field conditions has been disputed (Kreutzer, 1995; Løkke et al., 1996; Binkley and Hogberg, 1997; De Wit et al., 2001). Indeed, healthy trees have been found at sites where high soil solution aluminium 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 the effects of acid deposition on forests (Abrahamsen et al., 1993; Beier et al., 1998; Kreutzer and Weiss, 1998; Huber et al., 2004), have been inconclusive with respect to the effects of aluminium toxicity on root growth and nutrient uptake. Despite this criticism, a mean the critical Bc/Altot ratio of 1 is still often used in risk assessment (De Vries et al., 2015b).

5 Soil pollution

Soil pollution significantly affects human health and/or ecosystem functioning. Essential soil functions, such as the production of sufficient and safe food and the provision of clean water and a suitable habitat for soil dwelling organisms, can be impaired depending on the degree of soil pollution. To be able to assess the impact of soil pollution on soil health, meaningful indicators for the current state of soils and the associated risks from pollutants, as well as information about inputs and outputs of substances, are required. Once critical concentrations are exceeded, relevant soil functions can be impaired. To derive meaningful thresholds for pollutants in soil, the entire pathway — from emissions of pollutants to exposure — needs to be considered. This includes inputs to and outputs from soil, as well as processes that regulate concentrations in soil and adjacent relevant environmental compartments (e.g. water). This chapter provides an overview of some of the existing approaches for setting such thresholds, currently developed largely at national level. Options for harmonising thresholds are discussed, also emphasising the need for improved harmonising of risk assessment tools as a basis for soil screening values and risk assessment procedures.

Soil pollution affects various societal needs, as illustrated in Table 5.1. Clearly the impact that pollution has on each of these societal needs varies depending on soil conditions, pressures, future management and protection objectives. Here, especially, biodiversity and the filter function (in relation to water quality) are directly affected by pollution, whereas the quality of soil as platform for infrastructure is barely affected by pollution. Services such as storage of water and carbon can also be affected, albeit at much higher concentrations of pollutants than those that affect biodiversity, for example.

Considering the interaction between soil as carrier of pollutants and the functions related to them as part of societal needs, the impact of pollution decreases in the following order:

- **high** for biodiversity and filtering of pollutants;
- **intermediate** for crop growth;
- **negligible** for infrastructure.

The complexity of pollution as a soil threat lies in the fact that there are multiple interactions between relevant soil functions and even societal needs. For example, if biomass production and biological activity in soils are reduced through pollution, then it is likely that the ecosystem service 'carbon storage' will also be affected. Unravelling all these interactions is, at present, not considered fully, and most existing frameworks for soil protection consider single relationships between pollution and a specific function or need. In this chapter we will illustrate some of the underlying principles currently used by Member States. Each of these is based on data and specific scientific approaches and assumptions about acceptable risk levels.

Table 5.1 Relationship between soil pollution and key societal needs and soil functions

Societal need	Soil service	Impact
Biomass	Wood and fibre production	-
	Growth of crops	-
Water	Filtering of contaminants: water quality	-
	Water storage	-
Climate	Carbon storage	-
Biodiversity	Habitat for plants, insects, microbes, fungi	-
Infrastructure	Platform for infrastructure	Indifferent ([a])
	Storage of relocated material or artefacts (excavated geological material, sediments, cables and pipelines, archaeological material)	Indifferent

Note: ([a]) Land prices are lower if the soil is polluted, as remediation costs will be incurred.

5.1 Rationale: terminology and context

5.1.1 Diffuse pollution versus point source pollution

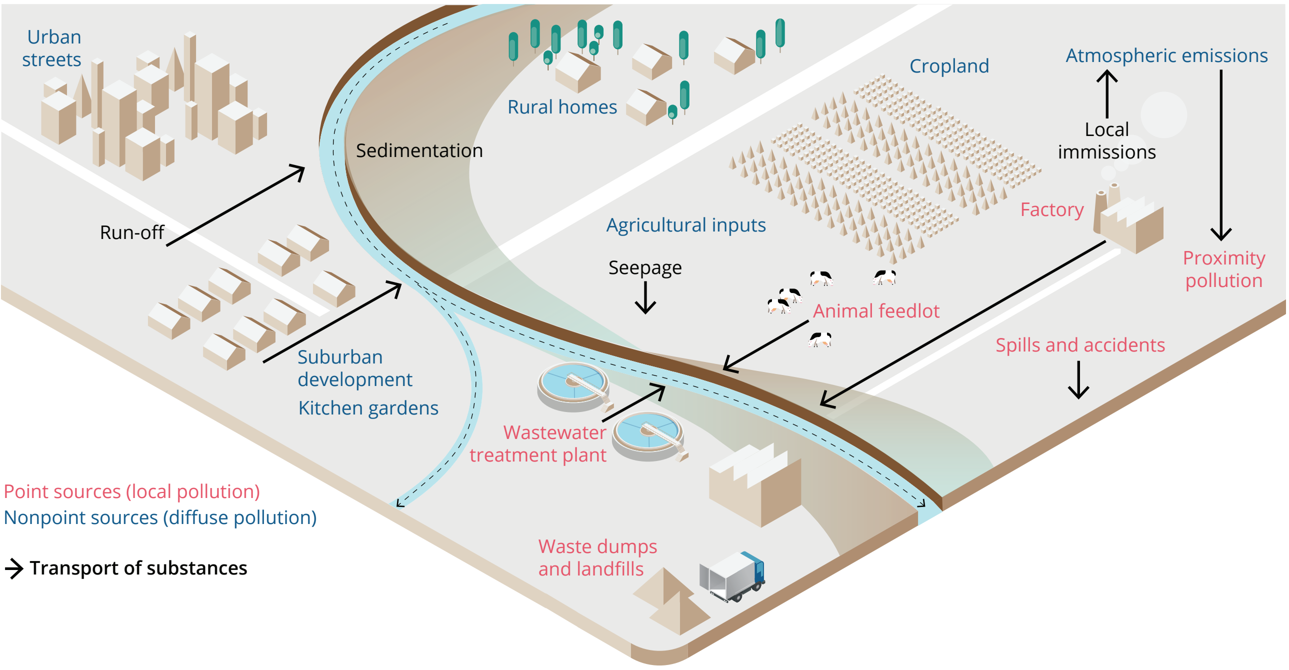
A distinction can be made between point source pollution and diffuse (or non-point) soil pollution (as illustrated in Figure 5.1).

In both cases, affected land suffers from the widespread application and distribution of pollutants (see Figure 5.1). Diffuse pollution commonly originates from a range of sources, including those related to management of agricultural land and atmospheric deposition (mostly from industry and traffic). It usually affects larger areas than those affected by point source pollution and is characterised by a relatively homogeneous pollution pattern. For atmospheric deposition, the link between the source of pollution and its destination often is not clear (with the exception of proximity pollution; see below). Former or ongoing deposition of polluted sediments in river floodplain soils is also a form of diffuse pollution, although the affected area is often confined to the area between dikes. According to the EEA, diffuse pollution is defined as 'Pollution from widespread activities with no one discrete source' ([18]).

A specific type of diffuse pollution is proximity pollution, which is a widespread form of diffuse pollution but originating from a single industrial source, outside the property boundaries of the industry (Van-Camp et al., 2004b). A typical example of proximity pollution is the regional impact of non-ferrous metal smelters in areas such as the Belgian-Dutch border zone De Kempen. There, elevated levels of cadmium and zinc are found up to 30km or 40km away from the smelter; this indicates that there is a spatial gradient in pollutant levels in soil, with high concentrations close to the source, and decreasing concentrations with increasing distance from it. Wind conditions determine the direction of this gradient: for example, in the case of De Kempen, it extends in a south-west to north-east direction because of prevailing south-west winds. The type of pollutants detected in soils are closely related to the main activity of the industry: in the case of smelters, metals such as cadmium, zinc, arsenic and lead.

In the case of arable soils, diffuse pollution is often the primary type of pollution, depending on agricultural practices and intensity of management. Aside from metals such as cadmium, copper and zinc, which are present in mineral fertilisers (cadmium), animal manure (copper, zinc), compost or sludges (mostly metals), pollutants of emerging concern are

Figure 5.1 Forms of pollution



Source: Brooks Cole Publishing 2005, modified.

([18]) https://www.eea.europa.eu/help/glossary/eea-glossary also increasingly detected in arable soils. Relevant pollutants in this case include medicinal residues (from both human and animal medicines, such as antibiotics and hormonal residues), nanoparticles or polyfluorinated compounds (PFCs, such as perfluorooctanoic acid, or PFOA) from sewage sludge, or microplastics present in some organic soil improvers. Usually, the inputs of these products are higher in arable cropping systems than in extensively managed forms of land use such as forests or pastures used for extensive grazing.

As summarised in Chapter 1, various policies address diffuse pollution. Examples of relevant EU legislation that addresses diffuse pollution with the aim of reducing inputs to soil include:

- Council Directive 86/278/EEC (Sewage Sludge Directive; EEC, 1986) regulating the quality and quantity of sewage sludge used in agriculture. In national legislation, limits are established on quality (via limits on the concentrations of pollutants in sludge) and quantity (via maximum application rates).

- Regulation (EU) 2019/1009 (EU, 2029) regulating EU fertiliser products by setting quality criteria for a range of fertilising products and components thereof. Criteria included are minimum requirements for the nutrient and organic matter contents (e.g. for organic fertilisers or organic soil improvers) and maximum limits for unwanted substances such as heavy metals and polycyclic aromatic hydrocarbons (PAHs). The level of these maximum limits and the number of chemicals regulated depend on the type of fertiliser or compound considered.

- Council Directive 91/676/EEC (Nitrates Directive; EEC, 1991) setting limits on the amount of nitrogen applied to agricultural soils via animal manure. Currently the maximum amount to be applied annually to soils via animal manure is 170kg of nitrogen. Limiting the use of manure indirectly also regulates the load of other pollutants (mainly copper and zinc, the most relevant in animal manure).

So far, pollutants of emerging concern, including perfluoroalkyl and polyfluoroalkyl substances (PFAS), nanoparticles or microplastics, have not been addressed at the same policy level. The reasons for this include the lack of data on the concentration of such chemicals in fertilisers and other products used in agriculture and the complex chemical behaviour of many of these compounds in soils. Furthermore, techniques for characterising the risk from most of these substances to relevant soil functions are still being developed; therefore, relevant risk-based thresholds are largely unavailable.

In contrast to diffuse pollution, point source pollution (also called local pollution) usually occurs at a smaller scale and often in a heterogeneous pattern, which is characteristic of the polluting activity (such as industry, waste, leakages and spills). In contrast to diffuse pollution, soils affected by point source pollution are often characterised by high concentrations of pollutants.

In the past, legislation at both national and EU levels has already resulted in a substantial decrease in emissions from such point sources. For example, countries that cover the broadest range of polluting activities in their national contaminated sites registers have approximately 10-12 national policies in place. Most policies include source-related actions (to prevent further contamination) and land management strategies. Examples of land management include traditional forms of remediation via removal of soil or *in situ* degradation (in the case of selected organic pollutants). Because of the high costs of such remediation measures, more research is directed towards improving soil conditions to reduce the toxicity of pollutants present in soil. Changing the land use to reduce human exposure and thus risk levels is a 'minimum effort' approach that will not change the potential risk the presence of the pollutant poses.

As a basic prevention measure, most EU countries have set a date after which point source emissions have to be reduced to acceptable levels to avoid a build-up of pollution in the surrounding areas in future. This is in line with the EU action plan *Towards zero pollution for air, water and soil*, which states that emissions of pollutants should be reduced to levels that no longer pose a threat to the environment or human health. Although in most countries emissions to soil have been drastically lowered, all EU countries are dealing with a substantial soil pollution heritage from the past. Table 5.2 lists the major characteristics of diffuse and point source pollution for the assessment of soil quality.

Table 5.2 Main characteristics of diffuse and point source soil pollution

Characteristic	Diffuse pollution	Point source pollution
Source	Largely ongoing in agricultural areas due to application of plant protection products and nutrients resulting in emission of metals, microplastics and/or nanoparticles. Partially historical in the case of floodplain soils or cases of proximity pollution. Immobile and persistent organic chemicals used in the past can still be found in specific forms of land use (e.g. DDT in areas with intensive fruit tree cropping).	Largely historical resulting from (historical) industrial emissions. Current emissions on land from large industrial installations (IED) are hardly reported (vs emission to air and water), but it is likely that the percentage of areas affected by ongoing point source pollution varies depending on the degree of implementation of existing legislation in individual Member States and required legal obligations to reduce emissions and/or remediate polluted sites.
Protection target	Mostly targeting arable cropping systems, including animal husbandry (grassland); more recently, adjacent terrestrial and aquatic ecosystems are also considered. Direct human exposure is considered in the case of proximity pollution.	Primarily focused on human health and on the soil ecosystem directly located at or near the polluted area. In addition, drinking water resources that are affected by leaching from soil or direct emissions into the water body are considered.
Procedural framework	Soil screening values and/or relevant risk limits in products are based on (1) at source level, i.e. quantity and quality of applied substances (sludges, fertiliser quality), (2) at effect level, i.e. risk limits for foodstuffs and drinking water quality. In many countries natural background levels are used as a first screening level even though these are not necessarily related to the risks of pollutants as such.	Tiered approaches for risk-based assessment, usually combining thresholds for soil and groundwater based on human health risks (exposure) and toxic effects on the ecosystem (ecotoxicological response of selected species). Risk assessment aims to ensure that the future intended land use remains below agreed risk levels.
Availability thresholds in receptors	Largely confined to nutrients (in surface water and groundwater), selected metals/metalloids (in plant products) and an array of plant protection chemicals used in agriculture. The number of other organic chemicals included in assessments of diffuse pollution varies strongly per country.	For most metals/metalloids, PAHs, aromatic compounds (including BTEX), volatile organic chlorinated compounds, mineral oil, asbestos and other commonly observed organic chemicals produced and emitted by industry; thresholds are available for human health based on exposure (TDI) and reference dose or soil organisms (ecological effects levels).
Policy	Largely acting via regulation of quantity and quality of inputs to soil (sources), either indirect (e.g. regulation of inputs of nitrogen and phosphorus) or direct (quality standards for pollutants in products used in agriculture). Specifically for sewage sludge current EU policy is based on preventing accumulation.	Largely based on remediation and/or soil management of affected sites to contain or reduce risk (including restriction of land use) at a local scale. Further development of emission control (zero pollution action plan)
	EU-wide limit values are in place that regulate the quality of mineral and organic fertilisers or soil improvers as well as sludges.	No common European regulation (except for WFD and IED — see Table 1.6); national remediation targets vary widely across the EU depending on the approach (and risk limit) applied.

Note: BTEX, benzene, toluene, ethylbenzene and xylene; IED, Industrial Emissions Directive; TDI, tolerable daily intake; WFD, Water Framework Directive.

During the early 1980s the discovery of multiple cases of extreme soil pollution triggered the development of soil quality guidelines or soil screening values (SSVs). Initially the aim was to protect (and remediate) soils at a negligible risk level, for example at the level of natural background levels (multifunctional approach). However, observing large areas affected by pollution and the high number of contaminated sites and the corresponding remediation costs has triggered the fitness for use concept. This implies that not all polluted sites need to be, or can be, remediated to achieve zero pollution, or zero risk levels. Instead, a predefined but variable risk level related to soil use was deemed acceptable depending on the intended land use. This approach has resulted in stringent SSVs for sensitive land uses (such as agriculture and nature or residential areas) and higher, less stringent screening values for forms of land use where exposure is less of an issue, as is the case for industrial sites. This system of what is now commonly called risk-based SSVs will be explained below.

5.1.2 Terminology

Current national soil pollution monitoring and risk assessment procedures (representativity, pathways, SSVs , planning instruments) are defined differently and are not comparable across Europe. To facilitate harmonisation, agreement on definitions of some key terms related to risk assessment and thresholds is needed. Table 5.3 lists a few relevant key words.

Table 5.3 Terminology important for soil pollution

Term	Definition
Background level	Level of pollutants in soil that can be found in the absence of 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. Background concentrations of most metals vary depending on parent material (rock type from which the soil developed). For a large number of human-made organic pollutants, background values are zero or close to zero, since they are not part of any soil forming mineral (e.g. microplastics, PFAS, most PAHs and dioxins). In some cases, background levels of selected organic compounds are not zero, e.g. in the case of some PAHs present in soils subject to forest fires, although these are exceptions.
Protection target (endpoint)	Here we refer to endpoints as the entity to be protected. At the highest level, two protection targets are distinguished, i.e. human health and ecosystem health (see Table 5.2) but specific sub-targets include water quality (for consumption or other uses, e.g. showers, swimming) and food quality (for specific products or groups of products). Common endpoints considered here include arable (food or fodder) crops, animal products, water quality, and terrestrial or aquatic ecosystems represented by a number of key species. In the case of human health, exposure models are used to convert intake from polluted soils via different pathways (food, water, air or soil) to a resulting total exposure that can be matched with a critical exposure value such as a tolerable daily intake.
Critical limit	These values specify exceedance limits for specific pollutants in endpoints (not in soil). Examples: water quality guidelines for drinking water; ecological thresholds to protect aquatic organisms in surface water bodies. Usually, such limits or thresholds are set at EU and/or national level. To convert critical limits in endpoints to corresponding screening levels in soil, transfer models are required.
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 abasis for thresholds (the latter may refer to, or be a part of, a legal framework).
Soil screening value	Soil screening values (SSVs) are the levels of pollutants **in soil** at which the corresponding concentrations in endpoints (e.g. quality standard for food or drinking water) are equal to the critical limits. SSVs therefore depend on the function considered and also depend on the soil type 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, which affects the transfer of most metals from soil to crops). Depending on the degree of protection desired, screening levels range 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). SSVs are, in contrast to risk limits, part of a legislative framework (however, there may be differences among Member States).
Transfer models	To convert critical limits in endpoints to corresponding risk limits or SSVs in soil, transfer models are needed. Examples include soil to crop models that can predict concentrations in crops based on the corresponding levels 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 pollutants and metals) influenced by specific soil properties. These models differ from **exposure** models used to calculate human exposure in that transfer models consider only specific **pathways** (e.g. the transfer of pollutants from soil to food).

5.1.3 Relevant groups of pollutants found in soil

Several soil pollutants, such as most metals, are naturally found in soils, but the levels are often increased by anthropogenic activities. Other pollutants are synthesised and brought into soils by a range of human activities. The types of pollutants found in soil, and how they affect soil health, is described below. Because of the large variety of substances applied to soil, a generic grouping is applied here that follows established principles of soil monitoring and risk assessment (FAO and UNEP, 2021). A more refined description of soil pollutants and their properties can be found in Van Gestel et al. (2022).

Metals and metalloids

Commonly regulated heavy metals include metals and metalloids such as arsenic (As), chromium (Cr), cobalt (Co), nickel (Ni), copper (Cu), zinc (Zn), cadmium (Cd), lead (Pb) and mercury (Hg). These can be toxic to either human beings or soil organisms even at low concentrations (e.g. Hg and Pb). On the other hand, elements such as Cu, Co and Zn are essential micronutrients that become toxic (in soil) only at high concentrations. A complicating factor is that metals such as Cu and Zn are essential for most arable crops, whereas they can be toxic at low concentrations for most aquatic or soil dwelling organisms.

For some metals and metalloids, notably Pb, Hg and Cd, specific policy measures are enforced, and their inputs to arable systems have decreased substantially. Pb inputs via air originating from fuel burning decreased by about 85% in the last 20 years of the last century in Europe (Lorenz et al., 2010). Inputs to agricultural soils via sludge, manure or mineral and organic fertilisers are legally regulated, either through standards that define maximum concentrations in such products (EU, 2019) or indirectly by setting standards in animal feed (in the case of Cu and Zn), which consequently affects the concentrations in manure.

Compared with most other metals, Cd is transferred from soil to plants relatively easy, and risk limits in plants (e.g. those set by the World Health Organization (WHO)) in crops such as rice, wheat or leafy vegetables can be exceeded at relatively low concentrations of Cd in soil. This can be an issue both in vegetable garden soils located in or close to urban areas and in arable soils in areas with elevated background levels or areas prone to diffuse pollution.

To avoid unacceptable accumulation in arable soils and subsequent transfer of Cd into food crops, an upper limit of 60mg Cd/kg P_2O5 in mineral phosphate fertilisers is now in place (EU, 2019) based on an EU-wide risk assessment considering the quality of wheat as the endpoint to be protected (Römkens and Smolders, 2018).

A more detailed assessment of diffuse pollution from heavy metals (and other substances mentioned below) can be found in Van Gestel et al. (2022) and in the zero pollution monitoring and outlook assessments (EEA, 2022b; JRC 2022).

Organicpollutants in soils

Plant protection products

Plant protection products (PPPs) are largely introduced to soil as part of common agricultural practices. Consequently, a series of PPPs, mainly herbicides, can be found in soil, groundwater and drinking water. Whether or not a chemical is of concern in groundwater and surface water bodies after being introduced to soil depends on its chemical stability (or persistence) and mobility, i.e. the degree to which chemicals can migrate with percolating water. Both stability and mobility are part of the registration procedure (aside from toxicity testing as such), and in most European countries the application and leaching of PPPs is regulated (type of chemical to be used as well as its application (Regulation (EC) No 1107/2009; EU, 2009). For most 'modern' PPPs, SSVs are not in place, since regulation is based on the principle that the concentration of the chemical will decrease to non-toxic levels within a prescribed time interval. Instead, limit values are developed for groundwater and surface water based on human health or toxicological criteria. However, despite the improved regulatory framework for new PPPs, concentrations can still exceed the detection limit (which often is used as threshold level) in surface water and/or groundwater. Especially in countries with shallow groundwater tables such as Belgium, Denmark, Germany and the Netherlands, many PPPs are found in most groundwater abstraction wells. Various applications also lead to a significant accumulation of metals (e.g. copper from herbicides). Soil screen values are available for PPPs used in the past that are less mobile and hence tend to accumulate in soil (e.g. for chemicals such as DDT and other persistent organic pollutants).

Other organic pollutants

Apart from PPPs, a range of other organic pollutants is found in soil and groundwater, in particular in urban areas. In agriculture, the use of sewage sludge (and to a lesser extent compost), is considered a prime source of organic compounds including PFAS, plastics and antimicrobials. Some of these organic pollutants (such as PFAS) are highly persistent in the environment and can thus bioaccumulate in the food chain. They can be naturally occurring (e.g. PAHs), or result from industrial processes (e.g. polychlorinated biphenyls, PCBs), or organochlorine pesticides such as DDT, dieldrin and hexachlorobenzene). A specific monocyclic variant of aromatic hydrocarbons is usually categorised as BTEX (benzene, toluene, ethylbenzene and xylenes), and is frequently found in urban soils and groundwater at or near large-scale cleaning facilities. Another important group of organic pollutants often detected in urban soils is that of volatile organic compounds (VOCs), including trichloroethylene, tetrachloroethylene, 1,1,1-trichloroethane and vinyl chloride. These substances enter soils through several industrial activities, including dry cleaning. Most VOCs are readily soluble in fat. VOC compounds are generally volatile and mobile.

Substances of emerging concern

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 typically have no regulatory standards. The Norman network ([19]) currently lists 860 substances in the aquatic environment (i.e. surface waters and to a lesser extent groundwater), of which some are prioritised, forming the basis for the first EU watch list of emerging pollutants, most of them organic (Commission Implementing Decision (EU) 2015/495; EU, 2015). Substances of emerging concern include PFAS, nanoparticles, antibiotics and other medicinal products such as anthelmintics ([20]). PFAS include more than 4,700 different substances (OECD, 2013), which are of very high concern because of their high environmental persistence and toxicity. Soil and groundwater pollution with PFAS has become evident in Europe (EEA, 2022a). Among other substances, 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 PFOA or perfluorooctane sulphonic acid (PFOS). A limit value of 0.1µg/l for each individual PFAS in drinking water (or a total concentration of 0.5µg/l for the entire PFAS group) has been introduced (EU, 2020). Based on preliminary human toxicological data (human tolerable daily intake, TDI), several Member States have also developed SSVs for PFAS (e.g. Netherlands and Germany). These, however, are not comparable, since the German values are based on a maximum concentration in a soil extract, whereas the Dutch values are based on measurements of the total concentration in the soil solid phase.

Research on the toxicity of many of the abovementioned compounds of emerging concern is ongoing. The derivation of screening values is challenging but urgent, because new emerging pollutants reach environmentally relevant levels (e.g. microplastics or nanoparticles). In addition, the conversion of applied products to secondary products with different properties and corresponding toxicity and their interactions with the soil matrix (mixing effects and interactions with co-pollutants) are challenging to address.

Pollutant mixtures

Soil and groundwater quality assessment is, at present, largely based on the evaluation of single pollutants. In most cases, however, different pollutants are detected at one contaminated site. As a consequence, humans and other organisms are generally exposed to more than one pollutant at the same time. For pollutants with the same toxicological endpoint (e.g. target organ) that have a common mode of action, dose addition is appropriate when assessing human health risks. If pollutants have the same endpoint, but have a different mode of action, response addition applies (Swartjes and Cornelis, 2011). The effect of combined exposure on organisms can be assessed using the multi-substance potentially affected fraction to measure toxic pressure on the ecosystem (Posthuma and Suter, 2011). Moreover, multiple pollutants may interact and alter their bioavailability, depending on soil properties and ageing (degradation products and metabolites).

5.1.4 Risk-based soil screening values

Background

The presence of pollutants in soil and their potential effects on the ecosystem and/or human health has been an increasingly important issue since the 1970s and 1980s. Before then, protection of the environment was not legally enforced in most countries. Soon thereafter it became clear that diffuse and point source emissions already had resulted in the widespread presence of pollutants. Initially many countries used rather low threshold (or screening) values. Often these were close to or equal to background values to provide the maximum degree of protection. This, however, would have had an enormous economic impact because of the high cost of remediation. This forced scientists to derive what are now called risk-based land management strategies. Rather than aiming to reduce levels of pollutants to minimum levels, remediation targets are now based on acceptable levels of risk for the intended land use.

This concept of risk-based land management has developed further since the 1980s and is applied with the aim of managing, or where needed remediating, polluted soil such that the functions or services outlined in Table 5.1 are maintained at a previously agreed level (see Box 5.1). This protection level depends on the function of the land envisaged and the risks arising from it. This resulted in the development of the risk-based SSVs. The key principle of this concept is that a critical value for a specific pollutant in soil takes account of chance and effects, in other words, the magnitude of

([19]) https://www.norman-network.net
([20]) An antiparasitic drug for livestock that kills or removes intestinal worms.

exposure or intake and acceptable exposure or acceptable concentration (with the level of acceptance involving a policy decision). For example, exposure models quantify the link between concentrations in soil and the (lifelong-averaged) exposure of human beings. In the case of ecosystem functioning, this means that the onset of adverse effects on organisms from pollutants can be related to a quantifiable concentration of that pollutant in either soil or pore water. The same is true for food production, in which the critical concentration in food or feed (e.g. as regulated by WHO or other legislation) is connected to a critical concentration in soil.

Risk-based land management can be applied for metals, metalloids and organic pollutants (e.g. PAHs, PCBs, VOCs). Risk assessment addresses all sources of pollutants (soil, air, drinking water, food and fodder) that are relevant for the exposure of the respective protection targets (endpoints).

Box 5.1 Key principles of risk-based land management and related soil screening values

To avoid excess exposure risks from pollution, risk-based land management was developed as restorative or remedial action triggered by the exposure of endpoints (i.e. consumers, the ecosystem or livestock). Several methodological steps identify how soil screening values are developed so that the correct management response can be triggered.

Step 1: Relevant critical limits for each type of 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 or can include multiple criteria if the land use includes multiple relevant endpoints. For agriculture, for example, it can include critical limits in food products, critical limits for animal health, and critical limits in nearby surface waters (e.g. for nitrogen and phosphorus). Each of the critical limits is then converted to a soil screening value, which represents the acceptable quality of soil below which its function is not affected by the level of pollutants it contains.

Step 2: This includes the actual assessment, as shown below. It involves comparing the actual quality in soil with relevant soil screening values. If the actual soil quality exceeds the relevant screening value (SV) (or minimum SV in the case of multiple protection goals), site-specific risk assessment follows (for a more comprehensive explanation, see Ehlert et al., 2013).

Step 1: Derivation of risk-based screening values (SSV) for soil based on critical limits in endpoint

Relevant *critical limit* in endpoint selected	Relevant pathway and data needed	Resulting *SSV soil*
Max allowed concentration in drinking water →	Soil — Solution sorption model Transport model Hydrological data →	Screening value drinking water
Max allowed concentration in food →	Soil — Crop transfer model Soil properties needed in model (e.g. clay, pH, org matter) →	Screening value agriculture

Step 2: Evaluation of current soil quality using risk-based SSVs

Relevant SSV	Assessment	Assessment soil quality
SSV agriculture →	Level in soil > SSV agriculture →	Not suitable, actions required
	Level in soil < SSV agriculture →	Suitable, land use poses no risk

Exposure assessment from soil pollution is complex, since it includes the fate and transport (vapour intrusion, uptake by plants), the intake rates of contact media (soil material, vegetables, inhaled air) and the metabolism in the human body (passage through the human gastrointestinal tract, absorption in the lungs). For example, in the case of exposure via food, the intake of chemicals such as Hg or As is largely determined by the quality of crops (including their processing and transport). In the case of Hg and As, the intake is largely controlled by the quality of seafood. A study on the contribution of Cd in food products grown in the EU by Rietra et al. (2017), for example, revealed that approximately 55% of the total intake of Cd via food was related to the Cd concentration in the soil. And lowering the average Cd concentration in the soil by 50% would result in an 18% reduction in the total intake of Cd. This shows that intake from food is relevant for human exposure to Cd and that this exposure can be related to soil quality. However, research has revealed that the relationships between Cd in soils and uptake by crops are complex and not always consistent for different crops and/or soil types. At present, only a few reliable models are available for use in risk assessment (e.g. those for wheat, rice and a selected number of vegetables). For most other metals and most organic pollutants, reliable soil to crop transfer models are not (yet) available.

Protection targets or endpoints

The overarching goal of soil protection regulations and procedures is to protect human health and the natural environment, i.e. terrestrial ecosystems, since the focus here is on soil health. To identify whether or not a soil represents a risk, i.e. whether specific functions or services attributed to the soil are affected by the pollutant present, it is imperative to compare the quality of the soil (in terms of pollutant concentrations) under investigation with a specific SSV, which is related to one or more specific endpoint(s) or protection target(s) (e.g. human health, the soil ecosystem, drinking water, food). Soil screening values are commonly linked to critical levels in soil via transfer models (De Vries et al., 2007; Römkens et al., 2018). Table 5.4 gives an overview of relevant endpoints and related critical limits (assessment criteria). In essence, the endpoint refers to the protection target.

Table 5.4 Overview of relevant compartments related to soil services to be protected and relevant assessment criteria

Relevant compartment	Protection target	Assessment criteria	Level of regulation
Arable cropping systems	Human health	Food quality standards in plant and animal products based on human TDIs	National, EU, FAO, WHO
	Animal health	TDI based on toxicological limit values	Recommended levels
	Ecosystem health	No effect concentrations ([a]) based on PAF ([b]) derived for soil or soil solution	EU, FAO, WHO
Urban soils	Human health and/or ecosystem health	TDI or excess lifetime cancer risk based on intake of plant products (urban agriculture) or intake of soil/dust. For ecosystem health, effect levels are applied	National, EU (EFSA)
Natural areas	Ecosystem health	No effect concentrations based on PAF in soil (terrestrial ecosystems) or water (aquatic ecosystems)	National, EU
Adjacent groundwater and surface water systems	Ecosystem health	No effect concentrations based on PAF in water	National, EU
	Human health	Drinking water and surface water standards	National, EU

Notes: ([a]) Note that the criterion used depends on the desired level of protection. For natural systems, no effect concentration (NOEC) levels are commonly based on the 5th percentile of NOEC concentrations of all organisms. For other soil functions this can be less strict, e.g. in the case of industrial land, an HC50 is applied as criterion for acceptance, equivalent to a level at which 50% of all species will be affected.
([b]) Potentially affected fraction of species and ecological processes.
EFSA, European Food Safety Authority; FAO, Food and Agriculture Organization of the United Nations.

Protection of both human health and ecosystems can be related to various soil services or functions. Here, the production service (growth and quality of crops), filter function (water quality) and habitat for soil dwelling organisms (as listed in Table 5.1) are the key services considered. These are part of four main entities that can be distinguished for soil protection (Table 5.4): arable cropping systems, urban soils, natural areas, and groundwater and surface water bodies. For each of the services considered, specific protection targets (or endpoints) can be described. The exposure of each protection target or endpoint is then related to different exposure pathways (from soil to endpoint). For example, in the case of arable cropping systems, human exposure is related to the concentration of pollutants in the harvested product and the total amount consumed. This procedure has been used to derive critical limits in food for a selected number of pollutants, for example Cd, Pb and As and selected organic compounds regulated by Commission Regulation (EC No) 1881/2006 (EU, 2006; with revisions for Cd (EU, 2021a) and Pb (EU, 2021b)). For groundwater and surface waters, exposure results from the direct consumption of polluted water as drinking water. The quantification of the role of soils in exposure assessments requires knowledge of the transport route of substances through the percolating soil solution (retention processes), from the (top)soil towards deeper soil layers or underlying water bodies. For VOCs, the vapour intrusion in buildings is relevant since this controls exposure via indoor inhalation of air.

Currently, the assessment criteria used (e.g. acceptable level in food) are set differently across Europe (e.g. by different authorities). For food quality criteria (e.g. EU, 2006, 2021a, 2021b), criteria are in place at EU level and implemented in most Member States as legally binding for arable products. The same is the case for quality of feed (EU, 2002) and surface water quality (Water Framework Directive; EU, 2000). However, criteria for ecosystem health have not been derived at EU level and it is up to Member States to decide what kind of criteria and what kind of protection levels are used to derive the final SSVs.

Risk reduction for local and diffuse pollution

When risk-based SSVs are exceeded, site-specific risk assessment is triggered to determine the need for and priority of risk management actions — ultimately remediation. Drastic and hence costly measures such as remediation are most appropriate for heavily contaminated sites, including brownfields, playgrounds, sites with VOCs present (which combine mobility with high toxicity) and allotments, where people have a lot of contact with soil 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, how pollution sources are managed becomes relevant. For diffuse pollution, often a source is still active, as is the case for pesticide applications. But for the risk assessment itself (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 source pollution.

Clearly, there is a distinction between soils affected by point source pollution and those affected by diffuse pollution, which explains why, until now, most soil remediation actions have been confined to point source pollution:

- Pollution levels observed in sites affected by point source pollution are often such that action is needed imminently. Often, the effects are obvious, such as degraded soil surfaces, visual impact on vegetation (or a lack of it), as is the case in many former mining areas, for example.

- Soil pollution levels from point sources often pose a direct threat to human health resulting from soil ingestion by children, pollution of drinking water, heavily polluted dust particles blowing into nearby housing areas or transfer into the food chain if the soil is used for local crop production, as is the case in or near city areas.

- Diffuse pollution, on the other hand, rarely results in pollutant levels at which the effects are immediately obvious (with the exception of urban allotments and proximity pollution; see below). The rate of accumulation is — in most cases — far less than that from point source pollution, and there are few examples of areas where diffuse pollution has required action (as it arises from continuous low dosage applications rather than spills or leakages).

- By nature, diffuse pollution affects large areas, which implies that possible measures (remediation, monitoring) affect large areas and, by definition, will be costly. Examples from areas affected by proximity pollution, such as the Belgian-Dutch border area of De Kempen, show that developing a regional approach can take decades and requires, in that specific case, cross-national harmonisation of risk assessment procedures.

- So far, diffuse pollution has not created urgent or visible issues with, for example, food safety or animal health beyond local sites (in the case of allotments) or selected areas affected by proximity pollution. In some of these areas (e.g. the Belgian-Dutch border area of De Kempen), regional SSVs are developed to protect the local population from excess exposure to cadmium and lead from home-grown food. In addition, recommendations for farmers have been developed to minimise the risk that pollutant levels in crops exceed national food quality standards. This can be achieved via a combination of soil management (pH control) and selection of crops. The observation that current pollutant levels are largely below SSVs targeting food safety can be misleading, because the slow but continuous build-up of pollutants in soil, as for cadmium or lead, can result in a slow but steady increase in exposure to such pollutants. This, however, is often difficult to quantify, since in most industrialised countries in Europe, food usually comes from a vast array of sources and few people depend on food grown in one place (with the exception of gardeners who have allotments and use them as their main source of vegetables and fruit). Nevertheless, Rietra et al. (2017) documented a relationship between the average cadmium concentration in soils in the EU and the exposure to cadmium via food. However, the impact of using PPPs and animal manure or mineral fertilisers on nitrate levels in groundwater and drinking water wells can be substantial. For the vast majority of pollutants of emerging concern, the knowledge base for deriving meaningful SSVs for food quality is still too small. Targeted monitoring of selected pollutants in soils and crops in areas of concern is the first step to evaluate whether or not such pollutants are actually transferred into food and fodder crops and can enter the human and animal food chain.

- Monitoring the impact of diffuse pollution on soil quality is difficult and typically would require long monitoring intervals (up to decades) to detect changes in contaminant concentrations in soil. This is mostly because of the low accumulation rate of metals in soil (see, for example, Römkens et al. (2018) for cadmium at the 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 trends for most pollutants is largely model-based.

- For many of the recently introduced pollutants of concern (e.g. medicines, PFAS, microplastics), diffuse pollution is a relevant source and can cover large areas. At present, however, regional data and risk-based limits in soils are largely absent or are in need of validation. In addition, robust analytical techniques to measure actual levels in soil are still being developed for some pollutants of concern, such as nanoparticles and microplastics. Nevertheless, there is growing concern that, if diffuse pollution continues, issues with emerging pollutants may become critical within decades, as is the case for microplastics (EU, 2018b) or PFAS (EC, 2020h).

The role of diffuse pollution has now been recognised as a potential problem at EU level and has already resulted in targeted policies. For example, the new Fertilising Products Regulation (EU, 2019) and the quality standards therein are at least partially based on a risk-based approach with the aim of minimising the long-term deterioration of soil health. There are also proposals for end-of-waste criteria for recycled materials such as compost and digestate (Saveyn et al., 2014) and, more recently, for materials such as biochar struvite and ash (Huygens et al., 2019). These proposals, however, largely target the quality of inputs to soil rather than assessing soil health with SSVs. Nevertheless, a large part of the current maximum limits included in the Fertilising Products Regulation originate from or are close to those from the German Bundes-Bodenschutz- und Altlastenverordnung (BBodSchV) and are at least partially risk-based.

Apart from the focus on the current condition of soils (e.g. agricultural soils), negative trends and expected future effects of soil pollution may also trigger (preventative) action, with the objective of avoiding exceeding SSVs under continuous pressure. Basically, two policy-driven approaches can be distinguished:

1. Future concentrations in soil should not exceed the defined SSV at any given point in time (or a predefined timeframe such as 2050 to establish a toxic free environment in the EU). Usually, risk-based limits are used to derive meaningful acceptable inputs to soil (e.g. in the case of Directive 86/278/EEC (EEC, 1986) on the use of sludge in agriculture). In some cases, background concentrations can also be used (except in the case of lithogenic anomalies), although this would inevitably lead to very strict (low) acceptable loads to soil.

2. Avoiding any accumulation of pollutants in soil is an alternative approach currently under discussion (also referred to as the 'stand-still' scenario) and introduced as part of the zero-pollution action plan. In the case of a stand-still approach, inputs to soil are not to exceed outputs from soil in order to maintain the current concentration of pollutants (or nutrients such as phosphorus). This approach is not risk based in that the current level is considered the relevant criterion rather than the level at which effects become unacceptable. For most relevant pollutants, however, current concentrations are largely below risk levels related to food quality or ecosystem health and a stand-still approach can therefore be considered sufficiently protective.

5.2 Indicators for soil pollution

5.2.1 Indicator definitions

The objective of an indicator for soil pollution is, in essence, to quantify the actual degree of pollution of soil or groundwater bodies by comparing the actual state (as defined by the measured concentration in soil or water) with the critical level indicated by the SSV or threshold in groundwater or surface water. The actual or intended use of the land or water body to be evaluated determines what risk level is appropriate and hence the magnitude of the respective SSV or water quality threshold. This means that, depending on the land use (as well as soil type or water quality), a site can be classified as 'at risk', whereas a similar area and similar pollutant concentration but a different land use could be classified as not at risk. It is therefore the case that SSVs for specific services such as 'nature' or food production are far lower (stricter) than those for industry.

The classification of the type of indicators for both diffuse and point source pollution and various types of chemicals (metals, nitrogen, phosphorous, organic pollutants) is based on the same principle, namely the risk-based approach using SSVs. Nevertheless, different types of indicators can be distinguished, depending on the nature of the pollution (point source or diffuse). A few examples are given to illustrate this below.

Selected indicators for point source pollution

With regard to the detection, management and remediation of contaminated sites, Freudenschuss et al. (2001) distinguished several sub-indicators (in this case statistical parameters), including soil polluting activity, number of contaminated sites, progress in the management of contaminated sites, expenditures on remediation and groundwater incidents. Since then, the work has been taken further by the European Environment Information and Observation Network (Eionet) Thematic Group Soil, based on questionnaires (EEA, 2022c).

The current indicator for contaminated land considers statistics on six site statuses representing some of the sub-indicators mentioned above (for details, see Payá Pérez and Rodríguez Eugenio, 2018). The current version of this indicator is based on the last Eionet questionnaire in 2016. Future updates may include polluting activities, dominant pollutants and spatial reference to regional administrative borders (number of sites per polluting activity and site status per NUTS 3 region ([21]); the proposal is currently under discussion and will address issues raised by Van-Camp et al. (2004b). The authors suggest the establishment of a **European Point Source Assessment System (EPSAS).** The development of such a register must be closely aligned with existing data collections on current industrial installations, reported to the EEA's European Pollutant Release and Transfer Register (E-PRTR), and data collections under the Mercury Regulation ((EU) 2017/852).

Indicators for soil pollution from diffuse sources

Currently, a proposal and agreement for an indicator on diffuse pollution is lacking, while placeholders for such an indicator are included in the indicator sets of the zero pollution action plan and the chemicals strategy for sustainability. Several national monitoring systems investigate the trend in metal concentrations and stocks in agricultural and forest soils; to a lesser degree, this includes organic pollutants.

Two kinds of indicators were suggested during several Eionet workshops (Freudenschuss et al., 2001):

- direct indicators related to a specific load of contaminants, such as the average pesticide consumption per unit area of agricultural land or the amount of sewage sludge applied per unit area of agricultural land;
- indicators based on a mass balance approach, such as the input-output assessment of heavy metals in arable cropping systems, and, based on this, the critical load of heavy metal content in soils related to different land uses.

An extended rationale for these parameters and indicators related to diffuse pollution can be found in Van-Camp et al. (2004b). They suggest that the following metals and nutrients could be realistically monitored, recommending 5- to 10-year sampling intervals:

- heavy metals (cadmium, copper, lead, zinc, mercury, arsenic, nickel and chromium);
- nutrients (nitrogen and phosphates).

These recommendations were evaluated and synthesised by Huber et al. (2008), as a suggestion for a European soil monitoring system. Because of the lack of soil data, the definition of an indicator on diffuse soil pollution has never been realised until now. The lack of representative soil data on actual pollutant concentrations (e.g. for heavy metals) has also prevented such assessments (Bünemann et al., 2018). Various recent European monitoring activities, in particular the LUCAS Soil survey (Toth et al., 2013; Reimann et al., 2014; Ballabio et al., 2018) are collecting harmonised data about basic soil properties (e.g. pH, clay content and organic matter) as well as heavy metals.

Table 5.5 provides an overview of the current set of indicators on diffuse and point source pollution.

All listed state and impact indicators require agreement about common criteria for the definition of thresholds (here SSVs). It is essential that any such agreement is based on a common terminology and definitions.

([21]) For more info on NUTS regions: https://ec.europa.eu/eurostat/web/nuts/background.

Table 5.5 Current set of indicators on diffuse and point source pollution

DPSIR grouping	Indicators (and sub-indicators)	Covered in this report (chapter) or elsewhere
State indicators		
Inorganic pollutants ([a])	Heavy metal contents in excess of thresholds	Chapter 5 (here)
	Critical load exceedance by heavy metals	Not covered here
Nutrients	Gross nutrient balance	Critical N and P limits (Chapter 3)
Persistent organic pollutants	Concentration of persistent organic pollutants	Principles of Chapter 5 apply
	Concentration of selected pesticides	Chapter 5 and ongoing research
Soil acidifying substances	Topsoil pH	Chapter 4
	Critical load exceedances by sulphur and nitrogen	Critical N limits (Chapter 3)
Emerging substances	Presence/amount of antimicrobials and plastics in the topsoil	Research ongoing
Pressure indicators		
Non-soil indicators related to land use intensity and pollutant inputs	New settlement area (urban fabric) established on previously developed land	Corine Land Cover
	Recycled land area	Copernicus Land Monitoring Service
	Area under organic farming	AEI: organic farming statistics
	Amount of mineral fertilisers (sub-indicators distinguish different qualities/product type)	AEI: mineral fertiliser consumption
	Amount of organic fertilisers (sub-indicators: sludges, compost, digestates, manure; if available, distinguish different qualities by chemical composition)	De Vries et al. (2022) CAPRI database
	Pesticide applications (sub-indicators specify different groups/substances/compounds)	AEI: consumption of pesticide; trends in use and risk of pesticide
	To be defined: inputs of plastics and antimicrobials	
	Location of installations (industrial facilities, mining, landfills)	Corine Land Cover
Pollution by point sources	Progress in management of contaminated sites (sub-indicators specify six site statuses) ([b])	Eionet Land and Soil indicator set ([c])
Impact indicators		
Ecosystems	Impact of soil pollution on ecosystems (above- and below-ground biodiversity and ecological processes) and on surface water, wildlife and crops)	Ongoing Eionet projects (ETC/HE)
Human Health	Impact of soil pollution on human health	

Notes: ([a]) Indicator fully discussed in this chapter.

([b]) Future refinements of this indicator may include other sub-indicators: polluting activity, (group of) polluting substance, spatial reference (NUTS 3 and/or functional urban area) of the Copernicus Urban Atlas).

([c]) See also Payá Pérez and Rodríguez Eugenio (2018).

AEI, agri-environmental indicator; CAPRI, Common Agricultural Policy Regional Impact model; DSPIR, drivers, pressures, state, impact and response model; ETC/HE, European Topic Centre on Human Health and the Environment.

5.2.2 Methodical references relevant for soil monitoring

Apart from the conceptual issues of what type of indicator should be used, methodological issues can often present an equally large obstacle to comparing or evaluating data from different sources. This relates to the entire chain from selection of sites to be monitored (either soil or land use monitoring) through procedural issues related to monitoring itself (e.g. sampling protocol) to analytical aspects and interpretation of data.

The following section considers some key methodical issues when collecting data on soil pollution:

- analyses of pollution sources and patterns (statistics, maps);
- sampling procedures (e.g. depth, sampling amount, field moisture and *in situ* determination of dynamic properties such as pH or EH (redox potential) levels at the time of sampling);
- sample transport and conservation;
- sample preparation and homogenisation;
- laboratory analysis and interlaboratory quality control;
- data interpretation.

The analyses of pollution patterns and the development of a sampling protocol are different for diffuse and point source polluted sites. Since diffuse soil pollution is characterised by a homogeneous pollution pattern, a limited number of samples and analyses of composite samples are appropriate. For point source polluted sites, several options are available for sampling, depending on the pollutant pattern. In the Netherlands, a preliminary, an exploratory and a 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 pollution is expected and for sites that are probably uncontaminated. The main objective of the exploratory investigation is to prove 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 pollution cost-efficiently. The main investigation is an iterative process: after each step, it has to be decided if the available information is 'fit for purpose'.

Sampling of soil and groundwater has also been standardised, i.e. for the sampling of soil (ISO 18400-104:2018 Soil quality — Sampling; ISO, 2018a) and groundwater (ISO 5667-11:2009 Water quality — Sampling; ISO, 2009). Depending on the country, additional national and/or regional protocols may also have been developed.

To achieve comparability of data from different sources (countries, laboratories) across the EU and Europe, an international protocol with common sampling and analytical procedures is needed. Error detection and quality control procedures also demand studies on spatial variability and method comparability (including ring analysis). Since data interpretation involves the matching of actual indicators with thresholds, the national risk assessment procedures underlying such thresholds need to be described and made available across Member States .

5.2.3 Dynamic indicators to detect future risks of diffuse pollution: critical load concept

In addition to the indicators used to characterise the current status of soils in terms of pollution, dynamic indicators are increasingly being developed to assess future levels and risks of pollutants in soil. This is particularly relevant for diffuse pollution, considering the differences observed in the health of soils affected by point source pollution versus those affected by diffuse pollution from an active source (such as the application of pesticides, manure or sewage sludge). In the case of point source pollution, the chemical quality of soils can be such that immediate action is required or, perhaps better, assessment tools should be able to detect where action is immediately required in order to reduce the risks to human and/or ecosystem health. For soils affected by diffuse pollution, however, the current health of most soils is not yet affected to such an extent that it poses an immediate threat. However, continuing the current inputs from diffuse sources (either the atmosphere or direct inputs from land use) can lead to critical pollutant levels being exceeded. This requires a different approach from that applied for point source pollution.

As stated above, diffuse pollution is often ongoing but, at present, has not yet caused serious issues with soil health (apart from selected cases of proximity pollution). For example, the current soil concentrations of copper and zinc in the Netherlands in terms of ecological risks are not of concern in the majority of arable soils (De Vries et al., 2004; Groenenberg et al., 2006). However, due to the rather high concentrations of both elements in manure (Römkens and Rietra, 2008; Deltares, 2018) and high application rates of manure in the Netherlands, both copper and zinc levels in soil are expected to increase because of the positive balance in most arable cropping systems. This in turn will lead to an exceedance of the critical threshold levels (SSVs) for both metals over time in terms of ecosystem functioning (De Vries et al., 2004 Groeneberg et al., 2006). The speed at which accumulation (or depletion) occurs depends on the combination of the load to soil and removal rate from the soil, which in turn depends on soil properties such as pH and organic matter. This means that accumulation is faster in high pH soils or soils rich in organic matter or clay. On the other hand, the risk of an increase in concentrations in groundwater is higher in low pH soils. The continued use of manure in combination with inorganic fertilisers, as well as the use of PPPs, has resulted in nitrate and pesticides leaching into groundwater bodies in several EU countries, thus affecting the quality of drinking water.

Since changes in the concentrations of pollutants such as metals in soils and groundwater bodies are typically small within a timeframe of a few years, the long-term impact can only be quantified using a dynamic model that includes all inputs to and outputs from soil. This approach can be used directly to assess where and when soils will be at risk, and also to calculate the maximum acceptable load to soils in order to avoid exceeding the threshold (in this case related to biodiversity). This approach is called the critical load approach and was developed for an array of pollutants, as well as nutrients and soil acidity, several decades ago by De Vries and Bakker (1996). At present, however, the resulting critical loads have not been implemented in national legislation in EU Member States.

This is partly due to the inherent high demand for data and process knowledge to enable such models to operate. Key processes such as retention in soil (controlled by sorption and degradation processes), leaching (which depends on the water fluxes through the soil in combination with retention) or changes in soil properties (e.g. pH or organic matter content) are complex to model at a regional or national scale. At present, the quality of integrated models to predict such changes over decades is still limited, and they are applied mostly for a certain metal including copper, zinc and cadmium. But even for cadmium, the uncertainties in the models used to calculate leaching, the dominant output from the soil, are large and the choice of model will affect the outcome.

This is even more relevant for most organic pollutants and an array of pollutants of emerging concern (including PFCs and microplastics) for which process-based models to predict crop uptake, leaching or degradation in soil are scarce, especially at national or even EU level.

5.3 Thresholds: soil screening values for soil pollution

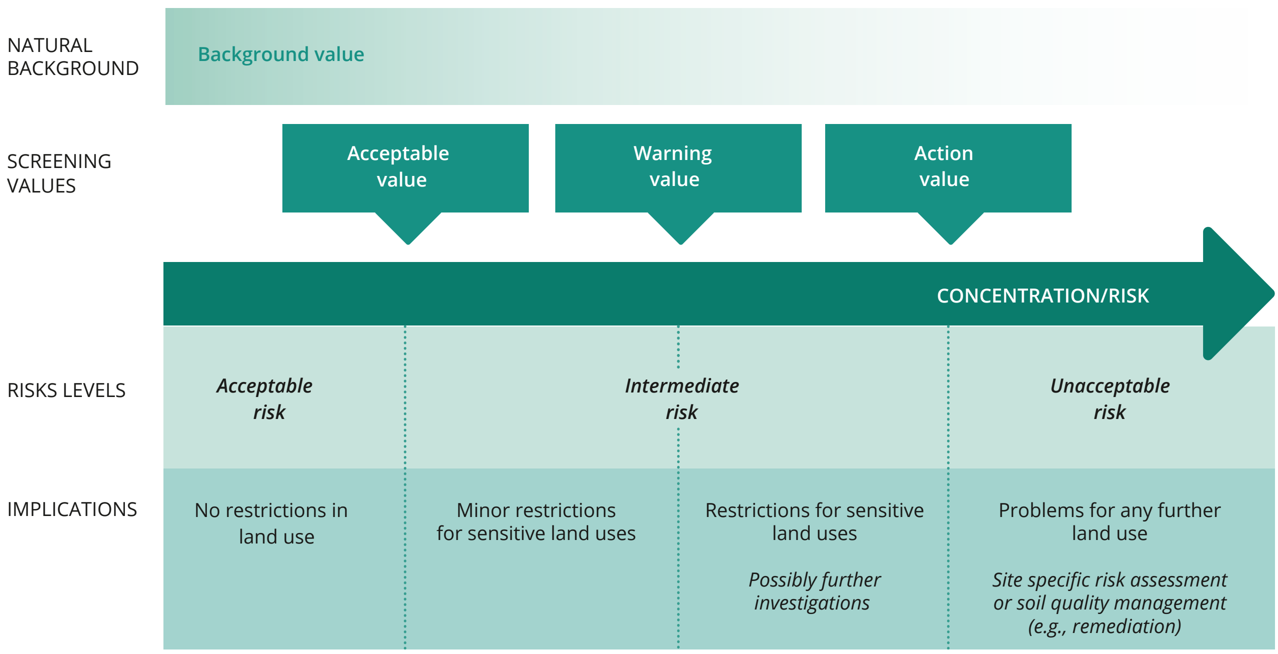
5.3.1 Knowledge base on thresholds for soil pollution

A wide range of thresholds for heavy metals and organic pollutants have been developed in many countries. As shown below, SSVs are a specific kind of threshold, generally derived from risk assessment methods (Swartjes et al., 2009). At present, most thresholds consider the critical endpoint to be protected, usually human health and/or the (soil) ecosystem. Other endpoints often used are groundwater, drinking water and surface water (Carlon and Swartjes, 2007). In some countries, wildlife, animal products or crops are considered endpoints. Current UK (soil guideline values), German (BMU,

2020) or Dutch SSVs, for example, are based on effects on humans and the ecosystem. Despite the differences in terminology used to address a risk limit or screening level in soil, the SSVs used in these countries do take a common approach in that risk assessment is at the core of the system and SSVs almost always depend on the actual land use. By relating exposure to acceptable exposure (e.g. TDI or excess lifetime cancer risk) of humans ([22]), a human health-based threshold can be derived. in the case of ecosystem protection, the potentially affected fraction of organisms as a risk-based threshold can be derived from species sensitivity distributions (Posthuma and Suter (2011) in the Netherlands; Martin et al. (2022) in the United Kingdom). In case of the Dutch approach, the minimum of the human health-based or ecology-based value serves as the final threshold for soil. Similar approaches have been adopted by other countries, but the underlying assumption in the models, and the variability in soil, climate and land use across Europe, have resulted in a wide range of SSVs (Swartjes et al., 2007).

'Threshold' as a general term in assessing the risks from soil pollution can be divided into background and SSVs as schematised in Figure 5.2. Not all kinds of national thresholds considered below are actually defined or applied in every country; rather, the schema in Figure 5.2 presents a generic nomenclature and guideline to which national thresholds can be referred.

Figure 5.2 Schematic representation of thresholds (adopted from the Heracles network)



Source: EEA.

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

The specifications of the thresholds presented in Figure 5.2 are as follows.

Background values

According to Reimann et al. (2005), the 'background value' is often used as a base value to evaluate whether a specific soil has been under the influence of anthropogenic pollution. The initial definition of background values refers only to concentrations defined by parent materials (Hawkes and Webb, 1962), which means it applies only to metals, not organic pollutants. ISO 19258 (ISO, 2018b), however, defines background concentration as 'concentration of an element or a substance characteristic of a soil type in an area or region arising from both natural sources and anthropogenic diffuse sources such as atmospheric deposition'. It thus also includes contributions from agriculture and even inputs to soils in urban areas and also relates to organic pollutants. Clearly, this distinction between a definition based on pristine, pre-industrial levels versus anthropogenically affected soils (agriculture, urban) can lead to large variation in what is considered background.

In some countries background values are assumed to pose no risk or negligible risk and are considered suitable for any type of land use. However, in the case of the ISO 19258, risk remains even at 'background' levels. Clear examples of this are elevated levels of lead in urban soils or levels of cadmium and zinc near former smelters. In the Netherlands such soils are explicitly excluded from the sampling to derive the background values.

Regardless of whether areas with diffuse inputs are included or not in the derivation of background concentrations, there is no relation between the background level itself and any specific risk. The background value merely reflects that the concentration in the soil is controlled by the parent material (and/or some additional increment due to diffuse pollution according to ISO 19258) from which the soil is developed. In the Dutch system, for example, background values are determined as the 95th percentile of values taken from 100 sites considered to be barely influenced by anthropogenic activities. The resulting background value, corrected for soil properties (clay and organic matter) is, for practical reasons, considered an acceptable upper level, despite not having been tested for effects on soil organisms, for example. There is much variability among countries, however, in the procedure to derive background values. This includes for example the cut-off percentile used to derive the background value, the population of measuring points and the level of stratification. Because of the variability of most pollutants in the parent material from which soils are derived, differences in background values related to soil type or geographical distribution can be quite large.

Acceptable value

The acceptable value generally reflects a negligible risk level as the protection level. The basic idea of an acceptable value is that there are no restrictions on land use, as long as the acceptable value is not exceeded. Acceptable values are sometimes also used as generic remediation targets.

Warning value

Warning values mark the lower end of the concentration range of what is considered 'intermediate risk'. Concentrations falling between the acceptable value and the warning value mark the range when (minor) restrictions on sensitive land use may be appropriate (e.g. growing 'sensitive crops' is not recommended). The value is also used as a trigger that initiates further soil or groundwater sampling to increase the reliability of the judgement on whether or not the action value is exceeded.

Action value

Action values mark the lower limit of concentrations above which unacceptable risk **can** occur. Exceedance of the first generation of action values (mostly derived in the early 1980s) often meant 'polluted soil' and required some kind of intervention (such as remediation). Currently, most countries have more advanced procedures based on **risk assessment frameworks**, in which the action value acts as a trigger for further, more detailed site-specific risk assessments in one or two additional assessment steps (Swartjes, 2019). Depending on the urgency of the risk as defined by modelling or experimental testing, action is required or not. The urgency of the risk is also related to the intended land use.

Terminology

The EU Member States use different terminology for all four thresholds, which complicates comparability**.** It must also be noted that not all countries have established risk-based SSVs, as shown in Figure 5.2; however, all countries include parts of the schema, so that nationally defined SSVs do have a place in a generic risk assessment nomenclature.

5.3.2 *Currently known soil screening values*

As discussed above, the data used and the concepts applied differ widely between Member States. It is therefore not surprising to find that SSVs for both metals (see below) and organic pollutants (see below) alike differ considerably. Below we summarise some of the most commonly regulated metals and organic pollutants.

Heavy metals

Data sources

When assessing the risk from metals in soil, the concept of deriving background, warning and action values has been applied in many EU Member States. Carlon et al. (2007) provide an overview of intermediate and critical risk levels for an array of metals in the EU. Here we present an update of these values for the most common metals considered in risk assessment (Table 5.6; the source of the data is given in Table 5.7). We selected cadmium, copper, lead and zinc and found more than 444 individual screening values, roughly 50-60 per metal and risk level. The values were initially retrieved from technical reports or policy and prescription documents at European, national or regional level and updated for Bulgaria, Czechia, Denmark and Germany. For some countries, the values might be outdated or not the most accurate available.

The full work by Carlon et al. (2007) has been summarised and included in a database by the European Topic Centre on Urban, Land and Soil Systems (ETC/ULS). The database is currently being expanded to include arsenic, mercury, nickel and chromium before it is handed back to the Eionet Working Group on Soil Contamination for review and updating. In parallel, supplementary information is being collected to understand the differences in how the SSVs are defined and derived.

Value ranges

The screening values presented in Table 5.6 reveal large ranges, which represent different stratifications:

- protection targets considered (human health (exposure), ecosystem health, arable crop quality);
- underlying risk limits in endpoints;
- methodologies to convert risk limits to screening values in soil;
- approaches to correct for land use or soil type; in some cases, e.g. for Wallonia, Belgium, only critical risk levels are available (called action values in Belgium).

Comparability between SSVs is also limited because of analytical differences between countries. In most Member States, intermediate or critical risk levels for metals are determined after extraction using strong acids (most commonly based on or equivalent to aqua regia, a mixture of concentrated nitric acid and hydrochloric acid). For these countries, the values can be quite comparable provided that the criteria listed above (risk limits used, protection target, etc.) are the same. In Germany, for example, SSVs for soils used for arable crops (called *Prüfungswerte* und *Maßnahmenwerte*, respectively; BBodSchV) are based on extraction using concentrated ammonium nitrate (DIN 19730: 06.97) which is believed to extract the amounts of metal (for arsenic, cadmium, lead, mercury and titanium only) that are available to plants. For grassland on the other hand, SSVs are based on extraction with aqua regia. This means that SSVs derived to protect arable crop quality are not comparable due to the method of extraction alone.

Levels of SSVs between countries also differ for different soil conditions, indicated by soil organic carbon class, texture, parent material group, land use and acidity (pH). However not all countries use a similar classification system, if at all, which again hampers the direct comparability of SSVs.

Interpretation of SSVs in Table 5.6

Interpreting Table 5.6 is best illustrated with an example. For cadmium, SSVs have been retrieved for 14 European countries and three regions. In accordance with the underlying principle of risk assessment, values are specific for a certain land use and specific texture classes or parent material. Other soil properties are included, such as saturated hydraulic conductivity and soil depth, as in the case of Poland. This creates not only differences in the level of SSVs between countries but also differences in limits within countries. Across all countries, the critical risk level for cadmium ranges between 1mg/kg (protected areas, Poland) and 30mg/kg (industrial land use, Brussels and Flanders region, Belgium; Slovakia); for intermediate risk, values vary by up to a factor of 100, between 0.4mg/kg (Slovakia) and 40mg/kg (Austria).

For copper, the range of variation is similar for the critical risk thresholds (30-1,500mg/kg) but much more extreme for intermediate values (factor of 1,500). A similar degree of variation is reported for thresholds for both lead and zinc (the critical values vary by a factor of 50), while the variation in intermediate values is the highest for lead (factor of 4,000).

To correctly interpret such differences, it is therefore imperative to always consider the goal of a specific SSV and the conceptual and methodological approach used at national level. Additional aspects such as stratification (land use) or correction using specific soil properties, such as pH or organic matter, also need to be considered if such SSVs are to be used beyond the national level (see also the explanation above of value range). In the 2000s, the Heracles network was founded with the purpose of stimulating the convergence of risk assessment procedures among EU countries (see Section 5.4.3 for details).

Table 5.6 Current screening values (SSVs) for cadmium and copper in soil for intermediate and critical risk levels (mg/kg)

	CADMIUM				COPPER			
Country/region	**Warning value**		**Action value**		**Warning value**		**Action value**	
	Stratum ([a])	SSV	Stratum	SSV	Stratum	SSV	Stratum	SSV
Austria	LU	1-40	-	10	LU	100-1,500	-	600
Belgium/Brussels	-	1	LU	2-30	-	40	LU	145-800
Belgium/Flanders	-	-	LU	2-30	-	-	LU	200-800
Belgium/Wallonia	-	-	LU	1.8-20	-	-	LU	53-600
Bulgaria	LU, pH	1.5-3.5	-	12	LU, pH	80-300	-	500
Czechia	LU, pH, text.	1.5-20	-	-	pH	150-300	-	-
Denmark	LU	5	-	-	LU	1,000	-	-
Finland	-	1	LU	10-20	-	100	LU	150-200
Germany	LU	2-20	LU	0.1-20	LU	1([b])	LU	1,300
Hungary	-	1	-	10	-	75	-	1,000
Italy	-	-	LU	1.5-15	-	-	LU	100-600
Lithuania	-	-	-	0.75-3	-	-	-	35-200
Netherlands	-	-	SOM, clay	13	-	-	SOM, clay	190
Poland	-	-	LU, SHC, z	1-20	-	-	LU, SHC, z	30-1,000
Slovakia	LU, text.	0.4-10	LU	20-30	LU, text.	30-500	LU	600-1,500
Slovenia	-	2	-	12	-	100	-	300
Sweden	LU	0.4-12	-	4	LU	100-300	-	1,000
	LEAD				**ZINC**			
Country/region	**Warning value**		**Action value**		**Warning value**		**Action value**	
	Stratum ([a])	SSV	Stratum	SSV	Stratum	SSV	Stratum	SSV
Austria	LU	100-300		500	-	300	-	-
Belgium/Brussels	-	120	LU	200-2,500	-	120	LU	300-3,000
Belgium/Flanders	-	-	LU	200-2,500	-	-	LU	600-3,000
Belgium/Wallonia	-	-	LU	120-1,840	-	-	LU	196-3,000
Bulgaria	LU, pH	60-150	-	500	LU, pH	200-450	-	900
Czechia	LU	300-400	-		-	400	-	-
Denmark	-	40	-	400	-	500	-	1,000
Finland	-	60	LU	200-750	-	200	LU	250-400
Germany	-	([b])	-		-	([b])	-	-
Hungary	-	100	-	750	-	200	-	2,500
Italy	-	-	LU	100-1,000	-	-	LU	150-1,500
Lithuania	-	-	LU	50-500	-	-	LU	75-1,200
Netherlands	-	15-590	SOM, clay	530	-	150-720	SOM, Clay	720
Poland	-	-	LU, SHC, z	50-1,000	-	-	LU, SHC, z	100-3,000
Slovakia	-	150	-	600	LU	2-500	-	3,000
Slovenia	-	100	-	530	-	300	-	720
Sweden	LU	80-300	-	800	LU	350-1,050	-	3,500

Notes: ([a]) Stratified according to: LU, land use, text., texture; SOM, soil organic matter; SHC, saturated hydraulic conductivity; z, soil depth.

([b]) Analysis based on concentrated ammonium nitrate (commonly, extraction with aqua regia is used).

The references for this table are available on request from the EEA; they are contained in a database of European SSVs (EEA and Eionet Thematic Group Soil, Working Group on Soil Contamination).

5.3.2.1 Organic pollutants

Organic pollutants are, more so than metals, characterised by a wide range of chemical properties and associated risks for humans and other organisms. In contrast to metals, most organic pollutants are absent from the soil parent material. Defining background levels as the levels present in raw earth materials, as for metals, would imply that the background levels of most organic pollutants are zero. For some pollutants, including certain PAHs, natural phenomena such as wildfires can result in regional increases in the concentrations of such compounds. Background levels vary depending on the definition of 'background', i.e. whether or not it includes the contribution of diffuse pollution.

Unlike for metals, the values initially compiled by Carlon et al. (2007) have not been updated. However, it can be seen in Tables 5.7 and 5.8, that SSVs differ depending on future, targeted land use.

Table 5.7 Screening values for potentially unacceptable risk (industrial soil use) for organic contaminants (mg/kg)

	Country/region						
Pollutant	**Belgium (F)**	**Belgium (B)**	**Belgium (W)**	**Finland**	**Italy**	**Poland**	**Spain**
Benzene	1	1	0.6	1	2	76.5	10
Ethylbenzene	70	70	76	50	50	130	100
Toluene	200	200	85	25	50	117.5	100
Xylene	190	190	20	50	50	77.5	100
Naphthalene	160	160	-	15	50	25	10
Anthracene	4,690	4,690	-	15	50	25	100
Benzo(a)anthracene	30	30	10	15	10	25	20
Benzo(g,h,i)perylene	4,690	4,690	100	-	10	52.5	-
PAHs (total)	-	-	-	30	100	110	-
Dichloromethane	3.5	3.5	-	5	5	-	60
Trichloroethylene	10	10	-	5	10	-	70
Tetrachloromethane	1	1	-	-	5	-	1
Hexachlorobenzene	55	55	-	2	5	-	1
Phenol	-	-	-	-	60	51.5	100
Cresols (sum)	-	-	-	-	25	51.5	100
Atrazine (p)	-	-	-	2	1	3	-
PCB	-	10.4	-	5	5	2.75	0.8
Methyl t-butyl ether	140	140	-	50	250	-	-
1,1,1-Trichloroethane	300	300	-	-	50	-	-
Benzo(a)pyrene	3	3	8.8	15	10	22.5	2

Note: B, Brussels; F, Flanders, W, Wallonia. For Finland, the upper guideline values are presented.
Not presented are countries where the intended land use is not clearly stated.
Until 2007, 13 countries had not reported any national SSVs for organic pollutants.
The list of substances in several national guidelines is larger than represented here.

Source: Compiled from Carlon et al. (2007).

Table 5.8 **Screening values for potentially unacceptable risk (residential soil use) for organic contaminants (mg/kg)**

	Country/region												
Compound	**AT**	**BE (F)**	**BE (B)**	**BE (W)**	**CZ**	**FI**	**IT**	**LT**	**NL**	**PL**	**ES**	**UK**	**DK**
Benzene	-	0.5	0.5	0.4	0.8	0.2	0.1	0.5	1	12.6	1	-	-
Ethylbenzene	-	5	5	28	50	10	0.5	5	50	38	20	41	-
Toluene	-	15	15	33	100	5	0.5	0.1	130	38	30	-	-
Xylene	-	15	15	10	30	10	0.5	0.1	25	18	100	-	-
Naphthalene	-	5	5		60	5	5	5	-	12.5	8	-	-
Anthracene	-	70	70		60	5	5	5	-	12.5	100	-	-
Benzo(a)anthracene	-	10.5	10.5	5	5	5	0.5	-	-	12.5	2	-	-
Benzo(g,h,i)perylene	-	3,920	3,920	15	30	-	0.1	-	-	10	-	-	-
PAHs (total)	50	-	-	-	-	30	10	5	40	30	-	-	40
Dichloromethane	-	0.35	0.35	-	-	1	0.1	2	10	-	6	-	-
Trichloroethylene	-	1.4	1.4	-	-	1	1	2	60	-	7	-	-
Tetrachloromethane	-	0.02	0.02	-	-	-	0.1	1	1	-	0.5	-	-
Hexachlorobenzene	-	0.1	0.1	-	-	0.05	0.05	0.5	-	-	0.1	-	-
Phenol	-	-	-	-	-	-	1	10	40	10.25	70	280	-
Cresols (sum)	-	-	-	-	-	-	0.1	-	5	10.25	40	-	-
Atrazine (p)	-	-	-	-	-	1	0.01	-	6	3	-	-	-
DDT (sum of DDT, DDE, DDD)	-	-	-	-	-	1	-	-	4	2.01	-	-	-
PCB	1	-	0.9	-	5	0.5	0	0.1	1	0.55	0.08	-	-
Methyl t-butyl ether	-	9	9	-	-	5	10	-	100	-	-	-	-
1,1,1-Trichloroethane	-	13	13	-	-		0.5	-	15	-	-	-	-
Chlorobenzenes (sum)	-		-	-	-		-	-	30	1.05	-	-	-
Benzo(a)pyrene	5	1.5	1.5	4.4	2	2	0.1	0.1	-	7.5	0.2	-	-
PCDD/PCDF (ng ITEQ TeCdd/g)	100	-	-	-	0.5	1×10^{-4}	1×10^{-5}	-	1×10^{-3}	-	-	-	-

Note: B, Brussels, F, Flanders; W, Wallonia; PCDD, polychlorinated dibenzodioxins; PCDF, polychlorinated dibenzofurans. For countries not represented, see also note for Figure 5-7.

Source: Compiled from Carlon et al. (2007).

Similarly to SSVs for metals, differences among Member States are considerable. This relates not only to the number of chemicals regulated but also to the absolute value for a given chemical. Differences in thresholds are particularly large for PCDD/PCDF with an apparent range from the strictest value of 1×10^{-5} (in Italy) to 100 (in Austria). This again illustrates how differences in assumptions, approaches and acceptable risk levels can result in extreme differences in the resulting thresholds for soil. To some extent there is more consensus on the type of model to be used in risk assessments for heavy metals than for organic pollutants. This also refers to the validation of such models, as the underlying database for metals is much larger than that for most organic pollutants.

5.4 Challenges and solutions to improve consistency of soil screening values across Europe

5.4.1 Factors affecting the variability of SSVs

Within the EU, there is a wide diversity of risk assessment tools (Heracles network ([23]); Carlon et al., 2007).

Table 5.9 provides examples of the factors that contribute to differences in risk assessment tools and approaches and, hence, to differences in SSVs among EU Member States.

Based on the conceptual differences listed in Table 5.9, the Heracles network developed an overview of key issues and differences in SSVs and underlying principles. Here, we list a selection of relevant observations:

- *Risk assessment models*: 11 countries developed national risk assessment models; five adopted existing models.
- *Background information on SSVs:* 50% of countries lack published and/or accessible technical information on national SSVs (see Goidts et al. (2018) as a recent example).
- *Receptors (endpoints)*: all participating EU countries primarily address the protection of human health, 11 out of 14 also address ecological receptors (endpoints), seven address groundwater and 3 address surface waters.
- *Pathways considered*: in total, 20 pathways of pollutants of endpoints are identified; of these only one is considered by all countries, and 11 pathways are hardly covered (by fewer than four countries).
- *Screening values in soil versus those in water:* current screening values of most countries refer to total content of pollutants in soil; in a few cases, concentrations in soil solution for protection of groundwater or transfer into plants pathways are considered.
- *Land use*: 12 countries adopted land use-specific SSVs (however, individual land use scenarios differ); only four countries consider soil properties (notably soil organic carbon, pH, clay content).

Additional examples of differences in approaches to deriving SSVs relate to the definition of a 'standard soil'. In order not to let soil variability limit the application of SSVs, the Dutch system, for example, has developed a generic SSV using a fixed set of soil properties. Such SSVs for what is called a 'standard soil' (a hypothetical soil with 25% clay and 10% organic matter) are to be used as a generic, first-tier national standard. A set of correction formulas then are to be used to convert the generic SSV to local or regional conditions (Wezenbeek et al., 2008). In other countries such as the United Kingdom, a similar approach has been established, albeit with different values to represent the standard soil. In the United Kingdom, for example, an average soil organic matter content of 3.4% is used as well as different ranges in soil parameters such as pH. In some countries, SSVs obtained for ecology are further corrected for the geochemical background (Martin et al., 2022), which results in a marked variability in the final SSV even within the country. In Wallonia, eight different soil standards are used because of the large variety of soil types encountered, including anthroposols (i.e. backfilled soils) (Goidts et al., 2018).

Despite these and other observed differences in approaches and assumptions, a significant achievement of the Heracles network is the agreement on the regulatory significance of SSVs, which enforce remediation action (further investigations, remediation).

([23]) Heracles: Network on Human Health and Ecological Risk Assessment for Contaminated Land in EU Member States (2005-2011).

Table 5.9 Factors contributing to differences in risk assessment tools and approaches, and to differences in SSVs among EU Member states

Conditioning factor	Role in risk assessment
Geographical factors	
Soil properties	(Bio)availability affecting ecology as well as filtering function
Depth of groundwater table	Vapour intrusion and, hence, human exposure through inhalation. It also affects the potential impact of soil quality on water quality, since contact time between soil and surface water is shorter in areas with high water tables
Climatic factors	
Time spent outdoors	Potential contact with soil and, hence, exposure through soil ingestion
Net annual water surplus	Leaching/accumulation is controlled largely by water flux (in combination with soil properties)
Temperature/rainfall	Climate controls crop types used by farmers and hence potential uptake of pollutants from soil
Cultural factors	
Self-supply gardening, subsistence farming Urban recreation	Exposure through the consumption of vegetables grown in kitchen gardens
Drinking water from private wells	Exposure through the consumption of groundwater as drinking water
Regulatory and political factors	
Prioritisation of economic and environmental values Complementarities with other existing laws Governance	Role and use of background values Definition of attributes in risk assessment Definition of acceptable risk (e.g. level of excess lifetime cancer risk) Authorities involved and responsible Land reclamation policies, spatial planning, drinking water protection
Scientific factors	
Scientific and analytical experience and cooperation Funding regime	Selection of protection targets and exposure pathways Method and use of background concentrations Toxicological data sources Acceptable exposure, exposure time, aggregate exposure Mixtures and non-soil sources

5.4.2 Generic critical levels in soil through back-calculation (transfer) of critical limits in policies

Existing critical limits in endpoints can serve as starting points for developing EU-wide applicable SSVs. The following are examples:

- critical limits in water based on ecological risk, as included in the Water Framework Directive (2000/60/EC; EU, 2000);
- limits in drinking water based on the protection of human health (Drinking Water Directive 98/83/EC; EU, 1998);
- maximum levels for certain pollutants (cadmium, lead and arsenic) in foodstuffs (Food Regulation (EC) No 1881/2006; EU, 2006);
- maximum levels in products used for animal feed (Directive 2002/32/EC; EU, 2002).

Such critical levels, in the case of human health as a protection target, are related to the broadly accessible information on TDIs. This knowledge is continuously updated, for example recently for cadmium whereby critical limits in food were reduced based on new experimental data on the long-term chronic effects of cadmium on human health. Such reductions in the TDIs are also being discussed for arsenic and lead.

The clear advantage of harmonised critical levels of pollutants in soils is that soil quality can be compared across borders. Note that the term 'critical level' is chosen here as a threshold back-calculated from critical limits in the policies listed above, and that SSVs remain specific to national risk assessments and policies (Römkens et al., 2022, in preparation).

The overview summarised in Table 5.11 shows that, at present, the approach to developing critical levels in soils would be applicable for a selected number of heavy metals (cadmium, copper, lead, zinc) and plant protection chemicals. For metals this relates to both human and ecological risks, as controlled by quality of food, fodder and water, whereas for plant protection chemicals the risks are largely ecological in both soils and (surface) waters.

5.4.3 Towards a stronger convergence of risk-based land management procedures

Risk assessment tools form the basis of risk-based land management. A risk assessment tool has been defined as 'a technical (scientific) instrument such as a model, equation, database, graph, manual or protocol that contributes to risk-based soil quality assessment' (Swartjes et al., 2009). Risk assessment tools form the basis, for example, for risk-based screening values and site-specific procedures for assessing priorities for remediation. It is widely acknowledged that many different risk assessment tools exist in the EU Member States for the same purpose. Consequently, risk assessments performed in different Member States result in widely differing risk estimates at comparable contaminated sites. The international research framework and network Heracles was active in the 2000s, in anticipation of the proposed soil framework directive (which was later called off). The purpose of Heracles was 'the improvement of the consistency of risk assessment tools for human health and ecologica risk-based soil quality assessment in the EU Member States'. Improving the consistency of risk assessment tools means that there must be neither a unique procedure for dealing with contaminated sites all over Europe nor a universal list of screening values among EU Member States. It does mean, however, that the technical part of risk-based soil health assessment should be based on a similar approach, when practically feasible.

Within Heracles, a joint approach for further developing risk assessment tools was proposed that would lead to a universal toolbox in Europe, which would allow a more consistent risk assessment approach, summarised as follows:

- Identify and harmonise risk assessment tools that must be similar throughout Europe (independent of geographical, cultural, climatic or policy factors): standardised risk assessment tools
- Identify risk assessment tools that do include geographical, cultural, climatic or policy factors: flexible risk assessment tools. Develop protocols or guidelines on how to use these risk assessments in the different Member States.
- Policy positions are determined at the national level. Examples of policy decisions include whether or not to include the soil ecosystem as a protection target (or, in more general terms, the selection of protection targets considered) and the acceptable excess cancer risk after lifelong exposure to genotoxic carcinogenic compounds.

It was further suggested that a repository of risk assessment tools in Europe be set up, which could lead to a toolbox with standardised and flexible tools (including guidelines for use) for EU-wide use in the future. See Swartjes et al. (2009) for details, in which 20 continuous, short term and long-term actions to create a harmonised toolbox for risk-based land management are set out. The design of such a toolbox is elaborated in Table 5.10.

Table 5.10 Design of a European toolbox for stimulating convergence in risk-based land management in the EU Member States

Risk assessment tool	Component to be internationally standardised/agreed
	1. Standardised tools
Human health risk assessment	Daily inhalation rates
	Tolerable exposure (reference dose or tolerable daily intake, which can be similar for a defined sensitive group of people); risk per unit of exposure
	Examples: average amount of soil ingested daily by children (Bierkens et al., 2011); dermal uptake from soil material models
Ecological risk assessment	Species sensitivity distribution ([a]) to determine ecologically-based soil health standards
	Determination of compound-specific bioavailable fractions (e.g. correction factors for the difference between intake and uptake of metals in the human body)
	Database with pollutant/compound characteristics (e.g. water solubility, vapour pressure, water-octanol partition coefficients)
Endpoint-specific risk assessment tools	Relevant EU-wide critical limits used in specific endpoints considered (e.g. critical limits in food, surface water or soil pore water concentrations)
	Description of relevant pathway and models needed to describe these to predict flows of pollutants to endpoints considered
	2. Flexible tools
Consideration of country- and site-specific conditions	Geographical conditions, e.g. for vapour intrusion models, which are dependent on soil type and groundwater depth
	Local management, e.g. home-grown vegetable consumption, food consumption data
	History of land management; national legal conditions

Note: ([a]) The species sensitivity distribution (SSD) is the empirical relation between soil concentration and fraction of species affected. The SSD gives the corresponding risk limit that can be used as an ecologically based soil health standard (Swartjes et al., 2009).

An additional aspect of harmonisation is the availability of representative **data at EU level**. Because of differences in monitoring approaches (e.g. analytical methods, sampling depth and frequency), results from national monitoring data may not be easily comparable. Nevertheless, for a risk-based approach across Europe, a large degree of harmonisation can be achieved if the standardised tools (Table 5.10) are agreed and made available. Regarding the comparability of SSVs, a Europe-wide assessment could be based on back-calculated generic risk limits (e.g. drinking water quality standards (see also Box 5.1), with acceptable levels in the soil solution, while national assessments could use countries' own SSVs. There will be considerable overlap, however, and national approaches will be more accurate.

Table 5.11 summarises some of the key elements relevant for risk assessment for several main groups of pollutants.

Table 5.11 Status of knowledge about various groups of pollutants in soil

Substance group	Representative 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 numer 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 model 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)	No, 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 ecosytem Food quality (?)	No	No experimental case studies only	No soil criteria

Note: EFSA, European Food Safety Authority; PNEC, predicted no-effect concentration; WFD, Water Framework Directive.

6
Soil biodiversity loss

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 the energy transformations and nutrient turnover of ecosystems. The aim of this chapter is to describe current approaches for defining loss of soil biodiversity at the European level. Species diversity may seem, by definition, a robust indicator of the health of soil communities. However, there is a general lack of knowledge about the status of soil biodiversity and its baselines. While an enormous number of soil-dwelling species are still undescribed, experimental evidence is also lacking for the critical role of functionally relevant (flagship) species and the effects of their loss. Rather than net species numbers, research is currently focusing on the interactions between functional groups of organisms and their abiotic environment. Although it is currently impossible to measure soil biodiversity quantitatively and accurately as a whole, or to assess its health or level of degradation, it can be approximated using combinations of sub-indicators.

An increase in soil biodiversity has positive impacts on almost all soil-related societal needs and soil functions (Table 6.1).

Table 6.: Relationship of soil biodiversity to key societal needs and soil functions

Societal need	Soil service	Impact
Biomass	Wood and 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	Indifferent
	Storage of geological material	Indifferent

6.1 Rationale for the indicator 'loss of soil biodiversity'

- 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. (2006b), the main functional groups of the soil food web are as follows:

- Earthworms consume plant residues and soil including (micro)organisms. Often, they form the major part of the soil fauna biomass with up to 1,000 individuals per m^2, 3,000kg fresh biomass per hectare or a few hundred kilograms of carbon (C) per hectare.

- Enchytraeids (pot worms) are relatives of earthworms with a much smaller size and a similar diet. Their densities vary between $10^2/m^2$ and $10^6/m^2$, with a biomass of up to 1kg C/ha.

- Mites (fungivores, bacterivores, predators) have a size of about 1mm, densities of about $10^5/m^2$ and a biomass of up to 0.1kg C/ha.

- Springtails or Collembola (fungivores, omnivores) also have a size of about 1mm. They reach densities of 10^3-$10^5/m^2$ and a biomass of up to 1kg C/ha.

- Nematodes (bacterivores, herbivores, fungivores, predators/omnivores) have a size of about 500µm, densities of 10/g to 50/g soil and a biomass of up to 1kg C/ha.

- Protists (amoebae, flagellates, ciliates) are unicellular animals with a size of 2-200µm, densities of about 106 cells/g soil and a biomass of about 10kg C/ha.

- Bacteria are usually smaller than 2µm, with densities of about 10^9 cells/g soil and a biomass of 50-500kg C/ha. Bacteria are the smallest and most numerous of the soil organisms.

- Fungi grow as networks of threads (hyphae) which usually have diameters of 2-10µm and reach total lengths of 10-1000m/g soil, and a biomass of 1-500kg C/ha.

  [The numbers given above vary considerably according to climatic conditions, soil properties and anthropogenic impact, especially at agricultural sites.]

The majority of soil processes and functions are driven by the soil biota listed above (i.e. communities of many different microbial and invertebrate species) — very often not by individual species or groups but through the close interaction of many different organisms (Ritz et al., 2009). Thus, any change in soil biodiversity directly affects soil ecosystem services (Breure, 2004). For instance, De Vries et al. (2013) demonstrated that adequate carbon and nitrogen cycling processes require a certain level of biodiversity, i.e. a minimum number of specific feeding groups (e.g. microbes and invertebrates), a minimum total biomass of the soil food web and a minimum biomass of the fungal, bacterial and root energy channels. According to Orgiazzi at al. (2016), in 14 out of the 27 EU countries investigated, covering more than 40% of the EU's soils, moderate-high to high potential risks to soil biodiversity can be expected.

In the absence of anthropogenic impacts, the occurrence and diversity of the abovementioned species groups is mainly determined by site-specific soil properties (soil habitat predictors), including biogeographical factors (e.g. climate) and land cover (vegetation). Rutgers et al. (2009) confirmed these relationships, i.e. that the biomass and/or numbers (abundance) of major groups of soil organisms vary depending on land use and soil type. To assess the level of biodiversity below which soil functioning would be hampered, several authors suggest the determination of threshold values (Van der Heijden et al., 1998; Liiri et al., 2002; Setälä and McLean, 2004; FAO and ITPS, 2015). However, first, a clear relationship between biodiversity parameters and indicators for specific soil functions must be established (Van Leeuwen et al., 2017). This begins with setting clear objectives for the quality criteria of individual ecosystem services for specific ecosystems and ends with the selection of the most appropriate indicator organism group(s) or species. Last but not least, it must be ensured that individual structural (i.e. diversity) and functional measurements have been performed according to international standards (preferably according to International Organization for Standardization standard methods; ISO, 2006a, 2006b, 2007.

Although there have been many initiatives on mapping soil biodiversity across Europe in the past, currently there is a lack of knowledge for establishing (site- or biotope-specific) soil biodiversity baselines (EEA, 2019c). Recently. Rutgers et al. (2019) highlighted two main constraints: (1) the lack of consensus on how to quantify the soil biodiversity provisioning function; and (2) the scarcity of data needed to map the distribution of soil organisms across larger scales. However, recently progress has been made at the global level: Van der Hoogen et al. (2019) for nematode community composition, Delgado-Baquerizo et al. (2018) for bacteria, Potapov et al. (2022) for springtails and Egidi et al. (2019) for fungi. Currently, in several EU Member States, work is ongoing on the distribution of soil meso- and macrofauna (in particular earthworms), often using DNA-based methods (e.g. Pérez-Losada et al., 2012; Taberlet et al., 2018).

The soil biota is primarily affected 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 of soil food webs and the community-related biomass of soil fauna (Tsiafouli et al., 2015) and the interaction network between soil bacteria (Karimi et al., 2019). Furthermore, soil faunal communities have fewer and taxonomically more closely related species, indicating reduced soil biodiversity. Bloem et al. (2006b) found that microbial biomass, microbial activity (respiration) and the number of soil fauna functional groups tend to be greater on organic and extensively managed farms. This observation is in line with the expectation that reduced physical and chemical impacts on the soil support the development of highly diverse soil organism communities (including an increase in abundance and biomass). For example, the number and diversity of species in the soil food web, such as nematodes, generally decreases with increased land use and management intensity. In some cases this happens intentionally; for example, applying an intensive rotation will decrease the abundance of potentially harmful organisms such as phytophagous nematodes. However, in general, intensive land use affects soil food webs negatively: diversity decreases — even individual body mass may decrease as larger organisms (such as earthworms) are worse affected than smaller ones (Tsiafouli et al., 2015). In the medium term (up to 4 years) under organic management, earthworms are found at two or three times the level found in conventionally managed fields (Blakemore, 2018). Because the relationship between management regime and soil biota is fairly stable across regions, agricultural policies may be steered to halt and/or reverse this loss of soil biodiversity.

6.2 Soil biological indicators: state of the art

'Loss of soil biodiversity' means that species richness (presence and relative abundance) and activity levels are reduced, so that soil processes (e.g. organic matter decomposition) and consequently soil functions (e.g. nutrient provision) are hampered. This requires an indicator that could monitor the presence (diversity) and amount (abundance) of key species and/or functional groups in the soil (based on Bispo et al., 2009; Rutgers et al., 2009, Bouchez et al., 2016). Accordingly, high species diversity combined with high species abundance within functional groups would then provide a greater contribution from organisms to ecosystem services in a spatially diverse habitat.

In recent years, several proposals for possible soil biodiversity indicators have been presented. The following section summarises these efforts in order to draw a clear picture of the feasibility of current solutions. This overview will also help to identify gaps in our knowledge needing further research and to steer further monitoring efforts.

6.2.1 Concepts for identifying soil biodiversity indicators

Huber et al. (2008), based on Bispo et al. (2007), proposed the use of three key indicators to assess the threat of potential loss of soil biodiversity and associated ecosystem functions: (1) diversity of earthworms; (2) diversity of collembolans; and (3) soil microbial respiration. The results are presented in Bispo et al. (2009); see also Section 6.2.2.

Breure (2004) proposed the following indicators: (1) microbial biomass (bacteria and fungi, which represent the largest amount of ecological soil capital); (2) nematodes (family level and feeding types), the relative and absolute abundance of which provides good information on the diversity and stability of the ecosystem; and (3) earthworms, because of their influence on soil properties and since they are (relatively) easy to determine taxonomically and to characterise ecologically. Soil respiration represents a major component of the soil carbon cycle and is an indicator of soil carbon storage, soil biological activity and overall soil quality (Lee and Jose, 2003).

Gaublomme et al. (2006) assessed soil microbial activity and functional biodiversity in forest soils, and how this can be described through indicators. They tested various parameters in a case study on 85 forest stands across the Flanders region. The most useful parameters for indicating microbial biomass are:

- Hot water extractable carbon (HWC): this represents the labile fraction of the total soil organic carbon (SOC) pool and thus includes microbial carbon and carbon and nitrogen sources that are readily available to microbes. A depletion in HWC indicates a deterioration of or decline in SOC.

- Amount of cultivable bacteria: this is greater in deciduous than in coniferous forest stands; mull humus form seems a good indicator of the amount of colony forming units.

Ritz et al. (2009) reviewed 183 biological 'candidate indicators', from which they selected 21 genotype-/phenotype- and function-based indicators for different trophic groups; of those, 13 indicators would currently be fully deployable in monitoring activities (see also Black et al., 2011). The indicator selection process is quite complex because the authors ranked biological indicators against ecological processes and soil properties associated with its functions. In addition, they considered the indicators' applicability in large-scale monitoring schemes. The following are 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; and fungi. These indicators are related to the structure of the microbial community and are determined using DNA yield (e.g. ISO, 2016).

- Indicators based on phenotypic methods, such as extractions, visual recordings or catches (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 and macro soil invertebrates (e.g. ISO 2006a, 2006b, 2007).

- Indicators based on 'functional' methods: substrate-induced respiration; potential enzyme activity (microbial biomass and total community activity) from the feeding activity of soil invertebrates.

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 (e.g. ISO, 2016). Nevertheless, the authors stress the importance of knowledge of the 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 to fully understand the sensitivity of these many different species (i.e. their potential) and how they correlate with soil functions. In addition, the variability of microbial and invertebrate communities across spatial (landscape) scales (soil types, etc.) and climatic zones (seasonality effects) must be considered, as well as management practices.

The approaches taken 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 types and management practices. Griffiths et al. (2016) selected 18 soil biological indicators from the literature and tested them at six experimental sites in various European regions. Apart from methods that address the diversity of individual groups (invertebrates and microbes), functional methods were identified that relate to different ecosystem services. However, further development and standardisation of methods (sampling and analysis), as well as inter-laboratory comparisons, are necessary to accompany the indicator measurements in monitoring. So far, there has been only one Europe-wide sampling programme covering almost 100 sites in which both structural and functional endpoints have been measured at the same time and place (Stone et al. 2016). Since 2018, a subsample of LUCAS Soil plots have been analysed for soil biodiversity using DNA-based methods (see Box 6.1; see also Orgiazzi et al., 2018).

Recently, Guerra et al. (2021) proposed essential biodiversity variables that they then related to policy needs (Convention on Biological Diversity, Sustainable Development Goals, Paris Agreement). Based on such variables, ecological indicators can be derived: soil health, soil biodiversity, soil carbon stocks, ecological vulnerability of soils, soil conservation value. The authors represent the Global Soil Biodiversity Observation Network (SoilBON ([24])), which operates under the Group on Earth Observations Biodiversity Observation Network (GEOBON), and which invites researchers globally to collect and share observational data on the condition of soil biodiversity and functions. While the suggested essential biodiversity variables have been discussed in previous frameworks including those cited above, Guerra at al. (2021) also recommend specific analytical methodologies:

- intraspecific genetic diversity (DNA extraction);
- abundance of species populations (DNA-based bacteria, fungi and 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 of nematodes; functional diversity of bacteria, archaea, fungi and selected invertebrates);
- soil biomass (substrate-induced respiration method);
- litter decomposition (litter bags, combined with incubation);
- soil respiration (oxygen consumed by microbial respiration);
- feeding activities of soil invertebrates (e.g. by the bait-lamina method);
- enzymatic activity (incubation followed by fluorescence measurements);
- soil aggregation (soil aggregate resistance index);
- nutrient cycling (amount and availability of nitrogen, carbon and phosphorus);
- habitat extent (bulk density and soil structure).

Following the vision of developing one or more holistic indicators for soil biodiversity, it becomes clear from Guerra et al. (2021), and their predecessors, that a large amount of information is available, and a clear idea exists of how measurable parameters can be systematically aggregated to develop indicators about soil biodiversity in support of environmental policies.

([24]) SoilBON project: https://geobon.org/bons/thematic-bon/soil-bon

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 using DNA was hampered considerably by the lack of trained taxonomists and by using morphological features alone. However, considerable progress has been made in the last decade, but it is only recently that some of these methods were standardised through the International Organization for Standardization (e.g. ISO 11063; see also Plassard et al., 2012).

These new methods are already in use (Aslani et al., 2021) and are to be used for nationwide soil biodiversity monitoring, for example in Germany (Römbke et al., 2022). In the near future, efficient, cost-effective and routinely applicable DNA-based methods for soil biodiversity monitoring systems are expected to be available for monitoring and evaluating soil biodiversity. Thus, in the foreseeable future, baseline and threshold data (see Section 6.3) will be generated for soil organism communities (ideally, for both invertebrates and selected groups of microorganisms, and probably also at the European level.

6.2.2 *Experience of applying soil biological parameters in soil monitoring*

Bispo et al. (2009) successfully tested the three Envasso project indicators in France, Hungary, Ireland and Portugal (namely: diversity of earthworms, diversity of collembolans and soil microbial respiration). To assess the diversity and abundance of the three indicators at the European scale, reference values or baselines are needed. Bispo et al. (2009) refer to work from the Netherlands (Rutgers et al., 2009) in which ranges for selected land use and soil type categories were developed. In the Netherlands, 12 biological indicators were measured at 300 locations in a 6-year cycle (Rutgers et al., 2009). Biological parameters included abundance of earthworms, nematodes, micro-arthropods and enchytraeids, bacterial biomass and DNA diversity; most parameters showed a clear pattern across gradients of land use intensity and soils. Table 6.2 presents some results from the French Bioindicator programme.

The French Soil Quality Monitoring Network initiative (Arrouays et al., 2002) allowed the characterisation of soil microbial communities across the country for microbial biomass and bacterial diversity under multiple soil types and land use types (Dequiedt et al., 2011; Ranjard et al., 2013; Terrat et al., 2017). More than 1,700 sites were characterised using standardised methods.

Römbke et al. (2016) analysed and compiled soil biological parameters from soil monitoring programmes in 15 European countries. Krüger et al. (2017) 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 mineralisation (laboratory incubation), metabolic potential of soil bacteria (physiological profiling), and earthworm abundance (extraction). They demonstrated that all the indicators they tested distinguished between the main land use types and enabled a rapid 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 more variable were the indicator values.

The LUCAS Soil survey, coordinated by the European Commission's Joint Research Centre since 2009, offers an open-access database, including soil physico-chemical properties, collected every 3 years in over 20,000 locations across Europe. Since 2018, a soil biodiversity component from 1,000 points has been included in the survey. It includes DNA metabarcoding of bacteria, archaea, fungi and other eukaryotes (e.g. invertebrates). The final aim is the characterisation of soil organism communities and the identification of indicator species and communities associated with soil properties (e.g. organic carbon), climatic conditions, threats (e.g. erosion) and land cover.

Table 6.2 Monitoring biological groups: results from the French soil monitoring network

Parameters/indicators	Indicator value	Site density	Source
Abundance of earthworms, nematodes, acari and the bacterial community, microbial biomass and earthworm species richness	Main land use (grassland, cropland, forest)	109	Cluzeau et al. (2012)
Macro-invertebrate abundance, collembolan abundance and richness, and nematode richness	Agricultural practice		
Biological soil quality index based on soil macro-invertebrate community patterns	Agricultural practice	22	Ruiz et al. (2011)
Earthworm community and species (abundance, biomass, functional structure and ecological traits)	Main land use, degree of soil pollution	13	Pérès et al. (2011)
Soil microbial abundance and diversity	Main land use and/or agricultural practices	>1700	Terrat et al. (2017)

6.2.3 Databases to support baselines

Rutgers et al. (2016) collected and harmonised existing earthworm community data from several European countries and combined the measured occurrences of earthworm taxa with environmental and climatic variables. Thus, they could predict earthworm abundance and produce biodiversity maps of earthworm abundance and numbers of taxa, which could serve as a reference data set for monitoring purposes.

Several global assessments of earthworms, springtails, bacteria and fungi have recently been published (see Section 6.1). Van den Hoogen et al. (2020) compiled a global nematode database. Soil nematodes are a good indicator of soil biodiversity because they play a central role in regulating carbon and nutrient dynamics, and they control soil microorganism populations. Tundra, boreal and temperate forests have the highest abundances (>2,000 nematodes/100g dry soil).

Edaphobase ([25]) is an archive of the distribution and ecology of soil animals (earthworms, small earthworms, nematodes, springtails, mites, centipedes, millipedes and woodlice) (Burkhardt et al., 2014). Edaphobase contains more than 500,000 observations, about 300,000 sites and 140,000 taxa (Römbke et al., 2012). The approach is to be modified in order to collect information on soil biodiversity in an extended version by connecting Edaphobase with the databases of other (mainly European) countries in the EUdaphobase project.

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 for the expected effect of soil biota under good conditions (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 their correlation with ecological niches and environmental parameters; thus, their geographical distribution could potentially be predicted from environmental data (Rutgers et al., 2016). Aksoy et al. (2017) provided the first overview of the diversity of soil animals and organisms in relation to the existing diversity of soils and their properties (see also Section 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 four categories: (1) soil nutrients; (2) soil biology; (3) soil structure; and (4) soil hydrology. These 37 attributes were used in a decision model to derive a qualitative assessment of the biodiversity function of soils. Given the large number of attributes required, data availability certainly limits the application of the model.

([25]) Edaphobase, a project under the German contribution to the Global Biodiversity Information Facility, GBIF-D (https:/portal.edaphobase.org). Contact to cooperate: https://www.eudaphobase.eu/contact.

Attributes are assessed in qualitative terms (high, medium, low categories), making data reliability less critical, on the one hand, while, on the other hand, needing expert knowledge for setting thresholds.

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. The following 14 attributes are analysed to determine soil biodiversity: soil texture, bulk density, groundwater table depth, pH, C:N ratio, N:P ratio, soil organic matter, organic carbon content, earthworm abundance and richness, nematode abundance and richness, bacterial biomass and fungal biomass. These attributes were measured from soil samples at various sites in Europe and combined with site, management and environmental attributes to quantify the functional capacity of the soils, evidencing the difficulty, but feasibility, of monitoring soil biodiversity across Europe. However, there is still a lack of standardised functional and structural methods (including soil biodiversity) to monitor and to quantify ecosystem functions and services (Rutgers et al., 2012).

Table 6.3 provides an overview of indicators related to soil biodiversity. Based on experiences in Germany focusing on species diversity, Toschki et al. (2020) proposed using different invertebrate groups (i.e. Enchytraeidae, Collembola, Chilopoda, Diplopoda and Oribatida) for the characterisation of three main land use types: forest, grassland and cropland sites. Only Enchytraeidae were useful for all land use types, but surely for the biological characterisation of soils more than one group of invertebrates is necessary. This is because not all groups naturally occur at all sites at similar and/or sufficient levels of diversity (e.g. soft-bodied organisms such as earthworms or enchytraeids do not thrive well in permanently dry soils).

Most of the methods used to determine the indicators have been standardised by ISO (Römbke et al., 2018). To assess the ecological condition of a given site, the 'reference approach' is recommended: a reference database is developed that contains the respective assemblages of species (earthworms, collembolans, 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 by comparing it with such reference assemblages. Details of 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 carried out within the last 20 years. The approach is operational and has two conditions: (1) the exact site-specific parameters for such a reference database need to be agreed; and (2) existing national monitoring pilot studies (see above) need to be extended and established in all countries, so that as many as possible representative soil conditions and their respective organism communities are covered in the database. Edaphobase and LUCAS Soil could serve as reference databases.

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 collembolans		●	
Microbial biomass	●	●	●
Diversity of nematodes	●		●
Soil texture	●		
Bulk density	●		
Groundwater table depth	●		
pH	●		
C:N ratio	●		
N:P ratio	●		
Soil organic matter	●		
Organic carbon content	●		

6.2.5 Additional aspects to consider in soil biodiversity monitoring

As well as any standard documentation on the site of soil sampling, we recommend collecting the following additional information:

- fabric of organic horizons (i.e. peat, forest floor): nature and arrangement of humus constituents (structure, consistency, character) (see Green et al., 1993);
- type of litter: plant species of origin, plant part (wood, leaf or needle, root), decomposition status;
- fungal mycelia and faunal droppings: distribution and abundance;
- roots: abundance and size;
- presence/abundance of common soil fauna, particularly earthworms, separating them into the functional groups endogeics (dwellers in the mineral layer), epigeics (dwellers in the litter layer) and anecics (vertical burrowers);
- horizon boundaries: shape and width.

Regarding the optimal measurement intensity (i.e. number of methods, sampling sites, etc.), Bispo et al. (2009) suggested that different levels of monitoring could be defined depending on the aim(s) of the monitoring programme. They proposed that at the first level (I) only samples for earthworms and collembolans are taken, while functional diversity and DNA analysis is done in fewer plots (level II). Only for specific questions (e.g. complex biological functions) would further samples be taken (level III). The sampling methodology and level will need to be adapted depending on the specific questions to be answered and the aims of the monitoring.

6.3 Baseline and threshold values

An indicator is useful only if its value can be unequivocally interpreted and reference values are available (Bünemann et al., 2018). Reference values for a given indicator could be either those of a native soil, which may however not be suitable for agricultural production, or those of a soil with maximum production and/or environmental performance (Doran and Parkin, 1994). In the Netherlands, for example, 10 reference soils for good soil biological quality were selected out of 285 sites that had been monitored for over 10 years (Rutgers et al., 2008). These reference soils represent specific combinations of soil type and land use (e.g. arable land on clay soil). Soil quality indicators at a given site could thus be compared with those at the reference site and with the mean value, and the 5th and 95th percentiles of all sites under a given land use, with the percentiles used to express the frequency distribution. An important drawback of this approach is that the reference site may not represent the optimum for all parameters (Rutgers et al., 2012).

6.3.1 Definitions

As demonstrated above, a baseline and threshold values are needed to monitor soil biological diversity. However, such values have barely been achieved yet. Huber et al. (2008) suggest that both baseline and thresholds need to be stratified by soil type and land use; Rutgers et al. (2018) introduce additional stratification by climatic zone and management practice. Huber et al. (2008) proposed a common approach to the derivation of baseline and thresholds in the Envasso project:

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

  Huber et al. (2008) suggest calculating reference scenarios as a baseline, consisting of minimum, maximum and mean values for each indicator, by land use, soil type and climatic or biogeographical region. Cluzeau et al. (2012) applied this approach in 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 highlighted that soil fauna and microbial biomass can be used as bioindicators.

  In a complementary approach, Horrigue et al. (2016) and Terrat et al. (2017) used biogeographical approaches which allowed defining models for the estimation of baselines for soil microbial biomass and bacterial diversity based on soil characteristics. The comparison of this baseline to a measured value allows estimation of the deviation from the baseline under particular land use/agricultural practices.

- **Threshold values**. These are defined by Schneiders et al. (2012), as a 'safe minimum standard of conservation': exceedance of the threshold implies irreversible changes in ecosystem conditions 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 naturally be 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).

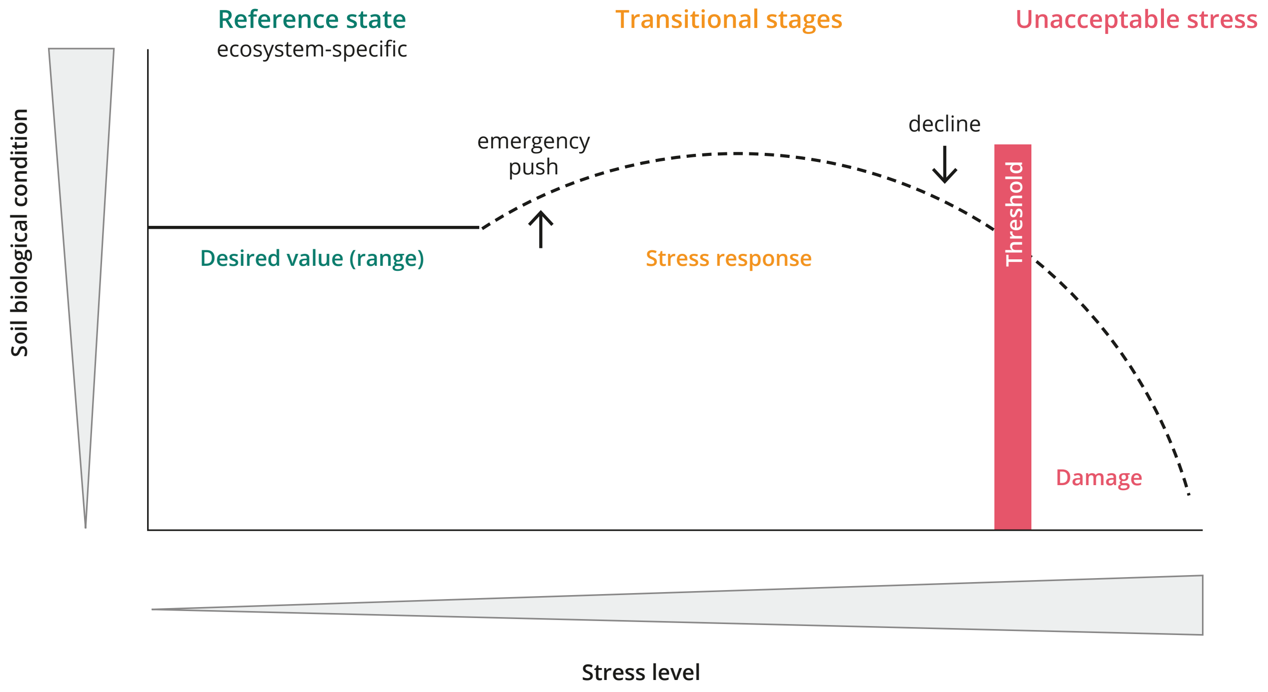
A more elaborate approach would aim to define a threshold as an unacceptable deviation from the baseline value or from the initial measurement. In the latter case, natural variations must be taken into account.

The same approach is suggested by Breure et al. (2004): initial information on status and trends from monitoring can be combined with ecological know-how and serve as a baseline or standard value for each indicator. Unacceptable (and natural) variations could be defined based on variations measured under regional, national and international monitoring networks. These data sets should be collated according to soil type, land use and climate.

More recently, Römbke et al. (2012) proposed another approach, which is briefly summarised in Figure 6.1. In this case, thresholds are based on species richness, but other biological endpoints or indices are possible. From a biological point of view, the limit of unacceptable degradation is not easy to determine, especially when looking at whole communities rather than one species. Figure 6.1 indicates the general relationship between soil biological condition (e.g. species richness) and the change in the diversity of soil organism groups at various levels of stress, ranging from an unaffected (reference) site to a site that has been severely affected 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, where ranges of, for example, species numbers for a given site or soil type were compiled (example: earthworms). The quality of the approach improves as the number of well-documented observations increases (ideally, at international level). As soon as an impact on soil biodiversity is observed (in this example, earthworm as flagship species), more detailed investigations are necessary, such as repeated sampling in different seasons.

Figure 6.1 **Baseline and threshold approach for soil biodiversity monitoring**



Sources: Römbke et al. (2012, 2016).

6.3.2 Operating ranges for key soil organisms

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

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 that are unsuitable for earthworms, mesofauna or microorganisms. Thresholds for temperature, texture, electrical conductivity, pH and land use change have been used to spatially delineate 'risk' areas for earthworms — and this approach can be extended to collembolans and probably other soil faunal groups. It seems that this approach is suitable for the mapping of soil biodiversity hot spots. However, considering the broad ranges as well as the large amount of species presence and abundance that cannot be explained by the scoring, a better differentiated approach is needed. By overlaying the earthworm abundance maps of Rutgers et al. (2016) with environmental spatial covariates, it is possible that Table 6.4 could be refined for earthworm diversity and also to provide a hypothetical value for earthworm diversity and abundance.

A similar approach was also suggested by Römbke et al. (2016) and Hallin et al. (2012). However, the latter authors proposed operating ranges for specific soil animals and microorganisms, defined as the range of a specific parameter (e.g. temperature, pH) that 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 various agronomic practices under different environmental conditions. The studies cited have shown that the diversity and composition of faunal and microbial communities affect ecosystem functioning under fluctuating conditions.

The conditions for ectomycorrhizal fungi in European forests were studied by Van der Linde et al. (2018). Because the growth of ectomycorrhizal fungi is strongly determined by the soil environment, thresholds could be determined for key environmental variables, correlated with the abundance of the fungi studied. The authors investigated 1,406 operational taxonomic units, from a total of 25,196 samples. Table 6.5 presents the results, which may serve as a baseline for assessing future changes in and resilience of forest fungi.

Table 6.4 Thresholds for environmental variables that may have a strong effect on soil biodiversity

Variable	Classes of parameters for scoring 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 to 500	>500	
Annual average temperature (°C)	<5	5-20	>20	
Soil biomass productivity	Poor	Average	Good	
Land use/land cover	Artificial	Arable	Permanent crops	Others

Source: Compiled from Aksoy et al. (2017).

Table 6.5 Environmental thresholds for operational taxonomic units (OTUs) of ectomycorrhizal fungi

Variable	Decreasing OTUs	Increasing OTUs
Throughfall nitrogen deposition	5.8kg N/ha/year	15.5kg N/ha/year
Forest floor pH	3.8	
Mean annual air temperature	7.4°C	9.1°C
Throughfall potassium deposition	6.9kg K/ha/year	21.7kg K/ha/year
Foliar N:P ratio	10.2	13.3

Source: Compiled from Van der Linde et al. (2018).

7
Soil erosion

In this chapter we discuss the assessment of soil erosion, with a special focus on soil erosion by water. We specifically address rill and interrill erosion and ephemeral gullying, which can be assessed through large-scale soil monitoring. In some parts of Europe, particularly the Mediterranean basin, permanent gullying and badlands are also important forms of degradation. Soil erosion refers to the loss of fertile topsoil after erosive rain events on sensitive soils, largely in the absence of sufficient vegetation cover. Based on modelling, about 13% of arable soils in Europe are affected by medium to high soil erosion rates. The main indicator for soil erosion is the rate of loss of topsoil mass, which is usually expressed in tonnes per hectare per year. While several generalised thresholds for unacceptable erosion levels have been developed, the concept of a tolerable rate of soil loss and the implementation of tiered monitoring is recommended here.

Soil erosion is the detachment, transport and sedimentation of soil particles by water or by wind. It has negative impacts on soil functions and (soil-related) ecosystem services (Table 7.1). Soil erosion is itself an important driver of other soil threats, especially an increase in flood risk and a decrease in biodiversity and soil organic matter (Stolte, 2016), also playing an important but underestimated role in soil organic carbon cycling (Chappell et al., 2016). While weather, and in particular heavy rainfall, is the trigger for soil erosion by water, in modern times agriculture, overgrazing, mining and infrastructure are the drivers of severe soil erosion (Stolte et al., 2016) due to unsustainable soil management. Soil erosion does not endanger soil functions and the provision of ecosystem services if the rates of erosion are less than the rates of geological soil formation. Unsustainable land management and climate change (increasing weather extremes) induce the acceleration of rates of soil erosion. Onsite effects of erosion often include crop yield losses: on average 4% per 10cm soil loss (Bakker et al., 2004) or 8% for severe soil erosion (>10t/ha/year; Panagos et al., 2018). Large erosion events are often accompanied by substantial offsite damage (river and dam sedimentation, including offsite pollution (Boardman, 2006)).

Table 7.1 Relationship of soil erosion to key societal needs and soil functions

Societal need	Soil service	Impact
Biomass	Wood and fibre production	-
	Growth of crops	-
Water	Filtering of contaminants	-
	Water storage	-
Climate	Carbon storage	-
Biodiversity	Habitat for plants, insects, microbes, fungi and any microfauna	-
Infrastructure	Platform for infrastructure	-
	Storage of relocated material or artefacts (excavated geological material, sediments, cables and pipelines, archaeological material)	-

Note: Soil erosion can increase the volume of soil sediments that need to be managed.

7.1 Erosion processes and challenges for soil monitoring

7.1.1 Types of soil erosion

Based on Huber et al. (2008) and Poesen (2018) various types of erosion, defined by agent, process, resulting landform and intensity, can be distinguished:

- **water erosion**: interrill (sheetwash), rill, ephemeral gully, gully and piping (subsurface) erosion resulting from surface run-off of excess rainwater or subsurface flow; gully erosion is a special and very intense form of water erosion in areas with concentrated run-off leading to large gullies as the typical landform;

- **wind erosion**: strong air movements displacing loose soil particles;

- **anthropogenic (technic) erosion**: including tillage erosion (onsite soil loss after tillage of sloping land), harvesting erosion (offsite losses of soil adhering to the crop during harvest, mainly of root and tuber crops), erosion caused by livestock trampling (onsite soil loss occurring after soil compaction, overgrazing and removal or reduction of vegetation cover).

Monitoring these different types of erosion is challenging, because they operate at different spatial and temporal scales (Stroosnijder, 2005). In addition, several processes can also occur in parallel, or trigger each other, such as livestock trampling and water erosion, water and tillage erosion, or water and harvesting erosion.

Mean rates of soil loss for arable land taken from studies at the continental scale (Europe) (Table 7.2) indicate that soil erosion by water and tillage erosion are the most severe erosion types in Europe. This overview of the mean rates for the whole of Europe masks the fact that the environmental and climatic conditions and management practices vary widely at the continental scale, leading to considerable differences in the magnitude and importance of the different types of soil erosion in different locations. For example, gully erosion in the Mediterranean region can account for 10-80% of total erosion on cultivated and grazed land, whereas water erosion can clearly dominate in temperate areas where erosive rainfall events are less frequent and intense (Boardman and Poesen, 2006). As discussed by Kuhwald et al. (2022), soil erosion by crop harvesting is important in regions with a large proportion of tuber and root crops in the rotation. In such hot spots, soil loss due to harvesting can be of the same magnitude as water and wind erosion. To be reliable and relevant, Europe-wide monitoring schemes need to address these regional differences in the importance and magnitude of the different types of erosion. The type of soil erosion that prevails under specific climatic conditions or management practices (e.g. growing tuber or root crops) can be most effectively monitored.

Table 7.2 Soil loss rates from different types of erosion on arable land in Europe taken from studies at a continental scale

Erosion type	Approach	Reference period	Mean soil loss rate (arable land) (t/ha/year)	Reference
Soil erosion by water	Empirical model (RUSLE2015)	2016	2.65	Panagos et al. (2020a)
	Upscaling plot data	-	3.6	Cerdan et al. (2010)
Soil erosion by wind	Empirical model (GIS-RWEQ)	2001-2010	0.53	Borelli et al. (2017)
Gully erosion	Observations and screen interpretation	2018-	unknown	Borelli et al. (2022)
Tillage erosion	Empirical GIS-based model	2000	3.3	Van Oost (2009)
Soil loss by crop harvesting	Upscaling study results	2000-2016	0.134 ([a])	Panagos et al. (2019)

Note: ([a]) Due to potato and sugar beet harvesting; recalculated for the total arable land area.

In addition to the erosion types listed in Table 7.2, Poesen (2018) lists others that deserve attention: subsurface erosion resulting in piping and tunnelling, land levelling, soil quarrying and trench digging. Some of these forms are restricted to special soil and geological conditions (subsurface erosion), or related to urban sprawl, and are often combined with soil relocation for construction projects (land levelling, trench digging). Soil erosion also occurs along the embankments of roads and railways, as well as a result of tourism, infrastructure or leisure activities (Seutloali and Beckedahl, 2015; Salesa and Cerdà, 2020). These special forms of soil erosion would likely not be targeted in a EU-wide monitoring soil scheme.

7.1.2 Soil erosion and ecosystem services

The role of soils in providing ecosystem services is introduced in Chapter 1.1. Soil-related ecosystem services affected by soil erosion include the provision of crops (crop growth), carbon sequestration, water filtration, water flow regulation, freshwater provision and the habitat function of safeguarding (soil) biodiversity (see Paul et al. (2021) for a list of soil-related ecosystem services based on CICES, the Common International Classification of Ecosystem Services). The control of erosion rates is a regulating ecosystem service in itself, representing a reduction in the loss of soil from vegetation covering the ground compared with bare soil (Guerra et al., 2014).

The impacts of soil erosion on soil-related ecosystem services can be assessed to estimate its negative effects on the provision of such services. Soil erosion causes the thinning of topsoil, with direct decreases in soil organic matter and water-holding capacity and subsequent negative long-term effects on various biological and chemical soil processes. Maintaining soil volume and topsoil is key to sustaining soil-related ecosystem services. This is also reflected in the study of the effects of erosion on ecosystem services by Steinhoff-Knopp et al. (2020) in northern Germany. There, sub-indicators for soil-related ecosystem services were quantified (crop provision, water filtration, water flow regulation, freshwater provisioning), using pedotransfer functions based on local soil properties, management and climate. The approach is site specific and considers the spatial variability of covariates that determine the degree of soil erosion. With the help of an evaluation matrix, the authors were able to quantify the potential supply of soil-related ecosystem services from degraded soils and evaluate the impact of soil erosion on such services.

Within this concept, the reduction in the supply of ecosystem services following erosion can be quantified based on the changes in soil properties caused by the loss of fertile topsoil from erosion. If the target of a minimum good status of potential ecosystem service supply is set, site-specific limits for tolerable erosion rates can be derived (see Section 7.3). It must be emphasised that an underlying matrix of sub-indicators requires validation for the varying soil and climatic conditions in Europe. The approach can be applied in European regions where spatial information on soil profiles/soil properties, with matching pedotransfer functions to assess soil-related ecosystem services and spatial estimates of erosion rates, is available. As a proxy (top)soil depth can be used to represent the sub-indicators, while research is needed to establish regional specific, reliable links between the two.

7.1.3 Onsite and offsite effects of soil erosion

The main onsite effect of soil erosion is the reduction in soil quality, induced by the loss of fertile topsoil (see Section 7.1.2), and this includes soil carbon (Lugato et al., 2016). However, these onsite effects are typically accompanied by offsite effects: the eroded material is transported by wind and water to adjacent and remote locations, along spatial gradients such as slopes and water run-off channels, sedimenting in catchments and river deltas, dams and other water-harvesting installations, harbours, and damaging buildings, adjacent properties and other infrastructure as a result of flooding (e.g. Verstraeten et al., 2006). Eroded soil is also known to clog up drainage systems, which could result in overflow and wash-out and subsequently their failure (WHO, 1991). If coupled with poor drainage maintenance, soil erosion can affect the lifetime of pavements (e.g. deformation, frost heave). Furthermore, road and sedimentation clearance operations are needed, causing significant cost.

Soil erosion can reduce water quality in rivers due to sediment and sediment-fixed contaminants (e.g. phosphorus); this can affect the public water supply. For example, the transport of soluble pollutants can occur through wash or sheetwash erosion (typically when topsoils are saturated and run-off is slow), which is difficult to model because erosion rates are generally low.

A recent regional-scale modelling study on lateral carbon fluxes (Nadeu et al., 2015) indicates that the lateral export of carbon from cropland through erosion may be of approximately the same magnitude as additional carbon sequestration in carbon-depleted eroded soils. Boardman and Vandaele (2022) reviewed the offsite effects of run-off from agricultural fields; they suggest that, in western Europe, such effects could be even more important that onsite effects.

7.1.4 Status of soil erosion by water

Soil erosion is among the eight soil threats listed in the European Commission's soil thematic strategy (EC, 2006a); it is one of most widespread forms of soil degradation, especially in agricultural areas. Hot spots include south-eastern and eastern Europe, as well as the Mediterranean region (Kirkby et al. (2004), based on predictions using the Pesera model). In the latter case, this is because of a long history of erosion in the Mediterranean basin, which has removed or degraded soils making them now more susceptible to run-off (floods) and erosion; the lack of proper soil cover from crops during the winter months also increases the risk of soil erosion by rainfall.

Several recent publications summarise the state of soil erosion in the agricultural areas of the EU (EEA, 2019a; Veerman et al., 2020; Eurostat, 2022). These reports essentially go back to the works of Panagos et al. (2015, 2020a). They estimated soil loss using the empirical prediction model RUSLE (Revised Universal Soil Loss Equation). According to Panagos et al. (2015), the mean soil loss rate in Europe's water erosion-prone lands (agricultural, forest and semi-natural areas) was found to be 2.5t/ha/year, resulting in a total soil loss of 970Mt annually. Roughly 25% of the EU land area shows erosion rates >2tha/ year, while about 6% of the agricultural area shows severe erosion (i.e. 11t/ha/year) (Panagos et al., 2020a).

Compared with the above results from European modelling, some authors have summarised erosion rates from upscaled local observations or plot measurements (e.g. Gobin et al., 2004). These results show that it is inaccurate to use average values rather than ranges of values, given that a few critical erosive events can increase the medians of otherwise insignificant erosion levels. Cerdan et al. (2010) conclude an average erosion rate for arable land across Europe to be 3.6t/ha/year ([26]), with a maximum rate of 17.4t/ha/year for vineyards. Assuming an average soil bulk density of 1.25g/cm^3, this translates into a loss of 0.2mm and 1mm of topsoil loss per hectare per year, respectively. Considering the typically slow rates of natural soil formation from weathering and other processes, estimated to be 0.05-0.5mm/year (Wakatsuki and Rasyidin, 1992, cited in Gobin et al., 2004), any soil loss of more than 1t/ha/year can be considered irreversible. Cerdan et al. (2010) estimate that 70% of the total erosion occurs in 15% of the area; thus, the rate of erosion varies considerably across Europe and is likely to occur at much higher rates than average in hot spots. Hence, standard estimates based on average values are not helpful in tackling soil erosion issues.

Darmendrail et al. (2004) present the results for 14 localities surveyed in the United Kingdom, totalling 4.8 million ha, and found an average annual erosion rate of 0.9t/ha. For comparison, the modelled rates of erosion in the United Kingdom, based on Panagos et al. (2015), amounts to 2.4t/ha/year across all land uses, and 1t/ha/year for arable land (cited from Evans and Boardman (2016); higher rates from modelling did not spatially correspond to the field measurements). For seven areas investigated in northern Germany, an average erosion rate of 0.8t/ha/year was found from field observations (Steinhoff-Knopp and Burkhard, 2018a, 2018b). Vandaele and Poesen (1995) applied a volumetric measurement method to obtain soil loss rates for two catchments in Belgium. The mean annual rates for a 3-year period were 6.75t/ ha and 10.25t ha ([27]). In a review of results for Spain, Benet (2006) cites 36 studies, which in total indicate the very large spatio-temporal variability of erosion processes, and the importance of harmonising and agreeing upon methods to measure and model erosion rates.

Prasuhn (2020), based on long-term monitoring of 203 fields in Switzerland, reported mean soil loss rates of 0.7t/ha/year (monitoring from 1997 to 2007) and 0.2t/ha/year (2007 to 2017). The author attributes this reduction in the rate of soil loss to changes in soil tillage practices and erosion control. Considering the variability of triggering weather events, accurate erosion monitoring is difficult: in a watershed in central Switzerland, between 1998 and 2007, 50% of all eroded material was related to six events (BAFU, 2017). For English lowlands, Evans and Boardman (2016) report that only about 5% of the observed area suffers from any level of erosion each year — depending on the risk category (the value is higher for more erodible soils) — and only a few events dominate the long-term averages, highlighting again the extreme variability of soil erosion in space and time. Empirical measurements from soil erosion plots show that most of the soil losses occur in few events. After two decades of measurements, Cerdà et al. (2021) found that five rainfall events (out of 470 in 11 years) contributed to 56% of the run-off. The effect on sediment delivery is even greater (Cerdà et al., 2018).

Climate change scenarios for the Mediterranean indicate high and increasing erosion risks, due to sparse vegetation, low soil structural stability, steeply sloping land and intense rainstorms (Cheviron et al., 2011). Taking an Intergovernmental Panel on Climate Change (IPCC) approach, Panagos et al. (2021) have estimated soil erosion projections in Europe for 2050 using 19 climatic models. Soil erosion as a result of rainfall is projected to increase by 16-26% depending on the RCP (representative concentration pathway) scenario.

7.2 Indicator specifications

7.2. Indicators for soil erosion by water

The soil erosion indicator aims to delimit and quantify the extent of land that is suffering from soil loss due to water erosion at such a level that proper soil functioning and the supply of soil-related ecosystem services is impaired ('tolerable soil loss rate'). The requirement is that the indicator can be regularly updated and applied across Europe to identify areas at risk. The indicator definition goes beyond purely model-based approaches and includes the collection of field data (ground-truth data on soil erosion). Current knowledge on indicators used to assess soil erosion by water is summarised so that a harmonised approach for Europe leading to a tiered monitoring approach can be developed.

([26]) The estimate is based on measurements from 81 experimental sites in 19 countries.
([27]) The report provides volumetric data in m³/t/ha; for comparability between catchments, a common bulk density of 1.25 t/m³ was used.

The main indicator for soil erosion by water is the soil loss rate, which is usually expressed in tonnes per hectare per year. Loss rates can be estimated using models (e.g. (RUSLE, PESERA) or measured using field- or plot-based monitoring approaches. In the absence of data on the actual soil erosion rate, various proxy or impact (sub-)indicators are used to estimate the severity of erosion, and/or to estimate a potential soil erosion rate. These may include for example, land use information, dimensions of erosion features, increased turbidity in run-off, amount of deposition and exposure of subsoil, amount of sediments or other indirect parameters such as changes in soil depth, reduced organic matter content, exposure of plant roots, and changes in soil texture. The set of (R)USLE factors can, for example, also be considered sub-indicators and mapped separately to differentiate natural and management-related contributors to soil loss rates.

An overview of soil-related indicators to support agri-environmental policies at European and global levels is given in Panagos et al. (2020b). The indicators are summarised in Table 7.3.

At the European level, the common agricultural policy addresses soil erosion by means of two sub-indicators: the estimated soil loss rate by water erosion and the area affected by certain rates of soil erosion by water (Table 7.3). In the EU Resource Efficiency Scoreboard, and the EU Sustainable Development Goal indicator set (EC, 2016), the area affected by severe erosion rates (estimated soil loss by water >10t/ ha/ year) is used as an indicator of soil erosion. Erosion is also included as one of the 28 agri-environmental indicators and is expressed as erosion rate at different administrative levels, since it is intended to monitor the integration of environmental concerns into the common agricultural policy at EU, national and regional levels. It is focused on agricultural areas and natural grassland and distinguishes between moderate (5-10t/ha/year) and severe (>10t/ha/year) erosion.

The data for the soil erosion indicators to support the current EU agri-environmental policies are estimated in the assessments of soil loss by water erosion carried out by the European Commission Joint Research Centre (JRC) using the RUSLE2015 model (Panagos et al., 2015, 2020a). Prediction models are capable of producing large-scale maps at national and European levels. RUSLE2015, implemented by JRC, guarantees that the estimation of loss rates is carried out by means of a standard EU-wide method and thus that the indicators are uniform across policies. At the same time, the indicators are mono-thematically bound to a central indicator (soil loss rate by water erosion) and limited to the results from the model. The indicators based on loss rates consider management but do not allow direct insights into the impact of management on soil erosion by water. Nor are they linked to the definition of site-specific tolerable soil loss rates to safeguard the provision of soil-related ecosystem services, and they do not include observations of soil erosion events and features (monitoring approaches). While monitoring approaches enable the acquisition of ground-truth data on the frequency and severity of soil erosion, direct measurements are limited to plots, fields and smaller investigation areas. Modelling and monitoring approaches can be combined in tiered monitoring to produce a database estimating the current status of soil erosion in the EU based on ground-truth and modelling (Section 7.2.2), thus enabling the definition of soil erosion indicators.

Table 7.3 Soil erosion indicators to support agri-environmental policies at European and global levels

<table>
<tr><th>Agri-environmental policy</th><th>Indicator</th></tr>
<tr><td>EU: common agricultural policy (CAP 2014-2020)</td><td>CAP context indicator 42: soil erosion:
a) Sub-indicator: estimated rate of soil loss by water erosion (t/ha/year)
b) Estimated agricultural area affected by a certain rate of soil erosion by water (ha)</td></tr>
<tr><td>EU: Resource Efficiency Scoreboard</td><td rowspan="2">Estimated soil erosion by water: area affected by severe erosion rate (>10t/ha/year)</td></tr>
<tr><td>EU: Sustainable Development Goals indicator set</td></tr>
<tr><td>EU: Agri-environmental indicators (AEIs)</td><td>AEI 21: soil erosion: mean estimated rate of soil loss by water erosion (t/ha/year) at various administrative levels (Member State, NUTS1, NUTS2, NUTS3)</td></tr>
</table>

Note: NUTS, Nomenclature of territorial units for statistics.

Source: Panagos et al. (2020b).

7.2.2 Two-level erosion monitoring and indicator definition

A two-level monitoring of soil erosion by water for arable land in Europe could address the different climatic, pedogenetic and agricultural conditions and focus on regionally relevant erosion processes. This section outlines such a monitoring approach and suggests the combination of large-scale modelling and long-term monitoring approaches at two levels considering scale and — subsequently — measurement intensity. The temporal dimension of any erosion monitoring programme must be long term because of the typically discontinuous character of soil erosion.

The two levels suggested in Table 7.4 are in essence not new. However, until now, a strategy has not been established to better integrate these levels; doing so would allow more accurate and better validated EU-level assessments.

Level 1 includes large-scale modelling to estimate the rates of soil loss by water and the impact of management on soil erosion. This modelling approach is already implemented to support various EU policies (Panagos et al., 2020a), and is needed to create harmonised information on soil erosion by water across Europe. Level 1 identifies hot spots of soil erosion that should be monitored at level 2.

Level 2 includes monitoring approaches (mapping, measurements) to create ground-truth data on soil erosion by water using standardised methods at the field to landscape scale focusing on agricultural parcels of land. The quantification (extent, frequency and severity) of water erosion (interrill, rill and gully erosion) through monitoring is generally difficult and more expensive and time consuming than modelling approaches. Focusing on relevant hot spots and standardising field survey methods is needed. This is also valid for the monitoring of mass soil movements (including landslides) and wind erosion, which are not covered by this report.

Often long-term monitoring data from run-off plots are used to directly measure soil loss by interrill and rill erosion (Maetens et al., 2012). Run-off plots, whether operated under natural rainfall conditions or in combination with a rainfall simulator, are a key element in soil erosion research. Measurements at run-off plots are suitable for studying rill and interrill erosion (Cerdan et al., 2010). They are often installed to compare different land uses and agricultural practices and to increase our knowledge of soil erosion processes. A database of run-off plots for Europe has been compiled by Maetens et al. (2012). Data from run-off plots are difficult to extrapolate to the landscape scale because additional processes take place at larger scales (Evans, 1995). However, Cerdan et al. (2010) used a simple extrapolation method for plot data across Europe to create a map estimating soil loss from sheet and rill erosion. In addition, run-off plot data were and are needed to calibrate erosion models. According to Boardman and Evans (2019), many models need to be better calibrated against real-world erosion monitoring data. A measurement-based monitoring using run-off plots is — for now — outside the scope of national- or EU-level monitoring, as the data obtained are mainly relevant for research on model development.

Table 7.4 Design for two-stage erosion monitoring in Europe

Monitoring level	Method	Result	Aim
Level 1 **Large-scale modelling**	RUSLE2015	Estimated soil loss rate by water erosion on arable land (t/ha/year) Estimated impact of management on soil erosion (C- and P-factor of RUSLE2015)	Pan-European information on soil loss rates by water erosion using a harmonised methodology; identification of soil erosion hot spots relevant to be monitored (level 2)
Level 2 **Monitoring at field to landscape scale** (focused on soil erosion hot spots identified in level 1)	Monitoring the appearance of soil erosion Standardised visual classification of the severity of each soil erosion event Standardised volumetric measurement of rill erosion	Frequency of soil erosion by water Severity of soil erosion Soil loss rate by rill erosion	Create ground-truth data on soil erosion by water using standardised field mapping methods Develop dynamic process-based models simulating the field- to farm-scale erosion processes

By contrast, level 2 of the proposed monitoring covers *in situ* soil erosion surveys on agricultural parcels, for example in hot spots, and such data provide ground-truth on the extent, frequency and severity of soil erosion by water. Comparable long-term monitoring programmes at field to landscape scale based on visual and volumetric measurements of water erosion have been described by Evans et al. (2016), Prasuhn (2011, 2020) and Steinhoff-Knopp and Burkhard (2018). Herweg (1996) and later Ledermann et al. (2010) and Boardman and Evans (2019) provide overviews of methods for assessing soil erosion at the field scale. In their reports the dimensions of rills and gullies are measured 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 are then added to obtain the total soil loss. Note that interrill erosion is difficult to estimate using this approach (significant measurement errors during field work can be expected). While such direct measurements are difficult to upscale, measurements are often combined with modelling; they are useful to develop, calibrate and validate predictions from modelling (Stroosnijder, 2005; Fischer et al., 2017).

The robustness of the data obtained from long-term monitoring schemes at field/farm to landscape scale is reduced by the necessarily simpler field-mapping methods (compared with run-off plots), but robust information on the extent, frequency and severity of soil erosion for larger areas can be collected. Monitoring at field to landscape scale also enables the capture of erosion processes not present in small-scale run-off plots (e.g. gullying). 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 across various European regions. Borrelli et al. (2022) showcase the LUCAS Soil survey, which offers a promising option to monitor gully erosion in Europe.

Because of the considerable variation in soil properties across large landscapes, arable land use systems (e.g. different tillage systems) and climate, a large and representative number of monitoring areas is required for level 2 erosion monitoring. Boardman (2006) suggests focusing on monitoring the effects of moderate to strong erosion events. Considering that about 70% of soil erosion occurs over only 15% of the total land area of Europe (Cerdan et al., 2010), hot spots play an important role when stratifying systematic inventories across large landscapes and regions.

An additional challenge for ensuring the representativity of monitoring areas is the dominance of different types of erosion in different regions in Europe (see Van-Camp et al., 2004a, table 2.3). The EEA (2000) conducted an analysis and mapping of areas threatened by soil erosion (hot spots) in Europe, in which broad zones with similar erosion processes were identified (hot spots map for water and wind erosion.)

Until now, (trans-)national programmes to monitor soil erosion by field surveys using a standardised procedure have been lacking. We suggest (1) the development of a standardised field survey method to monitor the extent, frequency and severity of soil erosion on agricultural land parcels based on current knowledge from existing long-term monitoring programmes and (2) the definition of monitoring areas (hot spots) in all countries that participate in such a coordinated and harmonised approach.

Based on soil erosion indicators, currently implemented across the EU and the proposed two-level monitoring approach, the following soil erosion indicators can be identified:

1. Indicators based on RUSLE2015:
   - Estimated rate of soil loss by water erosion (t/ha/year);
   - Management impact on soil erosion by water (RUSLE2015 C- and P-factors);
   - Area affected by intolerable soil loss rates estimated rate of soil loss by water exceeds tolerable soil loss rate ([28]);
2. Indicators from observation and measurement (monitoring): extent, frequency and severity of soil erosion by water in monitoring areas, based on visible and measured features in the field.

7.3 Critical limits

The Food and Agricultural Organization of the United Nations *Revised world soil charter* (FAO, 2015b) states that 'soil management is sustainable if the ... ecosystem services provided by soil are maintained or enhanced without significantly impairing either the soil functions that enable those services or biodiversity.' Since it is almost impossible to stop soil erosion completely, the concept of a tolerable soil loss rate is used to define the maximum acceptable level of soil erosion.

Various options for defining tolerable soil loss are described in the literature (e.g. Li et al., 2009; Verheijen et al., 2009), which have the following three main targets:

- maintain soil thickness/soil volume;
- maintain soil fertility (crop productivity);
- maintain the provision of soil-related ecosystem services.

To maintain soil thickness (i.e. soil volume) many authors suggest a strict steady-state concept (see table 3 in Verheijen et al. (2009)). Here, the tolerable soil loss rate is equal to the natural rate of soil formation to preserve soil thickness

([28]) The definition of site-specific tolerable soil loss rates is discussed in Section 7.3. Further research is needed to develop a reliable concept to define and estimate site-specific tolerable soil loss rates.

at a given point in time and space (Morgan, 1986). If soil loss remains below this threshold, soil management can be considered sustainable with regard to erosion, as the sustainable provision of soil-related ecosystem services, including crops, is safeguarded.

The implementation of this concept needs estimates of soil loss and soil formation rates. Natural rates of soil formation are variable and hard to measure. Soil formation rates found in the literature vary quite significantly, as early studies report rates of 0.05-0.5 mm per year (≈1t/ha/year) (Wakatsuki et al., 1992) or 1.4-2t/ha/year (Verheijen et al., 2009), while recent ones suggest that soil production can be 3.2-4.5t/ha/year (Egli et al., 2014). The relationship between soil formation and tolerable soil loss has recently been reviewed by FAO (2019). The study cites Montgomery (2007), who suggests using an average bulk density of 1.2g/cm² to convert a soil loss of 1t/ha to a loss of 0.08mm in depth. According to the same author, the soil naturally develops at an average rate of 0.173mm/year resulting in an increase of 2.2t/ha/year. Verheijen et al. (2009) used European data on soil formation to calculate a tolerable soil loss for Europe of 0.3-1.4t/ha/year (a reduction in depth of 0.02-0.11mm/year).

Implementing the strict steady-state concept of preserving soil thickness/soil volume to define a tolerable soil loss, rates of loss under 2t/ha/year are tolerable. Site-adapted or at least regionally adapted tolerable soil loss rates are needed to set reliable limits.

Maintaining soil fertility is a classic objective of agricultural policies and was an initial driver of soil erosion research. According to Li et al. (2009), Smith (1941) defined tolerable soil loss as 'the amount of soil that could be lost without a decline of fertility, thereby maintaining crop productivity indefinitely'. Wischmeier and Smith (1978) and Renard et al. (1997) include comparable definitions in the USLE/RUSLE handbook. They also give figures for tolerable loss rates identified in stakeholder workshops in the United States in 1959 and 1962, in the range of 2.24-11.21t/ha/year (converted from 1-5 t/acre/year). Renard et al. (1997) highlight the factors considered in defining tolerable soil loss: soil depth, physical properties affecting root development, gully prevention, in-field sediment problems, seeding losses, reduction in soil organic matter, and loss of plant nutrients. Often the tolerable soil loss rate to sustain soil fertility is combined with soil depth to define site-adapted thresholds: for example, in Switzerland, the maximum tolerable soil loss is 2t/ha/year for shallow soils of less than 70cm depth. For deeply developed soils, the threshold is 4t/ha/year (Schweizer Bundesrat, 1998, cited in Ledermann et al., 2008). A more refined approach, including soil depth as a proxy for soil fertility, was presented by Mosimann and Sanders (2004).

Based on developments in the soil function concept and the mainstreaming of the ecosystem services concept, recent research on tolerable soil loss rates focuses on the integration of soil-related ecosystem services (e.g. Steinhoff-Knopp et al., 2020). Further research is needed to develop methods to evaluate the impact of soil erosion on a wide range of ecosystem services across Europe.

As highlighted by Stolte et al. (2016), the establishment of potential thresholds for tolerable soil loss is still very controversial. There is a noticeable variation in terms of critical values but also lack of clarity on the definition of tolerable and critical soil losses. Further research should be encouraged to build a common, more solid basis for both of these aspects. Given the variability of soils and climatic conditions in Europe, threshold values for erosion rates should ideally be defined for different soil characteristics, land uses and climatic zones. Until such site-adapted limits for tolerable soil loss are developed, this report recommends, based on the limits applied in Switzerland, the following thresholds for tolerable soil loss rate: 2t/ha/year for shallow soils (<70cm depth) and 4t/ha/year for deeper soils.

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. This reduces hydraulic conductivity and rainwater infiltration, which reduces groundwater recharge, while increasing the likelihood of waterlogging. The last hampers rooting and soil biological activity. Consequently, soil biochemical processes are affected, including nutrient turnover and greenhouse gas emissions (nitrous oxide, methane). As a result, plant health, and thus food and fibre production, are adversely affected. Compaction is also known to trigger soil erosion. Operations at critical soil moisture levels, and the use of increasingly heavy machinery, cause compaction, which is particularly damaging in the subsoil. While topsoil compaction can be ameliorated through tillage and bioturbation from plants rooting and soil fauna burrowing, subsoil compaction is cumulative, often persistent and requires significant technical effort to alleviate. Therefore, subsoil compaction is often perceived as irreversible.

Soil compaction is primarily related to physical soil degradation, but interactions with chemical and biological properties and functions are evident (Table 8.1). Soil compaction occurs primarily if the internal soil strength (known as actual precompression stress) is exceeded by additional stress, for example from heavy machinery, dense trafficking and driving when soil moisture content is high. This exceedance results in plastic soil deformation, which negatively affects the soil functions and the provision of ecosystem services. Precompression stress indicates the soil's site-specific natural ability to bear and recover from external mechanical forces; it represents the condition in which soils are resilient and can be sustainably managed. Monitoring focuses on specific soil physical (functional) parameters that describe the mechanical behaviour of the soil.

Table 8.1 Relationship of soil compaction to key societal needs and soil functions

Societal need	Soil service	Impact
Biomass	Wood and fibre production	-
	Growth and quality of crops	-
Water	Filtering and buffering of contaminants, including supply of drinking water	-
	Water storage and availability, groundwater recharge, surface run-off and interflow	-
Air	Composition and exchange of soil gas with the atmosphere	-
Climate	Carbon storage and turnover, avoidance of greenhouse gas releases (e.g. nitrous oxide, methane)	-
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

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 (e.g. independent of the current soil moisture level), but also the operational conditions and limited knowledge of service providers appear to be additional pressures. An example of the need for specific knowledge are the risks of wheeling- and shearing-induced soil deformation, which depend on the kind of machinery used. The specific damage to soil is conditioned by the machine weight and contact area with the soil, the number of passes and the area covered, but also by shearing and soil smearing from wheel slip (Horn and Peth, 2011; Keller et al., 2019; Horn, 2021; Keller and Or, 2022). It is mainly the high wheel and axle loads of transport vehicles and harvesters that cause mechanical stress on a given contact area and which can exceed the resisting forces within soil: irreversible soil deformation and permanent compaction is the consequence — especially in the subsoil. Such effects are increased when soils are wet and weak and when field traffic efficiency is low (Duttmann et al., 2014). Repeated trafficking generally results in cumulative soil compaction to deeper soil depths and induces subsoil deformation of the pores and their functions if a given soil strength is exceeded by the applied stresses.

High livestock densities on pastures can also cause the exceedance of internal soil strength, particularly in the topsoil, and not only under wet soil conditions. This topsoil compaction is indicated by a less permeable platy soil structure, which in combination with shearing-induced puddling results in more intense soil deformation, initially up to about 30cm depth, but with a cumulative impact at even greater soil depths. Schroeder et al. (2022a) found that, based on the analysis of more than 500 soil profiles between 1980 and 2022, the formation of a platy structure in the subsoil increased, while the saturated hydraulic conductivity or air permeability decreased to values below a threshold. Furthermore, the vulnerability of highly fertile soil types such as luvisols during the last few decades has increased if physical soil properties such as hydraulic conductivity, air permeability or cohesion are considered.

Based on the European database 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 soil compaction. Graves et al. (2015) estimated the total annual cost of soil compaction in England and Wales at GBP470 million/year, corresponding to EUR540 million/ year (currency rate January 2019). Hence, the per hectare costs of soil compaction amount to approximately EUR140.2/year when related to the compaction-affected area, and about EUR56.4/ha/year on the basis of the total agricultural area. Other estimates suggest that between 32% and 36% of European subsoils are highly susceptible to compaction (Jones et al., 2012). Assuming an average yield loss of 4.5% (Graves et al., 2015) and that 35% of the arable land is affected by compaction (Oldeman et al., 1991; Graves et al., 2015; Schjønning et al., 2015; Brus and van den Akker, 2018), the value of yield losses for arable crops is estimated at EUR33 million/year. Eriksson et al. (1974) assumed yield losses due to compaction of 8% for soils with >40% clay, and 4% for soils with 15-25% clay, while yield losses for lighter soils were negligible. Considering significantly higher current machinery weight, yield losses may be higher nowadays, however, proper data are only available for small regions and are missing at EU-or European level (Keller et al., 2019).

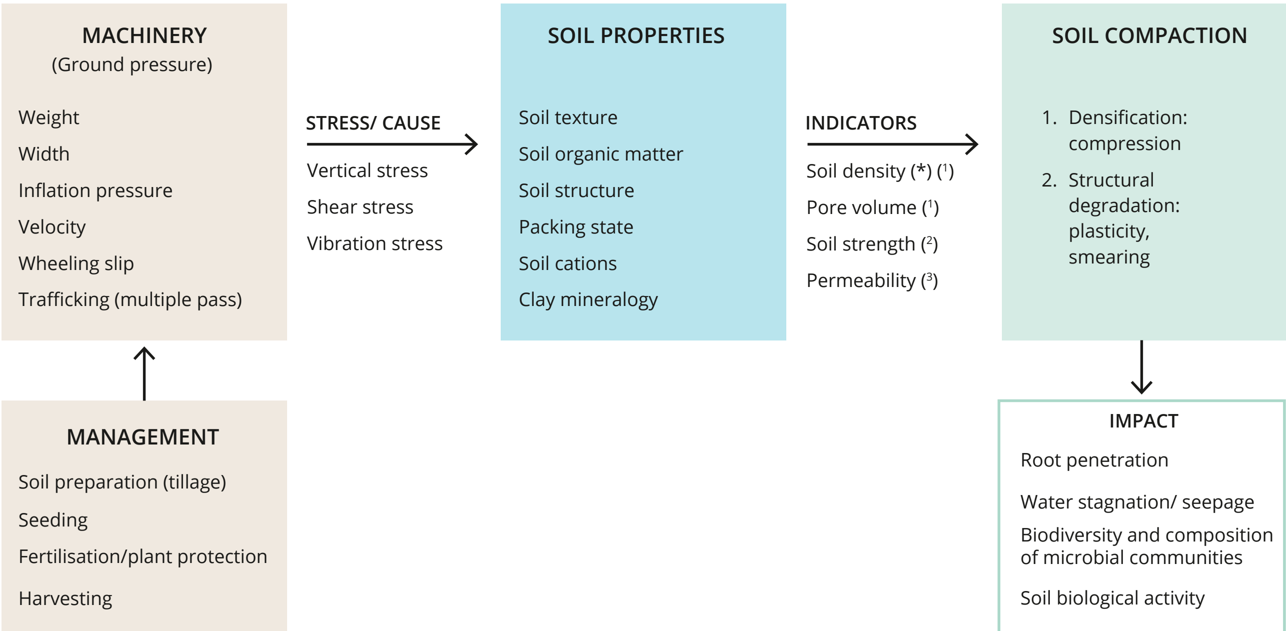
Mordhorst et al. (2020) quantified the compaction status of 342 soil profiles in northern Germany, including both natural and potentially anthropogenic compaction. Harmful subsoil compaction was determined in 20-40% of the area of (stagnic) Lluvisols and Stagnosols, of which at least 6-10% is caused by agricultural management; such degraded soils suffer from the lowest values for air capacity and saturated hydraulic conductivity. Van den Akker et al. (2013) calculated that about 43% of subsoils in the Netherlands are overcompacted, while, for the agricultural area in central Switzerland, Widmer (2013) estimates that about one third of the area may have critically high soil densities. Hakansson (1994) stated that, based on long-term wheeling experiments, a permanent yield decline of 5-10% must be considered, plus additional effects of climate change; the latter increases the uncertainty of obtaining high or normal yields.

Compaction increases the penetration resistance of the soil, while root growth and biological activity, like the frequency of earthworms, nematodes or collembolans, are reduced (Schrader, 1999; Gregory, 2006; Beylich et al., 2010; Horn et al., 2022). The extent to which these changes occur depends on the stress intensity and duration as well as the kind of stress applied (static or cyclic stress due to wheeling and induced shear effects). In addition, physico-chemical processes are affected, such as redox potential, mobility of ions and related effects on the pH value of the soil. Consequently, the microbial community composition can change from oxic to facultative anoxic to anoxic. Stress-induced formation of a platy structure favours horizontal fluxes in sloping areas, so that water erosion and stronger and higher floods can occur (Horn et al., 2019; see also reviews by Alaoui et al., 2018, and Van der Ploeg et al., 1999, 2002).

Trafficking and its effects on compaction receives increasing attention in developing solutions for 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 found in sugar beet or maize; in those crops, harvest involves more frequent trafficking with high ground pressures, at a time in late autumn when the soil water content is high, also at lower depths. Over the last 40 years, increases in stress-affected soil depth and decreases in rootability have been observed (Keller et al., 2019). Climate change impacts such as higher temperatures during the growing season in combination with less rainfall and reduced rootability will result in reduced crop yield or at least greater uncertainty in yield expectations. However, the cumulative effects of such impacts over the years must also be considered, for example reduced crop yields due to possible water shortages (Rulfová et al., 2017). The impact of these stresses depends on the soil's internal strength (defined as precompression stress), for which site-specific thresholds can be defined so that further deformation at higher levels of stress can be indicated and avoided (see Figure 8.1).

Without a doubt, soil compaction is a very serious problem, as it is not only associated with altered compositions of the soil phases (water, gas, solid) but it also affects the availability and accessibility of particle or pore surfaces for nutrient storage and carbon sequestration. The documentation of the soil compaction-induced decline in hydraulic conductivity or air permeability throughout the last four decades (Schroeder et al. 2022a, 2022b) is evidence for the long-term negative consequences of unsustainable soil management. The observed damage reduces the soil's resilience to climate change and enhances secondary effects, such as soil erosion and surface water pollution (Jones et al., 2003; Batey, 2009; Rogger et al., 2018; Horn, 2021).

Figure 8.1 Soil compaction processes, parameters and indicators



Notes: (*) Indicated by: bulk density, packing density, total porosity.
(1) Represent ENVSSO indicators (Explained in Huber et al. 2008).
(2) Mechanical resistence to failure under mechanical stress (soil mechanical strength): it is the maximun shear stress which soil can sustain; it increases with increasing soil bulk density, and decreases with soil-water and organic matter contents; silty texture tends to deform more easily, it is indicated by the soil's precompression strength and describes its compressibility.
(3) Determined by saturated hydralic conductivity.

Source: EEA.

8.1.2 Soil compaction processes

Observation of soil compaction and the definition and selection of the proper indicator(s) requires knowledge of the pressures on soil and its properties, and of the spatio-temporal responses in the soil (Table 8.2)

Table 8.2 Soil properties for use as indicators for soil compaction

Soil environment	Indicator	Soil property
Air regime	Air storage	Air capacity Bulk density
	Air flow	Air permeability Oxygen 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
Habitat for living organisms	Microbial composition	Diversity and community structure
	Abundance of functional species groups	Oxic/anoxic taxa and distribution (e.g. methanogens; sulphate-reducing bacteria or ectomycorrhizal fungi)
Physical soil regime: soil strength	Deformation status	Bulk density Proctor density ([a]) Average mean diameter of aggregates
	Stress strain ([b])	Precompression stress Crushing strength Shear strength Ratio of precompression stress to actually applied stress Stress propagation Changes in air, water, thermal flow processes and biological regimes due to stress strain and shear stress-induced distortion
Root functions	Rootability Nutrient availability	Root length and root surface density Penetration resistance

Notes: ([a]) 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.

([b]) Stress-strain describes the relationship between the effect of stress (machine weight and trafficking) on soil strength (syn. soil strain, 'change in size or shape'), indicated by an increase in soil particles per unit of soil volume and the change in pore structure (diameter and amount) (also called 'densification' or 'stiffness'). The comparison of soil strength and mechanical stresses (called the 'soil rigidity ratio') is used here as an indicator of soil compaction.

Sources: Ball et al. (1988), Beylich et al. (2010), Frey et al. (2011), Hartmann et al. (2014), Horn (2021), Horn and Fleige (2003, 2009), Jones et al. (2003), Keller et al. (2019), Langmaark et al. 1999), Lebert et al. (2007), Lebert (2010), Schjønning et al. (2003, 2016), Schrader (1999), Stepniewski (1980, 1981).

As primary, technically easy indicators, air and water storage can be derived and interpreted as showing the impact of soil compaction, in particular in the topsoil, while the external pressures and internal soil processes and their interrelation require more complex parameters and physical impact models, for example soil strength and/or deformation status. The strength of the soil indicates its capacity to resist stress; once that capacity is exceeded, compaction results, and this is then accompanied by visible changes in the physical, biological and physico-chemical soil properties and functions: in spring, when soils are wet, they are physically weaker and more susceptible to deformation by stress and wheeling-induced shear processes, particularly in the subsoil; during summer, soils dry out and the pore system is strengthened.

Soils have a natural range of strength, depending on:

- the parent material, and its physical (bulk density, texture) and chemical properties (soil organic matter content, calcium carbonate secondary oxides);
- the quality and quantity of the soil's reactive inner surface (cation exchange places);
- the soil structure based on its natural development (pedogenesis).

Compaction occurs when the applied stress overcomes the soil's natural strength and the mechanical rigidity limits of the soil's internal strength are exceeded: the soil 'fails'. This internal strength can be derived from 'stress-strain curves', which also define the precompression stress for soils under consideration (Horn and Fleige, 2009). This soil-specific relationship between stress and strain characterises the level of natural compression or stress prior to any compressed state. Only if this soil strength (defined by its precompression stress) is exceeded by the actual stresses applied do soil functions deteriorate. 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 observed not only by a loss of volume (soil subsidence) or increase in bulk density (densification) but also, and more importantly, by changes in sub-indicators that are also directly related to physical soil functions, namely:

- decreasing with compaction:
  - air permeability
  - gas diffusion
  - saturated hydraulic conductivity;
- increasing with compaction:
  - unsaturated hydraulic conductivity
  - heat flux

  [For comparability, the matric potential ([29]) is considered].

These sub-indicators depend on natural soil properties, including soil texture, soil structure, organic carbon and chemical properties, and can be either naturally low or high, or they can exceed acceptable values due to soil deformation. Physico-chemical parameters, such as redox potential, microbial composition and abundance are also altered (Horn, 2021); elevated greenhouse gas emissions from compacted soil (e.g. increased nitrous oxide or methane emissions) can be observed (Stepniewski, 1980, 1981; Ball et al., 1988; Haas et al., 2016; Horn et al., 2022). Compared to undisturbed soils, compressed and moist soils are colder in springtime while in late autumn they are warmer; both responses affect biomass growth, biological activity and carbon sequestration as well as nitrogen leaching.

Changes in soil strength often occur 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 with the applied stress (Horn et al., 2014). The extent to which these changes occur depends on the stress intensity and duration as well as the kind of stress applied (static = vertical loading or wheeling-induced shear and strain effects ([30])). Among other effects, plastic deformation and consecutive stress release 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 increases the risks of water erosion and flooding, especially in areas with prevailing fine-textured soils with a typically low infiltration capacity (Alaoui et al., 2018).

([29]) Soil matric potential indicates the soil water that is held by the soil matric (soil particles and pore space), and which becomes increasingly negative the finer the pore diameter. It also defines the plant available water range and the soil's air capacity or field capacity.

([30]) Static (vertical) loading results in three-dimensional soil displacement with a preferential dominance in the vertical direction; however, wheeling also induces lateral displacement and even tangential particle movement due to sliding. The latter causes the deformation or complete destruction of soil aggregates, a reduction in pore diameter and blockage or even complete closure of pores.

Beyond a threshold for soil strength, changes in soil properties are not or only partly reversible, and it takes decades of soil amelioration to rehabilitate soil functions. Any natural ameliorative measures to improve soil functions and crop yield require not only more sensitive (conservation) tillage management but also time. For example, natural drying induces crack formation and root penetration into deeper soil layers, and vertical pores are formed and inhabited by earthworms. It takes decades before soil structure is visibly improved as a result of such processes. So-called subsoiling, or deep ploughing, carries the risk of completely weakening the soil structure through homogenisation.

8.1.3 Topsoil and subsoil compaction

When the internal soil strength (precompression stress) is exceeded during wheeling, animal trampling or continuous loading, the soil is deformed down to the depth at which 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 20-35cm). However, while compaction in the topsoil can be mitigated through effective management (e.g. ploughing or chiselling) or through natural processes (e.g. soil biota activity, swelling and shrinkage), the damage to the subsoil is particularly relevant since, at these depths, compaction is cumulative and persistent over decades or maybe even centuries (Keller et al., 2019). Note that the often assumed 'curing' effect of freeze-thaw cycles are less effective in the subsoil (Hartge and Horn, 2016). Subsoil compaction is hence the main factor responsible for soil degradation, having a persistent impact on soil functions

8.2 Indicator specifications

8.2.1 Physical soil functional parameters and indicators

Indicators on compaction for soil monitoring were suggested by Huber et al. (2008) among others. In the absence of data on actual soil compaction, Huber et al. (2008) suggested spatially predicting the vulnerability of soils to compaction by (1) the actual water saturation or its binding forces within the pores (defined as matric potential), (2) the initial drainage condition and (3) the bulk density. However, such estimates provide only very rough information on where soils are overcompacted (Van den Akker et al., 2013). Therefore, parameters are suggested here that are sufficiently sensitive to quantify the degree of soil compaction and the consequent effects on soil functions.

The degree of topsoil compaction is difficult to clearly describe with thresholds because conditions are highly unstable and dynamic, for example the negative effects of mechanical seedbed preparation, followed by some recovery after the growing season, the use of cover crops, etc. The degree of topsoil deformation can thus be rather temporary; however, it can also be a warning sign that any continuation of current (harmful) practices is then likely to affect the subsoil. Topsoil compaction is primarily described using the parameters in set I, and can act as a warning sign that current practices do not sufficiently address the sensitivity of the soil. While topsoil compaction can be more easily alleviated, subsoil compaction must be completely avoided.

Parameter set I

The following section outlines parameters that can be easily measured or which are common in many soil surveys:

- bulk density, D_b;
- air capacity, AC;
- soil texture;
- visual features of compaction such as platy structure.

Bulk density
D_b defines a mass of dry soil material per unit volume. The values depend on texture, aggregation, organic carbon content, *in situ* water drainage and anthropogenic, geogenic or pedogenic processes. D_b is a parameter with high spatial and temporal variability. While bulk density (D_b) is compaction sensitive, it is nevertheless considered a rather unspecific parameter, because it describes only changes in volume but does not quantify the potentially negative impacts on pore functions. Thus, there is no direct link to soil strength or compaction. If bulk density is used because of its widespread availability in soil monitoring, additional (visual) information about, for example, texture, or soil structure is needed to gain a better qualitative judgement of compaction. Measurement of D_b can also be misleading because sampling in dry, strongly rooted and stony soils is difficult. Packing density is sometimes used instead of D_b. It is derived as a function of bulk density and clay content 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 that 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 (%) — is a measure of the degree of densification, which has a strong relationship with aeration and the 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, that 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 (1) comparing the current measurement with the initial measurement (as a reference value), (2) comparing the current measurement with an undisturbed site-specific value or (3) applying a threshold that can be expected for a specific soil (Wösten et al., 1999).

Visual or indirect soil evaluation

Spade diagnosis (VESS: visual evaluation of soil structure). VESS is a method for detecting changes in packing density ([31]); the method is described by Diez and Weichelt (1997, in German), and in more detail in Ball et al. (2017). The aggregate types and their arrangement can be described as a first indication of the ecological soil status by the visual analysis of cracks, their orientation and frequency, the actual aggregate shapes (e.g. coherent, subangular blocks or plates), and the aggregate surfaces (Babel et al., 1995).

Penetration resistance (penetrometer). The rootability of the soil is correlated with this indirect measurement. 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 in soils under conservation agriculture, especially under zero tillage, than for those under conventional management; soils become more rootable and macroscopically well aerated and at the same time mechanically very strong. Penetration resistance is best determined at 'field capacity'.

Parameter set II

This second set includes more complex soil physical parameters, which appear to have a strong dependency on the soil's actual water saturation and structure, as well as pedo-and anthropogenic processes. These indicators can be linked to the actual and dynamic gas, water and heat fluxes in soils, as they are sensitive enough 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 ratio = precompression stress (kPa)/actual soil stress (kPa);
- shear strength (kPa) (stiffness);
- hydraulic conductivity (K) (cm/d) and air permeability (K_l).

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 stabilising processes and of natural decompression (loosening such as bioturbation). It is derived from stress-strain curves as the 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 that it will suffer additional mechanical stress and long-term degradation of soil structure (Van den Akker et al., 1998; Horn and Fleige, 2003; Keller et al., 2019). The values for the precompression stress and the stress-dependent changes in these properties and functions are determined under laboratory conditions and often quantified when the soil is most sensitive (usually in early spring at matric potential values of pF1.8 = -60hPa), or when drying due to evapotranspiration reduces the soil water content (pF2.5 or -300hPa matric potential). The precompression stress, i.e. the soil strength, defines the threshold as a scale-dependent value for single soil horizons to bulk soils, soil distributions within a given geological origin up to country or continent scale, or, for example, for given land management practices. The pedotransfer functions for quantifying precompression stress are described in Horn and Fleige (2009) and Simota et al. (2005).

Contact area pressure

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

([31]) 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.

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 above 1.2 define rigid soil structure conditions with no compaction processes, while values below 0.8 define structure as irreversibly deformed. Values in between classify soil properties and functions as very susceptible to further soil deformation.

Shear strength

Shear strength or the stiffness of soil determines the binding forces between particles (texture) or soil aggregates that withstand rearrangement (= strain ([32])) due to smearing (also defined as slip). The pore functions within the soil will be affected by such particle arrangement.

Hydraulic conductivity (K) and air permeability (K_l)

The saturated or unsaturated hydraulic conductivity and the air permeability are sensitive parameters 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 and the air permeability also quantify the fluxes within the various pore diameters. The number of blocked pores that cannot contribute to mass exchanges affect the slip and smearing effects from densified aggregates. The primary soil data which are needed to derive K using pedotransfer functions, can be derived from existing databases such as regional or national soil mapping (Wösten et al., 1999; Ad hoc AG Boden, 2005; Simota et al., 2005).

To properly interpret the soil-related indicators of the two parameter sets, the external stresses applied by machines need to be monitored and set in relation to the internal soil parameters and the changes due to the applied stress. Combining soil strength and management-dependent pressure as an indicator allows us to define sustainability or resilience limits like those in Table 8.4 (see Horn et al., 2005; Horn and Fleige, 2011).

8.2.2 Suggestions for including compaction indicators in monitoring with different sampling intensities

Depending on the different sampling and analytical requirements of the indicators outlined above, different intensity levels for monitoring are recommended (Table 8.3). The latter allow the description of both topsoil and subsoil compaction, while the level of detail defines the degree of uncertainty and also the applicability of possible models or of scale-dependent pedotransfer functions. European, country-specific or local soil profile and management-dependent databases on soil strength and stress-dependent changes in physical, chemical, and biological properties and functions facilitate indexing the resilience and the performance of arable soils.

At level I, easily and commonly determined soil parameters are used to define the probability of soil compaction, while the application of more detailed measurement data appears at a higher tier (level II, likely to occur on fewer plots than 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 8.3 therefore provides an overview of the different levels of soil compaction monitoring. More detailed descriptions of key indicators are given above.

8.3 Critical limits

The issue of soil degradation due to compaction and deformation needs to be addressed in two ways:

([32]) Strain is a measure of deformation representing the displacement between particles at a given stress applied. It is defined as, for example, height change, void ratio.

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	In the field: hot spots with visible marks of compaction:		Representative sub-plots throughout a given field surrounding the plot centre
	e.g. reduced vegetation cover or growth, puddles	Proportion of affected area, e.g. per field, or per area around a representative observation point	
Direct and indirect monitoring of soil compaction	Morphological features (waterlogging, (platy) soil structure, rooting)		
	Precompression stress (estimated) ([a]) Soil rigidity ratio ([b])		Samples are measured at defined matric potential Contact area pressure of the machines and the actual contact area are determined
	Penetration resistance (PR) ([c]) (estimated with pedotransfer functions, PTFs)		Measurements of depth-dependent PR at a given matric potential
Basic soil physical parameters	Saturated hydraulic conductivity, air capacity, plant available water capacity (estimated with PTFs, e.g. Wösten et al. (1999), Schroeder et al. (2022b))	All basic soil physical parameters for PTF are measured	Tensiometer, sensors, actual soil sampling at defined depths Stress-dependent changes in the parameters are measured under in-field and under lab conditions
	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/$CaCO^3$ (measured — soil profiles)	
	Soil organic matter (measured)		
Biological parameters	Rooting estimated	Root density (measured)	
	Biological activity (bioturbation)	Diversity and community structure of soil microorganisms	
Depth	Soil surface, upper boundary of lower soil horizons (or simply topsoil and subsoil)	Refined depth classes/by genetic horizon	Depths of 40-45cm and 60-100cm
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 tyre widths of every vehicle, contact area
Seasonality of monitoring	Spring sampling (soil at field capacity)		Sampling at requested times throughout the year

Notes: ([a]) Precompression stress derived from PTFs for a given texture and aggregation, according 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 and confined shear tests are needed to determine both the internal mechanical strength and the shear strength of a given structured soil.

([b]) Ratio precompression stress/actual stress imposed by field traffic (see also Duttmann et al. 2014, 2022).

([c]) Establish reference sites from undisturbed, uncultivated sites.

- 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);

- by determining the ratio of incoming stresses to soil strength, and its effect on the physical, chemical and biological properties used to define soil degradation (Riggert et al., 2019).

To achieve both objectives, we suggest using the following parameters as indicators of compaction (see also Table 8.4):

- precompression stress;

- ratio of precompression stress to actual stress applied;

- air capacity;

- saturated hydraulic conductivity.

Although we focus on these four indicators, Table 8.4 sets out thresholds for both abovementioned sets of parameters. While the first set is based on easily measured or readily available soil data, the second set refers to well-defined physical units that are closely related to actual water saturation, soil structure, and pedotransfer and anthropogenic processes. The second set is therefore better suited to quantifying and documenting stress-induced changes in soil functions, such as water, gas and heat fluxes, as well as effects on biodiversity and physico-chemical processes such as changes in redox potential.

Table 8.4 Thresholds for soil physical parameters for detecting harmful subsoil compaction

Parameter	Explanation and thresholds	Soil sensitivity
	Parameter set I	
Bulk density	<1.2g/cm³ = very loose 1.2g/cm³ and 1.6g/cm³ = normal 1.6g/cm3 and >1.9g/cm³ = dense >1.9g/cm³ = very impermeable Based on DVWK 1997, 1998; see also Keller et al. 2019	Soils originating from clay > silt > sand; higher values are due to geological pre-stressing or anthropogenic impacts
Air capacity: air-filled pore volume	A low air capacity impairs root growth, reduces oxygen pressure in soil air and increases the formation of greenhouse gases. Below 5% air capacity at a soil matric potential of -6kPa, aeration or gas diffusion are mostly insufficient. With decreasing particle size, the pore volume increases, and soil aggregation and soil organic matter content increases. Values around 45% total pore volume are at least acceptable while those below 35% are generally defined as very critical irrespective of texture effects).	Soils originating from clay > loam > silt and sandy loam > sandy loess
Visual soil evaluations	Aggregate type and estimated bulk density Root growth/penetrometer Spade diagnosis The visual assessment of the soil as loose or dense based on aggregate size and strength, pore size and continuity, root density and distribution	Additional assessment for all soils
	Parameter set II	
Precompression stress (= internal soil strength)	Low precompression stress includes 'very low' (<30kPa) and 'low' (30-60kPa) internal soil strength, e.g. because of weak aggregation or wet soil conditions; soils are very sensitive to further deformation and decline in physical, biological and physico-chemical functions. At medium (60-90kPa) or high (90-120kPa) stress levels, sustainable soil management practices are especially necessary and effective.	All soils, but especially loamy, silty and clayey soils
Ratio of precompression stress to 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 deformed. Values in between classify soil properties and functions as very susceptible to further soil deformation.	All soils, but especially loamy, silty and clayey soils, at high water contents and weak levels of aggregation
Shear strength	Shear forces due to wheeling result in smearing: the shear strength is lower for less aggregated soils and decreases with increasing water content. Shearing is more pronounced at higher levels of wheel spin, especially when soils are moist.	All soils, but especially loamy, silty and clayey soils, at high water contents and weak levels of aggregation
Saturated/unsaturated hydraulic conductivity	Low conductivity is typical for stagnic soil conditions: delayed percolation reduces soil aeration and groundwater accumulation and increases surface run-off (critical values are defined as below 10cm/day).	
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. Critical values for air permeability <1µm² Diffusion coefficient (Ds) $<1.5\times10^{-8}/m^2/s$ (Bakker et al., 1987) or relative diffusion <0.005 for loamy soils and 0.02 for sandy soils.	All soils, but especially loamy, silty and clayey soils, at high water contents and weak levels of aggregation due to tillage or soil management

Sources: Arah and Ball (1994), Babel et al. (1995), Ball et al. (1988), Frey et al. (2011), Huber et al. (2008), Lebert et al. (2007), Stepniewski (1980, 1981), Schjønning et al. (2003, 2016); supplemented with additional information from 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 (including depth) and time. Apart from pragmatic tools using indicators that can be determined or obtained from national and internationally available databases, the actual soil functions' behaviour in response to mechanical stresses can be best assessed using *in situ* and laboratory measurements or derived from pedotransfer functions, which are available at regional, national or European levels: for example German method catalogue (Ad hoc AG Boden, 2005; Schroeder et al., 2022a, 2022b); hydraulic properties of European soils (Wösten et al., 1999). They can be used as input parameters for process-based models, which include more detailed mechanical properties.

8.4.1 Soil compaction models including pedotransfer functions

The prediction of soil compaction and shear-induced soil deformation using modelling is based on the current or assumed land use, the soil properties and climate. Several approaches and models can generate mechanical soil properties and related soil processes and functioning based on common soil physical parameters and indicators. Table 8.5 provides an overview of the most common models.

To generate meaningful and site-specific input data for models, well-defined parameters for local soil properties and in situ matric potential data need to be collected. Shear strength is then derived from pedotransfer functions based on soil texture, soil structure and the matric potential (related to the cohesion and angle of internal friction). At best, a minimum data set contains values for soils when highly sensitive (saturated with water in spring) and in drier conditions in summer (Horn and Fleige, 2003; Horn et al., 2005; Schroeder et al., 2022a, 2022b; see also www. soilcompaction.eu). The better differentiated site-specific and representative input parameters are, the more reliable modelling results become. Samples need to be taken for a representative number of sampling sites, at best more than 30 soil profiles down to 1m depth, while Schroeder et al. (2022a, 2022b) document a complete data set (level II, Table 8.3) based on more than 500 complete soil profiles including mechanical stress strain, shear stress relations and stress-dependent impacts on soil functions such as hydraulic conductivity and air permeability. This particular data set has been built over the four decades between 1980 and 2022, so that the impact on and trends in soil functions related to soil management can be predicted.

Schjønning et al. (2020) and Schjønning (2021) developed regression models of topsoil shear strength based on *in situ* measurements, while Imhoff et al. (2015) developed pedo-transfer functions of water retention curves and soil resistance to penetration, which consider plant growth by estimating the least limiting water range and critical bulk density.

The finite element coupled process model (FEM) requires well-defined soil mechanical data such as bulk modulus (i.e. a mechanical parameter describing homogeneous soil mechanical behaviour, e.g. elasticity), shear modulus or shear strength, which need to be site specific, either derived from very sophisticated triaxial tests or derived from stress strain and shear strain curves. The model predicts stress distribution as a function of soil strength as well as soil deformation and changes in pore continuity due to stress propagation.

The following models are all restricted to predicting soil stresses under wheel loads including three-dimensional stress propagation. However, the impact of the applied stresses on soil functions are usually not considered. Further soil compaction models with different boundary conditions (Socomo versus Terranimo) allow the prediction of stress impacts on soil properties; Terranimo includes precompression stress, is more advanced, and can also be applied for different spatial scales and for different kinds of farm machinery.

Table 8.5 Models to predict subsoil compaction

Model	Content	Sources
FEM (finite element method) 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 (2009)
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 tool for practitioners for site-specific data analysis of given soil properties and mechanical impacts	Stettler et al. (2014) www.soilcompaction.eu www.terranimo.dk
Scale approach	Combination of the field traffic model FiTraM and the spatially explicit soil compaction risk assessment model SaSCiA	Duttmann et al. (2022)
Pedotransfer functions	Regression models used to quantify the interaction between general soil properties and the dependent variable, e.g. precompression stress, as well as the stress-induced change in physical properties and functions such as air capacity and hydraulic conductivity	Horn and Fleige (2003, 2009) Schroeder et al. (2022a, 2022b)
	Prediction of the topsoil shear stress and stress distribution	Schjønning (2021), Schjønning et al. (2020)
	Pedotransfer functions of water retention curves and soil resistance to penetration	Imhoff et al. (2015)

8.4.2 Compaction verification tool

To evaluate the actual soil stability and the risk of stress-induced soil degradation, Zink et al. (2011) developed the compaction verification tool (CVT), which includes stress-dependent changes in soil functions described as indicators in Figure 8.2. The tool is based on measurements or estimates of saturated hydraulic conductivity (K_s) and air capacity (at -60hPa) as a function of actual stress applied within the virgin compression stress range (see also Tables 8.3 and 8.4). Suggestions for quantifying 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 >5%, K_s >10cm/day) represent soils that still function properly, assuming that the rigidity limits (precompression stress) are not exceeded and/or the texture, organic carbon content, etc., guarantee these values. The values in class II (air capacity >5% and K_s <10cm/day) and class III (air capacity <5%, K_s >10cm/day) define the 'precaution value' indicating intermediate compaction risk (no harmful compaction yet), while values for air capacity <5% and K_s <10cm/day in class IV are associated with a decline in yield due to lack of aeration, prevention of gas exchange and/or stagnant water problems and correspond to 'action values' (indicating unacceptable, harmful subsoil compaction).

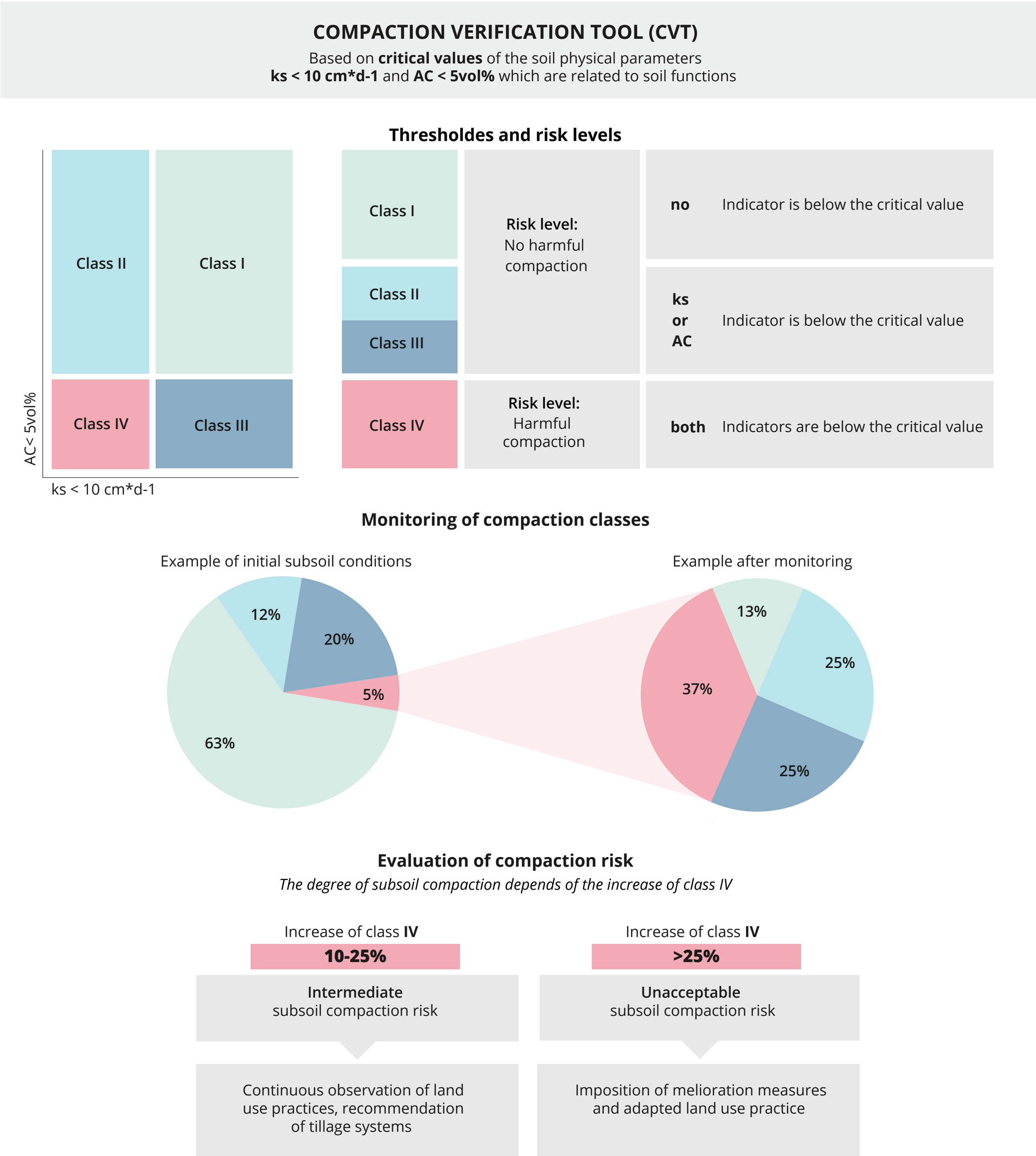
To promote sustainable soil management practices in agriculture and forestry, in particular to protect soils from degradation from the actual tillage systems used, tree harvesting and machinery impacts at given water contents, CVT was developed and built as a traffic light system: 'good' (class I) and 'acceptable' (classes II and III) (Riggert et al., 2019). The tool profits from modelling approaches as presented above (e.g. Terranimo), which describe the applied stresses in relation to soil strength.

It is likely that the monitoring will be applicable at all scales, and the necessary data on air capacity and saturated hydraulic conductivity are available for representative soil profiles (monitoring level III) or can be derived from existing databases (e.g. Wösten et al., 1999, or national soil mapping data sets) while the corresponding precompression stress data as threshold values can be derived from pedotransfer functions (Horn and Fleige, 2009) or detailed *in situ* measurements (level III). The quantification of stress implications for the two soil indicators (air capacity and hydraulic conductivity) beyond the precompression stress requires pedotransfer functions (Horn and Fleige, 2003, 2009) or site-specific measurements in combination with wheeling experiments (level III).

While the CVT allows mapping of harmful subsoil compaction, anthropogenic subsoil compaction still needs to be separated from natural compaction as a result of geogenic and/or pedogenic processes. Soils with stagnic or fluvic properties tend to have a high degree of natural compaction (46-65% of the fields mapped in a German case study area, compared with <13% of podsols and arenosols; Mordhorst et al., 2020). Anthropogenic compaction is found if the selected compaction-sensitive parameters — air capacity and saturated hydraulic conductivity — are larger in the subsoil than in the topsoil or if the saturated hydraulic conductivity in soil horizons with originally pedogenetically greater values now have values smaller than those in the deeper and more dense soil horizons (e.g. in stagnic luvisols and luvisols). The successful application of this threshold thus requires a horizon-specific analysis (topsoil/subsoil), calibrated with the knowledge from a large regional pedological database (level I and/or level II in Table 8.3).

The extrapolation of site-specific properties and stress impacts on changes in soil functions (as defined above) from wheeling impact has to be validated. Duttmann et al. (2022) offer such an approach by combining a newly developed field traffic model with a spatially explicit soil compaction risk assessment model. The risk assessment is based on the CVT approach and considers changes in mechanical properties as they depend on the matric potential (Rücknagel et al., 2015). With the help of GPS (global positioning system) data recorded by all farm vehicles involved in tillage, spraying and harvesting, the data serve to map wheeling intensity, and allow the spatially explicit mapping of scenarios for different wheel loads and predicted contact area stress. These data can subsequently be used for modelling soil compaction risk. Coupling the two models, FiTraM and SaSCiA, allows estimation of the spatially distributed soil compaction risk in the topsoil and in the subsoil, and even considers single field operations; it can also spatially specify the actual soil compaction and deformation status.

Figure 8.2 **Diagnosis of soil compaction based on threshold exceedance**



Source: Zink et al. (2009).

8.4.3 *Implications for monitoring*

The analysis of soil compaction and deformation status in Europe from the farm to country scale can be achieved irrespective of the model applied or the main soil properties and hydraulic functions such as pore size distribution and saturated hydraulic conductivity. These properties and functions are measured by some monitoring campaigns but can also be derived for the main soil types using pedotransfer functions. Information about input parameters, such as soil texture, soil structure and basic physical soil properties and functions, can also be derived from either soil monitoring or soil typological databases (e.g. derived soil parameters for soil mapping units. An enlarged European soil monitoring data set (e.g. representative national soil monitoring in LUCAS Soil) would further improve the accuracy of predicting soil compaction.

9 Soil sealing

Soil sealing is the destruction or covering of the soil by an impermeable material; this corresponds to an irreversible loss of soil and its biological functions and a loss of biodiversity. Between 2006 and 2015 the average annual soil loss due to soil sealing amounted to 429km² in the territory of the 38 EEA member and cooperating countries and the UK. Since the turn of the century, annual soil loss in Europe has ranged between 300km2 and 500km². This chapter presents the indicators available and discusses the implications of baseline and threshold definitions of soil sealing. In contrast to all other soil quality indicators presented in this report, baselines and thresholds for soil sealing are not soil science based but rather policy based (e.g. in relation to the 'no net land take' target).

Soil sealing fulfils the societal need for infrastructure but has negative impacts on all other societal needs and soil functions (Table 9.1).

Table 9.1 Relationship of soil sealing to key societal needs and soil functions

Societal need	Soil service	Impact
Biomass	Wood and fibre production	-
	Growth of crops	-
Water	Filtering of contaminants	-
	Water storage	-
Climate	Carbon storage	-
Biodiversity	Habitat for plants, insects, microbes, fungi and any microfauna	-
Infrastructure	Platform for infrastructure	+
	Storage of relocated material or artefacts (excavated geological material, sediments, cables and pipelines, archaeological material)	+

9.1 Rationale and status of soil sealing

9.1.1 Definition of soil sealing and imperviousness and their relation to land take

Imperviousness describes the covering of the soil surface with impermeable materials. Such areas are then incapable of being penetrated by air and water; thus, it describes by definition 100% sealing. In practice, soil sealing and imperviousness are used synonymously.

Soil sealing refers to the destruction or covering of soil by buildings, other constructions and layers of impermeable artificial material (asphalt, concrete, etc.). Sealed land is a subset of land take, i.e. land consumed by the development of settlements, infrastructure, and commercial and industrial areas. It is the most intense form of land take and is essentially an irreversible process (Prokop et al., 2011).

Imperviousness is a technical term used in remote sensing, in particular in the Copernicus land monitoring programme. An impervious surface reflects defined wavelength ranges differently from natural soil. Impervious surfaces can be artificial (anthropogenic) or of natural origin, for example rocks and glaciers.

Sealing is considered an irreversible damage to soil, since artificial surfaces are usually maintained for long periods of time, while the soil's natural physical structure and its chemical and biological capabilities have been deeply disturbed.

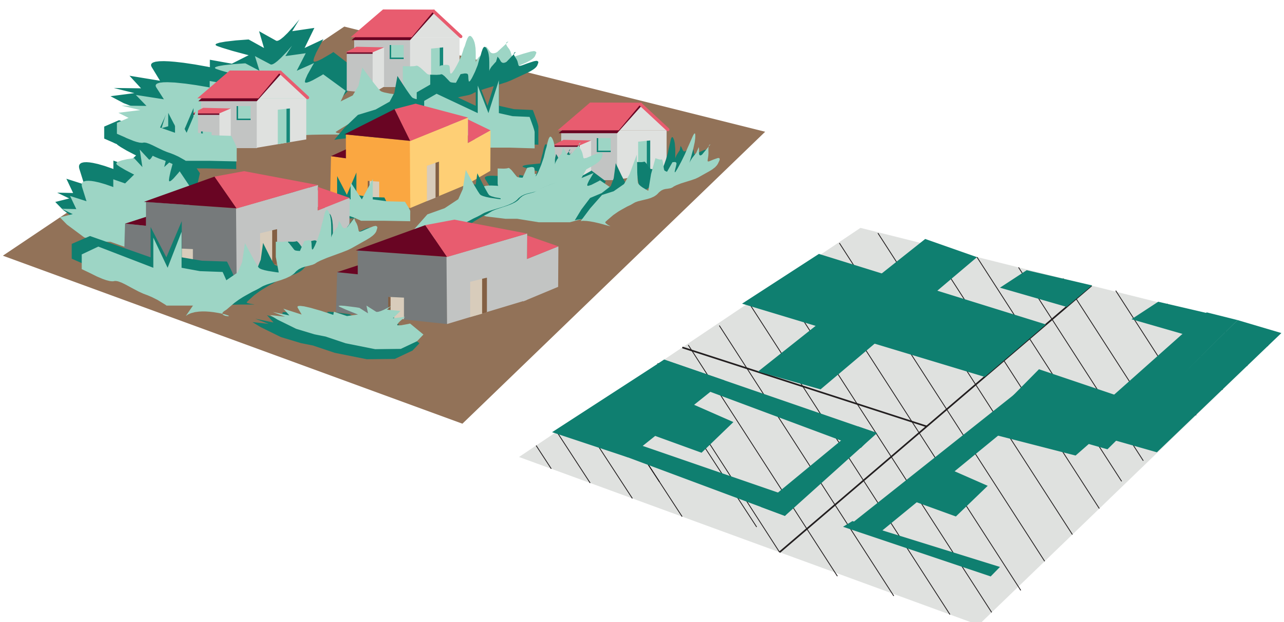
Soil sealing and land take go hand in hand: both are indicators of land degradation, which is accompanied by the loss of natural soil functions and ecosystem services (EEA, 2021a).

Land take (synonyms land consumption or artificialisation) can be defined as the increase in artificial areas over time and represents an increase in settlement areas (or artificial surfaces), usually at the expense of rural areas. This process can result in an increase in scattered settlements in rural regions or in an expansion of urban areas around an urban nucleus (urban sprawl). A clear distinction is usually difficult to make (Prokop et al., 2011). While land take largely occurs in urban areas, it also concerns rural areas to a certain extent (settlements, infrastructure). Land take indicates unsustainable land use, since it is usually realised at the expense of cropland or grassland, and in some cases also forest land, and these land uses guarantee important landscape functions (food security, recreation, climate balancing, etc.).

According to ETC/ULS (2019), artificial surface includes:

- urban fabric (continuous and discontinuous): private homes (including scattered agricultural buildings and cottages) and public buildings, including their connected areas (associated land, approach road network, car parks);
- industrial, commercial and transport areas;
- mines, dumps and construction sites;
- artificial non-agricultural vegetated areas (urban leisure parks, sport and leisure facilities).

Figure 9.1 The relationship between land take (left) and soil sealing (right, hatched surfaces)



Source: EEA (2021a).

Land take may also include parcels or surfaces that are not sealed (e.g. urban green areas, sport and leisure facilities) (Figure 9.1). Nevertheless, because both indicators estimate similar processes, there is a significant overlap in the areas affected by both land take and soil sealing. Sealing rates are usually lower in peri-urban areas with on average 10% sealing, and very high in core cities with on average 36% or more (Naumann et al., 2018). In Austria, the Federal Agency for Surveying and Mapping (BEV) found that sealing accounts for 32% of land take, and in Germany about 46% of the area consumed by land take is actually sealed (example cited from Prokop et al., 2011).

9.1.2 Status of soil sealing in Europe

Results from European land monitoring

For 2018, a European soil sealing layer with a precision of 10×10m, based on satellite data, is available. Earlier data sets for the years 2006, 2009, 2012 and 2015 are also available but with a lower resolution of only 20×20m. Therefore, the latest data set from 2018 cannot be compared with earlier data sets.

According to the latest and most precise data set, soil sealing in the EU and the UK amounted to an area of 97,903km^2 in 2018 and on average 192m^2 per inhabitant (EEA dashboard, 2021a). The sealing rate in floodplains amounted to 3.2% and in coastal regions to 3.9%. EU capitals were on average 50% sealed in 2010 (EEA, 2010). New sealing most commonly affects the soils on the outskirts of urban centres. Between 2012 and 2018 about 1,200km^2 of high and medium productive soils were lost to urbanisation (EEA, 2021a).

Results from national reporting on land take and sealing

In the context of the Land Use Land Use Change and Forestry (LULUCF) Regulation, all EU Member States report on conversions between different land use categories, including conversions to 'settlements', but also report on the trends in carbon pools within each category. Land cover is also reported under the United Nations Convention to Combat Desertification (UNCCD) 2018-2030 strategic framework, covering three indicators: trends in land cover, trends in land productivity or functioning of the land, and trends in carbon stocks above and below ground. Based on the UNCCD reporting for 2018 (which includes Sustainable Development Goal (SDG) indicator 15.3.1), 19 of the 27 EU Member States provide national reports, and 15 Member States have reported statistics using at least one of the three indicators. 'Artificial land cover per capita', later renamed to 'Imperviousness change rate', is also part of the EU SDG indicator set (though reported with EU-wide Copernicus data).

Prokop et al. (2011) compiled national reports on land take and sealing across the EU, but the systematic reporting of national data has not yet been established. For example, in Belgium, land take increased by 30% between 1985 and 2009, and, in Brussels, the percentage of sealed soil increased from 18% (1950) to 37% (2006). In Italy, the soil sealing rate has not decelerated in recent years, despite soil sealing limits being established at the municipality level:

- minimum values for the extension of permeable green areas, ranging from 15% in the town centre to 35% in residential areas (Brescia);
- 'surface permeability' according to land use classes, i.e. 30-40% permeability in residential areas, 70% for parking areas and 90% for green public areas (Padua);
- minimum standards for 'surface permeability' are 75% for private gardens and 15-50% for commercial areas (Parma).

BAFU (2020) presents an interesting case study for Switzerland:

- 7.5% of the national territory is artificialised, of which only about two thirds is officially zoned as building land.
- 4.7% of the national territory is actually sealed and hence devoid of biological soil functions.
- The area of sealed soils increased by 29% between 1985 and 2009, mainly accounted for by cropland and natural pastures. In the same period sealing also increased in wetland areas by 10% and in nature protected areas by 14%.
- The effect of sealing is larger than the area occupied by the actual construction, because additional adjacent terrain is affected by heavy compaction due to level infrastructure such as ramps, access roads, construction roads and agricultural barns.

9.1.3 Impact of urbanisation on the physical nature of soils

Urbanisation affects soils in various ways:

- Soils can be fully or partially covered. Removal of vegetated and biologically active topsoils is often the initial technical step of land take ('cut-off' soil profiles). What is often left, or imported from surface mines for construction, is the geogenic parent material, which then becomes compacted.
- 'Artificialised' soil in many city centres contains artefacts and debris from construction and waste (even after re-cultivation, such as in parks and green alleys). Such soils show irregular mixing, deep perturbation and sedimentation of artefacts, as well as a high degree of compaction.

- Soils can then also be completely sealed (asphalt) or covered with more or less impenetrable surfaces.

Based on Reto et al. (2006), urban soils can be classified as follows:

- local natural soils, such as recent, relict and fossil soils;
- traffic route soils (e.g. railway track ballast and paving cracks, where soil is formed by dust infiltration between the stones and sand);
- raw soils: soil on recent backfills or covered rocky layers;
- cultivated soils in gardens (hortisols), cemeteries (necrosols) and sewage farms;
- soils with gaseous or dust emissions/depositions (dry and wet deposits, dust infiltration and dust blown away);
- technogenic substrate soils;
- soils with thick and/or deep stratification, mixtures and compaction.

An indication of a soil consumed during land take is the change in its physical nature, i.e. the presence/mixing of artificial materials ('artefacts', such as building material, waste). Through land take and sealing, critical ecosystem services are lost or harmed, such as groundwater recharge, groundwater and surface water quality, agricultural production, biodiversity and recreation (EEA, 2021a). A detailed overview of the properties of different kinds of artificialised soils (soils affected by land consumption but not sealed) is provided by Cornu et al. (2021).

9.1.4 Sealing and ecosystem services

Depending on the degree of soil sealing, the ecosystem services provided by soils are affected to various degrees. If an area is completely sealed (100% sealing rate) all ecological soil functions, including the following, cease to be available:

- **Availability of nutrients.** The degree of imperviousness changes the allocation and accumulation of nutrients in soils. With increasing permeability, the availability of carbon, nitrogen and phosphorus also rises (Noe and Hupp, 2005; Pouyatet al., 2006; Pickett and Cadenasso, 2009).
- **Below-ground biological activity and diversity.** With increasing degree of sealing, soil microbial biomass carbon and nitrogen is reduced (Zhao et al., 2012). Urban green soils are especially important: some of them can still have an active soil life (soil fauna) like that of agricultural soils or forests (Ungaro et al., 2022).
- **Gas and water exchange.** In impervious soils, the exchange of gas, water and nutrients between the soil and other environmental compartments is heavily disturbed and most inhibited when fully sealed (Zhao et al., 2012).
- **Flood resilience.** Several authors confirm increased run-off due to increases in impervious surfaces: doubling over a 63-year period in Leipzig (Haase and Nuissl, 2007); 12% increase in run-off with a 12.6% increase in the area of sealed soil in Leeds (Perry and Nawaz, 2008). This increases the risk of flash floods following intense rain events (which are expected to become more frequent as a result of climate change).
- **Food security.** In the period 1990-2006, 19 Member States lost a potential agricultural production capacity equivalent to 6.1 million tonnes of wheat (Gardi et al., 2015).
- **Carbon cycle.** A large proportion of artificialised soils on allochthonous materials have very low soil organic carbon (SOC) content (especially soils related to road infrastructure and mining (Cornu et al., 2021). The importance of SOC as an ecosystem service for the city of Berlin has been demonstrated by Richter et al. (2020).
- **Human health.** Green urban areas with functioning soils contribute to cooling and air exchange in urban centres.

However, in reality, in a mosaic of different constructions, spatial arrangements and materials, sealed soils can still maintain some functions, for example the storage of water after infiltration from adjacent permeable fabric (Morgenroth, 2013). Porous pavements allow both higher infiltration and higher evaporation of water and have significantly cooler surfaces than fully sealed surfaces. Permeable pavements (e.g. concrete pavers with voids) and porous pavements (which are permeable over their entire surface) are increasingly used in spatial planning, as such materials help to mitigate the impact of paving on water and carbon cycles.

Urban soils and their ecosystem services receive increasing attention in sustainable urbanisation. The urban fabric represents a small-scale patchwork of very contrasting soil features, creating high short-term spatial variability (Vasenev et al., 2014). In some regions and countries, soil function evaluation has become an obligatory part of spatial planning, in particular when natural or productive soils are converted into building land. A mature soil evaluation procedure at the municipal scale (but also larger scales) for precautionary soil protection has been developed by the TUSEC project (Technique of urban soil evaluation in city regions — implementation in planning procedures; see details in Lehmann et al., 2013). As part of the URBAN SMS project, a handbook was developed that introduces measures for enhancing soil function performance and compensation for soil loss caused by urbanisation (Siebielec et al., 2010).

9.2 Indicator specifications

9.2.1 *Soil sealing and land take*

Soil sealing is usually calculated as a percentage (sealed area per total area) or as sealed area per capita for a given region or country. It can also be specified as square metres of sealed area per square kilometre of total area (applying a specific stratification) (see Table 9.2). As an indicator, soil sealing change, or imperviousness change is used, since it is the change rate that is of interest for spatial planners and policy commitments targeting no net land take.

Table 9.2 provides an overview of indicators relevant for monitoring soil sealing.

Table 9.2 Indicators used to assess soil sealing and land take

Indicator	Source	Explanation
Imperviousness	EEA Land and soil indicator set (LSI002) per area per capita (stratified)	European data set used in combination with other assessments, i.e. fragmentation, land recycling Example stratification: urban centre, peri-urban, rural
Land take	EEA Land and soil indicator set (LSI001) 8th Environment Action Programme headline indicator set, SDG target 15.3 Stratified, e.g. by major land cover category, urban protected areas, functional urban areas, urban floodplains	Used to report land take under the European and global environment programmes
Ratio of soil sealing to land take	Quality and quantity of sustainable land use and urban redevelopment	1 = total loss of all soil ecosystem services (except soil as a carrier for construction) 0 = no soil function is affected
Land recycling	Includes recultivation	
Sub-indicator	**Source**	**Explanation**
Cc/Co ratio ([a])	Tested in three Italian peri-urban areas, using multi-temporal SOC stock maps: SOC in built-up land and in natural soils (Cc/Co)	0 = unsealed land ≤0.5 low-intensity sealing 0.5≤2.0 medium-intensity sealing >2.0 high-intensity sealing
Sparse urbanisation prevalence index ([a])	Ratio of low-intensity sealed land to severely sealed land	Degree of anthropogenic sealing
Sprawl/densification ratio	Ratio of new sparse urban areas to urban densification area (trend in soil sealing change)	<1 tendency towards more compact urban forms >1 tendency to more diffuse urban forms
Share of undeveloped building land		>50% of cities ('population ≥50,000') have only a little (<10%) building land left while smaller towns have a larger buffer

Note: ([a]) Ratio between C in built-up land and organic C in soils (Cc/Co), according to Villa et al. (2018).

Land take in Europe is currently monitored based on Corine Land Cover (CLC) data. It looks at specific types of land cover change, i.e. the loss of agricultural, forest and other semi-natural and natural land towards urban and other artificial land use. CLC monitoring was initiated in 1985 (reference year 1990), and updates were produced in 2000, 2006, 2012 and 2018. The indicator LSI001 is part of the EEA Land and soil indicator (LSI) set. The CLC change data are produced with strong visual control of each spot, considering a 5ha minimum mapping area. While the CLC change map is very reliable, any land cover change of less than 5ha in size is not captured.

The last known update of land take was provided by the EEA in 2021, amounting to 539km2/year between 2012 and 2018; data are available on the EEA's land take dashboard (EEA dashboard, 2021b) and in an indicator assessment (EEA, 2019d).

The indicator on imperviousness and imperviousness change (EEA, 2020) is generated mostly through automated image classification of high-resolution satellite imagery, for the years 2006, 2009, 2012, 2015 and 2018; for 2018, only a status layer was generated, while, for the trend between all other monitoring intervals, a change layer is available (EEA 2020). LSI002 has a 20m resolution which is significantly higher than that of a CLC change map; the production algorithm builds on the correlation with a vegetation index. Impurities (thus uncertainties) are caused by seasonal changes in vegetation cover and the fact that not all non-vegetated areas are fully artificially sealed.

Details (interactive maps, data download, dashboard) of the Copernicus high-resolution layer on imperviousness and imperviousness change can be found on the EEA's website ([33]) and in references (EEA, 2019d; EEA dashboard 2019, 2021a).

The EEA indicator 'Imperviousness in Europe' (EEA, 2020) has been widely used as a soil sealing index (impervious soil coverage). The following sections describe the most common two methods to measure soil sealing. Both have their limitations, and it is therefore advisable to combine them.

9.2.2 Other indicators

The ratio between soil sealing and land take allows monitoring of the quality and quantity of sustainable land use and urban redevelopment. A low ratio indicates 'green' and extensive urbanisation or urban sprawl, which is currently typically happening in the urban fringes, whereas a high ratio indicates intensive, concentrated urbanisation (in urban centres), characterised by a high sealing rate.

Land recycling is based on the idea that land, once it has been artificialised and taken, must not be abandoned but should be reused as far as possible; it thus specifies that the land should return to non-artificial land categories (recultivation, or reverse land take). In order to understand the potential for land recycling, or the relationships between population growth in urban zones and sealing, i.e. in order to become land resource efficient, we need to know the share of the yet undeveloped building land ('building land stock') (see Table 9.2). Reliable measurements of the potential for land recycling in Europe are still unavailable but it is estimated to be high. Figure 9.2 depicts the three key types of land recycling: urban densification, grey recycling, and green recycling. Urban densification refers to construction on gaps between buildings and the increase of population density; grey recycling involves the construction of buildings or transport infrastructures on already developed land; green recycling is the development of green urban areas (EEA 2021a).

A European data layer for land recycling exists for the period 2006-2012 for 662 functional urban areas (Copernicus Urban Atlas), being core cities and their surrounding commuting areas (EEA, 2018).

Soil sealing and loss of productivity

The pattern of land use types is traditionally related to productivity classes: fertile land is cropped, while grassland and forests are managed on less fertile land or that with difficult topography. Fertile soils are very prone to land take and soil sealing, as they are situated in flat areas, where, historically, cities have emerged (EEA, 2021a).

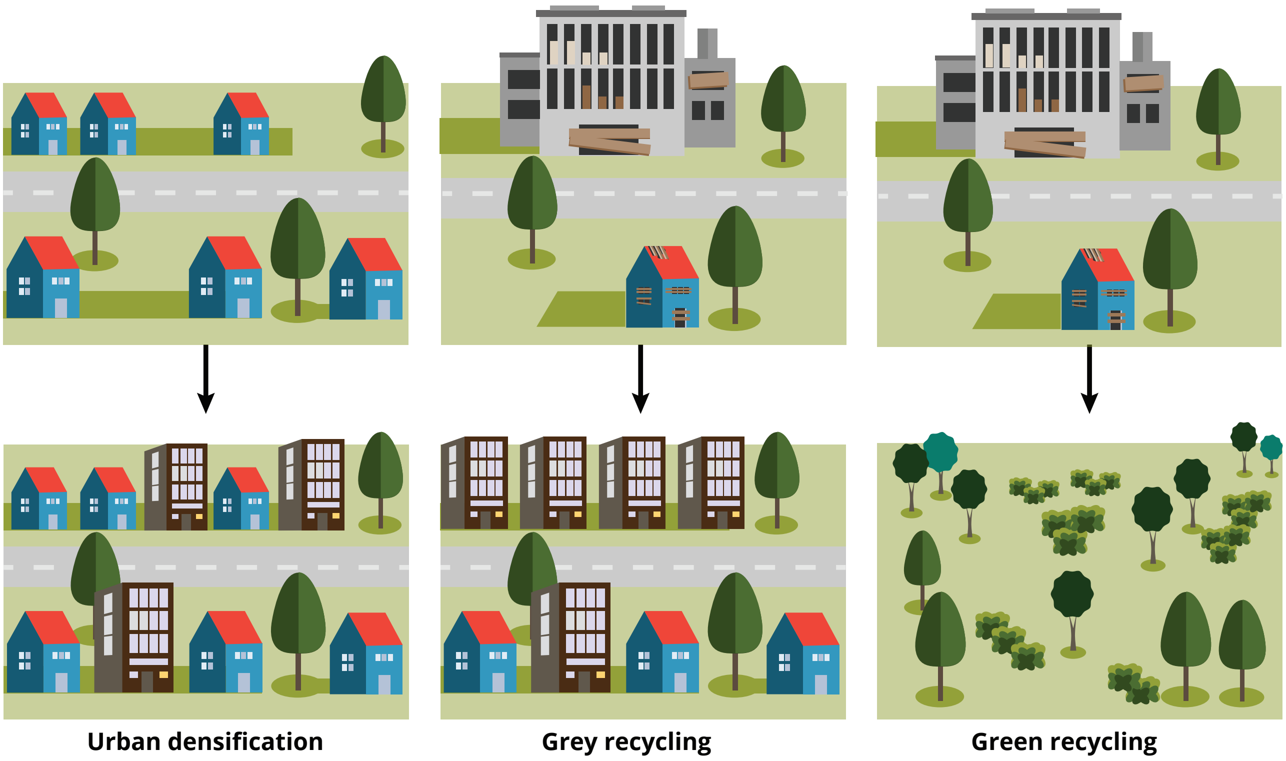
For the first time, the EEA has assessed the loss of biomass productivity due to soil sealing in 786 European functional urban areas for the period 2012-2018 (EEA, 2021a; EEA dashboard, 2021d).

To investigate land use efficiency, Haase and Lathrop (2003) suggest several **sub-indicators**, which can be derived from land take and/or sealing:

- density of new urbanisation;
- loss of prime farmland;
- loss of natural wetlands;
- loss of core forest habitat;
- increase in impervious surface area.

([33]) https://www.eea.europa.eu/data-and-maps/data/copernicus-land-monitoring-service-imperviousness-2

Figure 9.2 Land recycling types



$$\text{Land recycling (ha)} = \frac{\text{land densification + grey recycling + green recycling}}{\text{land take}}$$

Source: EEA (2021a).

9.2.3 Satellite methods

The most common method of measuring soil sealing is based on the different reflective behaviour of sealed and unsealed surfaces. This NDVI method (normalised difference vegetation index) quantifies vegetation by measuring the difference between near-infrared light (which vegetation strongly reflects) and red light (which vegetation absorbs). The NDVI method is used to measure vegetation, drought and also sealed surfaces.

Healthy vegetation (with chlorophyll) reflects more near-infrared and green light than other wavelengths, but it absorbs more red and blue light (thus, vegetation appears green). Satellite sensors such as Landsat and Sentinel-2 both have the necessary sensors.

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, which means no vegetation. An NDVI approaching +1 (0.8-0.9) indicates the highest possible density of green leaves.

At the European level, 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 is based on counts of pixels of impermeable soil cover (thus soil sealing), which are then mapped as the degree of imperviousness (0-100%). Imperviousness change layers have been produced as the difference between the corresponding reference dates and are presented as degree of imperviousness change (-100% to +100%):

- Data are available on a 3-yearly basis since 2006, namely 2006, 2009, 2012, 2015 and 2018.
  Data have a resolution of 20×20m, and since 2018 a higher resolution of 10×10m. Change layers are available for the periods 2006-2009, 2009-2012, 2012-2015 and 2006-2012. They are, however, based on a coarser resolution of 100×100m.

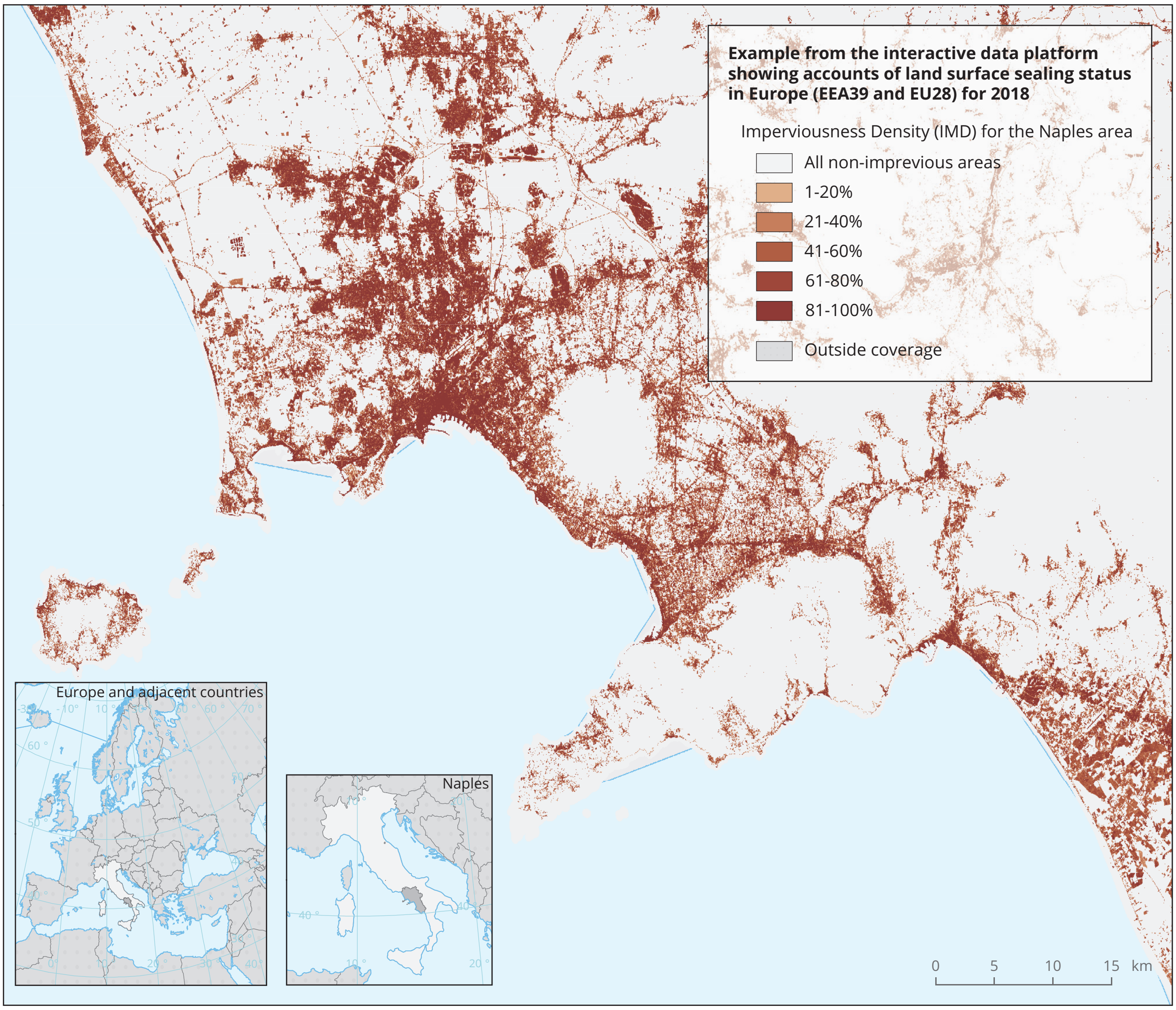
- The 2018 European data set on imperviousness change cannot be compared with earlier data sets due to its high resolution. It has been published in the format of interactive dashboards in combination with other assessments, i.e.:
  - soil sealing and ecosystem impacts (EEA dashboard, 2021a);
  - landscape fragmentation pressure in Europe (EEA dashboard, 2021c);
  - impact of soil sealing in functional urban areas (EEA dashboard, 2021d).

Based on the abovementioned data sets from the Copernicus Land Monitoring Service, the EEA publishes regular European assessments under the title 'Imperviousness and imperviousness change in Europe' (EEA, 2020). Data are available in an interactive format as maps and tables for the abovementioned reference years as absolute values or as changes for defined time periods (EEA dashboard, 2019; see Map 9.1 with an example).

Limitations of the method

Satellite methods are useful for identifying soil sealing when it comes to detecting actually sealed sites. However, satellite methods have limitations, in particular if tall vegetation conceals sealed areas or clouds disturb the reflection.

Map 9.1 Example from EEA's interactive data platform showing accounts of land surface sealing status in Europe (38 EEA member and cooperation countries and the UK) for 2018



Reference data: ©ESRI

Source: Copernicus (2018); EEA dashboard (2019).

9.2.4 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 by calculating average values from multiple sampling. 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.

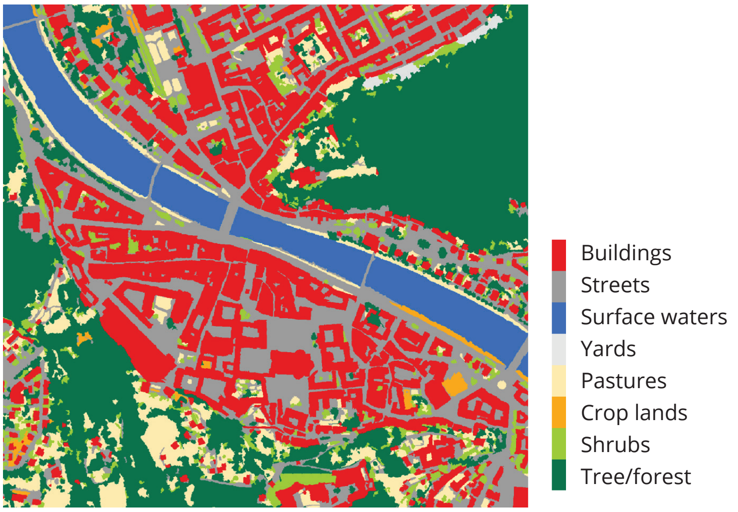
Figure 9.3 shows an example of this method. On the left-hand side standard sealing indices for specified land use classes are given. The right-hand side shows a map with the same land use classes depicted as coloured polygons. The overall sealing rate can be calculated by summarising the sealing rate of each polygon.

Limitations of the method

Cadastre data always depend on the precision of the data collection; the data set might lag behind in time as new data entries have not been integrated. A change in the nomenclature can also make the monitoring very difficult. With regard to aerial pictures, it is often the case that area-wide pictures or regular time series are not available.

Figure 9.3 Example of 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
Graveyards	35



Source: Monitoring of soil sealing in Austria. © Enriched LISA Landcover by GeoVille 2017.

9.2.5 Comparison of national and European monitoring of soil sealing

While several EU countries monitor land take at the national level on a regular basis, soil sealing is determined by only very few countries through surveys other than Copernicus. Table 9.3 shows the national soil sealing data available for the year 2015. National data generally reflect higher sealing rates, which leads to the conclusion that EEA-Copernicus data do not capture smaller structures and therefore underestimate soil sealing. Figure 9.5 shows three examples comparing the EEA-Copernicus data set with aerial pictures and indicates which structures were not captured by the EEA-Copernicus layer.

In the quality check of the 2015 high-resolution layer for soil sealing (Figure 9.4), it is obvious that smaller structures, such as dispersed single-family homes and smaller roads are not captured by this data set. However, it becomes clear that the higher resolution of the new data set (from 2018 onwards) overcomes this deficiency.

Table 9.3 Available national soil sealing data compared with Copernicus data, 2015

		Soil sealing (area, %)			
Country	Country size	National method		EEA/Copernicus	
Belgium (Flanders)	13,625km²	1,935km²	14.2%	1,212km²	8.9%
Austria	83,882km²	2,298km²	2.7%	1,475km²	1.8%
Luxembourg	2,593km²	176km² ([a])	6.8% ([a])	49km²	1.9%

Note: ([a]) Refers to the year 2018, as there are no data for 2015.

Figure 9.4 Comparison between EEA-Copernicus high resolution layers 'Degree of Imperviousness 2015-2018'

Small scale rural settlement and agricultural landscape in Austria (South of Obertrum am See).

2015 imperviousness, 20m pixel size. Omission errors occur where existing buildings are not captured as sealed, and small agricultural roads are below the minimum mapping unit (MMU).

2018 imperviousness, 10m pixel size. Buildings and settlements are much represented more consistently; some agricultural roads are still below the MMU.

Peri-urban Area west of Copenhagen, DK (Taastrup and Albertslund).

2015 imperviousness, 20m pixel size. Some possible omission errors occur where existing buildings are not mapped as sealed; large roads/highways are only partially captured.

2018 imperviousness, 10m pixel size. Buildings, settlements and road infrastructure are represented more consistently and in greater detail; more textural/structural detail in the settlement area.

Legend: 100% Imperviousness density

1% Imperviousness density

Source: EEA.

9.3 Baselines and target values

9.3.1 Land take Indicator

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. In addition, 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 rate of soil sealing or land take for a defined region or country. The rate for soil sealing or land take is usually expressed in an annual average 'hectares per day' (see Table 9.4 for some examples).

Only a few European countries have so far set baselines and target values for land take and hence implicitly soil sealing, and in most cases these targets are less strict than the EU target of 'no net land take by 2050'. So far only Belgium (Flanders), Luxembourg and Switzerland have set targets in line with the EU objective.

Even moderate amounts of land take will result in considerable negative consequences if continued over decades. We therefore suggest that all European countries set interim targets on their way to achieving no net land take by 2050. Interim targets would help to achieve a gradual reduction in both land take and soil sealing.

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 given in the second column of Table 9.4

Table 9.4 Current targets and baselines for soil sealing/land take in selected European countries

Target	Indicator	Source
Achieve no net land take by 2050	Land take (km^2) per 3-year period	EU Roadmap to a resource efficient Europe ([a])
To decrease land take gradually: 2016: land take 6ha/day (baseline) 2025 interim target 3ha/day 2040 final target 0ha/day ('land take neutral')	Average annual land take measured in hectares per day	Flanders Strategic vision of the spatial policy plan of Flanders ([b])
To reduce annual land take to a rate of 2.5ha/day by 2030 and to compensate unavoidable soil sealing	Average annual land take measured in hectares per day	Austrian government programme 2020-2024 ([c])
To reduce land take for settlements and traffic routes to less than 30ha/day by 2030 (at present 52ha/day as a 4-year average from 2016 to 2019)	Average annual land take measured in hectares per day	German sustainability strategy 2016 ([d])
To reduce land consumption from 1.3ha/day (average 2000-2006) to 1ha/day by 2020 and 0ha/day by 2050	Average annual land take measured in hectares per day	Luxembourg ([e])
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 ([f])
To stop net land (soil) take ('use') by 2050	Not yet defined	Switzerland ([g])

Notes: ([a]) EC (2011, p. 15, milestone 4.6).

([b]) *Strategic vision of the spatial policy plan of Flanders* (https://www.vlaanderen.be/publicaties/beleidsplan-ruimte-vlaanderen-strategische-visie-geillustreerde-versie), p. 36.

([c]) Austrian government programme 2020-2024 (https://www.bundeskanzleramt.gv.at/bundeskanzleramt/die-bundesregierung/regierungsdokumente.html), p. 104.

([d]) German sustainability strategy 2016 (https://sustainabledevelopment-deutschland.github.io/en/11-1-a).

([e]) Un Luxembourg durable pour une meilleure qualité de vie (https://environnement.public.lu/dam-assets/documents/developpement-durable/Un-Luxembourg-plus-durable-pour-une-meilleure-qualite-de-vie-2010.pdf), p. 35.

([f]) The law of agricultural and fishery modernization (https://artificialisation.biodiversitetousvivants.fr).

([g]) Schweizerischer Bundesrat (2020, p. 22). This target implements SDG target 15.3 and the Seventh Environment Action Programme (no net land take). Compensation measures included, however, are based on qualitative requirements and measures rather than area related. Soil sealing is used as an indicator for land take until a national soil functions map is available.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10
Operational soil indicators for the monitoring and evaluation of soil health

This report presents a review of largely well-known indicators of soil threats, their definitions and applications, and the parameters that need to be measured so that these indicators can be derived and monitored. It also brings together scientific information about critical limits, beyond which soils are clearly unhealthy and unable to achieve the quality desired, i.e. soils that cannot perform their functions as expected.

While current national and EU-wide monitoring instruments can provide some of the soil parameters needed for monitoring soil threats in a representative manner, for example soil carbon, some others are not systematically covered (e.g. soil physical parameters measuring compaction). This leaves great gaps in our knowledge about the state of the environment, and the role soils play, and interpretations often depend on highly uncertain predictions.

The discussions about the data requirements for the various European Green Deal environmental policies and the Eighth Environment Action Programme emphasise 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 mediators, bioreactors and buffers for many pressures affecting human health and ecosystem functioning. The information required can come only from soil monitoring — measuring accurately the inputs and outputs as well as the biological, chemical and physical transformation and transport processes in the soil. The parameters which describe these processes evolve into policy-relevant information by being aggregated into indicators and coupled to critical limits for the potential expected benefits and services soils provide for the living environment.

10.1 Soil health indicators

10.1.1 Risk-based soil health assessment using critical limits

This review has studied the current science on soil indicators, processes and functions. A synthesis has been developed with the aim of enabling assessment of soil health for a new EU soil protection policy. The risk-based approach taken here is well known in the context of local soil pollution, where risk assessment is used to trigger remediation. That approach has now been extended to other soil indicators (here: mostly soil threats), where critical limits are available under specific site and land use conditions. The risk-based approach builds on the concept that the harm caused by degraded soils to ecosystems and human health can be prevented, i.e. that soils can deliver the ecosystem services expected of them. Indicators and critical limits provide the necessary knowledge to enable decision-making where preventive and restorative action will be needed. The approach is presented in detail in Chapter 1.

Thresholds are given as critical limits once a specific protection target is involved, such as ecosystems and human health. The approaches to risk and thresholds depend on the underlying soil processes, available policy targets, land use, etc. Therefore, different kinds of operational critical limits were found:

- critical limits that relate to a complex soil threat indicator directly: erosion;
- critical limits that refer to measured parameters that comprise a composite indicator — soil pollution, nutrient loss, acidification, compaction and soil biodiversity — which can be derived in two ways:
    - back-calculation of policy thresholds/targets (nitrate in groundwater, drinking quality standards) into critical limits in soils (the maximum acceptable concentration at the point of sampling in the soil, so that the threshold in the groundwater or surface water is not exceeded);

- science-based functional limits of soil parameters (e.g. acidification, soil biodiversity);

- critical limits that refer to a single parameter that is at the same time an indicator: soil organic carbon (SOC).

10.1.2 Soil health indicators in this report

Soil threats are monitored by an indicator set that is well established, easily comprehensible and of limited complexity. The indicators feed on soil physical, chemical and biological parameters from monitoring networks. By applying critical limits to soil threats (and their parameters) and focusing on specific endpoints, the impact of soil degradation on ecosystem services can be assessed (and each soil function does not necessarily need to be separately investigated). Parameters and indicators need to be responsive to management and disturbance in a measurable way.

Table 10.1 provides an overview of the findings of this report. In a healthy, undegraded soil, fully capable of delivering its expected functions, none of the thresholds would be exceeded.

Based on Table 10.1 (and this review), a more complete approach can now follow, including:

- filling the remaining gaps in the approach (e.g. water storage, soil biological indicators);

- improving the regional representativity of thresholds (regional validation);

- completing indicators and thresholds for all land use types (since existing thresholds do not cover all land use types).

Table 10.1 Overview of soil threat indicators investigated in this report

Soil threat	Indicator	Thresholds	Comment
Soil organic carbon loss			
Cropland	Falling below optimal SOC level	Light soils: <1.2% SOC Medium soils: 1.2-1.9% SOC Heavy soils: >1.9% SOC	SOC: clay ratio (Johannes et al., 2017): optimum SOC content as 10% of the clay content/vulnerability limit
Nutrient loss			
Agriculture	Exceedance of critical levels of mineral nitrogen (agricultural land)	NH_3 in air: 1-3mg NH3/m^3 NO_3 in groundwater: 50mg NO_3/l N in surface water: 1.0-2.5mg N/l	Mineral N: sum of available NH_4 and NO_3
Forest land	N limitation based on exceedance of C:N ratio	C:N ratio 20-25 Leakage from forests: 1mg N/l	Forest floor organic layer
Agriculture	Falling below of optimal phosphorus	P concentration: 25-35mg/kg (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)	Forest floor organic layer
Acidification			
Agriculture	Exceedance of critical pH levels	1. pH<4.5-4.7 (critical) 2. pH<5.0-5.5 (avoid)	1. Risk of Al toxicity 2. Limited availability of Ca, Mg, K and P
Forest land	Exceedance of critical inorganic Al levels	Base cation (Bc):Al ratio = 1 (0.5-2.0)	Base cations are Ca^{2+}, Mg^{2+} and K^+

Table 10.1 Overview of soil threat indicators investigated in this report (cont.)

Soil pollution			
All land uses	Exceedance of screening values for critical risk from heavy metals and organic pollutants	Updated values for Cd, Cu, Pb and Zn (mg/kg) in this report: By country Database developed (Cd, Cu, Pb, Zn, As, Hg, Ni, Cr) Organic pollutants	Country-specific values vary broadly and are not necessarily comparable Stratification by land use and soil texture
Soil erosion			
Agriculture	Exceedance of actual rate of soil loss by water erosion	2t/ha/year for shallow soils (<70cm depth) 4t/ha/year for deeper soils (≥70cm)([a]) (soil loss tolerance)	Soil formation rate: 0.3-1.4 t/ha/year (Verheijen et al., 2009) Preliminary thresholds, derivation of site-adapted tolerable soil loss rates recommended The current indicator description in this report includes only soil erosion by water, whereas the threshold addresses all other erosion types
Soil biodiversity loss			
	Loss of soil biodiversity (sub-indicators)	To be developed: Exceedance of safe minimum standards of ecosystem conservation Exceedance of operating ranges (OR) for specific soil animals and microorganisms	Requires sub-indicators by species and/ or (functional) group
Soil compaction			
	Harmful subsoil compaction (sub-indicators)	Priority (sub)-indicators: Saturated hydraulic conductivity (Ks) <10cm/day Air capacity (AC) <5%	Exceedance of 'action values' (Zink et al., 2011) Secondary sub-indicators with available thresholds: bulk density, internal soil strength, air permeability and oxygen diffusion
Soil sealing			
	Sealed area per total land area	National targets to achieve 'no net land take'	

Notes: ([a]) Loss rates lower than 2t/ha/year are mandatory on soils adjacent to water bodies and/or soils with elevated levels of pollutants; such lower limits are needed to maintain water quality.
The current knowledge base covers a limited set of land uses and soil properties, for which thresholds are available; in the future, all relevant land uses and sites need to be covered.

10.1.3 Application criteria for the thresholds in this report

Some of the thresholds found here can be applied across larger gradients, countries and soil types (compaction, erosion, nutrients, acidification), although they have been largely developed for agricultural land, particularly cropland. It could be argued that the thresholds presented will be barely exceeded for other semi-natural land use types. However, such types of land use are exposed to atmospheric deposition, flooding, drought, fires and erosion; thus, unwanted inputs and effects occur, and thresholds could be exceeded. The thresholds presented here thus need additional validation and, in some cases, modification. For example, conditions for soils naturally rich (boreal) or low (Mediterranean) in SOC, require further analysis, since there is little evidence in the literature.

For this reason, the values presented here serve as orientation for identifying sensitive and degraded soils in the EU. Any definition of a legally valid threshold then requires a national validation procedure, the outcome of which either confirms the values presented here or results in new, region-specific thresholds. The latter would require a coordinated process among Member States, to ensure comparability.

10.1.4 Soil threats not covered in this report

Salinisation is characterised by an excessive increase of water-soluble salts in the soil. There is certainly a significant area of saline soils in Europe (primary — natural, and secondary — anthropogenic). Anthropogenic causes can be irrigation, hydrological modification, chemical additions and disposal of saline waste. Várallyay (2005) lists the input data required for the characterisation and risk identification of salinisation/sodification. Várallyay (2008) presents baseline and threshold values for soil salinisation, crop tolerance to salts, and the effects of salt on relative soil fertility. Salinity risk assessment methodologies are presented in Bloem et al. (2012).

Thresholds for wind erosion and other forms of soil erosion (this report has primarily focused on water erosion) are based on the concept of tolerable soil loss, and such thresholds consider all forms of erosion.

10.1.5 Outlook: definition and thresholds for soil functional indicators

Soil functional indicators are directly linked to a specific soil function and to ecosystem services (see Table 1.1). While there is still no agreement on an exhaustive matrix of such indicators and their thresholds, soil function assessments using pedotransfer functions have a long tradition and are an excellent basis for studying parameters and processes in the context of soil functions. A recent conceptual example of such an assessment is presented by Vogel et al. (2020). In that study, the potential and actual delivery of soil functions is predicted using static and dynamic soil properties, which are then scored to compare the performance of different soils. Among other functions, Vogel et al. (2020) also include the water storage function, which is not covered in this report. This is also one of the four functions investigated by Steinhoff-Knopp et al. (2020), who studied the effect of erosion on soil functions and ecosystem services. They suggested water storage capacity (mm) as an indicator for the water flow regulation function. Their approach is particularly interesting because it offers a local, site-specific approach to quantifying thresholds. A fully elaborated example to demonstrate the concept is yet to be presented.

10.2 Soil monitoring

The objective of soil monitoring in the context of the EU soil strategy for 2030 is to detect healthy and unhealthy (degraded) soils so that the rate of achieving the policy target (healthy soils by 2050) can be quantified. Thus, monitoring measures the success of halting degradation from unsustainable management (and climate change) — the non-deterioration principle — and the success of restorative practices (sustainable soil management).

10.2.1 Intensity of soil monitoring

In the run up to and follow on from the EU's 2006 soil thematic strategy, existing soil monitoring systems were reviewed, and the challenges for developing a common European monitoring system compiled (Van Camp et al., 2004c; Huber et al., 2008). Van Camp et al. (2004c) emphasised the need to develop a common baseline, to decide on a minimum parameter set, quality control, reporting and EU coordination. Interestingly, the parameters they suggested follow a tiered approach, covering all soil threats. This report investigates the most important soil threat indicators, offering an updated knowledge base (building on Huber et al., 2008).

Huber et al. (2008) presented an overview of 290 soil indicators for all soil threats, condensed into 60 selected priority indicators, as identified in the soil thematic strategy 2006. Twenty-seven of these indicators were tested against existing soil monitoring systems, with 20 being qualified to enter the envisaged European monitoring system (see also Table 1.4). Corresponding performance criteria were provided by Arrouays et al. (2008), including minimum detectable change and baseline and indicator thresholds where available.

More recent evaluations of existing monitoring systems in the EU have been conducted by Stolte et al. (2016), Van Leeuwen et al. (2017), Creamer et al. (2019), Bispo et al. (2020) and Faber et al. (2022). This work provides an extensive overview of existing national monitoring systems and concludes with minimum sets of parameters needed to describe soil quality and soil functions (in the case of the Landmark project; Creamer et al., 2019). The Landmark project concluded with the determination of optimal sampling densities to improve the representativity of the LUCAS Soil survey, which is particularly interesting for countries that lack monitoring systems.

Table 10.2 provides an overview of the physical, chemical and biological parameters needed to derive the soil threat indicators discussed in this report, based on threat-specific suggestions for monitoring approaches (in particular erosion, compaction, soil biodiversity). This list can easily be compared with, and combined with, knowledge about core indicator sets from the European studies mentioned above (e.g. Faber et al., 2022).

Soil monitoring in Europe faces the challenge of integrating different national and EU-wide soil surveys (for details see Bispo et al., 2020). Furthermore, different sampling regimes may exist within countries, such as between forest and agricultural soil monitoring. While these differences have their justification (e.g. sampling design addressing forest floors, sampling of peatland, maintenance of time series), linkages between surveys (observing changes in land use for certain sampling sites) and comparability across borders pose challenges for adapting the design and/or future implementation of these surveys. Example of such design modifications are:

- common terminology and methodologies (e.g. pedotransfer function)([34]);
- parallel laboratory analysis; development of conversion factors;
- spatial integration, e.g. through standardised delimitation criteria for spatial units;
- common representativity criteria (e.g. for organic soil or riparian soils).

With regard to different levels of sampling and analytical intensity, the term 'level' is preferred over 'tier', since the idea of sampling intensities largely follows the ICP Forests Expert Panel on Soil and Soil Solution (level I and level II forest soil condition monitoring).

([34]) For harmonisation of soil information, see also Baritz et al. (2012).

Three sampling intensities can be distinguished:

- **Level I**: sites where all general parameters are measured, e.g. large-scale topsoil surveys, with a central laboratory (LUCAS Soil, GEMAS (Geological Mapping of Forest Soils of Europe)) or based on a European network of closely calibrated national/regional laboratories (ICP Forests level I).
- **Level II**: investigations and monitoring of specific parameters and soil threats, e.g. types of erosion, soil biodiversity. Higher sampling densities allow improved identification of systematic errors, and higher sampling depth allows monitoring of subsoil processes.
- **Level III**: related to very specific problems, e.g. radionuclides, military sites, decontamination of specific industrial residues, 'hot spots' of anthropogenic or natural processes. In addition, local sampling and analytical capacity (e.g. analytics for farmers) can be involved and later integrated into larger scale surveys (involving local laboratories).

Table 10.2 Parameters for soil monitoring at different sampling intensity levels

Monitoring level	Level I	Level II	Level III
Soil threat		As for level I, and also	As for levels I and II, and also
Soil organic carbon loss	SOC and mineral carbon Total (organic) nitrogen C:N ratio Bulk density (derived with PTF) Texture class, stone content	SOC fractions Bioavailability of nutrients and pollutants GHG emissions Physical parameters (measured)	Refined local SOC monitoring Management types SOC cycling at ecosystem level (input/output)
Soil nutrient loss	Agricultural soils: • Total N, mineral N • Total P, available P: Pox/Al+Feox • Available K Non-agricultural soils: • C:N ratio, base saturation	Agricultural soils: • Cation exchange capacity • Base saturation Non-agricultural soils: • Soil solution concentrations	Agricultural soils: • Minor nutrients Non-agricultural soils: • As for level II
Soil acidification	Agricultural soils: • pH, clay content, SOC Non-agricultural soils: • pH, cation exchange capacity, base saturation	As for soil nutrient loss	As for soil nutrient loss
Soil pollution	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	Specific soil testing, e.g. reactive or available fractions, plastics, antimicrobials Balancing (inputs-outputs, e.g. modelling) to estimate/ validate accumulation	Very specific contamination problems, e.g. radionuclides, military contamination, large chemical facilities Site-specific risk assessment tools to predict actual and future effects (of specific risks such as food quality)
Soil biodiversity loss	Earthworms and Collembola	Parameters targeting functional diversity and DNA-based genetic diversity	Parameters describing complex biological functions (e.g. respiration, N and C mineralisation, microbial biomass)

Table 10.2 Parameters for soil monitoring at different sampling intensity levels (cont.)

Soil erosion (see also Table 7.4)	Modelling (using data on land cover/land use, geomorphological data, national soil data, rainfall)	Mapping visible soil erosion features Details on land use (e.g. ground cover)	Monitoring (measurements) of soil erosion (sediment loads): • Plot scale • Catchment scale • Sediment deposition in ponds, lakes or reservoirs
Soil compaction (see also Table 8.3)	Precompression stress (PTF) Soil rigidity ratio (PTF) Penetration resistance (PTF) Morphological features Soil organic matter (measured)	All basic soil parameters for PTFs are measured	Tensiometer, sensors at representative subplots Stress-dependent measurements
	Saturated hydraulic conductivity, air capacity, plant available water capacity (PTF)		As for level II, but with great sampling depth and more subsamples
	Soil texture/coarse fragments/ $CaCO_3$ (estimated) Rooting (estimated)		

Note: GHG, greenhouse gas; PTF, pedotransfer function.

Some soil threats such as compaction and erosion involve modelling, as well as other monitoring techniques, such as remote sensing, and more intensive sampling schemes to calibrate and validate the models while considering current land use and climate. Moreover, for monitoring SOC, remote sensing and modelling become more and more important and have the potential to better capture the variability of soil organic matter content, and to provide data at field/farm scale where the soil is managed (Castaldi et al., 2019).

The concept of sampling levels in soil monitoring was also discussed by the Eionet Task Force on Soil Monitoring, summarised and developed here:

- **Level I** could correspond to a large-scale Europe-wide sampling network, which consists of several country-specific constellations depending on the already existing national monitoring systems:
    - Several countries may adopt LUCAS Soil as their national system, while in others a combination of LUCAS and the established national monitoring system would create the most dense monitoring network**.** In the second case, comparability can be achieved only if the LUCAS analytical protocol is maintained at the national level (national samples from sites outside LUCAS Soil could even be provided to a central laboratory). A few countries may not see the need to supplement their own monitoring with LUCAS Soil. In that case consistency and comparability need to be established so that EU-wide evaluations based on LUCAS Soil (to cover all the EU territory in one time-efficient evaluation step) do not contradict national level I evaluations.
    - An alternative approach is to establish LUCAS Soil as the EU level I soil monitoring system, and any national activity would serve level II.

  National initiatives may help to fill remaining representativity gaps (e.g. as investigated by the Landmark project) by densifying the existing sampling grids. Sampling would be limited largely to the topsoil and include basic parameters, including macronutrients and metals; sampling density would be representative for all land use and soil (1:1 Mio) scale spatial combinations across Europe, based on international standards (CEN (European Committee for Standardization)/ISO (International Organization for Standardization)).

- **Level II** could then correspond to national monitoring networks, where there is a higher density of subsamples to encompass the local variability (thus avoid sampling errors). In addition, hot spots such as organic soils could be better captured, e.g. by narrowing sampling grids in wetlands. Sampling of deeper layers and a higher number of optional parameters (e.g. organic pollutants, emerging contaminants, soil biological parameters) are additional criteria for level II. Furthermore, soil biodiversity monitoring and organic pollutant monitoring have higher requirements for sampling, transport, storage and analysis, which are difficult to apply at level I (reference to monitoring levels for soil biodiversity; See section 6.2).

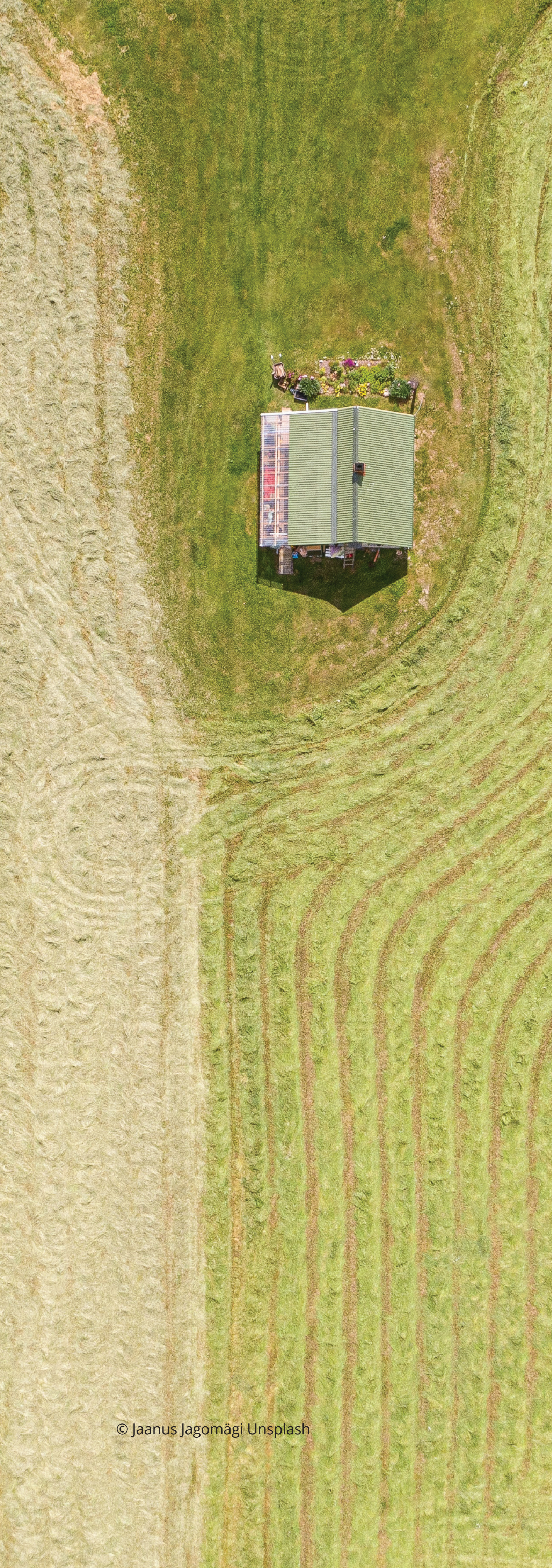
If countries adopt LUCAS Soil as their national monitoring system, they would supplement the LUCAS sampling points with an intensified (level II) sampling protocol (which still needs development; the experience of countries with intensive soil monitoring systems could help with this).

Level II serves a larger set of indicators and may be more accurate for detecting trends and calibrating and validating modelling, etc. However, the exact conditions for levels I and II still need to be elaborated between the European Commission and Member States, so that an integrated 'nested' system between levels I and II can be established. Table 10.2 may support this discussion. Certainly, the existing LUCAS Soil level I is already established, allowing an EU-wide harmonised topsoil assessment.

To allow integration between all sampling levels, agreed European protocols for sampling and analysis at levels I and II are needed. Experience of a combined scheme for sampling and analysis (EU standards/national standards) was collected during the repeated ICP Forests level I survey (Biosoil project under the Forest Focus Regulation ((EC) No 2152/2003)).

It can be expected that the representativity of the LUCAS Soil survey will be continuously improved in order to address emerging additional needs, while national expertise and specific plot extensions are needed to cover hot spots, deeper soil layers and other non-soil aspects of soil monitoring (e.g. crop quality, groundwater quality, vegetation composition, soil fauna).

The monitoring levels suggested here are not yet a fully operational level I and II system. This will still have to be developed. The options presented here do not fully correspond to the ICP Forests levels I and II; ICP Forests level II has much higher sampling density at the cost of reduced representativity with only about 10% of the level I plot density. The ICP Forests level II involves intensive monitoring of forest sites or forest stands with representative local sampling regimes (ecosystem monitoring) that can develop site-related element input-output balances.



10.3 Recommendations for soil monitoring and implementing soil-related indicators

Recommendation 1: We recommend that the critical limits presented here are investigated and validated and that regional specifications are developed where needed.

The risk-based approach to soil monitoring is not new, having been used for soil pollution, and thresholds have also been discussed for various indicators. However, the concept has now been expanded and elaborated for the monitoring of soil health, using critical limits for soil threat indicators and their physical, chemical and biological parameters. There are still gaps in current knowledge (for some land uses and some European regions); however, with additional validation, the approach is sufficiently well documented to serve as a methodical framework with critical limits as guidance values for Europe.

Soil is a complex medium, with highly variable local conditions because of its properties, historical and current land use, and climate. Many pressures affect soils, while their ability to function well — particularly if productivity is the only measure — is not properly understood or secured. In order to convince landowners and land users to take measures where soils are unhealthy, clear and well-defined criteria are needed to explain why a soil is considered unhealthy in terms of the various functions and services expected from soils. This is a challenge for participatory communication and awareness raising. Critical limits offer a transparent and plausible approach to determining whether a soil is healthy or not, and substantial investment in soil monitoring and regional validation of methods will be needed to succeed in cooperation with all stakeholders.

Recommendation 2: We recommend the development of a European soil indicator system based on the analysis of policy needs and experience from existing soil monitoring.

On that basis, level I and level II protocols for sampling and analysis can be developed, which use and integrate the experiences collected at EU and national levels.

Soil monitoring here is targeted to provide data on soil properties in a representative spatio-temporal approach, allowing the quantification and observation of pressure, status and impact indicators. Any soil monitoring system must be sufficiently robust and pragmatic to derive the necessary indicators. Because the different soil threats (and the underlying soil functions) require different input parameters and, in some cases, sampling approaches, not all indicators can be served by the same sampling design. However, at least for level I, the sampling requirements for a core set of parameters need to be agreed upon (see Creamer et al., 2019).

The following conditions for a European soil monitoring system need to be considered when developing protocols for sampling and analysis:

- **Core set of soil parameters**. As mentioned above, this has been widely analysed and suggested by different authors, so the list in Table 10.2 can easily be expanded for different indicators, and thus further specified. The work cited above is also based on the analysis of existing national monitoring systems and additional questionnaires (e.g. Faber et al., 2022), so that such a core parameter set is likely to find agreement and support. Table 10.2 contains some complex parameters that require evaluation functions to be derived (pedotransfer functions). There is still a need to revise, extend and further harmonise the European pedotransfer rules and functions database.
- **Indicators and parameters**. These must be clearly defined and comparable between different monitoring systems. Currently, national systems differ in their return intervals, sampling design (number and spatial arrangement of subsamples, sampling depth, representativity). In addition, several countries have not recently revisited their sampling locations and may even have stopped their monitoring programmes. A big effort is needed to integrate the reporting of indicators generated from these systems. Completed EU projects such as GS Soil and Landmark, national projects such as the Austrian LUCASSA, and experiences from European sampling systems, such as ICP Forests and LUCAS Soil, help to develop harmonisation tools and design options for an integrated system. As well as project consortia, expert networks exist (Eionet Thematic Group on Soil, stakeholder forum of the European Soil Observatory) that can help in building, testing and applying the required methods and tools.

Recommendation 3: We recommend the refinement of the soil threat approach through soil functional indicators and site- and land-use specific critical limits.

It has been demonstrated that the understanding of soil health requires information about how soils perform the functions expected from them. The current knowledge about soil threats enables an understanding of soil degradation by many stakeholders. However, soil functional indicators describe soil processes and their impact on ecosystem services in more detail (e.g. water dynamics in soil); a systematic hierarchy of soil functional indicators is still lacking. Investments into monitoring and research are also needed to significantly expand the knowledge base about critical limits in view of soil health.

- **Research challenge.** Knowledge about the impact of soil degradation on drinking water quality, human health, food quality, soil biodiversity and the ecological condition of ecosystems is still limited. Regarding the impact of soil pollution, the existing experience of the risk assessment of localised pollution must be expanded to risk assessment of diffuse pollution. Thresholds for all degradative processes and soil threats could follow a similar risk-based approach; for this, limit values should be defined in order to protect so-called 'endpoints', and to trigger restorative action. Thresholds are often defined at the level of protection targets (limit values in food stuffs, water quality), and transfer models are needed to determine corresponding critical limits at the point of measurement in the soil. Knowledge about such thresholds must be developed for key indicators, representative for all land uses, and comparable transfer models should be elaborated. Strategic partnerships and shared research infrastructures should be expanded; the successful European Joint Programme Soil (EJP Soil) serves as a blueprint for new research cooperations.

- **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, biodiversity, water and food security). Active engagement and agreement between the European Commission and Member States (and their neighbours) is needed to implement the new EU Soil Strategy 2030, with the ambition of achieving healthy soils by 2050, facilitated by a soil protection policy. At best, any new EU policy would re-emphasise already existing national efforts and highlight where such national initiatives still have gaps at the present time. Ideally, such a response would enable funding and support for national soil monitoring, which strives to be regionally representative. The Forest Focus Regulation could be a model case for establishing EU soil sampling as a demonstration project for soil monitoring.

10.4 Concluding remarks

Soil degradation is indicated by a reduction in or elimination of soil functions and thus the loss of the soil's ability 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 derive food from the soil and are in daily contact with it. Thresholds, i.e. critical limits, play an important role in understanding where soil functions have been harmed to a level that needs prevention, restoration or remediation; this can be indicated by the degree to which critical limits are exceeded, as shown in Figure 1.3. The loss of soil function may be indicated by reduced plant production (e.g. yield loss), reduced soil biodiversity or loss of soil stability (soil losses through erosion and landslides). However, except for soil erosion, soil degradation is generally not visible (e.g. chemical degradation). This means that proper sampling and analysis is fundamental for monitoring soil health.

Therefore, the approach to quantifying the degree of soil degradation by linking critical limits as thresholds and the current soil (functional) condition is a big step forward compared with the risk assessment schemes of the past. However, this method has several inherent challenges related to terminology, methodology and local conditions:

- While various indicators for soil threats have been proposed in the recent past, specifications for monitoring and evaluation are missing.

- There is still no consensus between countries on valid regional critical limits to be used as thresholds for specific soil functions.

- The methodology to link a specific threshold (via models) to the current condition of soil, or water, may differ between countries or groups of countries (e.g. Swartjes et al., 2009 about risk assessment of contaminated sites).

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

However, to evaluate the current soil status or the impact of relevant soil threats on the environment and human health, risk-based limit values are essential. At present, there is no consensus at EU level on unified critical limits for the listed soil threats. Although progress has been made, for example in developing effect-based critical limits for metals such as cadmium, lead, copper and zinc, national standards for soil protection within the EU still vary widely. One reason for not having a harmonised set of standards is the complicated interconnection between soil functions and site conditions (e.g. climate, soil fertility level), management measures (e.g. fertiliser management) and corresponding soil threats. In addition, views from Member States on targets or endpoints (e.g. water quality, food quality, ecosystem health) may also differ. This further complicates the quest for a harmonised generic approach. 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 Member State level) to effect-based critical limits (targeting, for example, human health as the endpoint).

Three recommendations are drawn from this review:

- **Available thresholds** need to be validated (confirmed and/ or improved) and gaps in indicators and thresholds filled.
- A European **soil indicator system is needed including a** protocol for **soil sampling and analysis**.
- Existing **soil monitoring** needs to be supported and improved as a tool to inform soil protection policies.

The corresponding maps for the test implementation of the indicators proposed in this report are available on request from the EEA.

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 Organization of the United Nations
ICP Forests	International Co-operative Programme on Assessment and Monitoring of Air Pollution Effects on Forests
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	United Nations Convention to Combat Desertification
WHO	World Health Organization

Other abbreviations

1OAO	One out, all out principle
AEI	Agri-environmental indicator
ASC	Achievable soil organic carbon sequestration
BAU	Business as usual
CAP	Common agricultural policy
Db	Bulk density
EAP	Environment Action Programme
EJP	European Joint Programme
Envasso	Environmental assessment of soil for monitoring (research project)
GAEC	Good agricultural and environmental condition (of land)
GEMAS	Geochemical Mapping of Agricultural and Grazing Land Soils (by EuroGeoSurveys)
Ks	Saturated water conductivity
Landmark	Sustainable management of land and soil in Europe (Horizon 2020 research project, 2015-2019)

LTE	Long-term experiment
LUCAS	Land Use and Coverage Area Frame Survey
LULUCF	Land use, land use change and forestry
MAES	Mapping and assessment of ecosystems and their services
MAOM	Mineral-associated organic matter
NDVI	Normalised difference vegetation index
NEC Directive	National Emissions Ceilings Directive
NRC Soil	Eionet National Reference Centre on Soil (since 2022 Eionet Thematic Group on Soil)
OR	Operating range
POM	Particulate organic matter
PR	Penetrometer resistance
PSI	Critical phosphorus saturation index
PTF	Pedotransfer function
Recare	Preventing and remediating degradation of soils in Europe (research project)
RUSLE	Revised Universal Soil Loss Equation
SDG	Sustainable Development Goal
STE	Short-term experiment
SOC	Soil organic carbon
SOM	Soil organic matter
SSV	Soil screening value
USLE	Universal Soil Loss Equation
VESS	Visual evaluation of soil structure
WFD	Water Framework Directive

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